

# Ecosystem Management:

Rare Species and Significant Habitats

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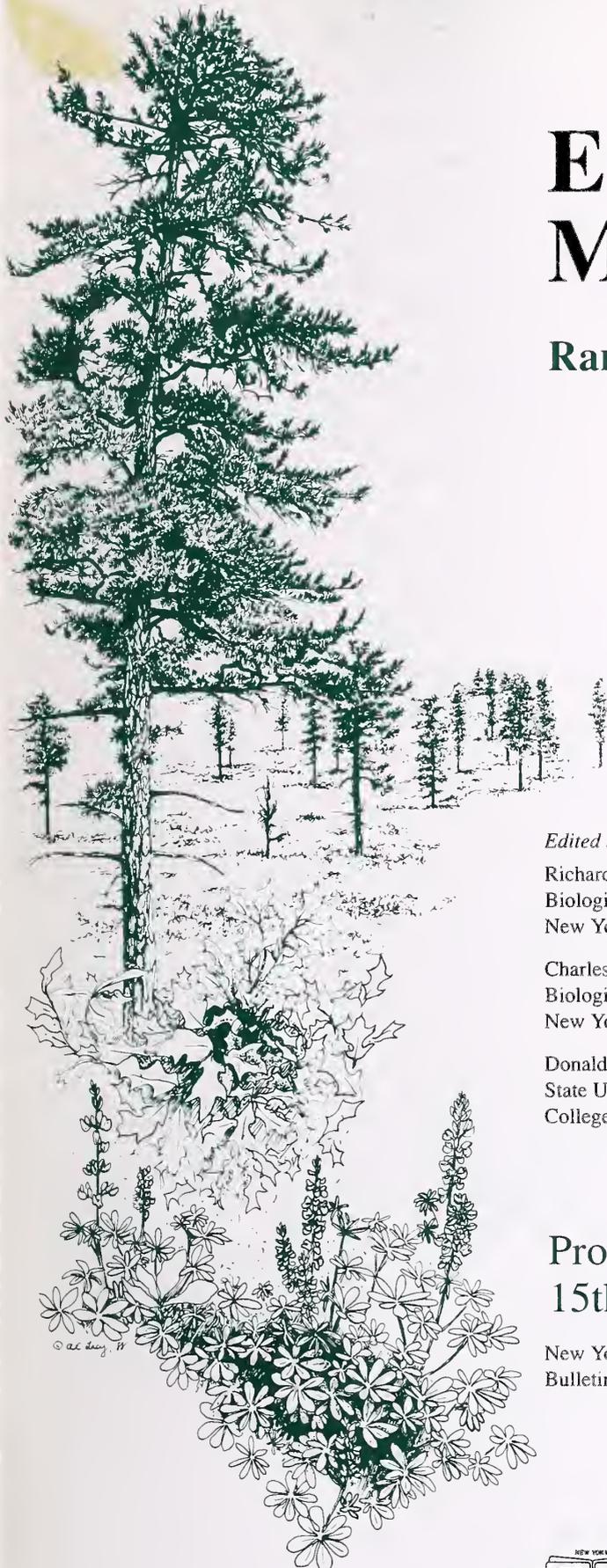
Donald J. Leopold  
State University of New York  
College of Environmental Science and Forestry

Proceedings of the  
15th Annual Natural Areas Conference

New York State Museum  
Bulletin No. 471



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## PREFACE AND ACKNOWLEDGMENTS

THE PUBLICATIONS comprising this volume are based on presentations made at the 15th Annual Natural Areas Conference/10th Annual Meeting of the Natural Areas Association, held at the State University of New York College of Environmental Science and Forestry, Syracuse, NY, from June 6 to 9, 1988. The Natural Areas Association is a nonprofit, professional organization for individuals involved in the identification, evaluation, management and protection of natural areas. Over 600 people attended the conference, where more than 100 papers, 25 posters and 10 workshops were presented.

The organizing committee consisted of Donald Leopold (cochair), Wendy O'Neil (cochair), Michael Birmingham, Alvin Breisch, Steven Clemants, Bruce Gilman, Thomas Lyons, Richard Mitchell, Pamela Otis, Horace Shaw, Charles Sheviak, Daniel Weller and Kenneth Wich. Sponsors included the Natural Areas Association, The New York State Museum, The Nature Conservancy - New York Field Office, New York State Natural Heritage Program, New York State Department of Environmental Conservation - Division of Fish and Wildlife and Division of Lands and Forests, New York State Office of Parks, Recreation and Historic Preservation, Community College of the Finger Lakes, The State University of New York College of Environmental Science and Forestry - Faculty of Environmental Science and Forest Biology and Office of Continuing Education, as well as the Center for Plant Conservation and Friends of the Burnet Park Zoo, Syracuse, NY.

Publication of these proceedings has been supported in part by contributions from the Faculty of Environmental and Forest Biology and ESF Continuing Education, State University of New York College of Environmental Science and Forestry, Syracuse, New York.

The cover art was donated by Anne E. Lacy. Our thanks to all concerned, especially to the authors who have participated in the writing of this volume.



# CONTENTS

Editors' Preface .....	Page iii
------------------------	-------------

## Chapter 1. GOALS AND STRATEGIES

<b>Setting Objectives — A Prerequisite of Ecosystem Management.</b> Richard L. Schroeder and M. E. Keller .....	1
<b>The Theory of Integrated Conservation Strategies for Biological Diversity.</b> Donald A. Falk.....	5
<b>Conservation Strategies: A Focus on Cooperative Action.</b> John D. Ambrose and Richard S. Mitchell .....	11
<b>Deer and Dauerwald Revisited: Aldo Leopold's 1935 Visit to Germany in Retrospect.</b> Warren W. Aney.....	14

## Chapter 2. HABITAT ASSESSMENT

<b>A Computerized Method of Priority Ranking for Natural Areas.</b> Linda C. Duever and Reed F. Noss.....	22
<b>Using Aerial Photography to Model Species-Habitat Relationships: The Importance of Habitat Size and Shape.</b> Jeffrey K. Keller.....	34
<b>Using Discriminant Analysis to Assess Critical Habitat for the Pawnee Montane Skipper.</b> Susan M. Galatowitsch .....	47
<b>An Approach for Quantifying Habitat Characteristics for Rare Wetland Birds.</b> JoAnn M. Hanowski and Gerald J. Niemi .....	51
<b>A Modified Point-centered Quarter Sampling Technique: a Tool for Plant Community Classification and Evaluation.</b> James G. Kooser and Wilson T. Rankin .....	57
<b>Stand Structure of an Old-growth Upland Hardwood Forest in Overton Park, Memphis, Tennessee.</b> James M. Guldin, J. Ritchie Smith and Lissa Thompson .....	61

## Chapter 3. MONITORING AND INVENTORY

<b>Radio Telemetry Techniques Applied to the Bog Turtle (<i>Clemmys muhlenbergii</i> Schoepff 1801).</b> James T. Eckler, Alvin R. Breisch and John L. Behler.....	69
<b>Response to Conspecific, Roadside Playback Recordings: an Index of Red-shouldered Hawk Breeding Density.</b> Glenn Johnson and Robert E. Chambers .....	71
<b>The Use of Departure Analysis to Protect Rare Species and Significant Habitats: An Example.</b> Owen R. Williams, Alice E. Johns and Stanley L. Ponce.....	77
<b>Status of Threatened Spruce Grouse Populations in New York: a Historical Perspective.</b> Robin P. Bouta and Robert E. Chambers .....	82

<b>The Raptor and Raven Community of the Los Medaños Area in Southeastern New Mexico: a Unique And Significant Resource.</b>	
James C. Bednarz, Timothy Hayden and Timothy Fischer .....	92
<b>Techniques Used for the Inventory of Rare Ecological Communities in New York State.</b>	
Carol Reschke .....	102
<b>Use of a County Soil Survey to Locate Remnants of Native Grassland in the Willamette Valley, Oregon.</b>	
Edward R. Alverson.....	107
<b>Preliminary Results of a Program to Monitor Plant Species for Management Purposes.</b>	
John Schwegman .....	113
<b>Monitoring <i>Gentiana saponaria</i> L. (Gentianaceae), an Endangered Species in Ohio.</b>	
Jennifer L. Windus.....	117
<b>Progress of Rare Plant Monitoring, Assessment and Recovery in Alberta, Canada.</b>	
Lorna Allen .....	119
<b>Attaching the New Jersey Natural Heritage Program Data Base to a Geographic Information System.</b>	
Thomas F. Breden, Jane E. Saks and Jack Schooley .....	122
<b>A Modern, Computer-accessed Flora of Pennsylvania: a Tool for Resource Managers.</b>	
Ann F. Rhoads.....	125

#### Chapter 4. DEMOGRAPHY AND SPECIES BIOLOGY: RARE ANIMALS

<b>The Dangers of Being Few: Demographic Risk Analysis for Rare Species Extinction.</b>	
Scott Ferson and Mark A. Burgman .....	129
<b>Viable Populations of Spotted Owls for Management of Old Growth Forests in the Pacific Northwest.</b>	
Mark S. Boyce and Larry L. Irwin.....	133
<b>The Red Wolf: Extinction, Captive Propagation and Reintroduction.</b>	
Donald E. Moore III.....	136
<b>Management Oriented Research: Marine Turtles in the Southeastern Unites States.</b>	
Nancy B. Thompson .....	140
<b>Assessing Cumulative Impacts to Wintering Bald Eagles in Western Washington.</b>	
Gary Witmer and T. A. O'Neil.....	144
<b>Western New York Bog Turtles: Relicts of Ephemeral Islands or Simply Elusive?</b>	
David E. Collins.....	151
<b>Criteria for the Introduction of the St. Croix Ground Lizard.</b>	
A.J. Meier, R.E. Noble and P.J. Zwank.....	154

#### Chapter 5. SPECIES BIOLOGY: RARE PLANTS

<b>The Management of Rare Plants: Suggestions Derived from Paleoecological Studies of Late-Pleistocene Floras.</b>	
Norton G. Miller .....	159
<b>The Biology and Management of <i>Potentilla robbinsiana</i>, an Endemic from New Hampshire's White Mountains.</b>	
B.T. Fitzgerald, K. Kimball, C. Cogbill and T. Lee.....	163

✓ <b>Genetic Diversity and Population Structure of a Rare Plant, Northern Monkshood (<i>Aconitum noveboracense</i>).</b>	
Philip M. Dixon and Bernie May.....	167
✓ <b>Species Biology of <i>Lindera melissifolia</i> (Walt.) Blume in Northeast Arkansas.</b>	
Robert D. Wright.....	176
✓ <b>The Significance and Management of Relict Populations of <i>Chamaelirium luteum</i> (L.) Gray.</b>	
James M. Utter and Ann W. Hurst.....	180
✓ <b>Recovery of the Endangered Virginia Round-leaf Birch (<i>Betula uber</i>): a Decade of Effort.</b>	
Terry L. Sharik, Peter P. Feret and Richard W. Dyer.....	185
✓ <b>The Establishment of a New Population of <i>Pediocactus knowltonii</i>: Third Year Assessment.</b>	
Peggy Olwell, Anne Cully and Paul Knight.....	189
✓ <b>Biological Considerations in the Management of Temperate Terrestrial Orchid Habitats.</b>	
Charles J. Sheviak.....	194

## Chapter 6. ENDANGERED BRYOPHYTES

✓ <b>Why Rare and Endangered Bryophytes?</b>	
Richard E. Andrus.....	199
✓ <b>Rare Species of Pennsylvania Mosses: An Assessment.</b>	
Harold J. Webster and Christine Manville.....	202
✓ <b>An Endangered Species List for Bryophytes: Endangered and Threatened Species of Ohio.</b>	
Jerry A. Snider.....	205
✓ <b>Endangered Bryophytes in New Jersey: Determination, Protection and Management.</b>	
Eric F. Karlin.....	208
✓ <b>Flora Protection: The Question of Rare Mosses in New York State.</b>	
Steven E. Clemants and Edwin H. Ketchledge.....	211

## Chapter 7. MANAGEMENT LAW

<b>An Overview of Natural Areas Laws.</b>	
Linda R. McMahan.....	219
<b>The Importance of Class I Air Quality Designation in Natural Area Protection.</b>	
Mary K. Foley.....	222
<b>The Massachusetts Wetland Protection Act: Protecting Rare Wildlife Habitat.</b>	
Scott M. Melvin and Steven M. Roble.....	224

## Chapter 8. MANAGEMENT IN PRACTICE

<b>Problems Encountered while Mimicking Nature in Vegetation Management: an Example from a Fire-prone Vegetation.</b>	
V. Thomas Parker.....	231
<b>Mine Reclamation: Opportunity for Critical Habitat Development.</b>	
Fred J. Brenner.....	235
<b>Ecological Planning and Resource Management: a Necessary Partnership for Small Lake Restoration.</b>	
Bruce A. Gilman.....	239

<b>Some Effects of Beaver Flooding on Rare Swamp Birch (<i>Betula pumila</i> L.) Vegetation in Connecticut: 1984 to 1987.</b>	
Thomas J. Rawinski and Beth Lapin .....	244
<b>Management Strategies for Increasing Habitat and Species Diversity in an Urban National Park.</b>	
Robert P. Cook and John T. Tanacredi .....	248
<b>Species Management for the Closed-lip Penstemon.</b>	
Linnea Hanson .....	251
<b>Ecology and Management of a Timber Rattlesnake (<i>Crotalus horridus</i> L.) Population in South-Central New York State.</b>	
Allen Peterson .....	255
<b>Creation and Management of Artificial Foraging Habitat for Wood Storks.</b>	
Malcolm C. Coulter .....	262

## Chapter 9. STEWARDSHIP

<b>Prescribed Burning and Mowing of Coastal Heathlands and Grasslands in Massachusetts.</b>	
Peter W. Dunwiddie and Caren Caljouw .....	271
<b>Inventory, Acquisition and Management of the Human-modified Landscape of the "Punkhorn."</b>	
Gregory M. Hellyer.....	276
<b>Northern Wild Monkshood in New York: Practical Problems in Resource Management.</b>	
Frederick J. Gerty, Jr. and Nelson L. Sears .....	288

## Chapter 10. AGENCY AND PUBLIC INVOLVEMENT

<b>Researchers, Agency Attitudes and Wilderness.</b>	
Lisa M.B. Harrington.....	293
<b>The Role of Local Government in Habitat Protection: a Long Island Perspective.</b>	
Murray C. Wade and Thomas W. Cramer .....	298
<b>Interagency Cooperation Using The Landowner Contact Approach.</b>	
Gary W. Burnett .....	304
<b>Participation by Local Cooperators: Management of Nuttall Oak, Rare in Illinois.</b>	
Guy Sternberg .....	306
<b>Maximizing the Effectiveness of a Natural Areas Program in Influencing Public Policy: The Maine Experience.</b>	
R. Alec Giffen and Harry R. Tyler Jr. ....	310
<b>Using Continuing Education to Inform the Public about Resource Management.</b>	
Robert J. Kent .....	313

# **Chapter 1.**

## **GOALS AND STRATEGIES**



# Setting Objectives — A Prerequisite of Ecosystem Management

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**Abstract:** Management implies movement toward desired end results. Therefore, primary prerequisites of ecosystem management are: clear definition of the components of the system to be managed and establishment of the desired end conditions for those components. Several steps are recommended in setting ecosystem management objectives: (1) objectives should be set early in the planning stages of the project; (2) managers should decide exactly what resources are of concern and focus the objectives on these specific resources; (3) objectives should be set within a "top-down" framework to force the consideration of larger scale constraints and needs; (4) objective statements should be made clear and concise, and stated in such a manner that progress in meeting those objectives can be measured; (5) the process of setting objectives is dynamic and should be kept flexible to respond to new information. It may be difficult to set clear objectives, due to imperfect knowledge or a lack of adequate data on the resources of concern. A great deal of additional research is needed to quantify and explain important population, community, and ecosystem level processes in order to make such decisions. In the interim, however, resource managers should make the best use of available information and conduct their management efforts within the framework of well-understood, measurable objectives.

Pages 1 - 4. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

"Ecosystem Management: Rare Species and Significant Habitats." The name of this conference confirms a growing desire to direct our management efforts toward the ecosystem level. We support this general trend and believe that land acquisition, land management and environmental impact assessments would benefit from increased emphasis on considerations at the ecosystem level. Caution is warranted, however, before we proceed too rapidly in this direction.

According to Webster's dictionary, management is defined as the "judicious use of means to accomplish an end." Therefore, a prerequisite of ecosystem management is identification of the desired end conditions, *i.e.*, setting objectives. Most natural resource organizations and agencies have basic goals that reflect their philosophical position and direction. For example, general goals might be related to maintaining biological diversity or managing specific wildlife populations. Although such overall goal statements are necessary, they do not provide a clear description of the desired end condition of a management effort. Such general goal statements need to be refined into specific and workable objectives.

The process of identifying specific objectives will become even more important as we consider larger and more complex natural systems. The increasing emphasis in areas such as landscape ecology, cumulative impact assessment and community modeling indicates a strong trend toward addressing higher levels of organization (Forman and Godron, 1986; Noss and Harris, 1986; Williamson *et al.*, 1986; Schroeder, 1987). Clearly stated objectives will be critical to guide decision making in these areas. The idea of setting objectives is certainly not new. In our experience, however, this essential step is too often overlooked in studies or projects at both the ecosystem and lower hierarchic levels.

## FACTORS TO CONSIDER IN SETTING OBJECTIVES

Crowe (1984) describes the planning process as a cycle of four questions. Where are we? Where do we want to be? How will we get there? And, did we make it? The question, "Where do we want to be?" refers

specifically to setting end objectives, and it should be clearly understood from the start that these objectives will later be used to measure progress. Without such specific objectives, it is difficult to assess progress or evaluate the success of management efforts. We recommend that the following process be followed when setting objectives (Figure 1).

### Set Objectives Early in Planning:

Objectives should be set early in the project planning if they are to have any chance of being realized. The entire management effort is affected by the choice of objectives, and all subsequent decisions reflect the objectives. Requiring that objectives be considered early forces biologists and decision makers to state which resources are of interest. These early discussions lay the groundwork to ensure that the stated objectives are adequately considered in the management effort. Failure to state objectives early in the planning process can

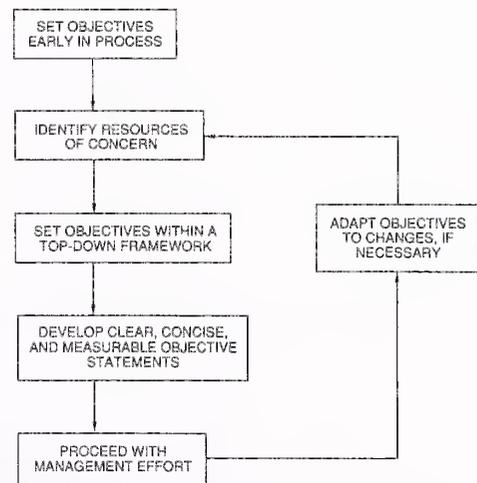


Figure 1. Flow chart of the objective-setting process for management efforts.

lead to misdirected efforts and inefficient use of time and personnel.

### **Identify the Resources of Concern:**

The phrase “ecosystem management” is, in itself, vague, and it could be argued that virtually every human activity is some form of ecosystem management. An important early step in setting objectives is to identify the management boundaries of the ecosystem specifically, *i.e.*, to set realistic boundaries on the management effort. Objective (goal) statements should be related to the particular resources of concern and should be attainable. For example, it is not possible to meet a general objective related to increasing population levels for all species of wildlife. Due to differences in life histories and habitat requirements, some species will increase and some will decrease in response to any given action. Nonetheless, our review of environmental impact statements and other assessment reports indicates that such all-encompassing and unattainable objectives are frequently stated or implied in these documents.

The process of defining objectives should force biologists to state what they think is significant and why. These statements should be clearly relevant to the overall goals or mission of the managing organization. Biologists might find it helpful to ask a series of questions to help identify the significant resources and to set realistic boundaries on ecosystem management efforts. The following checklist of questions is offered as a starting point in this exercise:

- Is the rarity of a species or community of concern?
- How important are endemic species/communities?
- Is the changing abundance of species/communities important?
- Is species richness of interest?
- Should ecosystem functions be considered?
- Are keystone species present?
- What species or communities are the best indicators of change in the ecosystem of concern?
- Can “natural conditions” be defined, managed for, and monitored?
- Are cumulative effects to be considered?
- Are there major landscape level changes to be considered?

An example of a study that illustrates this process concerns the development of a method for ranking habitats for oil spill response planning (Adams *et al.*, 1984). The study ranked coastal Louisiana habitats according to proposed protection priorities in the event of an oil spill. The overall goal was to minimize the ecological damage of an oil spill to the coast, but the process of ranking habitats required more specific objectives. Five ranking criteria were established as being relevant to minimizing the ecological damage of an oil spill. These were:

- (1) the time required for structural recovery of a habitat following an oil spill,
- (2) the length of time that oils persist in a habitat,
- (3) the extent of damage to a habitat resulting from attempted cleanup,
- (4) the rarity of the habitat, and
- (5) the number of socially important species supported by the habitat (this included commercial species, game species and endangered species).

These criteria provide the rationale for habitat protection and reflect specific objectives for minimizing ecological damage. The process of identifying these criteria is a valuable one in promoting discussions about what is important in the system of interest, and in leading to a clearer understanding of the biologist’s point of view. In reality, however, objectives must often be set within constraints

imposed by economics, politics or multiple-use demands.

### **Set Objectives within a Top-Down Framework:**

The elements of a “top-down” approach include an awareness of the relationship between national, regional and local project objectives and a consideration of cumulative and landscape-level concerns. A top-down approach provides a framework such that local objectives are set within the constraints of larger-scale needs. For example, an objective to manage for maximum species richness at the local (alpha) scale might result in lower species richness at a regional or national (gamma) scale (Samson and Knopf, 1982; Noss and Harris, 1986).

The significance of a community or the rarity of a species cannot be determined without consideration of the issue of scale. Is the community or species rare locally, regionally, or nationally? The extensive data bases on species and community distribution and abundance developed by The Nature Conservancy, various agencies, universities, museums and Natural Heritage or Natural Areas Programs have been used to address this question.

Consideration of landscape ecology and cumulative effects also reflects a top-down approach to setting objectives. Viewing projects or management efforts within the landscape context allows considerations of population demography, dispersal, and other biogeographical issues. Similarly, cumulative effects are often related to issues that occur across a broad geographic scale. For example, assessing the cumulative effects of development impacts on native wildlife species in bottomland hardwood forests would require a biologist to consider the impact of fragmentation of the habitat, changes in water quality, hydrology and many other types of disturbance.

### **Objectives should be Clear and Concise with Measurable Results:**

To be effective, objectives must be clearly worded and unambiguous, to ensure that there is no question about what is being measured or studied in the system of interest. Concise objectives focus the effort and provide the opportunity for a summary statement that is very useful when communicating with others. Presented in a measurable format, objective statements provide a yardstick against which progress can be evaluated. Conciseness and measurability are often best achieved by stating objectives in quantitative terms, whereas clarity requires only precise and unambiguous wording.

Barrett (1985) discussed objectives in his 19-step method to resource management problem solving. He stated that each objective should be specific and quantitative, and he provided the following examples of objective statements for a hypothetical project:

- The carrying capacity for an important indicator species should not be reduced by more than 10 individuals per 1,000 hectares.
- The biotic diversity of a particular guild should not be reduced by more than 10%.

Each of these statements meets the criteria of being concise, and each provides a quantitative measure against which projects or management efforts can be compared. Clarity could be improved by providing definitions for terms such as biotic diversity and carrying capacity, which are often interpreted differently by different individuals. Barrett suggests that objectives be developed by experts in the subject and study area.

In a review of community-level models used in impact assessment and land management, Schroeder (1987) noted that a “serious problem in the application of wildlife community models is the lack of clear objectives that can be tracked quantitatively with specific model outputs.” An example of an ambiguous objective statement is found in a procedure developed by Frye (1984) to provide a quantitative rating of

various plant community types in Texas. The procedure states that the components to be measured are those that “contribute to the ecological condition of the evaluated tract and resulting overall suitability for wildlife.” Although the procedure provides a concise objective statement and a quantitative output, it does not meet the criteria of clarity that would make it more useful for setting objectives. Terms such as “ecological condition” and “overall suitability” are ambiguous and subject to broad interpretation.

#### **Objective Setting Processes should be Flexible and Adaptable:**

Objective statements related to ongoing management efforts are, by necessity, limited by what we know of the system of interest. This is not to imply, however, that objectives must be static over time. The process of setting objectives should be flexible enough to allow objectives to be changed as either new information becomes available or as resource priorities change. With new information a biologist might add details to objective statements or add entirely new dimensions to previously limited objectives. Resource priorities might change through time due to political or biological reasons, and objectives need to reflect these changes. It is best to view the entire process as dynamic, rather than static. Changes in objectives, however, will have ramifications pertinent to ongoing studies or management efforts. The best strategy is to give ample consideration to objectives at the beginning of project activities and change them only for good cause.

### **POTENTIAL DIFFICULTIES IN SETTING OBJECTIVES FOR ECOSYSTEM MANAGEMENT EFFORTS**

Whereas the necessity for setting objectives is readily apparent, it also is important to acknowledge the numerous difficulties associated with doing so at the ecosystem level. The more one strives to be specific and quantitative about ecosystem management objectives, the more apparent these difficulties become. The stumbling blocks to setting ecosystem level objectives can be summarized as follows: (1) imperfect knowledge; (2) inadequate use of data; (3) conflicting objectives; and (4) institutional memory.

#### **Imperfect Knowledge:**

Our understanding of communities and ecosystems has increased tremendously over the past few decades. Despite these gains, much is yet to be learned in topic areas such as landscape ecology, minimum sizes of viable populations, critical ecosystem size, habitat fragmentation and sustainable productivity. As noted by Gottschalk (1975), in one of the first nongame bird symposia, biologists attempting to manage at the ecosystem level will often be frustrated when trying to develop specific, quantitative objectives. For example, consider an objective statement for preventing the loss of species as functional parts of a system (see Graul 1980; Conner 1988). Implementation of such an objective is difficult for two reasons: the phrase “functional part” is ambiguous and needs to be precisely defined, and the knowledge and data to implement such an objective are not readily available. Much additional research would be needed to quantify and explain important ecosystem-level processes. The process of setting specific, attainable objectives would be helpful in identifying future research needs in such a case.

#### **Inadequate Use of Data:**

Although greater research efforts in ecosystem management are always helpful, we see an even more compelling need to bridge the gap between research and applied biology. Existing knowledge is too

often not fully used by those making the decisions. Noss and Harris (1986) noted that with current levels of knowledge, biologists should be able to compare various conservation strategies in terms of their impacts on “native diversity.” Such comparisons might be well within the scope of existing knowledge, but, in our experience, such information is not often used by biologists making land-use decisions.

We need to consolidate existing species/community/ecosystem information into practical data bases for resource managers. Scott *et al.* (1987) proposed a structure for such a data base using a geographic information system. Their approach consisted of evaluating patterns of species richness, along with information on species abundance patterns, vegetative communities, levels of human disturbance and land-ownership patterns. They noted that data are available for such a system, but they have not been consolidated into a working format for managers.

#### **Conflicting Objectives:**

As noted previously, each piece of land cannot be managed for the good of everything. For example, management for forest interior birds might conflict with management for species preferring forest edges. Similarly, it might be impossible to manage for local wildlife diversity while at the same time managing for endemic species. Wagner (1977) noted that management at the ecosystem level requires that we manage for a desired mix of the various resources of concern. Selection of specific, quantitative ecosystem management objectives thus requires that we not only describe each objective, but that we describe the desired resource mix.

Where objectives conflict, it is often helpful to rank them. This ranking reflects human value judgments concerning the desired features of a community or ecosystem (see Powell, 1982). Systems such as the Multi-Attribute Tradeoff System (Brown *et al.*, 1986) can be used to quantify the differences in the rank order of various objectives. Those objectives that are highly rated should form the basis for the overall direction of the ecosystem management efforts of concern. The process of ranking conflicting objectives requires that resource managers document their logic for determining rankings. This documentation is extremely useful when communicating with others, and it also forces managers into critical thinking about what is important and why.

#### **Institutional Memory:**

Many of the concepts we have discussed have been addressed in one form or another in other documents (Phenicie and Lyons, 1973; Crowe, 1984). Why, then, do we see a lack of incorporation of these concepts in many environmental impact statements and resource management plans?

It seems that, as organizations or agencies undergo personnel changes and changes in leadership, there may be little effort given to transferring or updating ecosystem management objectives. Past objective statements may be overlooked or discarded and not replaced. If we are to make any progress, government agencies and others involved in ecosystem management must ensure that objectives are maintained, updated as necessary, communicated to new personnel and reiterated to biologists and managers involved in resource management activities.

### **CONCLUSIONS**

Setting objectives should be a prerequisite of any natural resource management activity. Management decisions generally reflect objectives of some sort, whether or not the objectives are explicitly stated. The value of clearly stated, quantitative objectives is that ecosystem management efforts can be directed toward a desired condition, and

successes and failures measured and evaluated. Based on these evaluations, management efforts can then be modified to correct deficiencies or updated to incorporate new knowledge.

Resource managers and field biologists should be required to conduct their management efforts within the framework of unambiguous, concise objectives. The very process of determining these objectives is instructive, in that setting objectives leads to a better understanding of the systems we are attempting to manage. In the process, data gaps are identified, research projects initiated, and more productive uses are found for limited personnel and financial resources.

The recent upsurge in interest in ecosystem-level management seems to hold great promise for solving many complex issues facing biologists and land managers. We do not view recent developments in this area as short-lived fads, but fundamental shifts in emphasis that have been unfolding over the past few decades. In this paper, we have not attempted to provide new approaches to this complex topic. Rather, our hope is that we have provided a strong reminder of the importance of conducting ecosystem management efforts within the framework of realistically bounded and clearly communicated objectives.

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# The Theory of Integrated Conservation Strategies for Biological Diversity

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**Abstract:** As the threats to biological diversity increase worldwide, conservation strategies (and programs based upon them) must be correspondingly strengthened. No single strategy sufficiently protects all levels of the biological hierarchy from the wide range of threats encountered. Moreover, the resources available to protect natural diversity are inherently limited. Because of the pervasiveness of environmental impacts and threats to species' survival, most conservation practice requires some active intervention into natural systems. Conservation decision-making should be approached as a question of optimal resource allocation, by protecting the greatest diversity for a given investment of conservation resources, with preference for strategies that are most cost-effective in protecting a desired biological entity. Thus, conservation has reached an era of diversity management, a significant change from the purist, preservation ideals of past centuries, and an approach requiring new kinds of information and programs.

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## INTRODUCTION: THE STRATEGIC CONTEXT

The North American continent has felt the impact of man for centuries. In 1583, the English mariner Sir Humphrey Gilbert complained in his journal describing Newfoundland that it was "an uncomfortable coast, nothing appearing unto us but hideous rocks and mountains, bare of trees, and void of any greene herbe." (Colbert, 1976). By 1850, over three-quarters of the land area of central New England — originally more than 95 per cent forested — had been cleared and converted to pasture and farmland or cut for ship building or construction timber (Tanner, 1974). Euroamerican settlers crossed the Continental Divide and reached the Pacific Northwest in 1805, and by mid-century the Anglo colonization of North America was extensive. In fact, it took little more than 100 years for the vastness of the continent to be permeated by human population or influence. By 1970, less than 6% of the total land area of the contiguous 48 states could still be considered truly wild (Fishbein, 1974). Each year, over two thousand square kilometers of remaining natural habitat, an area roughly equal to that of Rhode Island, is converted to agricultural or commercial use.

One measure of the biological impact of such an immense continental phenomenon is the anthropogenic extinction of species. Between 1790 and 1975 there were 200 documented plant extinctions in the United States, of which nearly half have occurred in this century (Mohlenbrock, 1983). Undoubtedly, more went unrecorded, and, according to a recent survey of regional botanists, as many as three times that number of extinctions could occur by the turn of the millennium (Center for Plant Conservation, in press). Moreover, virtually every major type of economic land use in the United States contributes in some way to the reduction of biological diversity (Falk, 1987b). The pace of land conversion, resource extraction and species extinction appear to be on the rise.

Given high rates of species and habitat loss, what do such data suggest for strategies to protect and conserve biodiversity? What kinds of measures can prevent species extinctions? How are we to choose what to do first, and where to place our priorities? The purpose of this paper is to provide a conceptual framework and theoretical basis for what may be called the "integrated conservation approach." Applications of this methodology have been provided elsewhere (Falk, 1987a; Falk and McMahan, 1988; Falk, in press).

## DIMENSIONS OF BIOLOGICAL DIVERSITY

There are many definitions of biological diversity, and therefore many different definitions of the goals of conservation. In the absence

of a rigorous and universally accepted definition of the object of conservation we are bound to miss important opportunities, and we may even work toward competing ends. Thus, a clear statement of goals is a prerequisite to effective action, and its formulation will require careful thinking about what we wish to conserve.

Most discussions of biological diversity emphasize description and measurement of diversity at the species level, while invoking the goal of preserving genetic diversity. Indeed, species and genetic diversity are two of the most important dimensions of biodiversity, but they are hardly the only relevant ones, and they are certainly not coextensive. Diversity exists at various levels of the genetic/ecological hierarchy as well: between individuals, populations, clines, ecotypes, communities and ecosystems. Each of these levels of biological organization embodies variation, not just within a taxonomic context but involving systems of interaction as well.

These distinctions are of great practical significance for conservation theory. For instance, if one is truly concerned only with global diversity at the species level, then on a per-dollar or per-acre basis it would be irrational to invest effort anywhere other than in the tropics, where indices of species diversity are high and land costs still relatively low. However, if one is committed to preserving global plant community types, then the geographic focus becomes worldwide, since communities vary richly and characteristically all over the globe. Similarly, if the focus of conservation is literally genetic diversity, then clinal and ecotypic variation within species must be of great interest, and the significant genetic polymorphism within some widespread taxa should be as important as the narrower range of variation found in many rare or endemic taxa.

These ideas are not meant to imply that any one of these levels of biological hierarchy is intrinsically more important or more suitable as a focus for conservation than another. On the contrary, sound conservation strategy must take into account the measures that will specifically address each level of organization. This is one of the more compelling arguments for an integrated approach.

## CONSERVING DIVERSITY AT THE GENETIC LEVEL

To illustrate the importance of devising conservation strategies to address specific levels of biological organization, consider the problems inherent in systematically protecting genetic diversity. Even within this level of the hierarchy there are tiers of organization. Genetic variation exists and may be analyzed at the level of the base pair, locus, allele, gene or gene complex, isoenzyme, and so on. Likewise, genetic variation may be measured or inferred by the use of enzyme

electrophoresis, chromosomal DNA or RLFP analysis, assessment of quantitative characters, morphological studies, and so on. Each of these tools and foci can produce a different profile of variation across a species or a population. Studies of the correlation between allozyme variation and quantitative traits continue to show an inconsistent correlation, which suggests that enzyme variation may not always be predictive for adaptive traits (Marshall & Brown, 1975). The best characters for study are either single-gene morphological traits or biochemical markers that are amenable to allozyme analysis, with confidence levels increasing as higher numbers of loci can be resolved.

No matter what the marker, such studies rest on a postulated general correlation between genetic variation and evolutionary potential (conversely, between extinction likelihood and limited genetic variation). But in general the relation of genetic diversity within plant populations to evolutionary potential and sustainable population size is still a matter in need of further research and badly in need of experimental verification (Barrett, in press; Huenneke, in press; Soule, 1986).

Once a hierarchic level and method of measurement have been selected, conservation efforts are still hampered by the lack of data about genetic variation in most wild plants, especially rare taxa. While a few groups of taxa have received some study at the population level (Hamrick, 1983; Hamrick *et al.*, in press), most non-economic, native species have received either little or no study of their population genetics. Among U.S. endangered species, few have received even basic electrophoretic study (Falk, in press). Thus, a conservation program seeking to conserve genetic diversity within one of these species would have either to generate the data by conducting an electrophoretic survey of heterozygosity in polymorphic loci, or to rely on theoretical models of the distribution and extent of genetic variation within groups (Hamrick *et al.*, in press; Center for Plant Conservation, 1986; Franklin, 1980). The problem with the latter approach is that valuable conservation funds may be committed without any certainty of the extent of diversity actually to be conserved. Where genetic variation between populations of a given species is concerned, this can have great practical consequences, since the additional investment in collecting time or acquisition of additional acres of habitat might multiply project costs significantly.

Without knowing what proportion of heterozygosity (by any measure) is found within each population, or how much is distributed among populations, it is impossible to determine how many populations should be sampled or protected (Falk, in press; Jain, 1975; Marshall & Brown, 1975). In the case of self-pollinating, annual species with isolated population distributions and limited seed or pollen dispersal, populations are likely to be highly distinct genetically. Consequently, more populations would have to be sampled to develop a complete profile of the species, or more populations conserved to protect the full range of genetic variation within the species (Bradshaw, 1975). Conversely, over 95 per cent of the variation (at an allelic frequency 0.05) in an outcrossing, widely dispersed generalist species may be found within a single population. The result is that the marginal additional genetic diversity within each successive population sampled or protected may be extremely low (Brown, in press; Center for Plant Conservation, 1986; Hamrick, 1983; Franklin & Briggs, 1980). Such data must be taken into consideration in developing conservation strategies if we are to allocate resources efficiently.

A comparable illustration is that of crop germplasm collections, where consideration of the actual genetic content of additional accessions has in many cases been so overlooked that many major crop collections (e.g. barley, wheat, sorghum and many legumes) contain three to five times the number of accessions needed to fully represent varia-

tion in the genome (Strauss *et al.*, 1988). Such high degrees of redundancy mean additional costs with no real gain in genetic diversity, and represent wasted resources that could be better directed toward conserving unprotected genetic resources.

These considerations apply equally well to the selection and design of a system of *in situ* preserves as to sampling for offsite genetic collections. The underlying principle of maximizing the genetic information per unit of conservation resource is the same. Furthermore, there is a normative issue built into such decisions: how much genetic variation is "enough" to conserve? If a sample of fifty plants of an outcrossing species captures 95 per cent of the alleles with a frequency 0.05, should that be considered an adequate sample? Implicit in this question is the ecological or evolutionary significance of rare alleles (Huenneke, in press; Marshall & Brown, 1975). A question unresolved in the literature is: by what standard may we conclude that the protection of 95 per cent of alleles at frequencies of 0.05 is biologically adequate? This is clearly an area in which the general uncertainties in models of plant population genetics and evolution make the design of biologically sound conservation programs exceedingly difficult. The Center for Plant Conservation (1988) has proposed to address this deficiency.

As in biological diversity, there is variation in structure and programs-implementation for conservation. If we are to devise effective conservation strategies, an understanding the biology of natural systems is not sufficient. We must also be able to analyze and describe threats to biological diversity, and to apply the appropriate range of tools and logic to any given conservation problem.

## DIMENSIONS OF THREATS TO BIOLOGICAL DIVERSITY

Although it is beyond the scope of this paper, the "taxonomy" of threats to diversity must be better understood if such threats are to be systematically addressed. Quite simply, conservation strategies can never be optimally effective if we do not have a firm grasp of the dynamics of species decline and extinction and the causes behind them. Except by chance, one cannot solve a problem one does not understand.

Threats to diversity fall into several natural groupings: causes related to competing land uses, pollution or degradation of abiotic systems, direct exploitation of biological resources and threats involving ecological processes. In the United States, competing land-uses provide the greatest number of threats and the greatest cumulative impact. For example, causes of decline cited in Endangered Species Recovery Plans by the U.S. Fish and Wildlife Service include every major type of economic land use in the United States (as well as many that are decidedly non-economic), primarily resource extraction of many forms and commercial and residential land development (Falk, 1987b). Other potential threats include air and water pollution, acid precipitation, stream siltation and contamination, overcollecting for private or commercial use and competition from introduced plants or herbivores, although, for many of these, empirical data are often anecdotal or absent.

What is significant about this array of impact sources on natural systems is that most are as affected by land management practices as by outright ownership. For example, federal agencies manage approximately 47% of the land west of the Rocky Mountains, including many species-rich areas of California, the Great Basin, Colorado Plateau, Sonoran Desert and the Pacific Northwest. At issue in these areas is not so much who owns the land, but how it is managed and used; multiple-use lands managed by agencies responsible for commodity production (e.g. National Forests, Bureau of Land Management lands) are

particularly susceptible to land-use decisions that adversely affect biotic diversity or the integrity of national systems.

In order for an analysis to be useful, threats to diversity should be sorted according to the level of the biological hierarchy that they affect. For instance, the fungal pathogen that attacks the conifer *Torreya taxifolia* Arn. in its natural habitat along the Appalachian River in Florida and Georgia affects that species severely, to the extent of preventing reproduction, but does not attack other species in the community (McMahan, 1989; U.S. Fish and Wildlife Service, 1987).

A further complexity in assessing the impact of various human activities is the difficulty in distinguishing natural and anthropogenically induced decline. The true endangerment status of species such as *Isotria medeoloides* (Pursh.) Raf. that undergo irregular, long-period fluctuations in population size may be extremely difficult to ascertain, especially in the absence of reliable baseline data. Correspondingly, where large amounts of potentially suitable but degraded habitat remain (as in many overgrazed areas in the Southwest and Intermountain regions), it may be nearly impossible to estimate the natural range and density of many rare species. Many federal land-managing agencies have only an incomplete inventory of the endangered species under their control; even fewer have any organized overall plan regarding them. Where such knowledge is lacking, we run the risk of treating a symptom (such as a reduction in population size) rather than a cause of the decline.

Assessment of threats and their use in devising conservation strategies are hampered by the inadequacy of our databases in representing site-specific and species-specific information. Even the best of the conservation databases in the U.S., such as the state Natural Heritage Inventories and the National Diversity Database of The Nature Conservancy, include only general or summary information about rates and causes of decline. To assemble more detailed information of this kind for all endangered species and communities would represent a truly massive undertaking that would probably exceed all available resources. The sections on threats to species survival in U.S. Fish and Wildlife Service Recovery Plans illustrate the minimal level of detail that would be required for such a project, but in many cases even this level is inadequate (Cook, 1988). Without such a system we must rely on much less comprehensive and systematic methods to set conservation priorities. Improvement of data relating threats and diversity should be part of any national program to protect biological diversity, since they provide a basis for integrated methods of conservation.

### THE CONSERVATION SPECTRUM

The early days of American conservation were often characterized by sweeping legislative measures encompassing immense areas in a single act. The creation of Yellowstone and Yosemite National Parks and the Adirondacks Forest Preserve epitomize this era of conservation. While legal protection of large areas is still possible (and in some cases critical, as in the case of the Alaska National Interest Lands and Conservation Act), land acquisition has increasingly turned to protection of smaller, high-quality interstitial preserves of exceptional ecological value. Moreover, since the U.S. government already controls approximately 27% of the land area of the continental U.S., acquisition of land *per se* is often not the critical step for biological conservation. For this reason, The Nature Conservancy and many other conservation organizations often now enter into cooperative management agreements concerning land held by other agencies to augment their outright acquisition activity.

During the past few years, there has been increasing emphasis on

land management as well as acquisition. This trend reflects an awareness of the need to intervene in biological processes (such as succession or fire) for conservation purposes, and a willingness to do so. The term "managed natural area" is no longer considered an oxymoron. However, management is an information-intensive process requiring precisely the kinds of data about biological systems and threats that are often lacking. Thus, managing for diversity compels lines of research and practice specifically directed toward conservation application (Soule, 1986).

Conservation increasingly consists not of a single tactic or objective, but rather an array of methods used in conjunction. Thus, the third component of the integrated conservation approach, following identification of the target level of biological hierarchy and assessment of the threat, is the selection of an ensemble of actions appropriate to a given situation. The range of measures, or "conservation spectrum", may be part of the palette of the contemporary manager (Figure 1). This menu of conservation practices is offered not to suggest rigid classes of intervention, but rather to illustrate the diversity (and continuity) of conservation actions. Also, it should serve as an antidote to the obsolete, dichotomous view of "*in situ*" vs. "*ex situ*" conservation, a polarity that has been transcended by the integrated model.

The term *wilderness*, correctly used, refers to whole ecosystems, including associated watersheds, airsheds and migratory pathways that are in a purely natural state, unaffected by human beings. Such areas are quite rare and probably should be considered to no longer exist in the United States, or indeed throughout much of the world (Jordan *et al.*, 1987; Turner, 1985). Increasing evidence of rapid and widespread colonization by man in North America following the most recent Pleistocene glacial recession suggests that human modification of ecosystems (perceived as "natural" by the culturally myopic early European settlers) may have been more pervasive than previously recognized. Perhaps the only true wilderness left in North America is the great expanse of boreal forest and tundra in Canada north of Great Slave Lake and the Athabasca River.

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WILDERNESS  
PROTECTED NATURAL AREA  
MANAGED NATURAL AREA  
RESTORED AREA  
PLANTATION  
NURSERY  
BOTANICAL GARDEN  
GREENHOUSE  
GERMPLASM BANK  
CELL OR GENE BANK

**Figure 1: The Conservation Spectrum.** Modes of conservation at the top end of this spectrum are characterized by low degrees of human intervention into large-scale systems, while conservation actions at the bottom end typically involve more intensive intervention and smaller-scale components of whole systems.

By contrast, many large *protected* areas have been legally safeguarded (perhaps fenced) but are otherwise only lightly managed. North American examples would include Wilderness Areas in the largest national and provincial parks, and regions such as parts of the Canadian Coast Ranges and northern Rockies, where overall anthropogenic impacts are still relatively light. Even in these areas, however, some management practices are used on a site-specific basis. The distinction between these areas and the next group may in some cases be

a matter only of degree. As with the categorization of any spectrum, there are many intermediate cases.

The vast majority of conservation land in the United States and Europe is best described as *managed natural area*. Management practices often directly involve modification either of succession (as in forest thinning and controlled burns) or species distributions (as with enhancement or reintroduction of populations). In both cases an understanding of the biology of the affected species and community is necessary if interventions are to be successful; it has already been noted such information is generally lacking, especially for rare species. The manager is thus too often forced to improvise or work empirically, and, although the results are frequently remarkable where rare species or fragile communities are concerned, a stronger foundation in biological knowledge would be comforting.

A type of intensive management practice involves ecosystem *restoration*. In terms of the intensity of human intervention, restoration exceeds what is customarily considered natural areas management. However, the differences again are often more in degree than in kind. The long-term management of a restored, prairie ecosystem may result in a community bearing a stronger resemblance to a natural area than does the intensive management of a historically documented remnant of original vegetation (such as the Bigelow Cemetery, a prairie remnant less than one-half hectare in extent, in Adams County, Ohio). Restoration poses a special set of practical and even ethical issues, then, that imply an entirely different relationship to nature (Jordan, 1986; Bonnicksen, 1988).

Although the maintenance of biological diversity is often not their primary objective, certain forms of large *plantations* and *nurseries* can serve conservation purposes. For instance, the U.S. Forest Service Institute of Forest Genetics in Placerville, California maintains genetic stock collections of most western conifers, including some provenance test plantings that are decades old. Although most large Forest Service reforestation projects are undertaken with timber production in mind, such massive plantings could easily be adapted to enhance biological diversity as well. In the case of rare trees such as *Pinus radiata* D. Don, offsite germplasm collections can help provide added insurance against further erosion of an already-depleted gene pool and provide breeding stock (Falk, in press).

*Botanical gardens* and *arboreta* likewise offer resources for the management and preservation of plant genetic diversity (Lucas and Syngé, 1977; Bramwell *et al.*, 1987; Elias, 1987; Falk, 1987a; Falk & McMahan, 1988). Many such institutions manage large areas of natural or semi-natural land in addition to cultivated areas; examples are: The Holden Arboretum (Mentor, Ohio) with over 3000 acres of woodland, and Pacific Tropical Botanical Garden (Lawai, Kauai, Hawaii) which owns and manages extensive areas of low- and mid-elevation forests on several islands. Many botanic gardens and arboreta have staff, facilities and programs specifically aimed at the propagation and study of native vegetation. These resources can and should be used to assist in efforts to protect and manage biological diversity. A growing body of examples illustrates the contribution of botanic garden research or conservation collections to onsite management of an endangered species (Parsons & Yates, 1983; Cox, 1987; Milne 1987; Rieseberg, 1988; Falk & McMahan, 1988). In some cases a botanic garden has played a central role in a truly integrated conservation effort for an endangered species (U.S. Fish and Wildlife Service, 1982; Cox 1987; Brauner, 1988). Botanic garden research programs, often executed in the *greenhouse*, can also help develop the very kinds of biological data that are useful for effective management of populations (Bowles *et al.*, 1987a, 1987b; Nicholson, 1987).

The end of the conservation spectrum, characterized by the most extreme manipulation of biodiversity, is the *germplasm bank*, including storage of seed, pollen, embryos and apical meristem. In terms of the biological hierarchy, these facilities are specifically designed for the maintenance of living plant genetic diversity in its most reduced form (Frankel & Hawkes, 1975). Germplasm banks enable large amounts of raw, genetic material to be maintained and made accessible for research. They also provide an economical insurance policy against failure of other conservation efforts for species whose survival in existing populations is uncertain (Harrington, 1970; Roberts 1975).

The concept of the spectrum hopefully will help to clarify relationships among different types of conservation. First, note that the steps along the spectrum are matched by a progressive shift in the level of the biological hierarchy being addressed, with a general trend from the ecosystem toward the allele. If one accepts the proposition that all levels of biological organization are worth conserving, it follows that deliberate application of a wide range of measures may provide the most effective, comprehensive and economical approach to conservation of biodiversity.

Correspondingly, it is implicit in the notion of a spectrum, as presented here, that one conservation approach should not be viewed as a substitute for another. Seed banks cannot take the place of conserving natural areas, not because they are ineffective conservation tools, but because they are designed for a different purpose. Botanic gardens do not substitute for ecosystem restoration. Rather, they are resources that can contribute to it by providing material for reintroduction or by performing research on seed germination, seedling growth and other relevant parameters. Given the magnitude of current and future threats to biological diversity, it is essential to employ any and all possible resources to their most appropriate purpose in a coordinated fashion. By applying tools such as seed banks and botanic gardens to their most appropriate level of biological hierarchy, cost-efficiency and effectiveness of conservation efforts may be dramatically improved. This is the core of the integrated conservation philosophy.

## INTEGRATED CONSERVATION STRATEGIES

The careful articulation of objective levels of biological hierarchy, threats to survival at each level and the spectrum of conservation measures available provides the theoretical groundwork for an integrated approach. With these ideas as the basis for a conceptual framework, it is possible to design a specific strategy for a particular conservation problem. Examples of such projects have been given elsewhere (Falk & McMahan, 1988; Cox 1987; Ferreira and Smith, 1987; Olwell *et al.*, 1987; McMahan, 1989), so the focus here will remain on the structural basis for the method.

In the terms outlined above, an integrated conservation approach may be defined as one which includes a determination of the biological entity to be protected, analysis of threats at each level of organization and selection of a range of complementary conservation actions tailored to address the problem. The measures implemented may be planned as a sequence over time. For instance, many ecosystem restorations are preceded by collection and propagation of genetically diverse stocks of plants, followed by long-term maintenance and management regimes. The conservation spectrum may thus be designed to unfold in its use over a period of months or years.

Integrated conservation strategies are inherently multidisciplinary and collaborative. No single agency can reasonably be expected to incorporate the full range of resources represented in the conservation spectrum. In fact, the need for efficiency in the use of conservation

resources argues for a certain degree of division of labor along the spectrum. By focusing its resources on a particular level of the biological hierarchy, an organization has the best likelihood of becoming expert at the methods utilized in its particular contribution. Thus, integrated conservation compels partnerships among organizations, with a preference for collaborations that provide complementary skills and resources. Botanic gardens should not single-handedly undertake species reintroductions; they should do so in consort with land-managing agencies. Likewise, land conservancies need not construct labs to study seed germination when gardens or universities already have the facilities that can provide the information they need under contract. Anything less than a full and devoted partnership is eventually likely to be inadequate to meet the challenge of conserving global, biological diversity.

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# Conservation Strategies: A Focus on Cooperative Action

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**Abstract:** Public awareness of the rapidly degrading state of our biosphere is becoming widespread, in terms of the physical environment as well as biological systems and diversity. In that context, it seems timely to consider the activities of agencies, plant and animal repositories and individual projects involved in diverse aspects of conservation, preservation, habitat reconstruction and land management, then to consider how they might function to complement one another more effectively in a coordinated program of interagency cooperation and action. It is also appropriate to question the harboring of endangered species in artificial environments like zoos and botanic gardens at the risk of reducing interest in conservation efforts aimed at wild populations. With current global concern for improving environmental quality and retaining diversity within viable, functioning biological systems, both natural and reconstructed, we also need to re-assess ongoing projects in terms of their effectiveness in further raising the awareness of the public and establishing a support-base for wiser and more integrated conservation objectives.

Pages 11 - 13. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Daily, we hear concerns about the degradation of our biosphere, the increasing rate of species extinction and loss of biodiversity threatening the viability of major biological systems. Since many of us are involved in institutional and agency projects, often narrowly focused on their individual components, it is important for us to re-examine their motives and conservation-effectiveness. How do we balance the concerns and costs associated with saving individual endangered species with the need to preserve or reconstruct their natural habitats? Is it really reasonable to allocate substantial monetary resources to botanic gardens and zoos, housing what Janzen (1988) calls "the living dead," ...individual plants and animals that are merely finishing out their lifespans with no remaining natural population to which they might return? Public awareness of such problems is often overlooked, yet improved understanding between environmental decision makers in government and the general public could go far toward assuring effective planning and satisfactory results.

## THE ROLE OF BOTANIC GARDENS

In recent years, many botanic gardens and arboreta have made commitments to conservation worldwide (Bramwell et al., 1988). Many such institutions directly manage natural areas under their control and may also serve as consultants on regional conservation lands. Others are involved in a wide variety of conservation activities, ranging from surveys and studies on vulnerable species in the field to experimental cultivation, propagation and maintenance of seed banks or conservation stands.

### Botanic Gardens and Conservation — Two Examples:

1) A prime example of a garden with a comprehensive conservation program is the Jardín Botánico Canario of Las Palmas, Canary Islands. It manages the Los Tilos Reserve, which contains one of the last remnants of a once extensive laurel forest (dominant tree species: *Ocotea*

*foetens*, *Ilex canariensis* and *Laurus azorica*). In co-operation with the insular government, the species of this forest are being propagated on-site and reintroduced into adjacent, long deforested lands that have since been colonized by exotics such as prickly pear cactus and century plant. At the garden, plant species endemic to the various Canary Islands are being propagated, placed in public displays and used as a focus for interpretive programs for school classes and the public. In addition, species in jeopardy are being maintained through a seed bank and made the subjects of studies to enhance their conservation status, including population and reproductive biology, cultural requirements and propagation.

2) The Arboretum of the University of Guelph has focused its conservation activities on the rare species of Canada's Carolinian zone, a narrow biotic zone north of Lake Erie where species of the deciduous forests of eastern North America reach their northern limits of distribution. Activities include field studies for status reports and subsequent studies of the biology of individual species. Information gathered by this program has provided a basis for action by the provincial ministry of Natural Resources to protect rare species habitat and provided input for a successful landowner contact program (Hilts & McLellan, 1984; Hilts, 1988). To complement the above *in situ* focus, *ex situ* conservation stands of many of the rare species under study are being established at the arboretum. Each individual within such a "gene bank" is fully documented, including details of the exact location of the parents of propagated seeds or cuttings and their biophysical habitat. Propagated plants then provide material for additional scientific study and tests to determine special cultural requirements to establish them in cultivation. In the future they will provide stock for possible reintroductions as well as for landscaping and other uses. A native, forest-communities collection is also being established to display rare species in naturalistic habitats to promote public awareness.

### *In Situ* Versus *Ex Situ* Protection and Beyond:

The development of *ex situ* collections of rare organisms prompts serious examination of their conservation role or lack thereof. While living collections can serve many short-term functions and reduce the immediate endangerment of species, they are neither efficient nor secure for long-term conservation purposes (Ashton, 1988). We should

<sup>1</sup>Published as contribution number 638 of the New York State Science Service.

consider the potentially counter-productive scenario of having "rescued" or "saved" a plant species by putting it into cultivation, after which serious interest in the remaining natural populations is decreased, leading to neglect of the problem and demise of the organisms in the wild. Since only a sub-sample of the gene pool of the species may have been preserved, a sufficient amount of genetic diversity to re-establish a viable, self-perpetuating population might not exist (Ambrose, 1987). Over the long term, we must question the worth of expending resources on collections of living specimens of species extinct in the wild, if this means neglecting programs to protect rarities and, indeed, entire natural landscapes.

Obviously all aspects of conservation deserve consideration, and a balance of full-spectrum actions should be considered: *in situ* and *ex situ* conservation, utilization of our ever-increasing knowledge of species biology and population dynamics, as well as efforts to instill in the public an awareness of the functions of environmental regulation necessary to preserve the intriguing diversity of our biosphere.

### **Exotic Introductions and Threats to Native Habitats:**

Botanic gardens and arboreta, as well as zoos and animal importers, have another responsibility that is less widely recognized. It concerns the introduction of exotic species that can potentially invade and disrupt natural ecosystems (Ambrose, 1988b; Huenneke, 1988). Botanic gardens need to assess their highly prized seedlings carefully after they have been grown from seeds collected during expeditions to biologically rich, exotic regions of the world. As they grow in their flats, we must ask: are we looking at yet another Japanese honeysuckle (*Lonicera japonica*) or purple loosestrife (*Lythrum salicaria*) with potentially dangerous invasive capabilities?

Much planting, even in rural areas for purported conservation or wildlife purposes, is often done with potentially weedy exotics. Notable examples are the introduced species: multiflora rose (*Rosa multiflora*), propagated widely as a wildlife-sheltering hedge, and kudzu vine (*Pueraria lobata*), planted as an erosion stabilizer, especially on road cuts throughout the southern United States. Both of these are aggressive species, once propagated and spread under the direction of governmental agencies, that are now considered noxious and destructive weeds over most of their introduced ranges. Much expensive effort has since been invested to eradicate them. Native species might have served their original purposes of introduction as well if not better, without threatening destruction of natural environments and personal property. Advice from the botanical and horticultural community is much needed in advance of such introductions to avoid repetition of such serious mistakes.

The impact of invading exotics appears to be most profound upon island ecosystems. Bermuda is an example where numerous plant and animal exotics have been intentionally or inadvertently introduced. The once-dominant Bermuda cedar (*Juniperus bermudiana*) has been reduced to occasional individuals by an introduced scale insect. The natural vegetation, now reduced to small, isolated sites, has been impacted by development and by highly invasive garden escapes, such as jasmine (*Jasminum simplicifolium*), Surinam cherry (*Eugenia uniflora*) and asparagus (*Asparagus* spp.). David Wingate, Bermuda's Conservation Officer, has launched a very ambitious and successful campaign to reclaim a series of sites harboring remnant natural vegetation. One sector at a time, exotics are being eradicated, and native species appropriate for the site are planted; these are often propagated from the few remaining individuals of a species remaining in the wild. This expanding project of habitat restoration and establishment of nature reserves has received considerable public support. An interest in

the islands' native flora for landscaping has also arisen, strengthened by the effects of a recent hurricane which has shown the vulnerability of many exotics used in landscaping.

## **INTEGRATING CONSERVATION ACTIVITY**

Falk (1987, 1988) has fostered the value of integrated strategies for conservation of endangered species. It seems timely to develop this thinking further. Consider first the threats, even to secure habitats, because they are often isolated, small islands of green. Secondly, consider all public and private agencies that independently manage networks of land, such as railway and utility rights-of-way, highway verges (see Drake & Kirchner, 1987), forestry and military reserves, parks, preserves, watersheds and other natural-area conservation lands. How do these units currently interact, and how might they pool their efforts? Ideally, there should be integrated efforts to preserve more than scattered fragments, and to connect these by corridors into a larger matrix of appropriately managed habitats (Ambrose, 1988a). A network of management bodies, all respecting their individual agency requirements, but seeking a common ground of connecting and enhancing fragmented natural landscapes (Noss, 1987) to enlarge the natural matrix, might greatly improve the state of our natural environment. Despite practical and political difficulties inherent in such a master plan, a combination of preserving natural area remnants and connecting them with a network of restored habitats could yield the high reward of a great area of diverse habitat. This would also provide space to manage for rarities and help reduce dangers to rare species. It could add meaning to short-term efforts to rescue or "ark" species that are in immediate jeopardy and ultimately contribute to the maintenance of a system with a high level of biodiversity.

In southern Ontario, river systems are regulated by conservation authorities within each watershed, rather than by politically defined regions. Historically, the authorities' mandate has been to manage the watersheds so as to reduce flooding damage, augment river flow during dry periods, and to regulate development and activity in floodplains and hazard zones. They are now considering new ways to manage the lands they directly control, and means to encourage and facilitate good management on private lands. In addition to a recent program to protect significant Carolinian habitats, authorities are also looking at the landscape heritage of their watersheds, both cultural and natural (Scott, 1979, 1988). This exemplifies the potential for a heightened awareness of local landscapes as units transcending political boundaries, that may be viewed as ideal sites for combined preservation and conservation land management. In order to strengthen native landscape restoration plans and raise the likelihood of their success, they may be coupled with the economics of promoting tourism.

## **HABITAT RECONSTRUCTION**

Interest in reconstructing natural ecological landscapes can be integrated into our existing matrix of natural areas, their management and linkage (Ruff & Tregay, 1982; Londo, 1983). Public parks have come under management review to allow naturalization (Granger, 1984), and some have been developed around the theme of reconstructed regional ecosystems, as with the Dutch Heemparks (Ruff, 1978), as well as certain nature or ecology parks (Berger, 1988; Wittkugel, 1988; Savage, 1988). Although these are reconstructions, such simulations of natural habitats can help to give the public an understanding and appreciation of nature. They also provide another option to typical urban landscaping, while serving as functioning extensions and links with existing

natural ecosystems. Comprehensive rethinking of landscape management and reconstruction practices has the potential to correct past mismanagement in part (Booth, 1988) and to help bring us into a sound, sustainable interaction with the global environment.

Preservation of rare organisms outside nature is not an answer in itself, and artificial "restoration" of habitats will, at best, only approach nature as it was; thus, these are not adequate substitutes for natural area preservation, but they may be combined with it to constitute an improved land management agenda. The environmental awareness that such projects bring to the public can also increase support for habitat preservation. In their proper context, zoos, gardens and reconstructed habitats may serve, through their educational function, to aid ultimately in the retention of biodiversity worldwide by promoting natural area appreciation and advancing the concept of global stewardship.

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# Deer and Dauerwald Revisited: Aldo Leopold's 1935 Visit to Germany in Retrospect

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**Abstract:** Over 50 years ago Aldo Leopold visited Europe to study intensive deer and forest management practices. In 1936 he published a two part paper in the *Journal of Forestry* describing incompatibilities between German deer and forest management and implications for American game and forest management. Current German forest and wildlife management is still afflicted with many of the problems and conflicts cited by Leopold in 1936. Furthermore, American forest and wildlife management is becoming more intensive, and neither following Leopold's advice nor incorporating lessons learned from his northern European experience.

Pages 14 - 19. *Ecosystem Management: Rare Species and Significant Habitats*. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

**“[There is an]...uncritical assumption, dying but not yet dead in America, that the practice of forestry in and of itself, regardless of what kind or how much, promotes the welfare of wildlife.” (Leopold, 1936a.)**

Aldo Leopold is considered the father of the American game management profession (Wild, 1979). Born in 1887, he received a master of forestry degree from Yale and began work in 1909 for the U. S. Forest Service in the Arizona Territory. During his 15 years in the southwest he wrote a game and fish handbook for district rangers, served as supervisor of Carson National Forest, became Chief of Operations for the Southwestern District, and helped create the country's first designated wilderness. In 1924 he became associate director of the U. S. Forest Products Laboratory in Madison, Wisconsin. Four years later he quit the Forest Service and went to work as a game and forestry consultant. In 1933 he published the classic text *Game Management* and occupied the newly created chair of game management at the University of Wisconsin, which he kept until his death in 1948.

Aldo Leopold helped found the Wilderness Society in 1935 and The Wildlife Society in 1937. His other well-known book, *Sand County Almanac*, was published after his death. In 1935 Leopold visited Germany and Czechoslovakia to observe forestry and game management practices. In 1936 he published “Deer and *Dauerwald* in Germany” in the *Journal of Forestry*.

In 1981 I also visited Germany. I was accompanied by local foresters to Kottenforst near Bonn, Arnsberger Wald in the upper Ruhr valley, Westerwald on the middle Rhine, and the southern Black Forest near Freiburg. I also visited, and subsequently corresponded with, forest wildlife researchers at Forsthaus Hardt near Bonn (Dr. Erhard Ueckermann), Justus Liebig University in Giessen (Prof. Dr. R. R. Hofmann), and Baden-Württemberg Wildlife Research Station (Dr. Herbert Kalchreuter).

This paper compares Leopold's 1935 observations with mine of 1981, and Leopold's recommended applications are contrasted with opinions and observations of several German wildlife scientists and managers. Current North American forest wildlife practices are also assessed in the light of Leopold's recommendations and the German experience.

## LEOPOLD'S VIEW OF THE GERMAN EXPERIENCE

Table 1 summarizes Leopold's description of the 900-year history of German forestry and game management. Leopold concluded this review with these words:

“...we have traced through a period of nine centuries the slow but inexorable growth of a system of silviculture incompatible with a natural and healthy game stand, and of a system of game management incompatible with a natural and healthy silviculture.” (Leopold, 1936b.)

Leopold reported that in 1936 there was still a continuing conflict between deer (primarily *Cervus elaphus* Linn. and *Capreolus capreolus* Linn.) production and forestry in Germany. Preferred deer browse species no longer occurred in deer forests he visited except where protected by fencing. Yew (*Taxus baccata* L.), for example, was virtually extinct. In spruce (*Picea*) and pine (*Pinus*) clearcuts, Leopold counted only five shrub and forb species, whereas in areas protected from deer he noted as many as 16 species.

He observed that most deer were artificially fed in winter, and summer feeding also occurred on “game acres” planted to oats (*Avena sativa* L.), rye (*Secale cereale* L.) or clover (*Trifolium*). Deer enclosure fences were opened at the appropriate season to give access to this summer feed.

Deer damage to forest and crops was noted:

“Artificial feeding, by keeping deer alive which would otherwise starve, of course enlarges the discrepancy between game density and natural forage, and thus also enlarges the variety and intensity of game damage to forest vegetation and to adjoining agriculture,...” (Leopold, 1936b.)

Leopold mentioned several protective measures, including bundling twigs around spruce trunks to prevent bark-stripping, tarring and bundling of young trees to prevent browsing, and game fencing.

“The most universal protective expedient is, of course, the game fence...wire or pole fences now attempt to keep the deer out of a large percentage of all reproduction areas. Fences also often parallel the exterior boundaries of the forest to keep the deer out of the adjoining fields. The deer are thus ground between the upper and nether millstone by fences which keep them out of all feeding areas.” (Leopold, 1936b.)

Finally, Leopold noted that although deer kill data were common, good censuses of deer populations were scarce. This made it hard to determine total size of the deer herd and percentage removed each year.

## LEOPOLD'S RECOMMENDATIONS

“We now have to deal with the ecological evidence of this conflict, the human motives involved, and the lessons which

TABLE 1.

German Deer and Forestry through Nine Centuries (as described by Leopold, 1936a).

Period	Forests	Wildlife <sup>a</sup>
The <i>Urwald</i> (old forest)	Hardwoods predominate Natural openings	Numerous, diverse fauna including predators
The Feudal forests (1100-1400)	Agriculture	Sport hunting Diminished game <i>Auerochs</i> , <i>Wisent</i> , wild horse extirpated
The Big High (1400-1618)	Wildlife damage	Game management by "foresters" Very high deer numbers Kills recorded, restrained Predator control
The Exploitation (1618-1700)	Clearcutting	Game decimated Bears and wolves increase
The Timber Famine (1700-1810)	Low Deer-maintained savannas	Deer increase Lynx, wolf extirpated
The Spruce Mania (1810-1914)	Spruce monoculture Deer damage heavy Podsolization Windfalls Insect epidemics	Breakdown of management Deer extremely low then increase Bear extirpated
<i>Dauerwald</i> (permanent forest, 1914-1935)	Selection, natural reproduction Mixed forest	Deer reduced, carrying capacity increased

<sup>a</sup>*Auerochs* or wild cattle (*Bos primigenius* Linnaeus), *Wisent* or European bison (*Bison bonasus* Linnaeus), wild horse (*Equus ferus* Linnaeus), deer (*Cervus elaphus* Linnaeus and *Capreolus capreolus* Linnaeus), bear (*Ursus arctos* Linnaeus), wolf (*Canis lupus* Linnaeus), lynx (*Felis lynx* Linnaeus), and introduced deer (primarily fallow deer, *Cervus dama*).

American game managers and foresters may draw from Germany's experience." (Leopold, 1936b.)

"An analysis of this predicament should be rich in lessons for all countries whose conservation policy is still in the making." (Leopold, 1936a.)

Leopold prefaced his recommendations by mentioning the "wood factory" concept imported into America from Germany. He warned that although this concept contained value, Americans needed to know how much it had been amended and superseded by newer and broader concepts of forest land use in Germany. The Germans came to realize that gains bought at the expense of soil health, landscape beauty, and wildlife were "...poor economics as well as poor public policy." (Leopold, 1936b.)

Leopold then recommended five applications to American practices as indicated by the German experience (Table 2). First, the better half of each forest should be devoted to intensive silviculture and the other half left to other uses, such as "...floral and faunal conservation." Second, deer predators may "cushion" the mistakes of game managers. Their total removal did not seem justifiable or necessary. Third, there should be a deep respect for natural species mixes and a deep suspicion of large pure blocks of any species, particularly those not indigenous to the site. He felt that large monocultures in America would reproduce both the German system and the German problem. Fourth, the problem

of dual jurisdiction (i.e., separate game and forestry agencies) should be examined. How could Americans stay out of trouble with their system of divided authority and responsibility, Leopold asked, when

Table 2.

American Application of Leopold's German Experience

Subject	Leopold's Recommendation
<b>Forest allocation:</b>	...develop an intensive silviculture on the better half of our forest soils...and leave the other half primarily to other uses." (Such as flora and fauna conservation.)
<b>Predator management:</b>	Predators "cushion" mistakes of game managers, total removal is not justifiable or necessary.
<b>Diversity:</b>	Deeply respect natural mixtures, deeply suspect large pure blocks of any species—especially species not indigenous to the locality.
<b>Resource Administration:</b>	Examine problem of dual jurisdictions over game. "If German foresters, with sole jurisdiction, can get themselves into such a fix, what can we expect when authority and responsibility are divided."
<b>Population and Habitat:</b>	Be generous in building up carrying capacity, stingy in building up stock.

"...German foresters, with sole jurisdiction, can get themselves into such a fix...?" (Leopold, 1936b.) Finally, there should be a good margin between the habitat's carrying capacity and wildlife populations. "...I plead for a generous policy in building carrying capacity, and a stingy one in building up stock." (Leopold, 1936b.)

## GERMAN FORESTS AND WILDLIFE SINCE 1935

Since Leopold's visit, Germany experienced another world war, the strain of foreign occupation and East-West division, and an unprecedented economic and social recovery. West Germany, or the Federal Republic of Germany (FRG), is now one of the richest and most stable socio-economic systems in the world. It has also maintained a large forest and wildlife base: FRG is 29% forested and contains estimated populations of 90 thousand red deer (*Cervus elaphus*), 1.7 million roe deer (*Capreolus capreolus*), and 50 thousand wild pigs (*Sus scrofa* Linn.) as well as substantial numbers of other game species (Ueckermann, 1979).

There is also considerable interest in outdoor, forest-oriented, non-hunting recreation. Only 0.4% of the population hunts, but 75% say they visit forests at least once per month (Erhard Ueckermann and Ernst Eick, pers. comm.). But at least some FRG wildlife scientists and foresters feel there has not been much improvement in forestry and wildlife management since 1935:

"...Leopold's paper is still, in many ways, a true assessment of [our] situation... Most of our foresters still cling to monoculture spruce (or pine) forests..." (R. R. Hofmann, pers. comm.)

"The situation has not changed much since Leopold's visit 50 years ago." (H. Kalchreuter, pers. comm.)

"Slow-growing trees have been replaced by quickly-growing species to suit the timber market... In spite of keen efforts, a large number of our forests are still far from having an ideal distribution of age classes... game damage... so increases that the demand for drastic control-shooting becomes imperative, and the forester perceives the only good deer being a dead one!" (E. Eick, pers. comm..)

Old growth forests and associated wildlife are scarce in the FRG. Visiting four different forest districts, I saw only one old growth stand. This was a 70 ha. reserve for capercaillie (*Tetrao urogallus* Linn.). Trees in this reserve were over 200 years old and snags were being retained.

Wildlife species that use old growth and mixed species forests are becoming more and more rare in Germany:

"*Auerwild* [capercaillie], *Birkwild* [black cock, *Tetra tetrix* Linn.] and *Haselwild* [hazel grouse, *Bonasa bonasia* Linn.] is nearly extinct because they cannot find their very special living space." (E. Eick, pers. comm..)

"Definitely capercaillie and hazel grouse disappeared from dense spruce plantations. On the other hand during the recent decades they also disappeared from ideal mixed-forest habitats..." (H. Kalchreuter, pers. comm..)

Ratios between cover area and open forage areas in FRG forests are very heavy to cover from a North American perspective. American managers of Rocky Mountain elk (*Cervus elaphus nelsoni* Bailey) speak about forests having 20% to 60% of their areas in forage (Thomas, 1979). A typical FRG forest should have only about 1.5% to 3.0% feeding area according to an Arnsberger Wald forester, Ernst Eick (pers. comm.). One prominent FRG researcher suggests that

0.15% to 0.5% feeding area is sufficient for red deer (Ueckermann, 1987). But this drastic contrast may be at least partly due to differences in adaptation between European and American subspecies of *C. elaphus*, as suggested by Geist (1982).

Most deer in the FRG are still being artificially fed. As Leopold noted, in many places this is necessary because previous natural forage areas are now cultivated farmland. Feeding is still used to reduce forest damage.

"Planting grazing areas and winter feeding are the main measures for improving feeding conditions... Planting permanent grazing areas and feeding red deer in winter, for example, is considered a useful improvement of upland habitats and areas with a high percentage of spruce... Feeding is usually carried out from November 1 to the middle of April... Winter feeding results in a distinct reduction of bark stripping damage." (Ueckermann, 1987.)

Also, census practices may still be inadequate. Eick (1985) reports that there is no method for determining the precise size of a game population. Ueckermann (1987) describes population estimation methods and problems:

"Population estimates are made by the hunting tenants from counts at winter feeding stations or from track counts after fresh snowfall. Red deer populations were often underestimated in the past... Gamekeepers may have made intentional mistakes to evade orders for the reduction of their herds. There are also cases of overestimated populations in order to achieve higher culls [harvest]."

Two recent phenomena have complicated forestry and wildlife management in the FRG. The first is a rapid increase in non-hunting forest recreation. More and more citizens are visiting their forests for recreational walking, skiing, collecting mushrooms, rock-climbing, etc.

"The managed forest has tended to become progressively transformed into a proper recreation forest, furnished with inter-connecting pathways and ski-tracks, which allow disturbance to increase erratically and in an overall manner during the entire year." (Eick, 1985.)

"[Our situation]...has recently been aggravated by much more public 'recreational' use of the forests, i.e., the game has literally no place to feed and digest in peace..." (R. R. Hofmann, pers. comm..)

The second phenomenon is *Waldsterben*, the new forest sickness attributed by many to acid rain and air pollution. This has resulted in pressure for some foresters to reduce ungulate populations so as to reduce tree damage and improve forest regeneration (Hofmann, 1985; H. Kalchreuter, pers. comm.).

There have been some improvements in forest diversity. For example, large areas clearcut for reparations following the last world war regenerated naturally as beech (*Fagus*) after good mast years (Eick, 1985). In some areas, hunters are killing more deer thereby reducing deer damage to forests. Common tree species can now be regenerated in these areas without problems, although some species still need fence protection [e.g., fir (*Abies*), maple (*Acer*) and oak (*Quercus*)].

"This costs money, but it seems to be the only possible compromise between the German desire to see game (as Leopold correctly stated) and the demand for keeping mixed forests." (H. Kalchreuter, pers. comm..)

## AMERICAN RESPONSE TO LEOPOLD'S RECOMMENDATIONS

Leopold visited Germany in 1935 during a time when new game and forestry laws were being written that still affect forest and game management in the FRG (Ueckermann, 1987; Eick, 1985). Many Americans besides Leopold were impressed by these changes. George W. Peavey, former dean of forestry and president of Oregon State College (now Oregon State University), praised German forestry in a 1935 speech. He said it was the most highly developed and intensive in the world, with forest lands being put to their best practical use, illustrating what should be done in Oregon (Oregon Department of Forestry, 1986).

Leopold's paper apparently received little notice. The Second World War may have diverted attention from this important paper. But this may still be the time to recall Leopold's observations, and I will examine each of his recommended applications in the light of present day perspectives.

First, have we devoted the better half of each forest to intensive silviculture and the other half to "floral and faunal conservation"? This is the crux of current National Forest planning in the U. S., that is, achieving a balanced allocation among the mandated multiple uses for federally managed forests. Using Oregon as an example, I find that current federal land management plans strike a balance close to Leopold's recommendation. In four representative National Forests in Oregon, an average of 40% of the land is proposed for intensive silviculture (see Table 3), but 40% of Oregon's forests are in private, state, or county ownership. These lands are predominantly devoted to intensive timber harvest. So, if Oregon is representative, Americans are devoting considerably more than half of their forests to intensive timber harvest, and much less than half is being devoted to "floral and faunal conservation". Leopold's recommended balance is not being met.

Second, have we maintained a population of deer predators? Again, using Oregon as an example, the last native wolf (*Canis lupus* Linn.)

and grizzly bear (*Ursus arctos* Linn.) were killed off within a decade of Leopold's warning (Bailey, 1936; Ingles, 1965); however, Oregon and most other western states are maintaining and managing healthy cougar (*Felis concolor* Linn.) and black bear (*Ursus americanus* Pallas) populations, and the coyote (*Canis latrans* Say) is ubiquitous and numerous. On a broader basis, predators other than the grizzly bear and wolf are generally holding their own or increasing in North America (Chandler, 1986b).

Third, have we maintained natural species mixes? Game management and forestry have been, almost by definition, anti-diversity. Game management favored species that were hunted, and forestry favored trees that could be sold.

"Foresters thought of a forest as a monoculture because all merchantable trees were of the same species. They sought to shift the composition of a given stand from variety to a single or dominant, commercially desirable, species." (Hays, 1987.)

By practicing "featured species" management (Thomas, 1979), game managers reduced species diversity. Deer, elk, pheasants (*Phasianus*) or trout (*e.g. Salmo*) have been favored over other indigenous species, and endangered and "glamour" species programs have had a similar management effect (Hays, 1987).

If Leopold planted the seed in 1936, the concept of managing for natural mixes took a long time to bear fruit. If we are what we call ourselves, game managers have started calling themselves wildlife managers only within the last 20 years. The Oregon State Game Commission was renamed the Oregon Wildlife Commission in 1972; the Washington Department of Game became the Washington Department of Wildlife in 1987; and six out of 13 western states still use the word "game" in the management agency's title. But there is encouraging evidence that diversity is becoming a serious management goal, by both foresters and wildlife managers:

"Emphasis on management for diversity in forest ecosystems will help insure the continued existence of the living components of the system—plants as well as animals." (Thomas et al, 1979.)

"It is the policy of the State of Oregon... To maintain all species of wildlife at optimum levels and prevent the serious depletion of any indigenous species." (Oregon Revised Statutes 496.012, adopted 1973.)

"Forest planning shall provide for diversity of plant and animal communities and tree species consistent with overall multiple-use objectives of the planning area." (Department of Agriculture Forest Service 36 CFR Part 219.)

"The addition of endangered species protection and nongame conservation to wildlife agency agendas is impelling state agencies to change their identity from one of game managers for sportsmen to ecological managers of all biological resources. This process is just underway." (Chandler, 1986a.)

"Nongame wildlife programs usually cite preservation of wildlife diversity and abundance as their primary goal... The most far-sighted wildlife administrators view nongame programs as an opportunity to develop comprehensive conservation measures for all wildlife." (Cerulean and Fosburgh, 1986.)

Fourth, have we seriously examined the validity of dual jurisdictions over game? In many states foresters and wildlife administrators both acknowledge this problem by signing agreements that jointly delimit and recognize federal Forest Service and state fish and wildlife authority and responsibility (Barton and Fosburgh, 1986). In these agree-

**Table 3.**

Land Allocation Area (ha. x 1000) in Four Representative  
Forest Plan Drafts  
(National Forest Preferred Alternatives).

National Forest	Total Area	Timber Harvest Level		
		Intensive	Reduced <sup>a</sup>	Precluded <sup>b</sup>
Umatilla	611 (100%)	238 (16%)	381 (62%)	134 (22%)
Malheur	591 (100%)	296 (50%)	242 (41%)	53 (9%)
Willamette	673 (100%)	282 (42%)	235 (35%)	156 (23%)
Umpqua	399 (100%)	258 (65%)	112 (28%)	29 (7%)
<b>Total</b>	<b>2274 (100%)</b>	<b>932 (41%)</b>	<b>970 (43%)</b>	<b>372 (16%)</b>

<sup>b</sup> Timber harvest reduced due to allocations for fish habitat, wildlife, scenic, big game winter range, recreation, etc.

<sup>c</sup> Timber harvest precluded due to allocations for designated wilderness, old growth, etc.

ments, the Forest Service is recognized as being responsible for habitat management and the state is recognized as being responsible for fish and wildlife species management on National Forest lands (e.g., Memorandum of Understanding, USDA Forest Service, Region 6, and the Oregon Department of Fish and Wildlife; signed and dated 1 July 1985). These agreements also require regular interagency meetings at all levels, cooperative relationships, free exchange of information, joint resolution of problems and conflicts, and mutual assistance and facility use.

In general, there does not seem to be much overt jurisdictional conflict between foresters and wildlife managers in North America. There seems to be general recognition that the states and provinces are responsible for most fish and wildlife management.

“A number of federal statutes establish cooperative wildlife programs in which the states’ right to regulate wildlife is recognized clearly and depended on to achieve management goals. Even wildlife on federal land is managed, in most cases, under state game and fish regulations...either because directed to do so by statute or because this is the most sensible and efficient approach.” (Chandler, 1986a.)

Contemporary published discussions of Forest Service and state wildlife responsibilities do not mention any problems due to dual wildlife jurisdiction (Barton and Fosburgh, 1986; Fosburgh, 1986; Gilbert and Dodds, 1987; Argrow, 1983; Johnson, 1983).

It appears that instead of jurisdictional uncertainty, we have an undocumented but effective system of checks and balances operating in North America. These checks and balances may be internal within an agency (as in the case of a Forest Service biologist reviewing a forester’s activities) or external between agencies (as in the case of a forestry agency reviewing a state wildlife agency’s activities). For example, the Forest Service played an important role in reviewing and influencing Oregon’s deer and elk population management objectives adopted in 1980.

Fifth, have we maintained a good margin between the habitat’s carrying capacity and wildlife populations?

Several years ago, a high-level Oregon wildlife administrator told me we must keep deer (*Dama*) numbers high enough so that we risk die-offs even during average winters. He said the hunting public is satisfied only when deer populations are that high. MacNab (1985) described this strategy and its outcome:

“The sagebrush (*Artemisia* [L.] spp.) steppe of western North America supports large populations of mule deer (*O.[Dama] hemionus* [Rafinesque]) that experience periodically harsh winters. Prior to the severe winter of 1983-84, antlerless deer harvests were small to zero in much of Eastern Oregon... Consequently, females were at high density, and deer died during the harsh winter... The implication of this is that the management objective was to maintain high deer densities to obtain bigger harvests later...”

McCullough (1987) put it in more general terms, writing that when a deer population is managed to maximum sustained yield there is a serious risk of habitat overuse: “It is not an immutable rule that more deer is better,” he concluded.

Failure to maintain a comfortable margin between habitat and populations may be even more evident with nongame wildlife species where our knowledge of habitat relationships is relatively meager. For example, Thomas *et al.*(1979) published a relationship between snag density and the viability of cavity nesting species populations. This

relationship suggests that cavity nesting species will have marginally viable populations when the combination of per cent of forested land area with snags and the intensity of snag management (as defined by Thomas *et al.*) drops below 40%.

In the 1987 draft Proposed Land and Resource Management Plan for Malheur National Forest (northeastern Oregon), the preferred Forest management alternative states: “Snag habitat would be retained to support 40 per cent of the potential population of cavity dwelling species Forestwide.” In a parallel plan, also released in 1987, the neighboring Wallowa-Whitman National Forest’s preferred alternative would provide large snag habitat at “27 per cent of optimum.”

For the Pacific Northwest Region of the Forest Service (Region 6), the March 1985 Supplement S3 to Forest Service Manual 2360 suggests that the 40% level be maintained for cavity nesting habitat, averaged over each subdrainage or management compartment of approximately 2833 ha.

In all cases, “per cent of potential population,” “per cent of optimum,” and “per cent level” equate to the combination described by Thomas *et al.* In none of these cases is there a stated intent to manage forestwide habitat for cavity nesting species at a level higher than marginally viable.

Recent European experience demonstrates how careful managers must be in protecting cavity nesting habitats under intensive forest management. Ahlén *et al.*(1978) reported that two endangered woodpecker (*Dendrocopos*) species in Sweden are highly specialized and have been adversely affected by habitat changes, primarily caused by modern forestry practices. The entire Swedish population of the middle spotted woodpecker (*D. medius* Linn.) was only 10-20 individuals in 1979 (Pettersson, 1980).

There are other examples where habitat requirements of a species or group of species has been defined, and habitat management is crowding these defined requirements, e.g., old growth forest management and the spotted owl (*Strix occidentalis* Xántus di Vesey) (Forsman and Meslow, 1986). In today’s sophisticated, computer rich environment, managers are tempted to use imperfectly understood habitat indices and wildlife population models to manage habitats and populations within uncomfortably close margins. According to Thomas (1986), managers have a tendency to endow simple observations about wildlife and habitat with unjustified qualities of precision, particularly once the observations have been entered into a model’s formula.

## CONCLUSION

Over 50 years ago Aldo Leopold visited Germany and noted some lessons that Americans could learn from the German forest and wildlife management experience. He tried to tell us in his 1936 paper what he thought these applications were. According to my analysis, we are only now re-learning some of these lessons:

Our current allocation of forest lands does not meet Leopold’s standards for providing a balance between intensive timber management and floral and faunal conservation. But system diversity, including the role of predators, is becoming a part of many forest and wildlife management policies and strategies. Resource managers are just beginning to manage whole systems rather than individual species.

Dual jurisdictions for foresters and wildlife managers may not be a hindrance to successful resource management; it may even be an asset. Still, we seem to be unable to maintain comfortable margins between habitat and wildlife numbers, and resource experts act as if intensity of effort can make up for uncertainty of knowledge. This is risky; as Thomas (1986) suggests, we are navigating a wild and largely uncharted

ed river: "...perhaps we need to slow down a bit and steer a bit more methodically."

### ACKNOWLEDGMENTS

Special thanks go to Dr. Jack Ward Thomas (Chief Research Wildlife Biologist, Pacific Northwest Research Station, USDA-Forest Service, La Grande, Oregon) for telling me about Aldo Leopold's 1936 paper and for otherwise encouraging and abetting this effort. My wife, Joyce Aney, also gets my thanks; her conversational German and social grace helped expand my limited formal contacts in FRG into productive professional and social dialogue. And my sincere appreciation goes to those German colleagues and friends who made my visit and subsequent investigations so fruitful: Prof. Dr. Reinhold R. Hofmann (Professor of Veterinary Medicine and Leader of the Working Circle for Wildlife Biology and Game Management at Justus Liebig University, Giessen), Ernst Eick (forester and ungulate management researcher, Möhnesee), Dr. Herbert Kalchreuter (former director, Baden-Württemberg Wildlife Research Station, Bonndorf), Dr. Erhard Ueckermann (leader, Nordrhein-Westfalen Research Station for Game Management and Wildlife Damage Control, Bonn) and Heinrich Brühne (hunting area lessee, Möhnesee).

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## **Chapter 2.**

### **HABITAT ASSESSMENT**

# A Computerized Method of Priority Ranking for Natural Areas

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**Abstract:** The Nature Conservancy has developed an efficient system for ranking the relative merits of different sites for preservation of particular species and communities, but there is no universally accepted systematic method for comparing the composite values of complex sites. We addressed this issue in preparing a natural area preservation priority list for Alachua County, Florida. Our system involves scoring sites on a ten-point scale according to each of six criteria: vulnerability, rarity, connectedness, completeness, manageability, and nature-oriented human use potential. The scores are derived from expert consensus on carefully defined qualitative ratings, then entered into a Lotus 1-2-3 program. This program permits the criteria to be weighted differently for evaluating sites taking into consideration different types of planning objectives, e.g., recreation area vs. wildlife sanctuary planning.

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## INTRODUCTION

Evaluation of natural areas for protection as reserves is a fundamental activity in conservation, and a topic that has generated a vast literature. Whereas early evaluations focused on scenic or recreational values (national park designations, for example), increasingly sophisticated and objective criteria have been developed over the last two decades (Usher, 1986), yet, despite extensive discussion in the biological conservation literature, no generally accepted methodology for evaluating natural areas exists.

In the United States, the best-developed conservation evaluation procedure is probably The Nature Conservancy's heritage approach (Jenkins, 1978, 1985; Pearsall *et al.*, 1986; Noss, 1987a). Heritage program inventories build data bases that are oriented around "elements" (primarily species and community types, but they also may include geological features, wildlife concentration areas, etc.) rather than sites. Focusing primarily on the rarity and vulnerability of elements at state, national, and global scales, the heritage approach allows identification of sites containing the best examples of the highest-ranked elements.

Although the heritage approach has been successful in locating, ranking and protecting endangered species habitats, and the best examples of rare and unusual community types, it has been less useful in evaluating complex sites with multiple community types (Noss, 1987a). With a few important exceptions, traditional approaches have not achieved full representation of characteristic regional ecosystems and landscapes within reserve networks (Noss and Harris, 1986; Hutto *et al.*, 1987; Noss, 1988).

Ideally, an evaluation procedure for natural areas should consider both the context and the content of each site, i.e., the surrounding habitats and proximity to other natural areas, as well as the populations and habitat quality contained within the boundaries (Noss and Harris, 1986; Noss, 1987b). An evaluation procedure also should be adaptable to address the divergent objectives of different land acquisition programs.

In this paper, we describe a computerized ranking system for natural areas developed by KBN Engineering and Applied Sciences, Inc. for Alachua County, Florida in 1987. The system is based on criteria that incorporate considerations of both site content and site context and that can be easily weighted for different planning objectives. The study was undertaken to provide information on important upland

sites as a background for county comprehensive planning. The Alachua County Department of Planning and Development needed information on ecological communities to complete the Conservation Element of the county's Comprehensive Plan and to assist the Alachua County Conservation and Recreation Areas Task Force (ACCARATF) in greenbelt planning.

## THE STUDY AREA

Alachua County surrounds the city of Gainesville in north-central Florida. The county's 231,035 hectares extend southward from the Santa Fe River, a spring-fed blackwater tributary of the Suwannee, into the rolling hills of the horse farm region north of Ocala. The northwestern quadrant of the county is karst topography with extensive farmland dissected by ravines supporting the southernmost remnants of Appalachian-type hardwood forest. Degraded sandhills that formerly supported longleaf pine/wiregrass communities dominate the southwest quadrant. Pine flatwoods, now mostly converted to slash pine plantations, cover the eastern half. Lakes, such as Newnans Lake, Orange Lake and Lake Lochloosa, and marshy prairies, such as Paynes Prairie, occupy much of the southeast part of the county, where hammock forests of live oak, cabbage palm and magnolia dominate the uplands.

Though the beauty and diversity of Alachua County's native landscapes had been praised for centuries by such writers as William Bartram, Marjorie Kinnan Rawlings and Archie Carr, there had never been a systematic inventory of the county's valuable natural areas. The state Heritage Program, the Florida Natural Inventory (FNAI), maintained a data base on the best sites for the rarest species and communities statewide, but their information on natural areas of local importance was incomplete. General land use maps showing the distribution of the major ecosystems and detailed habitat maps had been prepared for Paynes Prairie, San Felasco Hammock, O'Leno State Park, and other existing preserves. Wetlands had been mapped and given variable and debatable amounts of legal protection, but upland communities, such as sandhills, hammocks and flatwoods had been documented only in a piecemeal fashion.

## METHODS

We initially screened out most small areas on the assumption that bigger is better, knowing that larger tracts generally have better prospects for long-term viability. Bob Simons, a highly respected local forester/conservationist, examined the 1986 1:24,000 infrared aerial

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photographs of the county with John Hendrix of the Alachua County Department of Environmental Services. Drawing on their familiarity with the county, they identified sites greater than 20 hectares that appeared to be natural uplands and marked them on a set of 1:24,000 USGS topographic quadrangle maps. Bob then field-surveyed these sites and prepared site record forms and preliminary species lists describing each of the upland communities on each site. Meanwhile, we solicited information on important sites from local conservationists and biologists by telephone, mail and posting. We searched the FNAI files and transferred data on the locations of rare species and communities onto a second set of quad sheets, overlaying this information with the boundaries of the field-surveyed sites. We then requested printouts of FNAI's Element Occurrence Records, field surveys, and other relevant information for all areas in the vicinity of our field survey sites. We also examined references mentioned in the FNAI files and asked experts to recommend relevant literature.

As field survey forms, species data, reports, site recommendations, etc., came into the office, a file was begun on each site. Information on places that did not appear to have natural upland ecosystems was set aside. The remaining sites were screened at a meeting of the entire project staff. Those sites deemed to be unimportant were pulled out, and those too fragmented to be considered as potential preserves but still valuable as habitat were placed into a special group. The remainder went through the full priority-ranking process. Additional research was done to supplement information on the sites that came out in the top dozen after preliminary ranking. Additional field trips were made, so that all project team members had the opportunity to see each site. An overflight was made to look at the sites from the air and examine their landscape relationships.

We used six criteria for the formal ranking process. The first criterion, vulnerability, was incorporated because it is the key criterion emphasized in the Florida Department of Community Affairs' Model Conservation Element, a document illustrating the approach the state requires for conservation aspects of local government comprehensive plans. The next four criteria (rarity, connectedness, completeness and manageability) came from the October 22, 1986 draft of the Alachua County Conservation Element. After we began working with the ranking system, we felt that there was an aspect of ecological quality not addressed by the first five criteria, so we added a sixth: nature-oriented human use potential.

The vulnerability criterion addresses the likelihood of events that might degrade or destroy the site within the next few years. It is important to remember this short-term crisis time scale since a high score on this criterion raises a site's priority rank, whereas susceptibility to slow long-term degradation (evaluated under manageability) lowers the rank.

We ranked **vulnerability** based on our knowledge of human community development patterns, then had our assessments reviewed by county planners. A more precise score could be derived by interviewing property owners and examining real estate data, zoning maps, and planning documents.

The **rarity** criterion incorporates the rarity of each of the site's plant and animal community types, the rarity of the species for which the site provides habitat, and the uniqueness of its special features, such as geological formations or champion trees. Rarity is viewed on three scales: county, state and global. Conservation biologists generally consider rarity a good predictor of susceptibility to extinction (Terborgh and Winter, 1980).

**Connectedness** concerns how the site links with related elements of the landscape. Enhancing the links is a fundamental strategy for mit-

igating problems related to fragmentation (Harris, 1985; Noss and Harris, 1986; Noss, 1987b). Some questions asked are: does a site lie within, or constitute a link between, segments of an actual or potential wildlife corridor, a greenbelt, or a trail system? Is it an inholding or a buffer for another natural area? How do the habitats within it relate to those nearby?

In evaluating connectedness, we assume maintenance of existing and proposed natural areas and the loss of all others. Hence a tract that is now an integral part of an extensive natural system, but is surrounded by land likely to be developed, would receive a lower score than a site linked to others by restricted corridors of permanent preserve. We have attempted to base our judgments on evaluations of ecological linkages composed of comparable habitats, rather than on geographical linkages where, for example, two areas of protected wetland might be "joined" by a stretch of xeric preserve.

We assess connectedness on a regional basis. Thus, a site like Watermelon Pond that does not link very well to anything within Alachua County nevertheless receives a good connectedness score because it is an integral part of the Wacassassa ecosystem that extends through Gilchrist and Levy counties.

**Completeness** is basically an index of a site's ecological quality. We ask: are the ecological communities representative examples with a full complement of species? How diverse are the habitats, the flora and the fauna? To what degree has the site been degraded? Are the "missing" species gone forever, or is the basic integrity of the system still intact enough to offer a realistic potential for reintroductions?

The criterion of completeness combined both habitat diversity and species diversity. Diversity, in general, has perhaps been the single most common criterion in conservation evaluations (Margules and Usher, 1981). We found it difficult to mesh habitat and species aspects of diversity into a quality concept that we could score consistently, and, in the future, we will be inclined to separate these two very important assessments.

Since completeness was scored largely on the basis of diversity, few scores are at the top of the scale. This is because most of Alachua County's natural areas have already lost a number of species that were components of the presettlement ecosystem. The county no longer has any panthers, red wolves or red-cockaded woodpeckers, and species such as black bears and fox squirrels are restricted to fragments of their former ranges.

**Manageability** is an assessment focused on questions about long-term viability. Is the site big enough? Would its preservation and the maintenance of its species be compatible with present and future neighboring land uses? Are its degraded habitats in restorable condition? Would it be practical to conduct prescribed burning in fire-maintained habitats? Would there be problems with trespassers or neighbors? How expensive would it be to manage the land properly?

Manageability scores were reduced for fire-maintained communities near highways and airports because of the relative likelihood of future burning restrictions in those areas, for instance.

**Nature-Oriented Human Use Potential** ranking involves questions on the site's inherent suitability for human activities dependent upon non-destructive use of natural features. Is it a documented research site or especially appropriate for scientific studies? Does the site have a variety of habitats and transition zones through which a nature trail could be routed? Is it a beautiful place that would be aesthetically enjoyable for the public? How difficult would it be to construct and maintain trails and other facilities for passive recreation without damaging the environment?

This assessment is strictly limited to consideration of qualities

stemming from the site's inherent ecological characteristics. Social need parameters such as location in relation to population centers, recreation demand, etc., are excluded as being beyond the scope of an ecological evaluation, although they can be important factors in planning.

Scores based on the above criteria were linked to the quality levels described in Table 1. These descriptions were intended only to convey the concept of a quality level. Few sites fit the description for a certain score-level perfectly. Where part of a site rated high on a criterion and part rated low, we gave it an intermediate score related to the proportion and importance of the areas of different quality.

Our scores were based on the assumption that when sites are purchased or otherwise protected, their boundaries would be those we recommended. Since site boundaries change through the preservation/planning process, these ranking scores will not be as precise an estimate of relative quality. Excluding an important tract could drastically affect a site's score.

We reached our scoring decisions through an informal consensus procedure. We found that three natural area experts had little difficulty reaching agreement after sharing and discussing information on each site. Minor differences were easily accommodated by effectively expanding our five-point scale to a ten-point scale and occasionally giving intermediate scores, such as 2.5 and 4.5. Our scores tended to run high; this makes sense in perspective, however, since the sites that would have received low scores were screened out prior to the formal ranking process.

Computer runs were conducted (utilizing Lotus 1-2-3 software) to reflect several ranking scenarios and determine how the sites would compare according to different conservation planning philosophies.

**Table 1.**

Scoring System for Site Priority Ranking

### **Vulnerability**

1 — Preservation guaranteed by deed restriction, easement, or established regulatory authority.

2 — Respected by conservation-minded landowner. Some regulatory protection. Very low development potential.

3 — Owner has no sale or development plans. Heirs may be inclined to sell. Borderline case as to regulatory protection. Located in low-growth area. Marginal development site.

4 — Owner likely to sell or develop, but action not imminent. No significant regulatory protection. Located in high-growth area. Good development site.

5 — Slated for development or prime real estate currently up for sale. No significant regulatory protection.

### **Rarity**

1 — Common community types in poor to average condition. Habitat types widespread throughout county. No rare animals or plants. No significant occurrences of anything ranked higher than 4 on FNAI's state scale. No significant geological features or wildlife sites. No trees of extraordinary size or age.

2 — Typical community types still represented by extensive acreage in Alachua County. A few uncommon species, but no significant occurrences of anything ranked higher than 3 on FNAI's state scale. No major geological features or wildlife sites. No mature forests or outstanding examples of natural communities.

3 — Good examples of natural communities. Habitat types well represented statewide, but scarce in Alachua County. A few rare

species, but not many ranked 2 on FNAI's state scale and none ranked higher. Geological features or wildlife sites of moderate value. Some old growth, but no large tracts or stands of "living museum" quality.

4 — Excellent examples of natural communities, some of them scarce. A number of rare species, but none dependent upon this site for survival. Several species FNAI ranks 1 or 2 on state scale. No significant occurrence for a globally endangered (GI) species or community. Important geological feature or wildlife site. Extensive tract of old growth. One of the best sites of its kind in Alachua County.

5 — Rare community type. Extraordinary example of a natural community. Diverse array of superb habitats, several of them scarce. Many rare species, including a number FNAI ranks 1 or 2 on state and/or global scales. Critical habitat for a globally endangered species (GI). Unique geological feature or wildlife site. Nationally significant.

### **Connectedness**

1 — Isolated from natural habitats of significant size by a large expanse of unsuitable habitat or a virtually impenetrable barrier (from standpoint of organisms inhabiting site). No significant connecting corridors. Not situated strategically for interconnection of natural areas or trail systems.

2 — Isolated from natural habitats of significant size by a moderate expanse of unsuitable habitat. No significant connecting corridors. Not situated strategically for interconnection of natural areas or trail systems.

3 — Isolated from natural habitats of significant size by an expanse of marginally suitable habitat. Narrow connecting corridors. Useful situation for interconnection of natural areas or trail systems.

4 — Not broadly joined to large areas of natural habitat, but close or connected by significant existing or potentially restorable habitat corridors. Good situation for connection of natural areas or trail systems.

5 — Directly contiguous with large areas of natural habitat along extensive boundaries. Critical situation for interconnecting natural areas or trail systems.

### **Completeness**

1 — Poor habitat. Low species and community diversity. Seriously degraded. Too tiny and/or isolated to maintain normal flora and fauna.

2 — Fair habitat. Moderate species and community diversity. Degraded, but restorable. Might be capable of supporting populations of relatively tolerant species.

3 — Good habitat. Good diversity of species or communities. Slight degradation. Probably capable of maintaining populations of most typical species.

4 — Excellent habitat. Diverse species, communities, and successional stages. Practically all appropriate species except rarities and large predators present and thriving. Excellent potential for reintroduction of most missing species.

5 — Outstanding habitat. Diverse species, communities, and natural successional stages, including a number of rarities. Large enough to maintain long-term disturbance/succession matrix. Sizeable gene pools due to size and or links to similar habitat areas. Potential for retention or reintroduction of full normal flora and fauna, including large predators.

### **Manageability**

1 — Too small and/or degraded for maintenance or reestablishment of normal ecosystem processes, such as periodic burning or flooding. Highly vulnerable to uncontrollable external impacts. Probably beyond hope.

2 — Location and/or extent of degradation would make management difficult and expensive. Questionable whether protection/restoration programs would be fully successful.

3 — Could be maintained in or restored to good condition, but would require vigilant management. Location and/or historic use suggests chronic problems with trespassers and/or neighbors. Special programs such as exotic plant removal or hydrological restoration required. Difficult location for management.

4 — Habitats in good condition, but requiring regular attention, such as prescribed burning. Effective buffering from most external impacts possible. Location and surrounding land uses reasonably convenient for management.

5 — Low-maintenance habitat types in excellent condition. Inherently well buffered from most external impacts. Location minimizes problems with trespassers and neighbors and facilitates management access.

### Nature-Oriented Human Use Potential

1 — Unsuitable for passive recreation. Aesthetically unappealing. Little scientific or educational value.

2 — Suitable for limited passive recreation, but special management might be necessary to prevent adverse impacts. Pleasant setting. Useful site for school or nature center field trips or student research.

3 — Suitable for limited passive recreation. Attractive environment. Ecologically interesting enough to be a good outing destination for local groups like Audubon, Sierra, etc. Useful site for scientific research.

4 — Good for several types of passive recreation. Scenic. Suitable for nature trails and/or environmental center. Valuable site for scientific research. Special enough to be a popular regional recreation destination.

5 — Outstanding site for a variety of passive recreational uses. Excellent for nature trails and/or environmental center. Extraordinarily scenic. Important well-documented scientific study site. Features so exceptional site could attract national/international visitors.

## RESULTS AND DISCUSSION

Table 2 gives our recommended priority ranking for Alachua County's upland ecological communities. In weighting scores to give the most accurate overall assessment of the sites' relative importance, we felt it was important to give extra weight to the range of species and habitat types represented (completeness) and to the size of the tract. To accomplish this, we grouped the sites into small (a few hundred hectares or less), medium (500 to several thousand hectares), and large (thousands of hectares), then multiplied the completeness score by a size factor. For small sites we used a multiplier of one; for medium, three; and for large, five.

We feel that these systematically developed computerized ranks closely parallel to our overall professional assessments of the sites. On the basis of "gut feeling" alone, we were inclined to think that Mill Creek, Sugarfoot Hammock, and Hornsby Springs, all sites characterized by magnificent hardwood forests and minimal xeric habitat, should have made it into the high priority grouping, and we were not completely comfortable with Barr Hammock's high placement on the list. This type of site is comparable to a "straight B" student—good at everything, but not outstanding by any single criterion. Our results also paralleled "popular judgment," in that Prairie Creek was the subject of a major preservation effort by local conservationists and has now been earmarked for state purchase (along with Chacala Pond and part of Lochloosa Forest).

Table 3 disregards the sites' degree of endangerment and ranks them solely on quality. Note that Hornsby Springs, an outstanding tract maintained as part of a church camp and thus deemed relatively secure, jumps to a much higher place in this analysis.

Table 4 omits considerations of both endangerment and use potential, and lists the sites in order of scores on strictly ecological parameters. Buzzard's Roost, a small, botanically significant hammock with complex, karst topography, rises substantially in priority here.

Table 5 ranks the sites in order of their prospects for long-term viability. We felt the main determinants of this were connectedness, manageability, diversity and size, so we weighted the scores to emphasize those parameters. We assumed that the site was to be set aside, so short-term vulnerability had no bearing on this analysis.

Table 6 is weighted to reflect the usual concerns in establishing a greenbelt: threats to the parcels, recreational and educational potentials, management costs and continuity. Interestingly, although we did not take into account geographic location, about half the sites with high ranking on this list have been proposed as components of a greenbelt for the city of Gainesville. Only three sites from the bottom half of the list have been so suggested.

Table 7 shows how we think The Nature Conservancy (TNC) would rank these sites. TNC is most concerned with protecting rare species and uses a methodology that considers threats to the site and the prospects for successfully maintaining it, but it does not customarily incorporate context parameters such as connectedness. Sugarfoot Hammock, one of only two recorded sites for the taxonomically distinct Sugarfoot fly, *Nemapalpus nearticus* Young, comes to the top of the list. Hickory Sink, with perhaps (at the time of our ranking) the state's finest upland pine forest, a bat cave and species of rare invertebrates, ranks second.

Since there has been so much discussion in the literature about the pros and cons of wildlife corridors and natural area linkages (e.g. Noss, 1987c), we did a final ranking run (Table 8) omitting these considerations. In this analysis, Palm Point Hill, a small, isolated site with great habitat diversity, moves to near the top of the list. Gum Root Swamp drops close to the bottom. This site is an extensive area of flatwoods and swamp of modest ecological quality, but it forms a critical link between the Paynes Prairie-Prairie Creek-Lochloosa Forest systems and the wetlands to the north.

Taken together, these tables constitute a sensitivity analysis that shows that no one factor is overwhelmingly important in determining the ranking order of these sites. Eliminating consideration of a criterion may change a site's place in the order by three or four slots, but the general pattern remains essentially the same, even when the emphasis changes. The best sites stay near the top while lower quality sites stay near the bottom. To facilitate planning discussions, we grouped the sites into arbitrary high, medium and low priority categories, in line with the breakpoints between groups of scores in Table 2.

After formulation of a more detailed protection plan, taking into account the many non-ecological factors involved in finalizing boundaries, the rankings could be refined by incorporating diversity and acreage data for each habitat on a site into the completeness x size score.

In general, we are pleased with the ranking system and feel that it provides an acceptable method for evaluating relative priorities at the preliminary overview level.

## ACKNOWLEDGMENTS

Dozens of Alachua County conservationists assisted us with this

study. We particularly appreciate input from our fellow KBN study team members, Bob Simons and Jim Newman, from Bill Kinser and Latane Donelin of the Alachua County Department of Planning and Development and from John Hendrix and Mike Campbell of the Alachua County Department of Environmental Services.

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**Table 2.**  
Recommended Priority Ranking for Alachua County Upland Ecosystems

Rank	Site	CRITERIA							Site Total
		Vulnerability	Rarity	Connectedness	Completeness		Manageability	Nature-Oriented Human Use Potential	
					Criteria Score	Size Multiplier			
<b>HIGH PRIORITY</b>									
1	Prairie Creek	3.5	4.0	5.0	5.0	5.0	5.0	4.5	71.0
2	Santa Fe River	3.0	4.0	5.0	4.0	5.0	4.0	5.0	63.0
3	Lochloosa Forest	3.0	3.5	5.0	4.0	5.0	4.0	3.5	60.5
4	Barr Hammock	3.5	3.0	4.0	4.0	5.0	4.5	4.0	59.0
5	Watermelon Pond	4.0	3.0	4.0	3.5	5.0	4.0	4.0	55.5
6	Hickory Sink	4.5	4.5	4.5	4.0	3.0	4.0	3.5	54.5
<b>MEDIUM PRIORITY</b>									
7	Sugarfoot Hammock	5.0	5.0	3.5	3.0	3.0	4.0	3.5	50.0
8	Chacala Pond	3.0	3.0	4.5	3.5	3.0	4.0	4.5	49.5
9	Mill Creek	3.0	3.0	3.0	4.0	3.0	4.5	4.0	47.5
10	Hatchet Creek	3.0	3.0	5.0	3.0	3.0	3.0	3.5	45.5
11	Parchman Pond Scrub	5.0	3.0	3.0	3.0	3.0	4.0	3.0	44.0
12	Hornsby Springs	2.0	3.5	4.0	4.0	1.0	5.0	4.0	44.0
13	Kanapaha Prairie	4.0	3.0	3.0	3.0	3.0	4.0	4.0	44.0
14	Gum Root Swamp	2.0	2.0	5.0	3.0	3.0	3.5	3.0	43.5
15	Millhopper Flatwoods	4.5	2.0	4.5	3.0	3.0	3.0	3.0	43.0
16	South LaCrosse Forest	2.5	2.0	4.0	3.0	3.0	3.5	2.0	40.0
<b>LOW PRIORITY</b>									
17	Palm Point Hill	3.5	3.0	3.0	3.5	1.0	4.0	4.5	38.5
18	Fred Bear Hammock	5.0	3.0	3.0	3.0	1.0	4.0	3.5	38.5
19	Rocky Creek	3.0	2.0	4.0	2.0	3.0	3.5	2.5	38.0
20	Buzzard's Roost	2.0	4.0	3.0	3.0	1.0	4.0	3.0	37.0
21	Santa Fe Creek	3.0	2.0	3.0	2.0	3.0	4.0	2.5	36.5
22	North San Felasco Hammock	3.0	2.0	4.0	3.0	1.0	4.0	2.0	36.0
23	Shenks Flatwoods	3.0	3.0	4.0	2.5	1.0	3.0	3.0	35.5
24	Serenola Forest	4.0	3.0	3.0	2.5	1.0	3.5	3.0	35.0
25	Domino Hammock	3.0	3.0	2.5	3.0	1.0	4.0	3.0	34.5
26	Moss Lee Lake Sandhill	3.0	3.5	3.0	3.0	1.0	3.0	2.5	33.5
27	Beech Valley	4.0	3.0	2.0	2.0	1.0	4.0	3.0	33.0
28	Northeast Lake Altho Flatwoods	3.0	2.0	4.0	2.0	1.0	3.0	2.0	32.0
29	South Melrose Flatwoods	3.0	2.0	3.0	2.0	1.0	3.0	2.0	29.0
	Criteria Weight	1.0	2.0	3.0			1.0	3.0	1.0

**Table 3.**  
Priority Ranking for Alachua County Upland Ecosystems Based on Site Quality Considerations Alone

Rank	Site	CRITERIA							Site Total
		Vulnerability	Rarity	Connectedness	Completeness		Manageability	Nature-Oriented Human Use Potential	
					Criteria Score	Size Multiplier			
1	Prairie Creek	3.5	4.0	5.0	5.0		5.0	4.5	23.5
2	Santa Fe River	3.0	4.0	5.0	4.0		4.0	5.0	22.0
3	Hornsby Springs	2.0	3.5	4.0	4.0		5.0	4.0	20.5
4	Hickory Sink	4.5	4.5	4.5	4.0		4.0	3.5	20.5
5	Lochloosa Forest	3.0	3.5	5.0	4.0		4.0	3.5	20.0
6	Chacala Pond	3.0	3.0	4.5	3.5		4.0	4.5	19.5
7	Barr Hammock	3.5	3.0	4.0	4.0		4.5	4.0	19.5
8	Sugarfoot Hammock	5.0	5.0	3.5	3.0		4.0	3.5	19.0
9	Watermelon Pond	4.0	3.0	4.0	3.5		4.0	4.0	18.5
10	Mill Creek	3.0	3.0	3.0	4.0		4.5	4.0	18.5
11	Palm Point Hill	3.5	3.0	3.0	3.5		4.0	4.5	18.0
12	Hatchet Creek	3.0	3.0	5.0	3.0		3.0	3.5	17.5
13	Kanapaha Prairie	4.0	3.0	3.0	3.0		4.0	4.0	17.0
14	Buzzard's Roost	2.0	4.0	3.0	3.0		4.0	3.0	17.0
15	Fred Bear Hammock	5.0	3.0	3.0	3.0		4.0	3.5	16.5
16	Gum Root Swamp	2.0	2.0	5.0	3.0		3.5	3.0	16.5
17	Parchman Pond Scrub	5.0	3.0	3.0	3.0		4.0	3.0	16.0
18	Shenks Flatwoods	3.0	3.0	4.0	2.5		3.0	3.0	15.5
19	Domino Hammock	3.0	3.0	2.5	3.0		4.0	3.0	15.5
20	Millhopper Flatwoods	4.5	2.0	4.5	3.0		3.0	3.0	15.5
21	North San Felasco Hammock	3.0	2.0	4.0	3.0		4.0	2.0	15.0
22	Serenola Forest	4.0	3.0	3.0	2.5		3.5	3.0	15.0
23	Moss Lee Lake Sandhill	3.0	3.5	3.0	3.0		3.0	2.5	15.0
24	South LaCrosse Forest	2.5	2.0	4.0	3.0		3.5	2.0	14.5
25	Beech Valley	4.0	3.0	2.0	2.0		4.0	3.0	14.0
26	Rocky Creek	3.0	2.0	4.0	2.0		3.5	2.5	14.0
27	Santa Fe Creek	3.0	2.0	3.0	2.0		4.0	2.5	13.5
28	Northeast Lake Altho Flatwoods	3.0	2.0	4.0	2.0		3.0	2.0	13.0
29	South Melrose Flatwoods	3.0	2.0	3.0	2.0		3.0	2.0	12.0
	Criteria Weight	0.0	1.0	1.0	1.0		1.0	1.0	

**Table 4.**  
Priority Ranking for Alachua County Upland Ecosystems Based on Strictly Ecological Criteria

Rank	Site	CRITERIA						Site Total	
		Vulnerability	Rarity	Connectedness	Completeness		Nature-Oriented Human Use Potential		
					Criteria Score	Size Multiplier			
1	Prairie Creek	3.5	4.0	5.0		5.0	5.0	4.5	19.0
2	Hickory Sink	4.5	4.5	4.5		4.0	4.0	3.5	17.0
3	Santa Fe River	3.0	4.0	5.0		4.0	4.0	5.0	17.0
4	Hornsby Springs	2.0	3.5	4.0		4.0	5.0	4.0	16.5
5	Lochloosa Forest	3.0	3.5	5.0		4.0	4.0	3.5	16.5
6	Sugarfoot Hammock	5.0	5.0	3.5		3.0	4.0	3.5	15.5
7	Barr Hammock	3.5	3.0	4.0		4.0	4.5	4.0	15.5
8	Chacala Pond	3.0	3.0	4.5		3.5	4.0	4.5	15.0
9	Mill Creek	3.0	3.0	3.0		4.0	4.5	4.0	14.5
10	Watermelon Pond	4.0	3.0	4.0		3.5	4.0	4.0	14.5
11	Buzzard's Roost	2.0	4.0	3.0		3.0	4.0	3.0	14.0
12	Hatchet Creek	3.0	3.0	5.0		3.0	3.0	3.5	14.0
13	Palm Point Hill	3.5	3.0	3.0		3.5	4.0	4.5	13.5
14	Gum Root Swamp	2.0	2.0	5.0		3.0	3.5	3.0	13.5
15	Fred Bear Hammock	5.0	3.0	3.0		3.0	4.0	3.5	13.0
16	North San Felasco Hammock	3.0	2.0	4.0		3.0	4.0	2.0	13.0
17	Parchman Pond Scrub	5.0	3.0	3.0		3.0	4.0	3.0	13.0
18	Kanapaha Prairie	4.0	3.0	3.0		3.0	4.0	4.0	13.0
19	Moss Lee Lake Sandhill	3.0	3.5	3.0		3.0	3.0	2.5	12.5
20	Shenks Flatwoods	3.0	3.0	4.0		2.5	3.0	3.0	12.5
21	South LaCrosse Forest	2.5	2.0	4.0		3.0	3.5	2.0	12.5
22	Domino Hammock	3.0	3.0	2.5		3.0	4.0	3.0	12.5
23	Millhopper Flatwoods	4.5	2.0	4.5		3.0	3.0	3.0	12.5
24	Serenola Forest	4.0	3.0	3.0		2.5	3.5	3.0	12.0
25	Rocky Creek	3.0	2.0	4.0		2.0	3.5	2.5	11.5
26	Beech Valley	4.0	3.0	2.0		2.0	4.0	3.0	11.0
27	Santa Fe Creek	3.0	2.0	3.0		2.0	4.0	2.5	11.0
28	Northeast Lake Altho Flatwoods	3.0	2.0	4.0		2.0	3.0	2.0	11.0
29	South Melrose Flatwoods	3.0	2.0	3.0		2.0	3.0	2.0	10.0
	Criteria Weight	0.0	1.0	1.0		1.0	1.0	0.0	

**Table 5.**  
Priority Ranking for Alachua County Upland Ecosystems Based on Prospects for Long-Term Maintenance of Ecological Integrity

Rank	Site	CRITERIA							Site Total
		Vulnerability	Rarity	Connectedness	Completeness Criteria Score	Size Multiplier	Manageability	Nature-Oriented Human Use Potential	
1	Prairie Creek	3.5	4.0	5.0	5.0	x 5.0	5.0	4.5	59.0
2	Santa Fe River	3.0	4.0	5.0	4.0	x 5.0	4.0	5.0	51.0
3	Lochloosa Forest	3.0	3.5	5.0	4.0	x 5.0	4.0	3.5	50.5
4	Barr Hammock	3.5	3.0	4.0	4.0	x 5.0	4.5	4.0	48.5
5	Watermelon Pond	4.0	3.0	4.0	3.5	x 5.0	4.0	4.0	44.5
6	Hickory Sink	4.5	4.5	4.5	4.0	x 3.0	4.0	3.5	42.0
7	Chacala Pond	3.0	3.0	4.5	3.5	x 3.0	4.0	4.5	39.0
8	Mill Creek	3.0	3.0	3.0	4.0	x 3.0	4.5	4.0	37.5
9	Sugarfoot Hammock	5.0	5.0	3.5	3.0	x 3.0	4.0	3.5	36.5
10	Gum Root Swamp	2.0	2.0	5.0	3.0	x 3.0	3.5	3.0	36.5
11	Hatchet Creek	3.0	3.0	5.0	3.0	x 3.0	3.0	3.5	36.0
12	Hornsby Springs	2.0	3.5	4.0	4.0	x 1.0	5.0	4.0	34.5
13	Millhopper Flatwoods	4.5	2.0	4.5	3.0	x 3.0	3.0	3.0	33.5
14	South LaCrosse Forest	2.5	2.0	4.0	3.0	x 3.0	3.5	2.0	33.5
15	Parchman Pond Scrub	5.0	3.0	3.0	3.0	x 3.0	4.0	3.0	33.0
16	Kanapaha Prairie	4.0	3.0	3.0	3.0	x 3.0	4.0	4.0	33.0
17	Rocky Creek	3.0	2.0	4.0	2.0	x 3.0	3.5	2.5	30.5
18	North San Felasco Hammock	3.0	2.0	4.0	3.0	x 1.0	4.0	2.0	29.0
19	Santa Fe Creek	3.0	2.0	3.0	2.0	x 3.0	4.0	2.5	29.0
20	Buzzard's Roost	2.0	4.0	3.0	3.0	x 1.0	4.0	3.0	28.0
21	Palm Point Hill	3.5	3.0	3.0	3.5	x 1.0	4.0	4.5	27.5
22	Fred Bear Hammock	5.0	3.0	3.0	3.0	x 1.0	4.0	3.5	27.0
23	Shenks Flatwoods	3.0	3.0	4.0	2.5	x 1.0	3.0	3.0	26.5
24	Dominio Hammock	3.0	3.0	2.5	3.0	x 1.0	4.0	3.0	25.5
25	Serenola Forest	4.0	3.0	3.0	2.5	x 1.0	3.5	3.0	25.0
26	Northeast Lake Altho Flatwoods	3.0	2.0	4.0	2.0	x 1.0	3.0	2.0	25.0
27	Moss Lee Lake Sandhill	3.0	3.5	3.0	3.0	x 1.0	3.0	2.5	24.5
28	Beech Valley	4.0	3.0	2.0	2.0	x 1.0	4.0	3.0	23.0
29	South Melrose Flatwoods	3.0	2.0	3.0	2.0	x 1.0	3.0	2.0	22.0
	Criteria Weight	0.0	1.0	3.0		1.0	3.0	0.0	

**Table 6.**  
Priority Ranking for Alachua County Upland Ecosystems Based on Typical Greenbelt Considerations

Rank	Site	CRITERIA							Site Total
		Vulnerability	Rarity	Connectedness	Completeness Criteria Score	Size Multiplier	Manageability	Nature-Oriented Human Use Potential	
1	Prairie Creek	3.5	4.0	5.0	5.0		5.0	4.5	58.0
2	Santa Fe River	3.0	4.0	5.0	4.0		4.0	5.0	54.0
3	Hickory Sink	4.5	4.5	4.5	4.0		4.0	3.5	53.5
4	Sugarfoot Hammock	5.0	5.0	3.5	3.0		4.0	3.5	52.5
5	Barr Hammock	3.5	3.0	4.0	4.0		4.5	4.0	51.0
6	Watermelon Pond	4.0	3.0	4.0	3.5		4.0	4.0	50.5
7	Chacala Pond	3.0	3.0	4.5	3.5		4.0	4.5	50.0
8	Fred Bear Hammock	5.0	3.0	3.0	3.0		4.0	3.5	49.5
9	Lochloosa Forest	3.0	3.5	5.0	4.0		4.0	3.5	49.0
10	Palm Point Hill	3.5	3.0	3.0	3.5		4.0	4.5	48.5
11	Hornsby Springs	2.0	3.5	4.0	4.0		5.0	4.0	48.5
12	Kanapaha Prairie	4.0	3.0	3.0	3.0		4.0	4.0	48.0
13	Parchman Pond Scrub	5.0	3.0	3.0	3.0		4.0	3.0	48.0
14	Mill Creek	3.0	3.0	3.0	4.0		4.5	4.0	47.5
15	Millhopper Flatwoods	4.5	2.0	4.5	3.0		3.0	3.0	45.5
16	Hatchet Creek	3.0	3.0	5.0	3.0		3.0	3.5	44.5
17	Serenola Forest	4.0	3.0	3.0	2.5		3.5	3.0	43.0
18	Beech Valley	4.0	3.0	2.0	2.0		4.0	3.0	42.0
19	Domino Hammock	3.0	3.0	2.5	3.0		4.0	3.0	41.0
20	Shenks Flatwoods	3.0	3.0	4.0	2.5		3.0	3.0	40.5
21	Gum Root Swamp	2.0	2.0	5.0	3.0		3.5	3.0	40.5
22	North San Felasco Hammock	3.0	2.0	4.0	3.0		4.0	2.0	40.0
23	Buzzard's Roost	2.0	4.0	3.0	3.0		4.0	3.0	40.0
24	Rocky Creek	3.0	2.0	4.0	2.0		3.5	2.5	39.0
25	Santa Fe Creek	3.0	2.0	3.0	2.0		4.0	2.5	38.5
26	Moss Lee Lake Sandhill	3.0	3.5	3.0	3.0		3.0	2.5	38.0
27	S. LaCrosse Forest	2.5	2.0	4.0	3.0		3.5	2.0	37.0
28	Northeast Lake Altho Flatwoods	3.0	2.0	4.0	2.0		3.0	2.0	36.0
29	South Melrose Flatwoods	3.0	2.0	3.0	2.0		3.0	2.0	34.0
	Criteria Weight	3.0	1.0	2.0	1.0		3.0	3.0	

**Table 7.**  
Priority Ranking for Alachua County Upland Ecosystems Based on a Traditional Conservation  
Approach Similar to that of The Nature Conservancy

Rank	Site	CRITERIA							Site Total
		Vulnerability	Rarity	Connectedness	Completeness Criteria Score	Size Multiplier	Manageability	Nature-Oriented Human Use Potential	
1	Sugarfoot Hammock	5.0	5.0	3.5	3.0	4.0	4.0	3.5	27.0
2	Hickory Sink	4.5	4.5	4.5	4.0	4.0	4.0	3.5	26.0
3	Prairie Creek	3.5	4.0	5.0	5.0	5.0	5.0	4.5	25.5
4	Santa Fe River	3.0	4.0	5.0	4.0	4.0	4.0	5.0	23.0
5	Lochloosa Forest	3.0	3.5	5.0	4.0	4.0	4.0	3.5	21.5
6	Hornsby Springs	2.0	3.5	4.0	4.0	5.0	5.0	4.0	21.5
7	Barr Hammock	3.5	3.0	4.0	4.0	4.5	4.0	4.0	21.0
8	Fred Bear Hammock	5.0	3.0	3.0	3.0	4.0	4.0	3.5	21.0
9	Parchman Pond Scrub	5.0	3.0	3.0	3.0	4.0	4.0	3.0	21.0
10	Buzzard's Roost	2.0	4.0	3.0	3.0	4.0	4.0	3.0	21.0
11	Mill Creek	3.0	3.0	3.0	4.0	4.5	4.0	4.0	20.5
12	Watermelon Pond	4.0	3.0	4.0	3.5	4.0	4.0	4.0	20.5
13	Kanapaha Prairie	4.0	3.0	3.0	3.0	4.0	4.0	4.0	20.0
14	Palm Point Hill	3.5	3.0	3.0	3.5	4.0	4.0	4.5	20.0
15	Chacala Pond	3.0	3.0	4.5	3.5	4.0	4.0	4.5	19.5
16	Moss Lee Lake								
	Sandhill	3.0	3.5	3.0	3.0	3.0	3.0	2.5	19.5
17	Beech Valley	4.0	3.0	2.0	2.0	4.0	4.0	3.0	19.0
18	Serenola Forest	4.0	3.0	3.0	2.5	3.5	3.0	3.0	19.0
19	Domino Hammock	3.0	3.0	2.5	3.0	4.0	4.0	3.0	19.0
20	Hatchet Creek	3.0	3.0	5.0	3.0	3.0	3.0	3.5	18.0
21	Shenks Flatwoods	3.0	3.0	4.0	2.5	3.0	3.0	3.0	17.5
22	Millhopper Flatwoods	4.5	2.0	4.5	3.0	3.0	3.0	3.0	16.5
23	North San Felasco								
	Hammock	3.0	2.0	4.0	3.0	4.0	4.0	2.0	16.0
24	South LaCrosse Forest	2.5	2.0	4.0	3.0	3.5	3.5	2.0	15.0
25	Santa Fe Creek	3.0	2.0	3.0	2.0	4.0	4.0	2.5	15.0
26	Gum Root Swamp	2.0	2.0	5.0	3.0	3.5	3.5	3.0	14.5
27	Rocky Creek	3.0	2.0	4.0	2.0	3.5	3.5	2.5	14.5
28	Northeast Lake Altho								
	Flatwoods	3.0	2.0	4.0	2.0	3.0	3.0	2.0	14.0
29	South Melrose								
	Flatwoods	3.0	2.0	3.0	2.0	3.0	3.0	2.0	14.0
	Criteria Weight	1.0	3.0	0.0	1.0	1.0	1.0	0.0	

**Table 8.**

Priority Ranking for Alachua County Upland Ecosystems Based on "Simberloff Scenario" Disregarding Corridors and Linkages

Rank	Site	CRITERIA							Site Total
		Vulnerability	Rarity	Connectedness	Completeness Criteria Score	Size Multiplier	Manageability	Nature-Oriented Human Use Potential	
1	Prairie Creek	3.5	4.0	5.0	5.0	5.0	5.0	4.5	22.0
2	Sugarfoot Hammock	5.0	5.0	3.5	3.0	4.0	4.0	3.5	20.5
3	Hickory Sink	4.5	4.5	4.5	4.0	4.0	4.0	3.5	20.5
4	Santa Fe River	3.0	4.0	5.0	4.0	4.0	4.0	5.0	20.0
5	Barr Hammock	3.5	3.0	4.0	4.0	4.5	4.5	4.0	19.0
6	Hornsby Springs	2.0	3.5	4.0	4.0	5.0	4.0	4.0	18.5
7	Palm Point Hill	3.5	3.0	3.0	3.5	4.0	4.5	4.5	18.5
8	Fred Bear Hammock	5.0	3.0	3.0	3.0	4.0	4.0	3.5	18.5
9	Mill Creek	3.0	3.0	3.0	4.0	4.5	4.5	4.0	18.5
10	Watermelon Pond	4.0	3.0	4.0	3.5	4.0	4.0	4.0	18.5
11	Chacala Pond	3.0	3.0	4.5	3.5	4.0	4.5	4.5	18.0
12	Lochloosa Forest	3.0	3.5	5.0	4.0	4.0	4.0	3.5	18.0
13	Kanapaha Prairie	4.0	3.0	3.0	3.0	4.0	4.0	4.0	18.0
14	Parchman Pond Scrub	5.0	3.0	3.0	3.0	4.0	4.0	3.0	18.0
15	Beech Valley	4.0	3.0	2.0	2.0	4.0	4.0	3.0	16.0
16	Domino Hammock	3.0	3.0	2.5	3.0	4.0	4.0	3.0	16.0
17	Buzzard's Roost	2.0	4.0	3.0	3.0	4.0	4.0	3.0	16.0
18	Serenola Forest	4.0	3.0	3.0	2.5	3.5	3.5	3.0	16.0
19	Hatchet Creek	3.0	3.0	5.0	3.0	3.0	3.0	3.5	15.5
20	Millhopper Flatwoods	4.5	2.0	4.5	3.0	3.0	3.0	3.0	15.5
21	Moss Lee Lake Sandhill	3.0	3.5	3.0	3.0	3.0	3.0	2.5	15.0
22	Shenks Flatwoods	3.0	3.0	4.0	2.5	3.0	3.0	3.0	14.5
23	North San Felasco Hammock	3.0	2.0	4.0	3.0	4.0	4.0	2.0	14.0
24	Gum Root Swamp	2.0	2.0	5.0	3.0	3.5	3.5	3.0	13.5
25	Santa Fe Creek	3.0	2.0	3.0	2.0	4.0	4.0	2.5	13.5
26	Rocky Creek	3.0	2.0	4.0	2.0	3.5	3.5	2.5	13.0
27	South LaCrosse Forest	2.5	2.0	4.0	3.0	3.5	3.5	2.0	13.0
28	Northeast Lake Altho Flatwoods	3.0	2.0	4.0	2.0	3.0	3.0	2.0	12.0
29	South Melrose Flatwoods	3.0	2.0	3.0	2.0	3.0	3.0	2.0	12.0
	Criteria Weight	1.0	1.0	0.0	1.0	1.0	1.0	1.0	

# Using Aerial Photography to Model Species-Habitat Relationships: The Importance of Habitat Size and Shape

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**Abstract:** The habitat relationships of six species of breeding birds that are primarily associated with edge habitats were studied for five years on 23 plots on a 4850-ha wildlife management area in central New York. Habitat size, insularity, structural complexity, and measurements of the degree to which species habitats were clumped were derived using a technique for the quantification of edge and the spatial arrangement of habitat. Additional information was gathered on the quantity and vertical distribution of foliage. A variable related to habitat size was the primary correlate of the occurrence of five of the six species. The occurrence of one species, the alder flycatcher, was strongly negatively correlated with the insularity of its habitat. The average classification accuracy of jackknifed discriminant function models and tests of models for the six species was 89.3% and 82.7%, respectively. The results indicate that air photo analysis can be used to identify the threshold habitat size with optimal shape at which a breeding pair of a species will use a site. Application of this approach to the management of rare species is suggested.

Pages 34 - 46. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The relationship of various characteristics of habitat to distributions of wildlife has long interested ecologists (*e.g.* Saunders, 1936; Kendeigh, 1945). A number of different characteristics have been suggested as determining, or at least correlating with, the occurrence and abundance of species. Among these, vegetation structure (MacArthur and MacArthur, 1961), habitat size and insularity (MacArthur and Wilson, 1967), and productivity (*e.g.* Rabenold, 1978) have often been identified as influences on habitat selection.

In the past two decades, wildlife biologists increasingly have employed biometric models in their efforts to understand species-habitat relationships. Despite the variety of variables developed to quantify species-habitat relationships, and despite the fact that many species are described as being associated with "edge" habitat, few variables have been developed that attempt to quantify the habitat components composing an "edge" for a particular species. In addition, even fewer of these variables have described the 2-dimensional, spatial arrangement (see Wiens, 1974; Roth, 1976) of edge habitat on the landscape.

In a study conducted at the Connecticut Hill Wildlife Management Area (WMA) in central New York, between 1977 and 1981, I attempted to identify characteristics of habitats associated with 59 species of breeding birds that had been grouped into 19 guilds, based on their habitat associations and foraging characteristics (Keller, 1986). This effort employed both traditional measures and a technique developed for the quantification of edge and the spatial arrangement of habitat (Keller *et al.*, 1979). Six of the species in the study: alder flycatcher (*Empidonax alnorum*), black-and-white warbler (*Mniotilta varia*), black-throated blue warbler (*Dendroica caerulescens*), cedar waxwing (*Bombocilla cedrorum*), purple finch (*Carpodacus purpureus*), and ruby-throated hummingbird (*Archilochus colubris*) were considered to be the only members of their respective guilds, and five of the six were associated with specific edge types. This paper presents the results of attempts to model the relationships of these six species to variables that describe the size, insularity, productivity, structure, and heterogeneity of their

habitats. Implications of the types of variables most frequently included in the models and the application of modeling to the management of rare species are then discussed.

## METHODS

### Study Site:

The study was conducted at the Connecticut Hill Wildlife Management Area (WMA) in south-central New York, 25 km southwest of the Cornell University campus in Ithaca. Connecticut Hill is an upland area with elevations ranging from 396 to 640 m. It comprises 4,850 ha of second-growth, deciduous and coniferous-deciduous forest, abandoned oldfields, conifer plantations, and variously aged small clearcuts.

Twenty-three plots ranging in size from 1.0 to 24.0 ha were studied during the five year period 1977-81. These included 14 clearcuts, five oldfields, and four forests. Clearcuts and forests were categorized within general community type by age and size. Oldfields were categorized by structural type and size (Table 1).

### Bird Survey:

The breeding bird survey that included the six species considered here began in late May following the peak of migration. The spot-map census technique (Kendeigh, 1944; International Bird Census Committee, 1970) was initially the census method of choice because it is the most precise and most widely used method available (Robbins, 1978). However, because of the time constraint of a single observer censusing 23 plots, and because of the density of vegetation on clearcut plots, a modified spot-map technique was developed for this study.

A line transect (or transects) was established parallel to and midway between the longest sides of what were universally rectangular or L-shaped plots. Listening stations were established at 50-m intervals along the transect. I stopped at each station and recorded the azimuth in degrees, estimated distance, location in the habitat, sex (if identifiable), and activity of all individuals seen or heard during a 5-minute period. In all other respects, the techniques employed were those of the spot-map method.

All counts were conducted between 0500 and 0900 hr EDT from late May until early July. For each count, the time and direction of

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**Table 1.**  
Site characteristics of the 23 plots included in the study.

Plot	No. of years included in study	Plot age (yrs)	Stand Type <sup>a</sup>	Overstory trees left standing <sup>b</sup>	Size (ha)
*CCA-1	4	2-5	aspen-red maple	N	1.03
CCA-2	5	2-6	aspen	N	1.23
CCA-3	4	2-5	aspen-oldfield	N	2.80
CCA-4	4	2-5	jack pine/ash-red maple	N	3.15
CCA-5	4	2-5	aspen-red maple	Y	3.20
CCA-6	5	2-6	aspen-old field	Y	3.89
CCA-7	4	2-5	aspen-old field	Y	7.35
CCA-8	4	2-5	jack pine/ash-red maple	Y	10.46
CCA-9	4	2-5	jack pine/ash-red maple-oak	Y	13.82
CCA-10	4	2-5	aspen-old field	Y	24.15
CCB-1	5	4-8	NH-H/pincherry NH	Y	5.22
CCB-2	5	4-8	NH-H/pincherry NH	N	6.17
CCC-1	5	14-18	red maple-oak	N	2.20
CCC-2	2	24-25	red maple-white pine	Y	6.37
OFF-1	4	-	mowed field w/shrub plantings	NA	5.36
OFF-2	4	-	mowed field w/shrub and spruce plantings	NA	7.61
OFI	4	-	savannah w/red maple and white pine	NA	3.92
OFA-1	4	48-51	dense shrub-sapling-pole-timber oldfield	NA	7.49
OFA-2	4	38-41	open shrub-sapling oldfield	NA	7.70
F-SGA	5	55-60	second growth deciduous pole-sawtimber	NA	10.00
F-NHHB	4	75	sawtimber northern hardwoods-hemlock	NA	8.89
F-NHHC	5	100	sawtimber northern hardwoods-hemlock	NA	8.47
F-NHHD	4	120	sawtimber northern hardwoods-hemlock	NA	9.76

<sup>a</sup> If precut and postcut stand types differed, the value to the left of the slash (/) is the precut stand type.

<sup>b</sup> N = no; Y = yes; NA = not applicable.

\* Community Type: CC = clearcut; OF = oldfield; F = forest; SG = second growth; NHH = northern hardwood-hemlock. Increasing age within clearcut or forest: A-D

Oldfield Structural Type: F = frequently mowed (every 2-3 yr); I = infrequently mowed (every 5 yr); A = advanced woody invasion.

travel along transects were systematically varied to minimize sampling bias. Three to five counts were made in each area each year of the study with 8-14 d (median = 12 d) elapsing between successive visits to a plot.

Additional observations were made on each plot during the period from 0900 to 1500 hr EDT, because, due to the large number of plots, it was not possible to visit each plot the recommended minimum of six times during the early morning (International Bird Census Committee, 1970). These additional visits were used to corroborate records of contacts made during early morning visits, locate nests, detect raptor use, and locate inconspicuous or wide-ranging songbirds. Visits consisted

of noting the location of birds and nests while slowly walking along the plot boundaries or randomly traversing the plot. Frequently, stops of from five to 15 minutes were made and data were recorded as on early morning visits. The duration of visits was usually one hour, with larger plots visited proportionately more frequently than smaller plots to ensure equal coverage. Evening visits, between 1900 and 2200 hr, were made on forested plots to substantiate thrush (*Turdinae*) activity. Observations of a given plot never coincided with the date that plot was censused. In all years of the study, observation periods were continued until late July on clearcuts and oldfields. The number of censuses and observation periods combined totaled from six to nine for each

plot, each year. A plot of the species accumulation rate for all censuses conducted between 1977 and 1980 (89 plot years) demonstrated that >97% of all species were detected by the antepenultimate visit (Figure 1). Mist net samples collected in 1981 (Keller, 1986) further corroborated the efficacy of the modified spot-map technique.

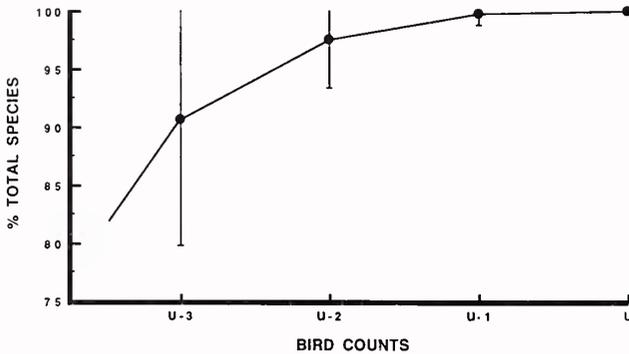


FIGURE 1: Bird species accumulation rates (with standard errors) for plots at the Connecticut Hill WMA censused between 1977 and 1980 (N=89) using a modified spot-map technique. Counts are identified as ultimate (U), penultimate (U-1), etc.

rated the efficacy of the modified spot-map technique.

The four forest plots did not have distinct borders and did not lend themselves as well to the modified spot-map technique as did the open-canopy communities. However, because of the time constraints on the observer and for the sake of consistency, these plots were censused in the same manner as the clearcut and oldfield sites.

After repeated checks, the average maximum distance on either side of the transect line at which birds on forest plots could be heard and identified competently was estimated to be 90 m. Sizes of the forest plots were estimated by multiplying the transect length by the transect width, 180 m, then adding to that the area a circle with radius 90 m, which represented the sum of the two hemispherical areas at each end of the transect. Sizes of all open-canopy plots were calculated from air photos at a scale of 1:2000 using a polar planimeter.

Registrations on the plot maps were considered to represent breeding pairs if there were at least two registrations of high territorial significance (e.g. song) that represented either a) two early morning visits to that plot, or b) one early morning visit and one observation period that occurred more than seven days prior to, or seven days after, that early morning visit. When a territory proved to be partly outside the study plot, the fraction of the territory estimated to be inside the plot border was used in calculating the density for that species (Johnston and Odum, 1956). The actual number of occupied territories on each plot then was used to calculate the estimated number of pairs per 40 ha of similar habitat.

#### Air Photo Analysis:

The structural complexity of the vegetation on the Connecticut Hill sites ranged from primarily horizontal heterogeneity on young clearcuts and oldfields to primarily vertical heterogeneity in forests. Although a variety of measures have been developed to quantify both horizontal and vertical heterogeneity (e.g. Wiens, 1974; Roth, 1976; Pielou, 1977), all suffer from at least one of a number of shortcomings that limit the ability to quantify the spatial arrangement (pattern) of multiple structural components, such as trees and shrubs, within a given area (for discussion, see Keller *et al.*, 1979).

To address this problem, Keller *et al.* (1979) proposed a method that

abstracts the information from air photos or other remotely sensed imagery of a site to an overlying grid of hexagonal cells. Each cell is classified as to habitat component type and numbered on a Cartesian (X,Y) coordinate system so that each cell can be located in 2-dimensional space in relation to every other cell.

A series of algorithms and APL computer programs were developed that employ this approach to provide a variety of measures of horizontal habitat structure. Attributes tabulated include areas of all contiguous cells of each type (called a "cluster"), lengths of all contiguous cell sides of each type (called an "edge"), gradient analysis by cell type or edge type, minimum linear distances between clusters or edges of the same type, and maximum density on the plot of specified edge types in increasingly larger "circular" samples. The collective name of this system is SPADIST (Spatial Distribution); for examples of program output, see Keller (1986).

For this study, air photo coverage of all plots was obtained on 22 May, 1977 and 10 June, 1980. Plots were photographed in stereo with 70 mm black and white film from an elevation of approximately 1100 m. Supplemental air photo information was obtained from the Resource Information Laboratory at Cornell. Stereo enlargements were printed at a scale of 1:5000. One photo of each plot was enlarged to 1:2000 for use as a base map because of the need for larger scale imagery in analyzing intra-community heterogeneity. During reproduction of the 1:2000 enlargements, ground truth measurements were used to correct for scale errors.

The two most important considerations in any application of this technique are the identification of an appropriate classification system for the habitat components on the site and the choice of a meaningful grid-cell size. Both should consider the biology of the bird species under study. Grid-cell size is dictated somewhat by the cost of analysis. Habitat components (Table 2) were as finely subdivided as resolution of the photos (1:2000) allowed. A cell size of 0.25 cm<sup>2</sup> (100 m<sup>2</sup> on the ground) was chosen as being the smallest workable cell size for this scale that still had biological relevance for this analysis. One hundred m<sup>2</sup> approximates the canopy of a moderately-sized tree; therefore, each cell classified individual structural elements that are likely to be used by birds in habitat selection (Hilden, 1965). Using the stereo imagery in conjunction with a hexagonal-celled grid overlying the 1:2000 base maps, each cell was identified to type and recorded on a separate grid map of each plot. Although it is possible to infer understory structure of closed-canopy communities from air photo information (cf. Roebig, 1979; Skaley, 1981), forested plots (F) were not included for some types of SPADIST analysis. SPADIST variables derived from the 1977 air photos were used in analysis of the 1977-78 bird survey data. The 1980 air photos were applied to 1979-81 bird survey data.

#### Habitat Types and Quantification:

In order to quantify the distribution and abundance of the habitats used by avian species, it first was necessary to identify the components that collectively describe the habitat type for a particular species. Habitat types were identified (Table 3) that corresponded to suspected habitat associations of five of the six species, based on a literature review (e.g. Kendeigh, 1945; Stein, 1958; Holmes *et al.*, 1979; Hamel *et al.*, 1982), discussions with colleagues, and my own observations. The alder flycatcher was thought to be most associated with saplings and tall shrubs adjacent to open areas (habitat #10, Table 3), the black-and-white warbler with deciduous sawtimber canopy adjacent to dense shrubs or sprouts (habitat 12), the black-throated blue warbler with northern hardwoods-hemlock canopy adjacent to dense shrubs or

sprouts (habitat 13), the cedar waxwing with mid-canopy foliage adjacent to open areas (habitat 15), and the purple finch with coniferous sawtimber canopy adjacent to shrubs or sprouts (habitat 16). Literature on the ruby-throated hummingbird and field observations did not suggest a particular habitat association for this species. Instead, other habitat types that were most highly correlated with the hummingbird in univariate analysis were used in subsequent multivariate analyses. These included open shrubs (habitat 4, Table 3), shrub/opening edge (habitat 9), and sapling/opening edge (habitat 10).

**Table 2.** Habitat component classification system for the Connecticut Hill WMA. Percentages are the proportion of a single map cell (actual area = 100 m<sup>2</sup>) represented by the type. See text for explanation of air photo analysis.

Habitat Component #	Description
1	>85% sprouts <sup>a</sup> or shrubs
2	10-85% sprouts or shrubs
3	10-85% sprout/sapling conifers
4	<10% sprouts or shrubs ( <i>i.e.</i> , open grass)
5	deciduous sapling/pole w/>33% sprouts
6	deciduous sapling/pole w/<33% sprouts and >67% canopy closure
7	deciduous sawtimber w/>33% sprouts
8	deciduous sawtimber w/<33% sprouts
9	conifer pole/sawtimber w/>33% sprouts or w/live branches to ground level
10	conifer pole/sawtimber w/<33% sprouts or w/dead branches to ground level
11	fruit trees or tall (>3 m) shrubs
12	bare ground
13	water
14	mixed deciduous-coniferous sapling/poletimber
15	mixed deciduous-coniferous sawtimber
16	deciduous sapling/poletimber w/<33% sprouts and <67% canopy closure

<sup>a</sup>Tree diameter classes:

Sprout:	<4 cm dbh (included seedlings, root suckers, and stump sprouts)
Sapling:	4-10 cm dbh
Poletimber:	10-20 cm dbh
Sawtimber:	>20 cm dbh

A set of six SPADIST programs that perform 2-dimensional gradient analysis of either solid (*i.e.*, areal) habitat components, such as canopies of trees, or edge components, such as interfaces between trees and adjacent shrubs, along each of the three major axes of the hexagonal-celled map was developed to quantify the degree to which habitat distribution is clumped. The output from these programs was analyzed using two techniques of gradient analysis. The first of these, the variance/mean ratio (V/M) (Pielou, 1977), calculates the variation in the number of cells or edges of type T (*i.e.*, designated habitat type) that occur within the bands perpendicular to the axis of the gradient analysis (see Keller *et al.*, 1980; Keller, 1986). The second gradient analysis measure was the Run's Index, I/L (Pielou, 1967), which is based on the number of consecutive bands along the gradient that contain type T cells or edges. Therefore, the two indices are developed from information that is in two separate and perpendicular dimensions in horizontal

space (*i.e.*, within bands for the variance mean ratio and across consecutive bands for the Run's Index). In order to better describe the 2-dimensional nature of habitat clumping and to mathematically accentuate differences in the degree of clumping, the two indices were multiplied by one another to create the variable VMRI (Table 4).

As an alternative to VMRI, a 2-dimensional measure of habitat clumping and potential territory size used the arguments of Covich (1976) that in an energetics context, a circle is the optimal shape in horizontal space for an all-purpose territory. I measured the maximum diameter circle (MDC) that fit within each solid habitat type on each plot directly from the 1:2000 hexagonal-celled habitat maps.

In order to measure the size of edge habitats in a manner comparable to MDC, an edge-scanning algorithm (ESCAN) was developed. This program locates areas on the map with the highest density of edges (m/m<sup>2</sup>) for a given habitat type T within a series of increasingly larger "circular" (actually hexagonal) samples. A variable called the diameter of the equivalent area circle (DEAC) was calculated by this program. The DEAC was defined as the diameter of a circle with an area equal to the actual area sampled by ESCAN in which the highest density of habitat type T edges occurred on the plot. An example of ESCAN output is found in Keller (1986). The density of edges was calculated as an edge index (EI), where EI = the number of edges of type T in the sample ÷ (area sampled)<sup>1/2</sup> (Patton, 1975). Based on samples of ESCAN, values of EI and DEAC were calculated for all habitat types composed of edges (see Table 3).

#### Foliage Profiles:

Foliage-height profiles (*i.e.*, the proportion of foliage in each 1 m interval above the ground) were determined for each plot using the camera technique described by Aber (1979) in order to achieve a more detailed description of vertical foliage distribution than could be obtained by air photos alone. A method then was developed that used a series of Pearson Type I distributions (Elderton and Johnson, 1969) to extrapolate the foliage profile estimates from 1979 to all five years of the study (Keller, 1986). Estimates of the changing vertical distribution of foliage over time were made for all clearcuts less than 20 yrs old. The oldfields (OF), forests (F), and clearcut CCC-2 were assumed to have constant foliage profiles over the course of the study.

#### Leaf Area Estimation:

Leaf area production (Whittaker *et al.*, 1974) was used as an estimator of primary productivity. Estimation of leaf area was as described by Aber (1979). As with the foliage profile data, it was desirable to produce estimates of the year-to-year changes in leaf area production on clearcuts. Examination of the data from this study and from Aber's (1979) work indicated that the pattern of leaf area regeneration following cutting could be modeled by the Mitscherlich curve (Snedecor and Cochran, 1967). The leaf area estimates and the foliage profile estimates then were combined as a means of describing the vertical distribution of available foliage (*e.g.* variables LALOW, LAMID) at a given time during succession (Keller, 1986). The Mitscherlich curve and its inverse were also used to derive estimates of changes in MDC and DEAC, respectively, for each habitat type over the course of the study. This was reasonable because the changes in MDC and DEAC values between the 1977 and 1980 air photos were essentially measures of the rate of canopy closure in each habitat type (Dr. Douglas Robson, Cornell Biometrics Unit, pers. comm.).

#### Data Analysis:

A correlation matrix was produced for all of the original variables in

**Table 3.** Definition of habitats using the habitat component classification system described in Table 2. For consistency and ease of reference, habitats were numbered as per Keller (1986) and a subsequent manuscript (in preparation). Habitat type 4 is a "solid" habitat type. Habitat types 9, 10, 12, 13, 15, and 16 are "edge" habitats. Each 2XN matrix below represents all the combinations of adjacent habitat components (Table 2) composing an edge habitat. For example, 1 over 2 means dense shrubs (type 1) adjacent to open shrubs (type 2). See the Methods for a discussion of the relationship of the habitats to the species in the study.

Habitat number	Habitat description	a Habitat components composing the habitat	Vertical <sup>b</sup> Profile
4	open shrub	2 3 11 16	0-3 m
9	shrub/opening	1 1 1 1 2 2 2 2 3 3 4 4 5 2 3 4 16 5 7 9 11 5 7 5 7 16	0-3 m
10	sapling/opening	2 2 3 5 <sup>c</sup> and + 1 1 <sup>d</sup> or + 1 <sup>e</sup> 5 11 5 11 5 11 2	0-3 m
12	canopy/shrub	interior <sup>f</sup> 1 1 5 5 7 7 8 7 8 8	0-3 m
13	NHH/shrub	1 5 7 15 15 15	0-3 m
15	mid-canopy/opening	interior 1 1 2 2 2 3 3 4 4 4 5 5 5 6 6 11 11 5 6 5 6 11 5 6 5 6 11 11 12 16 11 16 12 16	3-7 m
16	coniferous canopy/shrub	1 1 3 3 5 5 9 10 9 10 9 10	0-7 m

<sup>a</sup> Numbers refer to habitat component numbers in Table 2.

<sup>b</sup> Portion of vertical profile from which leaf area estimates were used to calculate the variables DEACLA and EIDCLA (Table 4).

<sup>c</sup> Used for jack pine clearcuts (CCA-4,8,9) in 1977-78.

<sup>d</sup> Used for all CCA plots other than jackpine clearcuts in 1977-78, for all CCC and OF plots in all years, and for CCA-4,8,9 in 1979-80.

<sup>e</sup> Used for all CCA plots in 1979-81 except CCA-4,8,9 and for both CCB plots in all years.

the data set (Keller, 1986). This allowed comparisons of the degree of correlation between various dependent and independent variables and between two or more independent variables correlated with the same dependent variable. If two or more independent variables were correlated significantly both with a particular dependent variable ( $r > .5$ ,  $p < .001$ ) and with each other ( $r > .6$ ,  $p < .001$ ) and if one of the variables was judged less biologically significant in determining species occurrence, it was deleted from the list of variables being considered. This procedure was intended to reduce redundancy within the data set and to reduce the risk of obtaining spurious correlations, as can occur when the number of variables included in the analysis is too great. This procedure resulted in the inclusion of the variables listed in Table 5 in subsequent analyses.

Two methods were used to analyze the relationships of species occurrence and density to the various independent variables used to characterize the habitats. The first of these, the BMDP application of Stepwise Discriminant Function Analysis (SDFA) (Jennrich and Sampson, 1983), was used to examine species occurrence. In this analysis, the data set was split, a model was built that would predict species occurrence from a portion of the data, and then the model was tested, using the remaining portion of the data. To achieve the objectives of the SDFA model-building/testing scheme, a random procedure was identified for splitting the data set. Due to the site tenacity of many bird species (*e.g.* Harvey *et al.*, 1979; Bedard and LaPointe, 1984; Sikes and Arnold, 1984), plot years were deemed not to be independent. Therefore, plots (all years included) were classified into nine strata according to major physiognomic or physiographic characteristics of the 23 plots (Appendix I). In order to maintain a scaled-down version of the original plot population, all but one of the plots from each stratum were selected randomly for each iteration of the model-building/testing procedure. All years of information associated with

the selected plots were used to build a discriminant classification model for each species. The remaining plots were used to test the discrimination power of the model. The main difference between the random splitting procedure used here and the base BMDP procedure is that the former allows variable selection (S), as well as discriminant functions (DFA), to differ from one iteration to the next. Iterations of the procedure were run until the standard deviation of the test's classification accuracy became constant (D.S. Robson, pers. comm.). No fewer than six nor more than 21 iterations were performed.

The results of this model-building/testing procedure then were compared with the results of an SDFA of the six avian species using the entire data set. This allowed evaluation of the overall analyses in terms of how representative they were, how sensitive they were to changes in the sample, and how well certain independent variables predicted occurrences of particular species.

The second analytical technique, Stepwise Maximum r-Square Improvement Multiple Regression (MAXR Regression) (SAS User's Guide: Statistics, 1982), was used to evaluate the relationship of the density of the six species in question to the independent habitat variables.

## RESULTS

The occurrence of the six species of breeding birds on each plot from 1977-81 is shown in Table 6. Cedar waxwing, ruby-throated hummingbird and alder flycatcher were the most common species. They occurred in 40%, 26%, and 23%, respectively, of all plot years and on 45%, 57%, and 43%, respectively, of all plots. In contrast, the black-throated blue warbler, purple finch and black-and-white warbler occurred in only 15%, 12%, and 10%, respectively, of all plot years, and on only 22%, 17%, and 22%, respectively, of all plots.

**Table 4.** Independent variables used in the analysis of avian species-habitat relationships.

Variable	Description or Reference
CONCOR	Presence or absence of a connecting corridor to the nearest area where the species in question was observed holding a territory.
DEAC	Diameter (m) of the equivalent area circle in an "edge" habitat type;
DEACLA	DEAC $\times$ leaf area in an "edge" habitat type;
DIST	Distance (m) to an area as described above for CONCOR.
EIDCLA	Highest edge index (from ESCAN) $\times$ DEAC $\times$ leaf area in an "edge" habitat type;
EIDEAC	Highest edge index (from ESCAN) $\times$ DEAC in an "edge" habitat type;
H20EDGE	Edge length (# cell sides) of types 2:13, 4:13, and 5:13 (see Table 2).
LA1M	Leaf area ( $\text{cm}^2/\text{m}^2$ ) from 0-1 m;
LAHIGH	Leaf area ( $\text{cm}^2/\text{m}^2$ ) $>7$ m;
LALOW	Leaf area ( $\text{cm}^2/\text{m}^2$ ) from 0-3 m;
LAMID	Leaf area ( $\text{cm}^2/\text{m}^2$ ) from 3-7 m;
LALOWMID	LALOW + LAMID
MDCLA	Maximum diameter (m) circle in a "solid" habitat type $\times$ leaf area in the habitat;
NUMHAB	Number of habitat components actually present on each plot within a given habitat type.
PLOTEI	Plot edge index = # edges of habitat type (T) on the plot $\div$ (plot area) $^{1/2}$ .
PROPNH	Proportion of foliage profile samples containing northern hardwood species.
SOLRAD	Incidence of potential direct beam solar irradiance; Kaufmann and Weathered (1982).
STREAM	No stream, stream $\leq 1$ m wide, stream $>1$ m wide (values = 0,1,2 respectively)
VMRI(1-16)	Variance/mean ratio $\times$ inverse of Run's Index;

Only the black-throated blue warbler and the ruby-throated hummingbird were ever observed in any of the old growth forests, and the warbler never occurred on any of the early successional, aspen or jack pine clearcuts (CCA). The black-and-white warbler, black-throated blue warbler and purple finch also were never found in any of the old-fields (OF). All species but the purple finch occurred for at least 3 yr on the 4 to 8 yr-old northern hardwood/hemlock clearcut with residual overstory, CCB-1.

Table 7 lists simple correlations between the habitat variables and bird occurrence. Five of the six species had their highest positive correlations with one of the four variables that included DEAC, the measure of the size of the area on a plot with the highest density of edges of the habitat type thought to be most associated with the species in question (see Tables 2, 3, and 5). Variables including DEAC were strongly correlated ( $r > 0.30$ ,  $p < .01$ ) with all species (23 of 25 correlations) with the exception of the correlation of DEAC with the occurrence of the black-and-white warbler and ruby-throated hummingbird. The only highest correlation not associated with a DEAC-related variable was that of the black-and-white warbler with the plot edge density index (PLOTEI). The black-throated blue warbler also had a co-highest correlation with this variable along with DEACLA. Another measure of habitat size, MDCLA4, was strongly correlated with the flycatcher, waxwing, finch, and hummingbird.

Two other variables that were strongly positively correlated with at least three species were the amount of leaf area from 0 to 3 m (LALOW) for the waxwing, finch, and hummingbird, and the habitat component richness observed in a designated habitat type (Tables 2 and 3) on the plot (NUMHAB) for the flycatcher, waxwing, and the two warblers.

The highest negative correlations were those of the waxwing with high canopy, northern hardwoods vegetation, and the presence of a stream, and of the flycatcher and black-throated blue warbler with the distance of the plot from similar habitat.

#### Multivariate Analyses:

Measures including the size of the edge habitat (DEAC) of a species were the most frequently selected variables for five of the six species in iterations of the stepwise discriminant function procedure (Table 8). A size-related variable (DEAC) that also included the amount of leaf area (LA) in the bird species' suggested habitat was commonly included ( $>40\%$  of all trials) in models for all species. Another measure of the degree of habitat clumping/size, VMRI, was a common variable for three species. The relative amount of habitat edge for the species on the entire plot, PLOTEI, was the most common variable for the black-and-white warbler and was a common variable for the ruby-throated hummingbird. The only common variable not related to habitat size was the distance from similar habitat (negative correlation), which was included in all but one trial for the alder flycatcher.

The relationships of the flycatcher to its two most common correlates in SDFA (Table 8) are depicted on logarithmic scales in Figure 2. Although the distribution of this species was strongly positively correlated with the combination variable EIDCLA (see Table 4 for description), plots located farthest from occupied similar habitat did not support this species, regardless of EIDCLA values.

Collectively, the variables included in the models explained a large

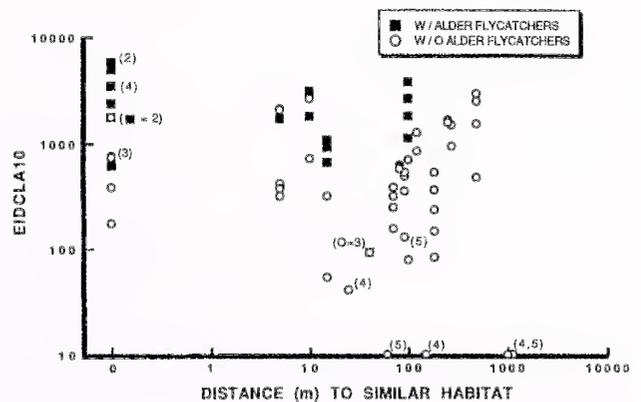


Figure 2: Alder flycatcher occurrence as a function of habitat quality (EIDCLA10) and distance of the plot from similar habitat. EIDCLA10 = highest edge index (EI)  $\times$  size of the edge habitat (DEAC)  $\times$  leaf area (LA) in sapling/opening habitat (see Tables 3 and 4). Numbers in parentheses are the number of observations of values at the adjacent point.

portion of the distribution patterns of the six species studied (Table 9). The average classification accuracy of all jackknifed discriminant function models for the entire data set, iterative trials, and for iterative tests of the models, was 89.3%, 89.3%, and 82.7%, respectively.

The second multivariate analysis, MAXR Regression of species density with the habitat variables (Table 10), produced results very similar to the discriminant function analysis. Variables including the

**Table 5.** Variables (Table 4) used in the multivariate analysis of species occurrence and density for six species of breeding birds at the Connecticut Hill WMA. Numbers in the table refer to habitats described in Table 3.

Variable	Species					
	Alder Flycatcher	Black-&-White Warbler	Black-throated Blue Warbler	Cedar Waxwing	Purple Finch	Ruby-throated Hummingbird
CONCOR	10	12	13	15	16	4
DEAC	10	12	13	15	16	
DEACLA	10*	12*	13	15	16*	9,10
DIST	10	12	13	15	16	4
EIDCLA	10	12	13*	15*	16	
EIDEAC	10*	12*	13*	15*	16*	
H20EDGE				+		
LA1M	+	+	+	+	+	+
LAHIGH	+	+	+	+	+	+
LALOW	+	+	+	+	+	+
LAMID	+	+	+	+	+	+
LALOWMID			+			
MDCLA						4
NUMHAB	10	12	13	15	16	9
PLOTEI	10*	12	13	15	16	9,10
PROPNH			+#			
SOLRAD	+	+	+	+	+	+
STREAM						+
VMRI	10	12,13*	13	15	16	4

\* Variable used in maximum r-square improvement regression only.

# Variable used in stepwise discriminant function only

size of the edge habitat (DEAC) for the species accounted for the most variability in cedar waxwing, purple finch, and ruby-throated hummingbird abundance. An example of the relationship of habitat size to bird abundance is shown for the cedar waxwing in Figure 3. One or more of the DEAC-inclusive variables were included in the regressions for all six species and all four variables containing DEAC were included in regressions for the finch and the two warblers. The relative amount of habitat edge for the species on the plot (PLOTEI) was selected in five regressions and explained the most variation in black-and-white warbler abundance. Distance to similar habitat (negative correlation) was the primary variable in the regression for the alder flycatcher.

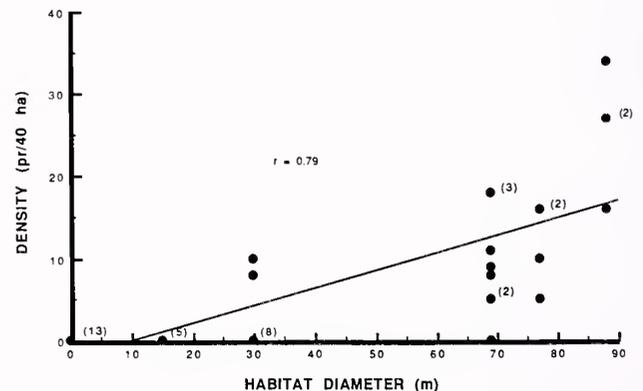
All regressions were highly significant ( $p < .0001$ ) and  $R^2$  values averaged .62. The regression model was very good for the black-throated blue warbler ( $R^2 = .92$ ) and the cedar waxwing ( $R^2 = .81$ ), good for the purple finch ( $R^2 = .63$ ) and black-and-white warbler ( $R^2 = .60$ ), and relatively poor for the ruby-throated hummingbird ( $R^2 = .41$ ) and the alder flycatcher ( $R^2 = .34$ ).

## DISCUSSION

### Species-Habitat Models:

It has long been suggested that vegetation structure is the most important proximal cause of habitat selection (Hilden, 1964). Associations of the bird species in this study with vegetation structure have been described by a number of authors (e.g. Stein, 1958; Kendeigh, 1945; James, 1971). It was based on such suggested habitat

associations, the recommendations of colleagues, and my own field observations that the habitat types described in Table 3 were proposed as correlates of the distributions of the six species discussed here. Several new techniques based on aerial photographic analysis then were used to quantify the spatial arrangement of these habitats more thoroughly than had been done previously. The results (Tables 7-10) suggest that the habitat types and the variables derived to describe



**Figure 3:** The density (pr/40 ha) of Cedar Waxwings versus habitat size as measured by the variable DEAC (see Table 4) on plots at the Connecticut Hill WMA between 1977 and 1981. The numbers in parentheses are the number of observations of values at the adjacent point.

**Table 6.** Presence (+) or absence (-) of six species of breeding birds on 23 plots at the Connecticut Hill WMA between 1977 and 1981. A blank space indicates the plot was not surveyed that year. Plot mnemonics are described in Table 1.

Plot	Alder Flycatcher		Black-&-White Warbler		Black-throated Blue Warbler		Cedar Waxwing		Purple Finch		Ruby-throated Hummingbird														
	77	78	79	80	81	77	78	79	80	81	77	78	79	80	81	77	78	79	80	81	77	78	79	80	81
	Year																								
CCA-1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CCA-2	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	+
CCA-3	-	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
CCA-4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	-
CCA-5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	+	+	-	-	-	+	+	-
CCA-6	-	-	+	-	-	-	-	-	-	+	-	-	+	+	+	-	-	+	+	+	-	-	+	-	-
CCA-7	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	+	+	+	-	-	-	-	-	+
CCA-8	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	-	-	-
CCA-9	-	+	+	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	+	+	-
CCA-10	-	+	+	+	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	+	+	+	-
CCB-1	+	+	+	+	-	+	+	+	+	+	+	+	+	+	-	-	-	-	-	-	+	+	+	-	-
CCB-2	-	-	-	-	-	-	-	-	-	-	+	+	+	+	+	-	-	-	-	-	+	-	-	-	-
CCC-1	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-	-
CCC-2	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
OFF-1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
OFF-2	+	-	-	-	-	-	-	-	-	-	+	+	+	+	-	-	-	-	-	-	-	-	+	+	-
OFl	-	-	-	-	-	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-
OFA-1	+	+	+	+	-	-	-	-	-	-	+	+	+	+	-	-	-	-	-	-	-	-	+	+	-
OFA-2	-	-	+	+	-	-	-	-	-	-	+	+	+	+	-	-	-	-	-	-	-	-	-	-	-
F-SGA	-	-	-	-	-	+	+	-	-	-	+	+	+	+	-	-	-	-	-	-	-	-	-	-	-
F-NHHB	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
F-NHHC	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
F-NHHD	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

<sup>a</sup> The cedar waxwing was classified as a member of a multispecies guild on young clearcuts (CCA,CCB) based on observed foraging behavior and plant community structure; see text.

them provide good first-cut estimates of some of the habitat features important to these species.

In particular, the size of the habitat type identified *a priori* as peculiar to each species was consistently important in determining the occurrence of that species (Table 8). Because the species were all considered to be associated with various types of edges rather than contiguous habitat types, such as the alder flycatcher with sapling/opening edge (Table 3), the size of the habitat was considered to be that circle diameter that contained the highest density of edges of the type "appropriate" to the species (see DEAC in Table 4). This relationship suggested that, even in areas where patches of appropriate habitat for the species occurred, the habitat must have a minimum (threshold) size and clumped arrangement (optimally a circle) to be occupied by a breeding pair of the birds (Figures 2 and 3). Thus, it appears that at the scale of territories for individual breeding birds, habitat shape, as well as size, is an important consideration. If this argument is extrapolated to the design of nature reserves, it suggests that reserve shape and the shape of habitat patches within reserves may be critical, at least for small reserves, due to the negative influence of "edge" effect (Ambuel and Temple, 1983) on reserve interior species. The importance of shape in larger reserves has yet to be demonstrated (for an excellent discussion, see Simberloff, 1986). In addition, knowledge of the threshold size for species occupation of a site can contribute toward an

estimation of the reserve size necessary to maintain a minimum viable population of that species.

The only case where habitat size was not the most highly correlated variable with a species occurrence was with the alder flycatcher. On plots supporting relatively small sapling/opening habitats (*e.g.* CCA-2,3,6), the pattern of occurrence of this species was erratic (Table 6). When only a single territorial male was present, alder flycatchers tended not to occupy sites in consecutive years (7 of 11 cases). On more isolated plots with larger sapling/opening habitats (*e.g.* CCA-5,7), it often appeared that apparently suitable habitat was unoccupied (Figure 2). Two documented explanations for these patterns may be applicable. First Fretwell (1972), and later Wolff (1980), and O'Conner and Fuller (1983) suggested that fluctuations in regional population levels can lead to unoccupied, but apparently suitable, habitat when regional populations are low. In this case, decreasing population levels lead to local extinctions of small populations in marginal habitats, and only the best habitats remain occupied. A second, and not mutually exclusive, explanation suggests that some species tend to form territorial clusters (*e.g.* Sherry and Holmes, 1986) around previous territory holders (*e.g.* Hilden, 1965; Alatalo *et al.*, 1982; Stamps, 1988) that often may be older, previously successful breeders (Greenwood, 1980; Dawson, 1984; Liberg and von Schantz, 1985). The strong negative association of the alder flycatcher with distance of a plot from occupied habitat,

**Table 7.** Simple linear correlations (r) between bird species presence and habitat variables for 23 clearcuts, oldfields, and forests sampled between 1977 and 1981 at the Connecticut Hill WMA in central New York. Signs and levels of significance were identical for correlations of the habitat variables with density (pr/40ha) of the six species. Correlation values for density averaged 15-20% less than for species presence.

Variable <sup>b</sup>	Species <sup>a</sup>					
	ALFL	BAWW	BTBW	CEWA	PUFI	RTHU
CONCOR <sup>c</sup>	0.20 *e	- <sup>d</sup>	-	0.52 ***	-	+
DEAC	0.52 ***	0.27 *	0.79 ***	0.86 ***	0.40 ***	0.28 **
DEACLA	0.54 ***	0.37 ***	0.91 ***	0.52 ***	0.55 ***	0.32 **
DIST	-0.30 **	+ ***	-0.45	-	-	-
EIDCLA	0.61 ***	0.43 ***	0.90 ***	0.52 ***	0.70 ***	0.41 ***
EIDEAC	0.57 ***	0.34 **	0.85 ***	0.80 ***	0.47 ***	0.34 ***
H20EDGE	0.28 **	-	-	0.38 **	-	+
LA1M	0.33 ***	+	-	0.73 ***	0.27 **	0.29 **
LAHIGH	-0.26 **	-	+	-0.69 ***	-	-0.23 *
LALOW	0.26 **	0.27 **	0.29 **	0.66 ***	0.36 ***	0.35 ***
LAMID	-	+	0.35 ***	-	-	-0.24 *
MDCLA(4)	0.45 ***	+	-	0.71 ***	0.44 ***	0.32 ***
NUMHAB	0.51 ***	0.50 ***	0.85 ***	0.83 ***	0.24 *	0.21 *
PLOTEI	0.49 ***	0.64 ***	0.91 ***	0.56 **	0.50 ***	+
PROPNH	-	0.28	0.59	-0.60 **	- ***	- ***
SOLRAD	0.27 **	+ *	-	+	+	0.23
STREAM	-	-	-	-0.38 **	-	-
VMRI	+	+ +	0.56 ***	+	0.44 ***	0.23 *

<sup>a</sup> ALFL = Alder Flycatcher; BAWW = Black-&-White Warbler; BTBW = Black-throated Blue Warbler; CEWA = Cedar Waxwing; PUFI = Purple Finch; RTHU = Ruby-throated Hummingbird.

<sup>b</sup> The variable LALOWMID was generated as the sum of LALOW + LAMID (Table 4) in later analyses and was not included in the simple correlations.

<sup>c</sup> For variables such as CONCOR, DEAC, and DIST, which were measured for different habitat types, see Table 5 for a listing of the habitat type(s) that applies to each species.

<sup>d</sup> + = positive correlation but  $p > 0.05$ ; - = negative correlation but  $p > 0.05$ .

<sup>e</sup> \* $p < 0.05$ ; \*\* $p < 0.01$ ; \*\*\* $p < 0.001$

even when that plot appeared to contain suitable habitat, is consistent with this social facilitation concept.

The most common variable in discriminant function analyses for three of the six species was a cross-product of habitat size with habitat leaf area and/or habitat edge density. Variables with size and leaf area were commonly included in models for all species. Leaf area was considered a measure of primary productivity (Whittaker *et al.*, 1974) and an indicator of secondary productivity (*e.g.* arthropod abundance; Blenden *et al.*, 1986). The significantly low (chi-square test,  $p < .01$ ) number of species correlations with leaf area independent of habitat size (*e.g.* LALOW) emphasizes the importance of the interaction of habitat size and productivity. The lack of correlation of leaf area with species occurrence in SDFA models may be because leaf area in the vertical layer selected was truly unimportant to the species. However, I feel the more likely explanation is related to the fact that the habitat types assigned to each species already reflected the range of leaf area appropriate for the species. This was indicated by the relatively high correlations ( $r > .37$ ,  $p < 0.001$ ) of leaf area (LA) in the habitat with DEAC for five of the six species. Thus, it appears that after the size and size-related variables were included in the models, the independent leaf area variables were unable to explain sufficient additional variance to be added to the models.

**Table 8.** Variables (Table 4) and their proportion of occurrence in repeated trials of the Stepwise Discriminant Function model building/testing procedure for explaining the presence/absence of 6 species of breeding birds at the Connecticut Hill WMA. The number of trials of the procedure is in parentheses.

	Alder Flycatcher (N=11)	Black-&-White Warbler (N=10)	Black-throated Blue Warbler (N=11)
*EIDCLA10	1.000	*PLOTEI12	.700
*(-)DIST10	.909	EIDCLA12	.500
NUMHAB10	.364	VMRI12	.400
LA1M	.182	*LALOW	.300
VMRI10	.182	*LA1M	.300
LALOW	.091	*DEAC12	.200
(-)LAMID	.091	LAMID	.200
DEAC10	.091	DIST12	.200
		*NUMHAB 12	.100
		(-)LAHIGH	.100
	Cedar Waxwing (N=15)	Purple Finch (N=15)	Ruby-throated Hummingbird (N=15)
* DEAC15	.667	*EIDCLA16	1.000
DEACLA15	.400	*VMRI16	.800
NUMHAB15	.267	*DEAC16	.733
PLOTEI15	.200	(-)LAMID	.333
LA1M	.133	UMHAB16	.133
H20EDGE	.067	*LOTEI16	.067
		(-) DIST16	.067
		(-) CONCOR16	.067
		LA1M	.067
* DEACLA10	.600	*DEACLA10	.600
VMR14	.400	VMR14	.400
PLOTEI9	.400	PLOTEI9	.400
MDC4	.133	MDC4	.133
MDCLA4	.067	MDCLA4	.067
PLOTEI10	.067	PLOTEI10	.067
LALOW	.067	LALOW	.067

\* Included as a variable in Stepwise Discriminant Function Analysis of all plots combined.

**Table 9.** The percentage of correct predictions of the presence or absence of six species of breeding birds on study plots at the Connecticut Hill WMA using Stepwise Discriminant Function Analysis. Classification accuracy when using the entire data set for model building is included for purposes of comparison. All values for the entire data set and for models are jackknifed estimates. The expected percentage of correct predictions for each species is 50. For calculation of expected values and a detailed discussion of the data splitting and model building/testing procedure, see Methods.

Species	Entire Data Set	$\bar{x}$ for Iterative Data Splitting				U <sup>a</sup>
		Model	Trials	Test	Trials	
Alder Flycatcher	88.7	88.8	***	82.1	** (N=11)	.483
Black-&-White Warbler	91.8	89.1	***	83.2	* (N=10)	.503
Black-throated Blue Warbler	95.7	98.4	***	95.9	*** (N=11)	.103
Cedar Waxwing	93.3	88.7	***	80.9	6NS <sup>b</sup> 9p<.001	.249
Purple Finch	95.9	96.7	***	90.4	*** (N=15)	.287
Ruby-throated Hummingbird	70.1	74.3	**	63.7	7NS 8p<.05	.342
	$\bar{x} = 89.3$	89.3		82.7		

<sup>a</sup>Wilks' Lambda or U statistic.

<sup>b</sup>Numbers on the left of the column equal the number of trials; NS = not significant.

\* All trials p<.05

\*\* All trials p<.01

\*\*\* All trials p<.001

**Table 10.** Variables included in models for the density (pr/40ha) of six species of breeding birds at the Connecticut Hill WMA using MAXR Regression. Variables are listed in order of decreasing variance explained (F value) within the model. The significance of each variable is listed in the right hand column under each species. All variables were significant (p<.05). All models were highly significant (p<.0001). R<sup>2</sup> values are shown in parentheses and averaged .616.

Alder Flycatcher (.337)		Black-&-White Warbler (.596)		Black-throated Blue Warbler (.920)	
	P		P		P
(-) DIST10	.0024	PLOTEI12	.0001	VMRI13	.0001
DEACLA10	.0054	LALOW	.0001	DEAC13	.0001
PLOTEI10	.0255	DEACLA12	.0001	EIDEAC13	.0001
		EIDEAC12	.0001	DEACLA13	.0001
		LA1M	.0001	EIDCLA13	.0001
		DEAC12	.0002	PLOTEI13	.0001
		EIDCLA12	.0005		
Cedar Waxwing (.808)		Purple Finch (.632)		Ruby-throated Hummingbird (.405)	
	P		P		P
EIDEAC15	.0074	EIDCLA16	.0001	DEACLA9	.0001
PLOTEI15	.0212	EIDEAC16	.0001	LALOW	.0001
		DEACLA16	.0001	VMRI4	.0002
		VMRI16	.0001	PLOTEI9	.0043
		DEAC16	.0259	(-) DIST4	.0089
				DEACLA10	.0212

## Model Development and Improvement:

Despite the apparent explanatory power of the S DFA models developed here, they are very likely not to be the best models for use with these species, even within the context of this data set. For example, the ruby-throated hummingbird occurred over a wide range of plant community types (Table 6), and, because of its wide range of habitat associations, it was not assigned a habitat type distinct from any other species or guild. Instead, variables describing several of the identified habitat types, particularly shrub/openings and sapling/openings (Table 3), were used in the analyses for the hummingbird. Thus, the only way to determine whether more descriptive variables exist for this species, in the context of this study, would be to test systematically the ability of various combinations of all of the habitat variables in the original data set (Keller, 1986) to explain the hummingbird's occurrence.

In the original study, 16 habitat types (with several measures of each) were identified for 59 species in 19 guilds resulting in over 200 variables. Furthermore, most of these variables are simple linear variables. Some of the species' relationships with these variables may actually be curvilinear, so, variables could be transformed (*e.g.* squared, cubed) to describe more adequately the curvilinear relationships with species occurrence. These steps might have been useful in the case of the hummingbird; however, this suggestion assumes that sampling error was minimal and that variables in the model were the causes of the poor correlations.

For a species as inconspicuous as the hummingbird, sampling error may have been a factor contributing to the relatively poor correlation of this species' occurrence with the habitat variables. Clearly, a number of alternative explanations are possible for this result. For the black-throated blue warbler and purple finch, simple, linear relationships provided very accurate classification (Table 8), and further small improvements in the models may have been costly and time-consuming to obtain. In addition, there is no guarantee that a statistically better model will be more biologically interpretable.

A potentially more productive avenue for model improvement when the initial model has good classification accuracy at one locale is to attempt to validate the model with data from samples collected independently of those used in model development (Capen *et al.*, 1986; Mosher *et al.*, 1986). This approach is taken routinely with the development of Habitat Suitability Index (HSI) models by the U.S. Fish and Wildlife Service (for examples see Verner *et al.*, 1986). In some cases, models that are quite good in one region are not equally applicable to the same species in another part of its geographic range and combinations of models from the two areas may be poor predictors of the species occurrence in either area (Mosher *et al.*, 1986). This suggests that some models may need to be developed at a regional rather than continental level. This seems reasonable because the edaphic and biological (*e.g.* potential competitors or prey) conditions that help determine a species distribution may change greatly across its geographic range. Of course, consideration of such things as variations in climate and competitors can be included in a model, and sometimes these can explain significant portions of the variation in species distributions (Haefner, 1981). However, such inclusions may make model development unnecessarily complex and expensive. Capen *et al.* (1986) summarized a number of the pitfalls associated with model development by stressing the need for adequate sample sizes (*i.e.*, much larger than the number of variables), careful selection of variables, and avoidance of multicollinearity of variables. Noon (1986) agreed and added that poor experimental design is often a problem, in that data are often collected in such a way that they do not meet assumptions of the models used to analyze them. He also agreed that models developed from local

studies often perform poorly when applied to other areas. This opinion reinforces the need for model validation.

When little is known about the habitat relationships of the organism, several systematic attempts may be required to develop the most appropriate and widely applicable model(s).

As noted above, this can be an expensive process. Public education may be appropriate and necessary to promote interest and gain monetary support for the management of selected species. Once a program is initiated, the first step is identification, from the literature or expert opinion, of variables correlated with the species distribution. We saw several examples of this preliminary step demonstrated by participants in the 1988 Natural Areas Conference who noted the association of certain rare plants with particular soil types. Subsequent steps include the quantification of additional habitat variables associated with the species, statistical development of the model, and model refinement through validation. The model then can be used in a predictive fashion as a management tool. For example, if species-habitat models are developed that incorporate variables identifiable from remotely sensed imagery, such as in this study, then that imagery can be used to locate additional areas with similar habitat characteristics (*e.g.* Davis and DeLain, 1986). These areas then can be surveyed to determine if previously unknown populations of the species in question exist there. In locations where the species historically occurred, but currently does not exist, the model can be used to evaluate potential causes of extirpation. For example, the size of the habitat may have been reduced by successional changes in the vegetation, or the site may have become increasingly isolated due to changes in adjacent land use. In either case, the model may provide information to help decide whether habitat manipulation and/or species reintroduction are viable management alternatives. Similarly, models can form a basis for decisions regarding the management of *in situ* populations facing potential habitat loss or alteration.

It seems clear that, as human populations and the accompanying demands for land continue to grow, more quantitatively based management strategies will be needed to insure the persistence of dwindling populations of wild species. Habitat models, such as those described here and in many of the references cited, are a first step toward greater accountability in the decision-making process regarding species of special concern.

## ACKNOWLEDGMENTS

The study was supported in part by Pittman-Robertson funds administered through the New York Cooperative Wildlife Research Unit. Computer time was provided by the Cornell College of Agriculture and Life Science, and by Bill Youngs of the Cornell Department of Natural Resources. I thank my former chairman Milo Richmond for advice and technical support, Eric Bollinger for useful discussions of statistical matters, Bill Wischusen for help with the graphics, and Charles Smith and my wife, Lorraine, for reviewing earlier drafts. Elaine Depew and Nancy Bowers typed the manuscript.

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## APPENDIX I

Plot strata used when randomly splitting the data set for use in the discriminant function model building and testing procedure.

Plots <sup>a</sup>	Strata
CCA-1, CCA-2	smallest plots
CCA-3, CCA-6, CCA-10	aspen clearcuts
CCA-4, CCA-8, CCA-9	jackpine plantations prior to cutting
CCA-5, CCA-7	most isolated clearcuts
CCB-1, CCB-2	northern-hardwoods clearcuts
CCC-1, CCC-2	mid-succession clearcuts
OFF-1, OFF-2	open fields
OFA-1, OFA-2	advanced shrub-sapling stage old fields
FNHHB, FNHHC, FNHHD	mature northern-hardwoods-hemlock forests
OFI <sup>b</sup>	"savannah" old field
FSG-A <sup>b</sup>	second growth deciduous forest

<sup>a</sup> See text and Table 1 for explanation of plot mnemonics.

<sup>b</sup> OFI and FSG-A were the only representatives of their respective strata and were always used in the model.

# Using Discriminant Analysis to Assess Critical Habitat for the Pawnee Montane Skipper

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**Abstract:** The pawnee montane skipper (*Hesperia leonardus montana*), a federally listed threatened species, is restricted to the South Platte River drainage in the Front Range of central Colorado. Construction of the proposed Two Forks Reservoir will result in the elimination of an estimated 22% of suitable skipper habitat and 18% to 33% of the total population. Habitat suitability was assumed to be a function of the distribution of *Liatrix punctata*, the species' nectar source. To test this assumption, habitat data (tree cover, tree density, *Liatrix* density, grass cover) collected in the same areas as skipper observations were studied with discriminant analysis. The results show that the distribution of the species is a function of both sapling tree density and abundance of *Liatrix*. These two variables are not correlated. The discriminant function correctly predicts the presence or absence of the species in 78% of the observations. Only 15% of the sites below the reservoir line were predicted to be unsuitable, compared to 48% above the reservoir line. These data demonstrate that habitat suitability differs above and below the reservoir line and suggest that mitigation sites should be selected based on the basis of forest structure as well as nectar source distribution.

Pages 47 - 50. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The Pawnee Montane Skipper (*Hesperia leonardus montana*) is restricted to less than fifty square miles in the Front Range of Colorado, southwest of Denver. The species was listed as threatened by the U.S. Fish and Wildlife Service in 1987 because of its restricted distribution, anticipated impacts from development of Two Forks Reservoir, and habitat damage from off-road vehicle use. No critical habitat was designated for the skipper at the time of listing, because collecting and vandalism could potentially become threats upon public disclosure of location information (Federal Register, 1987) and because previous studies on factors limiting species distribution were inconclusive.

A distribution survey, population census, and habitat suitability study were conducted in 1985 and 1986 by consultants for the Denver Water Department, the project proponent for Two Forks Reservoir. A description of suitable habitat for the species was developed from a qualitative comparison of occupied and unoccupied sites. Quantitative comparisons of habitat above and below the reservoir line were completed for *Liatrix punctata*, the adult nectar source, which was the only vegetation component assumed to limit the species' distribution. *Bouteloua gracilis*, the host plant for the larvae, does not appear limiting because it is widespread throughout the study area. *Liatrix punctata* was found to be significantly more abundant within the Two Forks inundation zone.

The results of these studies formed the basis for assessment of critical habitat for the pawnee montane skipper by the U.S. Fish and Wildlife Service during the listing process. That agency issued a biological opinion that the Two Forks Reservoir project was not likely to jeopardize the pawnee montane skipper if conservation measures (mitigation), including acquisition and management of suitable habitat, were fully implemented (Buterbaugh, 1987). Although a habitat description developed by Denver Water Department consultants for selecting suitable habitat included an estimate of overstory and understory tree cover, shrub cover, grass cover, *Liatrix punctata* density, and *Bouteloua gracilis* cover for occupied habitat, no analysis was conducted to assess whether vegetation characteristics (other than *Liatrix* density) limit the species' distribution.

Habitat requirements for butterflies are more complex than the necessity of the presence of host and nectar plants (Vane-Wright and

Ackery, 1984). For example, incident radiation, quality of food plants, and perching sites are among factors documented to influence habitat suitability in butterfly species. Selecting suitable sites for mitigation for the pawnee montane skipper requires evaluation of as many vegetation characteristics as possible to determine which are influential in limiting species distribution in pawnee montane skippers.

Discriminant analysis can be used to classify observations into one of two or more groups, based on a set of measurements. Discriminant analysis can also identify which variables contribute to making the classification. (Affifi and Clark, 1984). The data set from 1986 was reanalyzed to derive a discriminant function to describe vegetation components that effectively predict the presence or absence of the skipper and whether these are important for assessing habitat suitability above and below the inundation line of the proposed Two Forks Reservoir.

## METHODS

The study area is located in the South Platte River Valley of Jefferson and Douglas Counties, Colorado. Elevations in the study area range from 6200 to 7500 ft. Open ponderosa pine communities at lower elevations grade into Douglas fir forests at higher elevations. Pikes Peak granite is the substrate in the study area.

The 1985-1986 butterfly study was conducted as part of the preparation of the Systemwide Environmental Impact Statement by the U.S. Army Corps of Engineers. The study area, comprising most of the species range, was stratified into forty-six areas to ensure sampling throughout the species' range. Half of the areas occur above the reservoir line, and half are within the proposed impact area. A fixed belt-transect, 400m by 50m was randomly located in each stratum. Skipper occurrence and habitat characteristics were recorded for each transect.

Sampling for the butterfly was conducted during peak flight activity, in late August, between 9:00 a.m. and 4:00 p.m. MDT. Data were collected under clear to light overcast conditions. All *Hesperia leonardus montana* individuals observed were recorded by location along the transect. The following vegetative characteristics were determined for each transect as well: density of *Liatrix punctata* stems, the per cent cover of *Bouteloua gracilis*, per cent cover of other grasses, tree density of ponderosa pine and Douglas fir, by size classes for both species, and aerial shrub cover. Tree size classes were established as individu-

als less than 12.7 cm diameter breast height (dbh.), 12.7 to 22.9 cm dbh, and greater than 22.9 cm dbh.

The BioMedical Data Processing statistical program (1985) was used to develop the discriminant function based on the vegetation variables. Log transformations were used to stabilize variances for cover and density of Douglas fir, total sapling density, and *Liatris punctata* stem density. One transect with ten times the stem density of *Liatris* was considered an outlier and removed from the analysis.

## RESULTS AND DISCUSSION

### The Discriminant Model:

Discriminant analysis separated occupied and unoccupied areas most effectively using sapling tree density (individuals less than 12.7 cm d.b.h.) and *Liatris punctata* density:  $DF = 3.01177(\log TS) - 1.8967(\log LP) + 2.26286$ . When tree sapling density increased and *Liatris* density decreased, habitat was considered less suitable. Both of these variables contributed significantly ( $p < 0.05$ ) to the model. No additional variables improved the discriminant function. The discriminant function correctly predicted 68.8% of the areas without butterflies and 82.8% of the areas with butterflies. Misclassifications of presence or absence of the species may have resulted from inadequate selection of variables in the experimental design or from population dynamics. The fact that the discriminant function more often categorizes unoccupied habitat as suitable and occupied habitat as unsuitable may indicate that population dynamics is a factor limiting the effectiveness of the model. For example, movement patterns can change dramatically with environmental conditions, especially those that affect the abundance of available food (Ehrlich, 1984). Empty, suitable habitat may also be the result of temporary low numbers of butterflies.

The discriminant variables, *Liatris punctata* stem density and tree sapling density, are not correlated; however, significant correlations do exist between these two variables (used in the model) and variables not significantly contributing to the model (Table 1). Tree sapling density is positively correlated with total tree cover, Douglas fir cover, Douglas fir density, ponderosa pine density, and the density of mid- and large-size trees. Tree sapling density is negatively correlated with shrub cover and blue grama cover. High tree sapling density appears to be an indicator of openness of habitat, with open habitat suitable for the species and closed habitat less suitable. Figure 1 shows the relationship between tree sapling density and total tree cover within the study area. Peet (1981) describes a similar montane woodland type from the northern Front Range that is dominated by ponderosa pine, with some Douglas fir in the overstory. A dense cohort of small Douglas fir trees apparently became established after cessation of periodic fires at the time of settlement. None of the stands in the northern Front Range study or in this study are old enough to have reached a steady state structure. Therefore, the relationship between sapling tree density and overstory vegetation cannot be extrapolated. Tree sapling density may not be expected to continue to increase if canopy closure occurs and reduces the light available for seedling establishment. So, the effectiveness of tree sapling density as a discriminating variable may change as succession proceeds.

The Pawnee Montane Skipper's requirement for open habitat may be a result of a physiological limitation, behavioral patterns or distribution of the host or food plant. Similarly, the endangered heath fritillary butterfly (*Mellicta athalia*) that occurs on the British Isles is apparently restricted to woodland openings and grasslands. Conifer plantings degraded the suitable habitat for that species by reducing the abundance of the host plant (Warren, 1987). The host plant of the pawnee

montane skipper, blue grama, is widely distributed at low abundance throughout the study area, negatively correlated with tree sapling and Douglas fir density. If Douglas fir continues to encroach in previously open ponderosa pine communities, blue grama may be expected to become limiting to the butterfly within its current range, although only five of 45 transects did not have blue grama. The sample used for the discriminant analysis is probably inadequate to detect or predict the limiting role of blue grama for skipper habitat suitability. This relationship deserves further investigation.

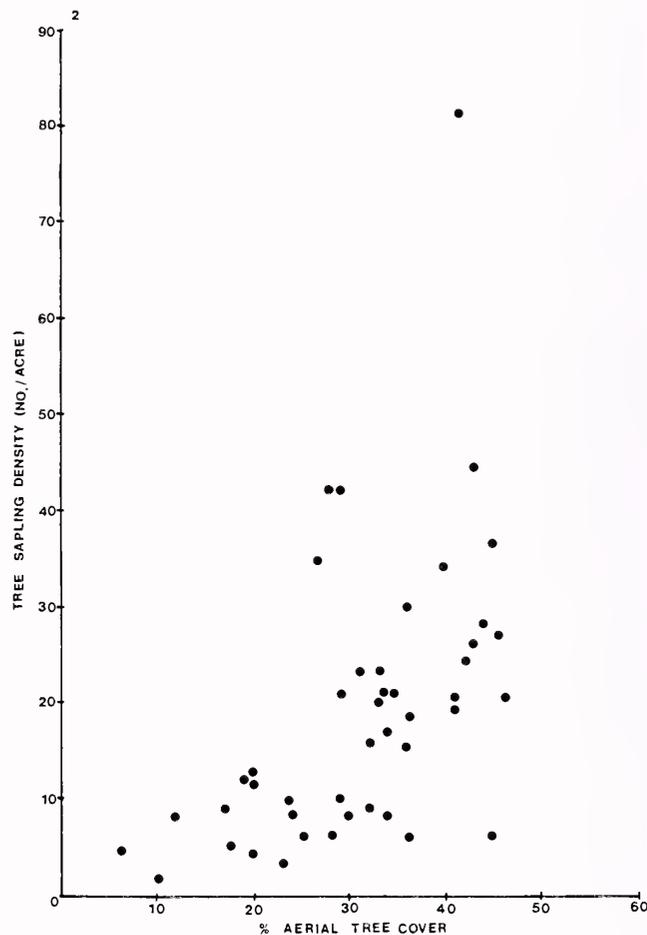


Figure 1. Tree sapling density plotted against aerial tree cover for the study area.

*Liatris punctata* stem density is positively correlated with aerial tree cover, ponderosa pine cover, density of ponderosa pine, overstory tree density, and blue grama frequency. *Liatris* stem density is negatively correlated with aerial shrub cover. *Liatris* is more typical of the Great Plains and lower foothills, reaching its elevation limit where cooler Douglas fir forests replace ponderosa pine woodlands. The correlations suggest that *Liatris* is associated with ponderosa pine in the study area. Like tree sapling density, *Liatris* density has a positive relationship with overstory vegetation characteristics that may change as forest structure changes. *Liatris punctata* is important to the pawnee montane skipper as a nectar source, but it may also be a general indicator of habitat differences between open and closed forest types in the discriminant model.

**Table 1.** The correlations among the twelve habitat variables considered in the discriminant analysis are present. Statistically significant coefficients are underlined ( $p=.95$ ).

	TRE	PPI	PPS	TS	DPI	DPS	T2	T3	SHR	GRA	BGR	LP
Total Tree Cover	TRE 1.00											
Ponderosa Pine Cover	PPI <u>.74</u>	1.00										
Douglas Fir Cover	PPS <u>.44</u>	-.15	1.00									
Tree Sapling Density	TS <u>.61</u>	.23	<u>.43</u>	1.00								
Ponderosa Pine Density	DPI <u>.45</u>	<u>.56</u>	-.11	<u>.53</u>	1.00							
Douglas Fir Density	DPS <u>.40</u>	-.14	<u>.76</u>	<u>.64</u>	-.11	1.00						
Mid-Size Tree Density	T2 <u>.54</u>	<u>.39</u>	.15	<u>.64</u>	<u>.61</u>	<u>.29</u>	1.00					
Large-Size Tree Density	T3 <u>.47</u>	<u>.51</u>	.13	<u>.44</u>	<u>.63</u>	.13	<u>.67</u>	1.00				
Shrub Cover	SHR <u>-.47</u>	<u>-.48</u>	-.10	<u>-.53</u>	<u>-.41</u>	-.19	-.29	<u>-.43</u>	1.00			
Grass Cover	GRA -.15	-.13	-.09	-.11	-.14	-.17	-.05	.24	.16	1.00		
Blue Grama Cover	BGR -.16	-.01	-.21	<u>-.28</u>	-.01	<u>-.29</u>	-.08	-.04	.10	<u>.53</u>	1.00	
Liatris Density	LB <u>.32</u>	<u>.42</u>	-.00	.16	<u>.40</u>	-.18	.09	<u>.37</u>	<u>-.37</u>	<u>.24</u>	.37	<u>1.00</u>

### Habitat Suitability:

The graphic model of the discriminant function, presented as Figure 2, shows the relationship between habitat suitability and proposed reservoir impacts. Eighteen of the thirty sites predicted to be suitable by the discriminant model occur below the proposed reservoir inundation line. Only three of fifteen "unsuitable" sites occur below the inundation line. Clearly, a higher proportion of suitable habitat exists below the reservoir line than above the reservoir line. Opportunities to mitigate the loss of habitat from reservoir development may, therefore, be limited.

Although *Liatris* and tree sapling density are uncorrelated, both may be related to elevation. High elevation sites within the study area tend to be cooler, favoring seedling establishment, particularly of Douglas fir. Lower elevation sites are warmer, more open, and more favorable for *Liatris punctata*. Elevation differences in microclimate have been shown to result in differences in time available for daily flight activity in *Colias* (Kingsolver, 1983) as well as differences in the structure and composition of the vegetation.

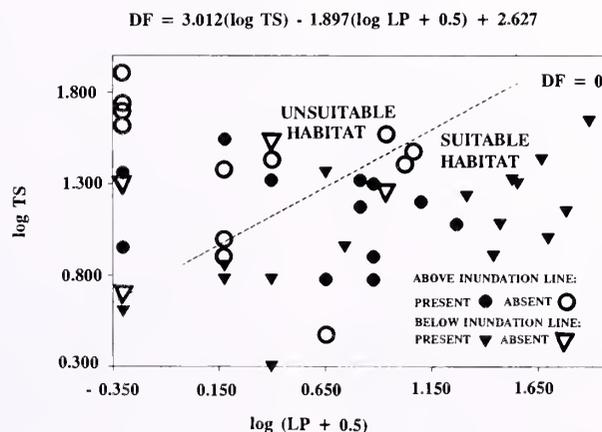
Elevation specificity in butterflies can be a result of competition and selection for reproductive isolation, as well as a habitat response (Gilbert, 1984). *Hesperia leonardus pawnee* occurs at lower elevations from the plains-foothill border into the Great Plains (Scott and Stanford, 1982). Lower elevation limits of *Hesperia leonardus montana* could potentially result from a response to the presence of *Hesperia leonardus pawnee*. However, *Hesperia comma ochracea* occurs with *Hesperia leonardus montana* and does not appear to segre-

gate (ERT, 1986a). So, intra-specific segregation does not appear to explain the upper limit of *Hesperia leonardus montana* or differences in butterfly distribution above and below the inundation line.

### Habitat Management:

The discriminant model correctly predicted 82.8% of the suitable habitat and should be useful for selecting mitigation sites. Maintaining suitable habitat may require active management, particularly on cooler or higher elevation sites. Open canopy, apparently a requirement for the pawnee montane skipper, is characteristic of fire-derived woodlands in the Rocky Mountains (Crane, 1982). Tree ring data and charcoal in the soil suggested that densely forested stands similar to those in this study are of post-fire origin on the Front Range (Peet, 1981). Fires may have occurred at intervals of five to 25 years in some ponderosa pine woodlands (Crane and Fischer, 1986).

Logging for mine timbers, fuelwood, and lumber first occurred along the Front Range in the 1860's (Alexander, 1986). Mature trees (greater than 100 years old) were found across the study area indicating that selective cuts rather than clearcuts occurred (ERT, 1986). Selective cutting probably did not replicate the effects of fire. Natural fires that occurred prior to settlement are assumed to have been ground fires that created open savanna habitat (Crane, 1982). Logging may have been less patchy than fire and modified the overstory more. The importance of the scale of woodland openings to the species is unknown. Prescriptions for active habitat management will require further study of *Liatris* patch size required by the pawnee montane skipper.



**Figure 2.** The graphic model of the discriminant function shows the relationship between predicted habitat suitability and proposed reservoir impacts.

*Hesperia comma* (silver-spotted skipper), considered 'vulnerable' in Britain, was found to be sensitive to the quality of its host and larval food plant, *Festuca ovina*, and the structure of its grassland habitat (Thomas *et al.*, 1986). Like the pawnee montane skipper, the silver spotted skipper depends on a widespread host plant that apparently does not limit the species' distribution. Small *Festuca* plants on xeric sites with high incident radiation were preferred for egg-laying in the silver-spotted skipper. Similarly, the comparative effects on habitat quality in the understory that may be caused by fire versus logging have not been explored for the pawnee montane skipper.

The effects of habitat fragmentation are also not accounted for in the discriminant model. Figure 3 shows the current distribution of suitable habitat and its expected distribution after the development of Two Forks Reservoir. Much of the potential habitat is continuous at low elevations, but it will be broken into discontinuous higher elevation fragments with the rise of the water, and suitability of the isolated habitat units selected for mitigation cannot be predicted from the model. Studies have not been conducted to determine the mobility of the pawnee montane skipper and the probability of its colonizing unoccupied, suitable habitat or highly fragmented habitat.



**Figure 3.** A: The current distribution of potentially suitable habitat. B: The expected distribution of potentially suitable habitat after the development of Two Forks Reservoir.

### SUMMARY

Reanalysis of the data used in the Systemwide Environmental Impact Statement with discriminant analysis characterized critical habitat for the pawnee montane skipper. The discriminant model presented in the analysis demonstrates that habitat suitability differs above and below the Two Forks Reservoir inundation line and may limit opportunities for mitigation. The model isolates two vegetation vari-

ables, tree sapling density and food plant density, that effectively predict the presence of the pawnee montane skipper and should be used to select mitigation sites. However, further study is needed on the insect's specific habitat quality requirements, the role of fire in maintaining open ponderosa pine communities, and spatial dynamics, before active management prescriptions can be developed.

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# An Approach for Quantifying Habitat Characteristics for Rare Wetland Birds

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**Abstract:** Vegetation characteristics were quantified in breeding areas of three wetland bird species that have special concern in Minnesota or nationally: American Bittern (*Botaurus lentiginosus*), Yellow Rail (*Coturnicops noveboracensis*), and Sharp-tailed Sparrow (*Ammodramus caudacutus*). We used discriminant function analysis to explore which habitat characteristics differed between species and to determine whether habitat measurements taken within individual territories could be assigned to the respective species. Two discriminant functions were significant ( $P < 0.01$ ) and accounted for 100% of the variation. On the basis of this analysis, an average of 67% (range 43% - 90%) of the territories sampled could be assigned to the correct species group. Densities of shrubs, cattails and grasses/sedges were the most important variables in discriminating between the habitats used by each species. Based on these habitat variables and plant species composition, each bird species was found to occupy a different wetland microhabitat. American Bittern and Yellow Rail territories were associated with higher densities of shrubs and cattails than Sharp-tailed Sparrow territories, while Yellow Rails tended to be associated with higher densities of grasses and sedges than American Bitterns. Annual variation in habitat characteristics measured within these areas was probably due to phenological differences involving different sampling times between years, variable weather patterns, and regional differences in habitats selected for analysis each year. This quantitative approach may be useful for describing habitats of rare species and for identifying potential habitats necessary for the survival of rare or threatened birds.

Pages 51 - 56. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Population declines of many bird species over the past 100 years can often be attributed to loss or changes in habitat characteristics or complexes required by those species (Evans and Probasco, 1977; Suring and Knighton, 1985). For example, destruction or manipulation of wetlands throughout North America has had a particularly profound effect on endemic wetland bird species (Fredrickson, 1985). Unfortunately, it is difficult to manage or maintain appropriate wetlands for many species because of a lack of understanding of their specific habitat requirements. Often, wildlife managers or experienced naturalists recognize habitat components or configurations that a species may need within its breeding habitat, but objective, quantifiable methods for determining habitat requirements and incorporating this information into a management framework are limited.

Several authors have described breeding habitats of the wetland species that we considered here: American Bittern (*Botaurus lentiginosus*), Yellow Rail (*Coturnicops noveboracensis*), and Sharp-tailed Sparrow (*Ammodramus caudacutus*). American Bittern habitats were described by Bent (1926) and Mousley (1939). Yellow Rail habitats have been described by Morris (1905), Peabody (1922), Walkinshaw (1939), Terrill (1943), Lane (1962), Stahlheim (1974), Stewart (1975), Savaloja (1981), and Bart *et al.* (1984). Sharp-tailed Sparrow habitats were described by Rolfe (1899), Bowman (1904), Breckenridge and Kilgore (1929), Woolfenden (1956), and Murray (1969). However, none of these studies has quantified the habitat characteristics in the breeding territories of these species; therefore, predicting habitat suitability for species and management of areas where they are known to occur are now based on subjective judgments.

The species treated in this study represent a diverse group, yet all are found in wetland habitats and all have populations that are of special concern in either Minnesota or the United States (*e.g.*, populations have declined in the state or the status of the species is unknown) in Minnesota (Minnesota Department of Natural Resources, 1983). The American Bittern has been named on the Blue List, an "early-warning system" for birds (Tate and Tate, 1982; Tate, 1986).

The specific objectives of this study were to: (1) quantify the habitat

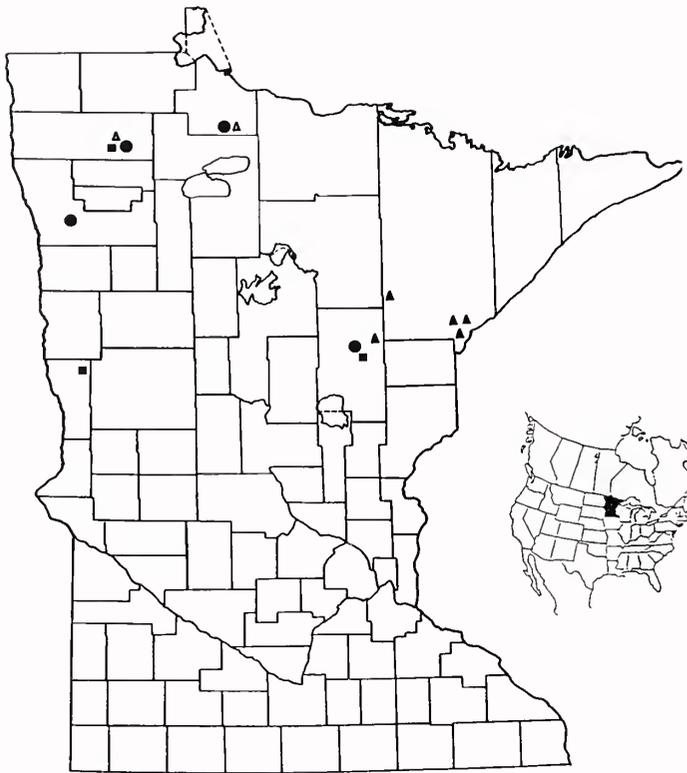
characteristics of breeding areas occupied by each species; (2) identify interspecific differences and similarities in wetland habitats that these species occupy during the breeding season; and (3) determine whether our methods could be used to predict suitable breeding habitat for a species.

## STUDY AREAS

All territories were located in northern Minnesota (Figure 1). We completed field reconnaissances in all areas where the species had been reported during a five year period (1981-1985) from records completed by the Minnesota Ornithologists' Union. Areas identified and visited included: (1) Agassiz National Wildlife Refuge in Marshall County; (2) Nature Conservancy prairie preserves in Clay and Wilkin Counties; and (3) several wetlands in Aitkin and St. Louis Counties. Northern Minnesota is near the southern edge of the breeding ranges for the Yellow Rail and Sharp-tailed Sparrow, but near the center of the American Bittern's breeding range (Peterson, 1980; Robbins *et al.*, 1983).

## METHODS

We visited potential nesting areas for American Bittern, Yellow Rail, and Sharp-tailed Sparrow in early June of 1983, 1984, and 1985. If the species was present, we marked the area with flagging. Vegetation sampling was delayed until early July to avoid jeopardizing nesting success. Thus, habitat variables measured here represent characteristics that were present late in the nesting cycle. Sampling in July allowed us to standardize sample collection time between years, but it could have resulted in sampling when individuals were no longer present. However, we felt that standardizing the time of sampling without disturbing nesting birds was more important than sampling during the breeding season (June), especially for rare species with possible sensitive nesting requirements. In addition, a major objective of the study was to develop a technique to determine potentially suitable breeding habitats for these species and not to document habitat characteristics selected by them.



**Figure 1.** General locations of Yellow Rail (●), Sharp-tailed Sparrow (■), and American Bittern (▲) territories in 1983, 1984, and 1985.

We sampled 27 areas and 270 points within areas occupied by the three bird species. Four habitats were sampled for the Yellow Rail and Sharp-tailed Sparrow in both 1983 and 1984, and two areas studied for each of these in 1985. We also sampled four American Bittern areas in 1984 and three in 1985. To control for potential regional differences between species habitats and to encompass a variety of breeding areas in Minnesota, we sampled areas from all regions of the state where these bird species are known to occur (Figure 1).

Vegetation was sampled using methods previously used in peatlands of northern Minnesota (Niemi and Hanowski, 1984; Niemi, 1985), modified from Wiens (1969) and Wiens and Rotenberry (1981). Ten point samples were made at 10-m intervals (see Mueller-Dombois and Ellenberg, 1974) along a randomly selected 100-m transect within the breeding areas of each species and the following characteristics were estimated (Table 1): 1) per cent ground cover, defined as the percentage of green vegetation < 10 cm high in a m<sup>2</sup> surrounding the point; 2) density and vertical distribution of grasses/sedges > 10 cm high; 3) density, vertical distribution, and species composition of forbs (plants > 10 cm); (4) water depth; and (5) density, vertical distribution, and species composition of vegetation > 30 cm high that is persistent in the habitat from year to year (e.g., shrubs and cattails).

Each variable was tested for normality and homoscedasticity of variances (Sokal and Rohlf, 1981) and log transformations were used to normalize distributions and stabilize variances when needed. Because we were interested in describing interspecific differences, a discriminant function analysis (DFA) was run to identify habitat characteristics that were different among species. DFA was also used to calculate the probability that each territory belonged within the habitat space defined for a species. This allowed us to determine the percentages of territories measured for each species that were correctly classified as suitable for that species. We examined correlations of all habitat variables prior to performing the DFA. This was done because we had relatively small sample sizes for these species, and it was necessary to limit the number of variables in the analysis. For example, it has been suggested that the minimum sample size required for a DFA must exceed the multivariate dimensionality by a factor of three to achieve adequate stability in discriminant loadings (Williams and Titus, 1988). Therefore, we reduced our set of variables threefold (from 18 to 6). Although this slightly exceeded the minimum recommended ratio, we felt that all six variables chosen for the analyses were required to distinguish species territories adequately.

A t-test was used to test differences in species means for the six variables used in the DFA (see Table 2). In cases where this test was significant ( $P < 0.05$ ) a paired comparisons test was completed for the variable. A chi-square test was used to test for interspecific differences in plant species composition among all species' territories, and, if significant, among species pairs.

**Table 1.** Description of habitat variables collected at each point within territories of three wetland bird species.

Variable	Description (unit)
Overall height	Estimate of the average height of vegetation in 10 m <sup>2</sup> surrounding point (m)
Vegetation height	Tallest vegetation in m <sup>2</sup> surrounding point (cm)
Ground cover	Percent of green vegetation less than 10 cm high in m <sup>2</sup> surrounding point (%)
Water depth	Depth at point (cm)
Persistent vegetation density	Density of persistent vegetation > 40 cm measured by the point-centered quarter method (stems/5m <sup>2</sup> )
Persistent vegetation height	Average height of the four persistent vegetation species measured for the persistent vegetation density variable (cm)
Persistent vegetation hits	Number of hits measured in 0-30 cm, 31-60 cm, and 61-100 cm height intervals
Forb density	Density of forbs > 10 cm high measured by the point-centered quarter method (stems/m <sup>2</sup> )
Forb hits	Number of hits measured in 0-30 cm, 31-60 cm, and 61-100 cm height intervals
Grass/sedge density	Density of grasses/sedges > 10 cm high measured by the point-centered quarter method (stems/m <sup>2</sup> )
Grass/sedge hits	Number of hits measured in 0-30 cm, 31-60 cm, and 61-100 cm height intervals

**Table 2.** Mean (or median) for habitat variables gathered in breeding territories of three bird species in 1983, 1984, or 1985 and grand mean for all samples. F-values and result of t-test for variables included in the discriminant analysis (see Table 4) are also shown.

Habitat Variable	Yellow Rail				Sharp-tailed Sparrow				American Bittern			F-value
	1983	1984	1985	X	1983	1984	1985	X	1984	1985	X	
N	40	40	20		40	40	20		40	30		
Overall Height (m)	1.3	0.8	1.2	1.1	1.2	0.8	1.1	1.0	1.5	1.1	1.3	1.54
Vegetation Height (cm)	130.0	95.4	141.8	122.4	111.3	105.8	148.6	121.9	146.2	121.4	133.8	
Ground Cover (%)	15.7	28.5	3.8	16.0	13.8	7.8	8.2	9.9	7.3	10.6	8.9	0.23
Water Depth (cm)	7.6	4.0	26.2	12.6	7.7	0.5	26.8	11.7	8.3	12.8	10.6	0.36
Persistent Vegetation	95.7	32.2	103.9	77.3	44.6	41.7	72.8	53.0	98.8	94.5	96.7	
0-30 cm	16.8	29.6	2.9	16.4	18.6	13.5	1.8	11.3	12.3	4.7	8.5	
Grass/sedge 31-60 cm	14.2	24.6	15.7	18.2	15.9	16.3	13.6	15.3	9.2	10.7	9.9	
Hits 61-100 cm	10.6	1.6	20.0	10.7	8.9	7.5	20.5	12.3	3.8	12.1	7.9	
0-30 cm	0.5	0.7	0.3	0.5	0.4	1.4	0	0.6	1.0	0.6	0.8	
Forb 31-60 cm	0.1	0	0.2	0.1	0.2	0.1	0	0.1	0.2	0.3	0.2	
Hits 61-100 cm	0.5	0	0.1	0.2	0	0	0.1	0.03	0.1	0.1	0.1	
Persistent 0-30 cm	0.5	0.1	0	0.2	0	0.5	0	0.2	1.1	0.1	0.6	
Vegetation 31-60 cm	1.0	0	0.3	0.4	0	0.2	0.3	0.2	1.8	0.2	1.0	
Hits 61-100 cm	1.7	0	0.1	0.6	0	0	0	0	2.4	0.3	1.4	
Grasses/sedges												
Density <sup>M</sup> (m <sup>2</sup> )	220.1	450.1	402.4	357.5	148.2	219.6	177.8	181.9	111.1	117.2	114.2	4.92*
Forb Density <sup>M</sup> (m <sup>2</sup> )	8.3	9.8	0.2	6.1	0.1	51.1	0.3	17.2	3.6	4.1	3.9	0.08
Persistent Vegetation	29.5	0.1	78.4	36.0	0.1	0.1	0.8	0.3	167.0	3.9	85.5	8.03**
Density <sup>M</sup> (5 m <sup>2</sup> )												

M Medians

\* P < 0.05

\*\* P < 0.01

## RESULTS

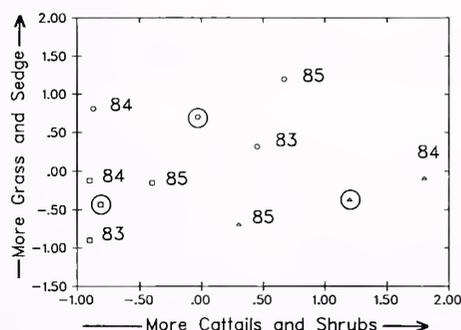
### Interspecific Habitat Differences:

All species occurred in wetland habitats during the breeding season, but, based on quantitative data, each species occupied a distinct set of microhabitats within the wetland complexes. Each species occupied an area that was different ( $P < 0.05$ ) from both other species for at least one measured habitat characteristic (Table 2). Yellow Rail territories had more ( $P < 0.05$ ) persistent vegetation than Sharp-tailed Sparrow territories and a higher density of grasses/sedges ( $P < 0.05$ ) than American Bittern territories (Table 2). In addition, American Bittern territories had more persistent vegetation ( $P < 0.001$ ) than Sharp-tailed Sparrow territories (Table 2). Each species had a distinct plant species composition in its territory ( $P < 0.01$ ) (Table 3). Despite overall differences in vegetation, plants of the mint family (Lamiaceae) were shared as the most common forb species found in American Bittern and Sharp-tailed Sparrow territories, and willow (*Salix*) and cattail (*Typha*) were the most common persistent plants found in all territories (Table 3). By contrast, loosestrife (*Lysimachia thyrsoiflora*) was the most common forb species found in Yellow Rail territories.

### Predictive Discriminant Analysis:

Discriminant functions 1 and 2 (DF 1 and DF 2) in analysis of habitat differences were significant ( $P < 0.05$ ) when using each species defined as a group. We interpreted DF 1 (Table 4) as a vegetation gradient that ranged from open grass/sedge (e.g., wet prairies) to cattail-

and shrub-dominated wetlands (Figure 2). This axis primarily separated wetlands occupied by Sharp-tailed Sparrows (relatively low densities of cattails and shrubs) from wetlands occupied by the American Bittern (relatively high densities of cattails and shrubs) (Table 2, Figure 2). DF 2 was related with grass/sedge density and primarily separated areas where the Yellow Rail was found from areas occupied by the American Bittern and Sharp-tailed Sparrow. Sharp-tailed Sparrows were found in areas with relatively high densities of grasses/sedges ( $> 220$  stems/m<sup>2</sup>), while medium densities where American



**Figure 2.** General interpretation and mean centroids for three wetland bird species [Yellow Rail (●), Sharp-tailed Sparrow (■), and American Bittern (▲)] in a space determined by discriminant functions 1 (X-axis) and 2 (Y-axis) for 1983, 1984, and 1985.

Bittern and Sharp-tailed Sparrows were found, were less than 220 stems/m<sup>2</sup> (Table 2).

Sixty seven per cent of the territories were correctly classified as belonging to the correct species. Yellow Rail and Sharp-tailed Sparrow territories were assigned correctly to their respective groups 70% and 90% of the time. In contrast, American Bittern territories were classified correctly only 43% of the time. If the territory was not correctly classified to its own group, it was most often found to correspond to the Sharp-tailed Sparrow group.

**Table 3.** Percentage of individual forb species (A) and persistent vegetation species (B) identified within the territories of three bird species and results of chi-square test. Only those plant species with a frequency > 5% in one category are included. All others were grouped in the others category.

	Yellow Rail	Sharp-tailed Sparrow	American Bittern
N	400	400	280
<b>A. Forb Species</b>			
<i>Calla palustris</i>	4	-	9
<i>Chamaedaphne calyculata</i>	7	-	4
<i>Potentilla palustris</i>	1	2	6
<i>Apocynum</i> sp.	5	1	-
<i>Zizia aurea</i>	-	-	8
<i>Lysimachia thyrsoiflora</i>	23	-	13
Lamiaceae	21	44	22
<i>Galium</i> sp.	9	13	7
<i>Petasites</i> sp.	1	-	6
<i>Solidago</i> sp.	5	5	3
<i>Equisetum</i> sp.	3	6	-
Others	21	29	22
Chi-square	96.6***		
<b>B. Persistent Vegetation</b>			
Apiaceae	2	6	-
<i>Typha latifolia</i>	58	35	36
<i>Phragmites communis</i>	-	5	3
<i>Salix</i> sp.	39	46	45
<i>Betula pumila</i>	-	-	7
<i>Populus tremuloides</i>	-	5	1
Others	1	3	8
Chi-square	47.8***		

\*\*\* P < 0.001

#### Annual Habitat Variation:

The overall mean position (average discriminant scores for all points) for each species in the discriminant space was generally consistent from one year to the next. Areas where the Yellow Rail and American Bittern were found showed a high degree of annual variation in their position along DF 1, whereas the Sharp-tailed Sparrow showed less variation in the areas it occupied each year (Figure 2).

**Table 4.** Summary statistics for discriminant function analysis using six habitat variables.

Habitat variables	Standardized Discriminant Function Coefficients	
	DF1	DF2
Persistent Vegetation Density	0.90	0.48
Grass/sedge Density	-0.29	0.98
Eigenvalue	0.69	0.32
Wilks lambda	0.44	0.75
X <sup>2</sup> of Wilks lambda	19.1**	6.6
Variation explained	68.2	31.8

\*\* P < 0.001

\* P < 0.01

## DISCUSSION

### Quantitative Versus Qualitative Habitat Descriptions:

It is difficult to compare our data with other studies, because previous accounts of breeding habitats of these birds are mainly qualitative. For example, breeding habitats have been described for Yellow Rails as: (1) large marshes of mixed-sedge and bulrush with cattails in deeper water in Minnesota (Savalaja, 1981); (2) small boggy areas with grassy hummocks and water-filled depressions less than three acres in size in Manitoba (Lane, 1962); (3) fens or boggy swales fed by springs in North Dakota (Stewart, 1975); and (4) monotypic stands of *Carex lasiocarpa* with standing water and procumbent, matlike canopies of dead vegetation in Michigan (Bart *et al.*, 1984). Similar accounts of breeding habitats are available for Sharp-tailed Sparrow such as: (1) bulrush wetlands in North Dakota (Krapu and Green, 1978); (2) extensive marshy areas with cordgrass (*Spartina pectinata*) as the predominant plant, but with no sedges and sourdock that are used as song perches (Murray, 1969); and (3) very wet, boggy marshes in Minnesota (Roberts, 1932). Some authors have used wetland classification systems (*e.g.*, Cowardin *et al.*, 1979) for classifying habitats occupied by wetland breeding birds. For example, American Bitterns breed in semi-permanent ponds, stock ponds, seasonal ponds, intermittent streams, dugouts, and permanent streams (Weber *et al.*, 1982).

Qualitative descriptions of avian habitat preferences may be useful to initially identify potential breeding areas for a species, but they do not provide information on specific habitat features that a species may require for breeding (Roth, 1979). It is evident that standardized quantitative methods are needed to define subtle habitat features that may be required for breeding by certain birds (James and Shugart, 1970). These data are especially needed for those species with status of special concern (Kantrud and Stewart, 1984). The power of a quantitative method, such as the one used here, is that it can be used to predict whether an area is potentially suitable breeding habitat for a given species.

### Use of DFA to Predict Habitat Suitability:

Among habitat management goals for wildlife species of special concern are: (1) to predict whether an area is suitable for a particular species; and (2) to identify habitat characteristics missing from an area that potentially render it unsuitable as breeding habitat for a given species. DFA is a multivariate, statistical technique that can possibly be used to accomplish both goals. For example, DFA can be used to identi-

fy habitat characteristics in each species' territories that are different from one another, and it also allows one to calculate the probability that a vegetation sample belongs within the range of habitats occupied by a species. We suggest that this might be considered equivalent to predicting the probability that an area provides suitable habitat for a species.

Data presented here are used to illustrate a quantitative method for assessing habitat suitability for breeding birds and are not to be construed as delimiting the definitive range of breeding habitats for these species. However, use of the quantitative method (DFA) to predict habitat suitability for three wetland species considered here proved encouraging for two species, because it correctly classified high percentages of Yellow Rail and Sharp-tailed Sparrow territories correctly. These values (70% and 90%) are similar to the values of 86 to 91% reported by Rice *et al.* (1983) for bird species in riparian habitats. This information also can be used to identify how specific or general the breeding habitat requirements are for a given species. For example, the position of Sharp-tailed Sparrow territories in the discriminant function space was less variable annually than for the other two species. Based on our field observations and on the DFA, this species appears to be more selective in choosing the habitats it uses for breeding. In contrast, the Yellow Rail and American Bittern appear to be able to breed in a wider variety of wetland habitats.

We consider this analysis to be exploratory and, hence, this data base is probably too small to use on a wide regional scale. More territories need to be measured over a larger portion of the breeding ranges of these species to encompass a greater proportion of the variability in habitats used by each. A real test of this method would be to sample plant communities of unknown occupancy by the birds, then to use the prediction model to try to determine whether the areas are suitable breeding habitat; finally a census should be carried out to determine their presence or absence. We expect, however, that the bird species have small, variable populations and that some suitable areas, as identified by DFA, may not be colonized. For example, Bart *et al.* (1984) identified several areas in Michigan that had vegetation characteristics thought to be suitable for the Yellow Rail, but that species was not found there. They reasoned that these areas may have been suitable habitat, but the species may not have been found because populations were low, and rails are apparently gregarious, forming loose colonies (Morris, 1905; Terrill, 1943; and Lane, 1962). Sharp-tailed Sparrows have also been reported to occur in colonies (Murray, 1969). Other factors (*e.g.*, behavior and wintering habitat) may also determine whether suitable habitats are colonized by these species.

#### Interspecific Habitat Differences:

We found interspecific differences in the community characteristics and plant species composition of habitats where the three species were found during the breeding season. The DFA, using two habitat variables, was successful in separating all three species' territories. Areas where the American Bittern and Sharp-tailed Sparrow were found were most dissimilar, compared with areas where Yellow Rail and Sharp-tailed Sparrow were found. This was also consistent with our observations in the field. Yellow Rail and Sharp-tailed Sparrow often co-occur in wetland complexes in northwestern and central Minnesota. However, we believe that each species occurs in a distinct microhabitat within these wetland complexes. Yellow Rails select areas with higher densities of shrubs and cattails as compared with Sharp-tailed Sparrows. We always observed some forms of persistent vegetation (*e.g.*, cattails and shrubs) in all Sharp-tailed Sparrow habitats; their presence is likely to be associated with their use as song perches (Murray, 1969).

#### Perspectives Concerning Wetlands:

In the early 1900's, wetlands were considered to be "unproductive and an economic waste" (Palmer, 1915). Losses of wetlands in the prairie pothole region were estimated at 51% between 1900 and 1955 (Schrader, 1955) and at 75% between 1850 and 1977 (U.S. Dept. Agriculture, 1980). It is not surprising that many endangered, threatened and rare bird species use wetlands during a portion of the year, and their currently precarious status is at least partially a by-product of the loss of wetland habitats. Little quantitative information exists for wetland habitats used by bird species that have status of some concern. It is possible that subtle habitat characteristics separate suitable and unsuitable habitats for many of these species. These subtleties must be defined before we can fully ascertain the impact of losses and properly manage remaining habitats for wetland bird species (Fredrickson and Taylor, 1982; Kantrud and Stewart, 1984). In summary, we have described a potential approach for identifying suitable habitats for three species. This information provides a possible framework for predicting whether a given wetland is suitable or unsuitable for a particular species and, hence, whether an area should be protected for potential use by a target species.

#### ACKNOWLEDGMENTS

Funding for this investigation was provided by the Minnesota Department of Natural Resources Nongame and Minerals Divisions. We thank L. Pfanmuller and N. Aaseng for their help in implementing the study and P. Collins and G. Grunwald for their assistance in collecting field data. J. Blake, F. James, R. Newman, J. Pastor, and J. Verner offered constructive comments on an earlier draft.

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# A Modified Point-centered Quarter Sampling Technique: A Tool for Plant Community Classification and Evaluation

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**Abstract:** The Ohio Natural Heritage program has used a modified point-centered quarter sampling method to gather data on the structure and composition of forested plant communities in the state. During the 1987 field season, the performance of this method was tested. Three forested communities, representing a gradient of species richness and diversity, were selected from Ohio's state nature preserve system. DBH and species identity were recorded for every canopy tree within a 1.5 ha area in each community to derive absolute data on per cent composition (both weighted by tree DBH and unweighted), species richness and mean DBH. Each area was then re-sampled, using the modified point-centered quarter method, carried out by three experienced surveyors, and three inexperienced surveyors. A total of nine inexperienced surveyors were used to prevent inexperienced surveyors from gaining experience with subsequent samples. Experienced surveyors, as a group, generated data sets significantly more similar to the absolute data than inexperienced surveyors. Estimates of mean DBH, and per cent composition weighted by DBH, did not differ significantly from absolute data for most experienced surveyors. There was no significant trend in the performance of the method over the range of diversities studied. Staff with adequate training in the method are able to generate data sets that accurately picture the structure and composition of forested communities.

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## INTRODUCTION

The evaluation of plant communities often serves as a "coarse filter" by which potential nature preserves may be selected (The Nature Conservancy, 1982). Some sort of standardized method of evaluating plant community examples is needed to assure that quality examples are selected. Field methods used to collect data upon which such evaluations and acquisitions are based should give an accurate picture of plant community structure and composition, be easily learned and applied in the field, and be efficient in terms of time, energy and money.

Many conservation organizations gather only qualitative data such as a simple species list, with subjective notes on the relative abundance or dominance of species. It is often argued that agencies charged with preservation on a state or region-wide level only have time to gather such qualitative data. Quantitative sampling techniques may be more time and labor intensive, but they yield more statistically reliable and repeatable results than do qualitative techniques. Some quantitative sampling techniques, such as relevé synthesis (Almendinger, 1988), may yield reliable data when sampling is done by experienced personnel, but these methods may be difficult to learn quickly. Further, the degree to which a single large plot is truly representative of an entire community example depends wholly on the experience of the surveyor in placing such a plot, a complication that can render relevé data quite subjective. Subjectivity may be removed by taking a random sample of vegetation in a community occurrence, but completely random techniques are often time consuming.

The Division of Natural Areas and Preserves, Ohio Department of Natural Resources, has used a modified point-centered quarter method for gathering plant community data since 1982. This paper presents a description of that method and the results of tests to determine how well the method depicts the actual structure and composition of forested plant communities; also tested were the effects of surveyor's experience on the accuracy of data generated by the method.

## SAMPLING TECHNIQUE

The Division's modified point-centered quarter method, which we refer to as the basic quantitative method, or BQM, was adapted from the method of Cottam and Curtis (1949), by Anderson (1982). A concise description of the unmodified method appears in Mueller-Dombois and Ellenberg (1974).

The surveyor's first task is to identify the boundaries of the plant community to be sampled. This step does not require the surveyor to identify the actual community type, but only to recognize borders of vegetation types where they occur. Samples are then taken within more or less discrete vegetation units.

The first sampling point is selected at random, within the constraint of avoiding community edges. A transect direction is then selected, usually along a compass heading, and the transect is placed so it will not parallel any obvious gradients in the community. Data are taken at 20 points in each community. The distance between points is paced rather than measured and based on the density of the vegetation and the length of the surveyor's legs. Points must be sufficiently spaced to assure that the same trees are not sampled at consecutive points. Thus, communities with widely dispersed trees require a larger point to point distance than would very dense communities. The 20 points may be placed along several transects, in effect forming a grid, to properly space the points and avoid sampling beyond community boundaries.

At each point, the surveyor mentally draws a line perpendicular to the transect, dividing the area around the point into quarters. In each quarter, the canopy tree closest to the point is recorded by species and size class. We define canopy trees as those trees with a DBH > 10 cm. Size classes are recorded in 20 cm increments, beginning at 10 cm, and are indicated by letter codes starting with a = 10-30 cm. In order to save time and allow for use of only one surveyor per site, exact DBH and point-to-tree distances are not measured as they are in the unmodified method. Each surveyor estimates DBH visually, and calibrates his or her "visual template" by measuring several trees at the beginning of, and several times during, the field season.

Finally, all herbaceous species and tree seedlings (defined as tree species less than 1 m in height) rooted in a 1 x 1 m plot located in any one of the four quarters at each point, are recorded by species. All shrubs and saplings (defined as tree species  $\geq$  1 m in height, and < 10

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cm DBH) rooted in, or whose branches reach over the same 1 m<sup>2</sup>, are recorded as present. All three layers are thus rapidly sampled, preserving both horizontal and vertical composition data.

### DATA REDUCTION

The raw data are first reduced to species-frequencies and per cent composition figures. Frequency is calculated as the percentage of points at which the species appears. Per cent composition is calculated as the percentage of total stems ( $n = 80$  in a 20 point sample) accounted for by species  $x$ . This is easily calculated by multiplying the total number of stems of species  $x$  by 1.25.

Per cent composition and frequency do not express dominance. In order to estimate dominance, we multiply the number of stems of species  $x$  in each size class by a factor that takes into account that size class, so that individuals in each successive size class are given twice the weight of those in the preceding class. Thus, for species  $x$ , the number of stems in the 10-30 cm size class is multiplied by 1.25, the number in the 30-50 cm class by 2.50, etc. A species' weighted per cent composition (WPC) is the sum of the weighted number of stems in each size class. WPC values are summed for each community, to yield a total WPC for each stand. An estimate of mean DBH is obtained by multiplying the number of stems of all species in a size class by the midpoint of that size class. Frequency is the only figure calculated for shrubs, saplings, seedlings and herbs.

### METHODS

Three different forested communities on three Ohio nature preserves were selected to test the performance of the BQM. These stands represented a gradient of diversity and species richness including a low diversity Appalachian oak forest, a middle diversity beech-maple forest, and a relatively high diversity beech-oak-red maple forest (Table 1). Forest types follow Anderson (1982). Within each community, a 1.5 ha area was marked off, and all canopy trees  $\geq 10$  cm DBH in the area are recorded and their DBH values recorded, to obtain absolute data on community composition and structure.

**Table 1.** Absolute data from the three communities sampled. Shannon-Wiener diversity ( $H'$ ) and weighted per cent composition (WPC) are dimensionless. Mean DBH is in cm, and was calculated using diameter measurements on all stems  $\geq 10$  cm DBH.

Site	Community Type		
	Appalachian oak	beech-maple	beech-oak-red maple
$H'$	2.5	2.4	2.7
No. spp.	11	15	18
WPC	196.4	284.0	205.7
mean DBH	34.3	37.5	41.0

Each community was then sampled, using the BQM, by three inexperienced and three experienced surveyors. Inexperienced surveyors had no previous experience with the BQM, but possessed plant identification skills. Three different inexperienced surveyors were used for each of the three sampled communities ( $n = 9$ ) to insure inexperienced surveyors did not gain experience by sampling several communities.

Each experienced surveyor had performed county-level natural area inventories, using the BQM, for at least 1 yr.

Per cent composition, weighted per cent composition and mean DBH were calculated for the absolute, and for each surveyor's data sets. Per cent similarities, based on total WPC, were calculated and used to compare surveyor's data to absolute data. Per cent similarity was calculated following Bray and Curtis' (1957) method:

$$c = 2w/(a+b)$$

where:  $a$  = the sum of the WPC's of stand A.

$b$  = the sum of the WPC's of stand B.

$w$  = the sum of the lesser values of WPC's for those species common to both stands.

Per cent similarities were arc-sine transformed, and the transformed data were lumped into experienced and inexperienced classes. Differences between classes were tested by ANOVA (Snedecor and Cochran, 1980). Total WPC, and WPC for the characteristic dominant species in each community, generated by individual surveyors, were compared to the absolute data, using the Chi-square goodness-of-fit test (Snedecor and Cochran, 1980). Characteristic dominants were defined as those species that occur in at least 70% of all the data sets of a community type. Those sites in which the dominant's WPC represents a large proportion of the total WPC are considered good structural examples of the type. Mean DBH estimates generated by each surveyor were compared to the absolute data, using a two-tailed t-test (Snedecor and Cochran, 1980).

**Table 2.** Similarity coefficients expressing the degree of similarity between absolute community data and BQM data generated by inexperienced and experienced surveyors. ANOVA a shows the results of ANOVA performed on data from both surveyor classes. ANOVA b was performed on experienced surveyor's data only, using community type as the class variable.

Community type	Surveyor Class			Experienced		
	1	2	3	1	2	3
Appalachian oak	0.87	0.82	0.88	0.96	0.99	0.98
beech-maple	0.97	0.98	0.96	0.98	0.96	0.96
beech-oak-red maple	0.81	0.94	0.56	0.99	0.90	0.90
Mean similarity for class	0.89			0.96		

Analysis of Variance (a)				
Mean	df	Square	F	p
Source				
Between classes	1	342.085		
Within classes	16	65.949	5.187	0.037
Total	17	82.192		

Analysis of Variance (b)				
Mean	df	Square	F	p
Source				
Between classes	2	25.59		
Within classes	6	21.66	1.18	> 0.25 ns
Total	8	47.25		

## RESULTS

Similarity coefficients for both experienced and inexperienced surveyors were 0.80 or higher for all cases but one (Table 2). All experienced surveyors had similarity coefficients greater than 0.90. Data sets with coefficients of at least 0.80 are thought to represent samples drawn from the same population (Bray and Curtis, 1957). Experienced surveyors, as a group, generated data sets with significantly higher coefficients of similarity than did inexperienced surveyors (Table 2, ANOVA a).

Significant differences between surveyor and absolute WPC values were found for six inexperienced and three experienced data sets (Table 3). Experienced surveyors generated data that accurately pictured structure and composition in nearly 70% of the cases.

In six cases, there were significant differences between characteristic dominant WPC values generated by inexperienced surveyors and the absolute values (Table 4). Two values generated by experienced surveyors differed significantly from the absolute data, thus data sets taken by experienced surveyors give a more accurate picture of dominant composition.

**Table 3.** Total WPC values for all surveyor's data, and the absolute data, shown by community type and surveyor class. Significant differences between surveyor's and absolute data for each community type are shown by lower case letters, where \* =  $p < 0.025$  and \*\* =  $p < 0.005$ .

Surveyor class	Community Type			
	Appalachian oak	beech-maple	beech oak-red maple	
Inexperienced	1	152.0 **	203.0 **	228.0
	2	200.3	183.5 **	204.5
	3	141.4 **	184.0 **	81.8 **
Experienced	1	183.0	191.0 **	184.0
	2	183.7	201.3 **	203.4
	3	221.0	270.7	170.4 *
Absolute	196.4	284.0	205.7	

Inexperienced surveyors incorrectly estimated mean DBH 5 out of 9 times, while only two experienced surveyor's estimates showed any significant difference (Table 5). Provided a surveyor has an accurate "DBH template" stored in his or her mind, the BQM will generate an accurate estimate of a stand's mean DBH.

Similarity coefficients from experienced surveyors, particularly data from surveyors 2 and 3, suggest a slight trend toward greater similarity between surveyor and absolute data as species richness declines. Similarity coefficients were quite high (.99 and .98 for experienced surveyors 2 and 3, Table 2) for the low richness Appalachian oak community (no. species = 11, Table 1), and somewhat lower (.90 for both surveyors, Table 2) for the higher richness beech-oak-red maple community (no. species = 18, Table 1). To examine differences in similarity values between species richness levels, an ANOVA was performed using community type as the class variable, and only similarity coefficients from experienced surveyors. No significant differences were found (Table 2, ANOVA b). Accuracy of data generated by the BQM does not appear to be affected by species richness.

## DISCUSSION

Our experienced surveyors had at least one year of experience using these methods in the field. Given that amount of training with the methods or more, surveyors can employ the BQM to generate data that accurately depict the structure and composition of forested plant communities. Parameters such as weighted and unweighted per cent composition and mean DBH may be accurately estimated and used to classify and rank examples of plant communities for preservation purposes.

**Table 4.** WPC values for the characteristic dominant species for all surveyor's data, and the absolute data, shown by community type and surveyor class. Significant differences between surveyor's data and absolute values for each community are indicated by lower case letters where \* =  $p < 0.05$ , \*\* =  $p < 0.01$  and \*\*\* =  $p < 0.005$ .

Surveyor class	Community type			
	Appalachian oak	beech-maple	beech oak-red maple	
Inexperienced	1	120.0 ***	153.0	180.0 ***
	2	177.7	139.0 *	172.2 ***
	3	118.8 ***	153.9	57.4 ***
Experienced	1	159.0	162.0	149.0
	2	131.2 *	131.3 *	140.7
	3	136.7	171.6	115.5
Absolute	159.0	167.0	129.2	

Ranking of a plant community example is usually done to indicate the example's importance to a preservation effort's goals. Comparisons are made between examples of a given community type, not on a community type to type basis. Several numerical scoring systems have been developed to rank plant community examples as potential natural areas (e.g. Tans, 1974). Most ranking systems assign a numerical value

**Table 5.** Estimated mean DBH values calculated from surveyor's data, and absolute values for mean DBH of all trees  $\geq 10$  cm DBH within the sampling area for each community. Mean DBH values which differ from the absolute values are indicated by lower case letters, where \* =  $p < 0.005$ , and \*\* =  $p < 0.005$ .

Surveyor class	Community type			
	Appalachian oak	beech-maple	beech oak-red maple	
Inexperienced	1	34.2	41.8	32.5**
	2	40.5	37.2	49.5**
	3	43.0*	41.3**	41.5
Experienced	1	31.3**	37.5*	42.5
	2	40.1	34.0	39.8
	3	37.3	34.1	40.0
Absolute	37.5	34.3	41.0	

to a community quality, which has been subjectively ranked. Thus, while these systems may appear highly quantitative and objective, it is often not the case. Most systems rank stands by their naturalness, lack of anthropogenic disturbance, rarity of the community type in the state or region, diversity, stand size and preservability. While many of these factors do not lend themselves to measurement in the field, several may be easily quantified. Noss and Harris (1986) differentiated those ranking-factors which are intrinsic, measurable properties of a community (they called these "content"), from those properties that essentially effect preserve design, such as stand size, shape and isolation (called "context"). The Division of Natural Areas and Preserves uses quantitative data generated by the BQM to assess a stand's content. Context is assessed by examining a stand's position in the landscape, its size, shape, degree of isolation and the rarity of the community type within the state and physiographic region.

The Division is beginning to rank areas on the basis of quantitative differences in the collected data. Characteristic dominant WPC is used to examine the composition of a stand. When the species WPC represents a high proportion (75% or greater) of the stand's total WPC, the stand has a composition typical for the type.

Mean DBH is a criterion often used to compare examples of a forested community. All other criteria being equal, the stand with the greater mean DBH is probably the older, or at least more mature, example of that type. Surveyors must, however, keep in mind a site's capability of supporting large diameter trees, and not overemphasize the simplistic connection between tree diameter and "old growth" forest conditions (Juday, 1988). Only where careful consideration is given to a site's tree-growth potential, may mean DBH be used to make meaningful comparisons between community examples.

Species composition data can be used to assess a stand's naturalness in terms of the relative abundances of conservative and invasive or weedy species. Understory shrub, herb, seedling and sapling data may all be used to assess naturalness, the level of anthropogenic disturbance, alien invasion and stand stability. Careful field notes, combined with accurate BQM data, may also be used to assess disturbance history.

Forested communities often have indistinct boundaries. Consider a slope in the unglaciated Appalachian plateau. Typically there will be a gradient from a xeric, often oak-dominated situation on the ridge top to more mesic conditions down-slope. Qualitative data gathering methods, and some quantitative methods such as relevé synthesis, demand that the surveyor make an *a priori* decision as to the type and location of the communities on the landscape. The relevé method in particular stresses the importance of proper placement of the sample within the community, a process called entitiation (Almendinger, 1988). The single, large plot that makes up the relevé must be placed in the "most representative" portion of the stand. Qualitative and relevé data are thus wholly dependent upon the classification used to stratify the samples, whereas the BQM allows the surveyor to take several contiguous 20 point samples if needed. These may then be lumped or segregated, as indicated by the data, to facilitate classification and ranking, eliminating the need to select the most representative portion of a stand, a process that requires a great deal of field experience.

The BQM provides accurate estimates of several parameters useful

in the classification and ranking of plant communities. The data collected represent a stratified, systematic sample of a plant community. As such, they may be statistically tested, subjected to ordination, and more robustly analyzed than can qualitative or more subjectively gathered quantitative data. The accuracy of an individual surveyor's BQM data may be tested against previously sampled areas, and any biases, such as a propensity to confuse species or select mostly larger diameter trees, may be identified. These biases may be difficult or impossible to detect using qualitative or relevé data. The skills necessary to perform the BQM are also easily learned, and may be more quickly acquired than would a broad knowledge of the types and inherent within-type variability of plant communities in an area.

## ACKNOWLEDGMENTS

We thank D.M. Anderson for devising the BQM and training us in its use. Valuable field help was provided by the central office staff of the Division of Natural Areas and Preserves, B.L. Kooser and K.M. Morland. Helpful suggestions on an earlier version of the manuscript were made by D.M. Anderson, R.J. Garono, B.L. Kooser, R.M. McCance Jr., T.L. Sharik and an anonymous reviewer. Funding for this research was provided by the Division of Natural Areas and Preserves, Ohio Department of Natural Resources.

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# Stand Structure of an Old-growth Upland Hardwood Forest in Overton Park, Memphis, Tennessee

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**Abstract:** The Overton Park Forest is an urban woodland with an impressive large-tree character. There is a marked under-representation of overstory tree species in the regeneration and sapling classes in this forest, however. In particular, oak regeneration is abundant, but saplings and poles of oak are greatly under-represented; in addition, there is a total absence of yellow-poplar (*Liriodendron tulipifera*) regeneration in this study. Three non-native woody perennials and a native grape appear to impede the gap-phase ecological processes by which these overstory species characteristically regenerate. The abundance of native understory trees, due in part to the exclusion of natural disturbances, may also play a role in the suppression of overstory tree regeneration in this urban forest. It is recommended that research on the enhancement of sapling recruitment by elimination of non-native vegetation, and perhaps by controlling native understory vegetation, be undertaken, to determine the best way to maintain the large-tree old-growth character of this forest.

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## INTRODUCTION

Overton is the most prominent park in Memphis. The 138.4-ha park and associated city parkway system were planned for the newly-established Memphis Park Commission by Kansas City landscape architect George Kessler in 1901. The spirit of innovation inherent in the establishment of a city parks commission and a park system at the turn of the century was reflected both in Kessler's foresighted and time-honored design, and in the thorough and competent execution of his plans. It is noteworthy that most of Memphis' early eastward development was mediated by Kessler's parkways and by Overton Park.

Today, the park exemplifies the City Beautiful movement of landscape architecture, with its romantic and pastoral themes. It lies slightly west of the metropolitan center of the city in the Midtown neighborhood, and serves a diverse regional clientele. To the residents of Midtown, it is a neighborhood park. Their visits are characterized by walking, jogging and bicycling. To Memphis, it is a major city park, with amenities that include a nine-hole golf course, recreational greensward for outdoor play and picnic facilities. To the inhabitants of the upper reaches of the lower Mississippi basin, it is a regional park, housing the Brooks Museum of Art, the Memphis College of Art and the Memphis Zoo and Aquarium. Nationally, Overton Park is associated with the setting of a major precedent in environmental law, because its presence halted the development of the I-40 corridor through the center of Memphis in the 1970s.

The predominant natural feature of the Park, instrumental in both its general appeal and its legal defense, is the 70.8-ha Overton Park Forest. This upland hardwood forest is renowned for its stately large-tree character; the largest trees exceed 150 cm in diameter at breast height (dbh., measured at a height of 1.37 m above ground), and are approximately 175 years old (Guldin, 1987). This forest lies within the Mississippi Alluvial Plain, and falls within the Outer Coastal Plain and the Southeastern Mixed Forest Provinces, Subtropical Division, Humid Temperate Domain (Bailey, 1976, 1978).

Because of the large trees, the forest is commonly called an old-growth forest; however, this should not convey the impression that the

forest is undisturbed. At the time of park establishment, the forest was part of a holding known as Lea Woods, and was associated with a local farm (Hopkins, 1987). Disturbances in Lea Woods were probably limited to farm-related woodlot activity such as grazing and unregulated selective cutting for fuelwood and timber products. The frequency or intensity of these disturbances is not known. Since its establishment as a park, the only deliberate disturbance has been the felling of dead trees which, if not obstructing trails or roadways, are left to decompose in place.

In mature upland hardwood forests in natural-area and rural locations east of the Mississippi River, overstory species, predominantly *Quercus*, are either under-represented or absent in the understory (Oosting, 1942; Bray, 1956; Den Uyl, 1961; Christensen, 1977; Harcombe and Marks, 1978; Hemond et al., 1983; Lorimer and Krug, 1983; Eickmeier, 1988; Parker et al., 1988; Harrison et al., 1989). This under-representation has been attributed to stand maturation and to the absence of gaps of suitable size in the overstory (Christensen, 1977; Oliver and Stephens, 1977; Harcombe and Marks, 1978). Given the importance of gaps in understory recruitment (Bray, 1956; Pickett and White, 1985), the consensus from these studies is that overstory oak species will decline in importance over time, due to insufficient replacement through recruitment from the sapling component.

In similar, mature, upland hardwood forests in urban areas, the lack of replacement of overstory oak species and the eventual decline in their importance have also been observed (Stalter, 1981; Airola and Buchholz, 1982; Profous and Loeb, 1984; Rudnický and McDonnell, 1989). Researchers generally attribute failure of sapling recruitment of the oaks to ecological factors frequently reported under rural conditions. In addition, they note the adverse influences of an urban environment on natural development, such as restriction of wildfire and other natural disturbances, altered wildlife populations, pollution and intensive recreational activity with adverse effects on soil properties.

In the Overton Park Forest, one's first impression is that non-native kudzu [*Pueraria lobata* (Willd.) Ohwi], honeysuckle (*Lonicera* spp.), and privet (*Ligustrum* spp.) occur in the very gaps where one would expect to find saplings of overstory species. Non-native vegetation has

not previously been implicated regarding under-representation of oak saplings in old-growth upland hardwoods. If the interactions of native species alone may lead to under-representation of the oak sapling component in old-growth forests, then additional competition from aggressive non-native species can almost surely result in a greater suppression of seedlings than has been previously reported.

## METHODS

In 1986, the Memphis Park Commission undertook an environmental plan to reconcile conflicting pressures concerning Overton Park, and to plan for future development. As a component of this plan, we conducted an assessment of stand structure within the Overton Park Forest. In the summer of 1987, we superimposed an 80.5 m<sup>2</sup> grid on the 70.8-ha forest. At each grid point, we conducted a nested-plot, stratified sample as follows. Regeneration, defined as trees between 0.3 m and 1.37 m in height, was sampled using two 4 m<sup>2</sup> circular plots located 6 m due north and south, respectively, of plot center. The sapling component, trees with a dbh larger than 0 cm but less than 9.1 cm, was sampled by 2.5-cm classes, using two circular 40 m<sup>2</sup> plots similarly located north and south of plot center. The pole component, trees with dbh ranging from 9.1 cm to 24.2 cm, was sampled by 2.5-cm classes using a circular 0.04-ha plot. The mature tree component (trees with dbh larger than 24.3 cm) was sampled by 2.5-cm classes using a 0.081-ha plot. The inventory resulted in an 8.45% sample of mature trees, a 4.23% sample of the pole component, a 0.73% sample of the sapling component, and a 0.074% sample of the regeneration.

For analysis, species were grouped as follows:

- the red oak component (*Quercus*, subgenus *Erythrobalanus*) including black oak (*Q. velutina* Lam.), cherrybark oak (*Q. falcata* var. *pagodifolia* Ell.), northern red oak (*Q. rubra* L.), southern red oak (*Q. falcata* Michx. var. *falcata*), Shumard oak (*Q. shumardii* Buckl.), water oak (*Q. nigra* L.), and willow oak (*Q. phellos* L.)
- the white oak component (*Quercus*, subgenus *Leucobalanus*), including white oak (*Q. alba* L.) and post oak (*Q. stellata* Wang.)
- the yellow-poplar component (*Liriodendron tulipifera* L.)
- the ash (*Fraxinus* spp.) and elm (*Ulmus* spp.) component
- the hickory (*Carya* spp.) and maple (*Acer* spp.) component
- the cottonwood (*Populus* spp.) and American sycamore (*Platanus occidentalis* L.) component
- the minor overstory species component (*Prunus serotina* Ehrh., *Liquidambar styraciflua* L., *Sassafras albidum* (Nutt.) Nees., *Nyssa sylvatica* Marsh., and others)
- the major understory tree species component (*Cornus* spp., *Ostrya virginiana* (Mill.) Koch., *Carpinus caroliniana* Walt., *Asimina triloba* (L.) Dunal., and others)
- the minor understory tree species component (*Aesculus pavia* L., *Aralia spinosa* L., *Viburnum* spp., *Bumelia lanuginosa* (Michx.) Pers., and others)

Importance values were calculated by averaging the relative density (stems per ha) and relative dominance (basal area per ha), for the sapling, pole and mature tree component, as well as for 5.1-cm dbh saplings, by species component. The importance value (IV) of the regeneration component is based only on relative density, because these individuals technically have no dbh, and no cover estimates were taken. In the graphs of importance value versus dbh, the importance value of the regeneration component is plotted at -5 cm dbh on the x-axis for purposes of graphical comparison with the larger size classes.

We determined the area of the forest currently occupied by canopy gaps from 1:300 aerial photographs using a Leitz polar compensation planimeter with optical tracer. We then visited the gaps, and assessed the presence of competition using per cent cover estimates in 25% intervals, for the major non-tree species competing with trees — native grapevines (*Vitis* spp.), and non-native kudzu, honeysuckle, and privet.

## RESULTS

### Forest structure:

Sample data show the Overton Park Forest to have 9,967 trees per hectare, of which 9,737 are in the regeneration and sapling component (Table 1). The red oak and white oak components account for nearly 36% of mature trees, and 14% of the regeneration, but only 0.4% of the saplings and 1.1% of the poles. *Liriodendron* is prominent in the overstory, and comprises over 26% of the mature tree component; however, only 4.6% of the poles and 0.1% of the saplings are yellow-poplar, and there was no yellow-poplar regeneration found during this inventory.

**Table 1.** Trees per hectare by species component and size component, Overton Park Forest.

Species Component	Regen.	Sapling	Pole	Mature Tree	All size classes
<i>Quercus</i>					
( <i>Erythrobalanus</i> )	565.9	7.7	1.3	23.5	598.5
<i>Quercus</i> ( <i>Leucobalanus</i> )	245.2	7.7	0.3	4.2	257.5
<i>Liriodendron tulipifera</i>	0.0	1.9	7.0	20.5	29.5
<i>Fraxinus-Ulmus</i>	207.5	212.4	15.8	5.2	440.8
<i>Carya - Acer</i>	283.0	432.4	34.9	4.7	755.0
<i>Populus - Platanus</i>	0.0	1.9	0.0	1.3	3.3
Minor overstory spp.	528.2	164.1	37.9	17.4	747.5
Major understory tree spp.	3,753.7	2,747.1	53.1	0.8	6,554.9
Minor understory tree spp.	377.3	201.0	1.3	0.2	579.8
All species	5,960.7	3,776.3	151.7	77.8	9,966.6

Both the *Fraxinus-Ulmus* and the *Carya-Acer* components exhibit increasing proportions of stems per hectare from regeneration and sapling stages through pole components (4% to 6% to 10%, and 5% to 12% to 23%, respectively), and these two components comprise 7% and 6% of the mature trees, respectively (Table 1). The minor overstory trees account for 25% of the poles and 22% of the mature trees, but for only 9% of the regeneration and 4% of the saplings. The understory tree species account for 72.7% of the plants in the regeneration and sapling components, 36% of the pole component, and 71.6% of all trees, but they comprise only 1.3% of the mature tree component.

The genera *Quercus* and *Liriodendron* have a combined importance value of almost 70% of the mature tree component (Table 2), but their importance values as poles (7%) and saplings (0.7%) are extremely low. The combined importance values of the *Carya-Acer* component, the *Fraxinus-Ulmus* component, and the minor overstory component comprise nearly 34% of the sapling class, almost 60% of the poles, but only 27% of the mature trees. Importance values of the understory species show a decline with increasing size, from 65% in the sapling

component to 33% in the pole component, and less than 1% in the mature tree component.

**Table 2.** Importance values (per cent) by species component and size component, Overton Park Forest.

Species Component	Regen.	Sapling	Pole	Mature Tree	All size classes
<i>Quercus</i> (Erythrobalanus)	9.49	0.15	1.20	38.55	23.27
<i>Quercus</i> (Leucobalanus)	4.11	0.15	0.23	5.78	3.97
<i>Liriodendron</i>	0.00	0.36	5.83	25.59	11.18
<i>Fraxinus Ulmus</i>	3.48	8.53	11.33	4.99	4.48
<i>Carya - Acer</i>	4.75	19.21	21.96	4.76	6.88
<i>Populus - Platanus</i>	0.00	0.06	0.00	2.18	1.16
Minor overstory species	8.86	6.04	26.43	17.21	10.42
Major understory tree spp.	62.97	60.89	32.09	0.79	35.57
Minor understory tree spp.	6.33	4.60	0.93	0.14	3.07

In the transition from regeneration classes to sapling classes, the importance of understory species declines slightly, from 69.3% to 65.5%. Overstory species that exhibit decreased importance include both the red oak and white oak components, which both drop to a very low importance, and the minor understory species, which decline slightly. Yellow-poplar, which was not represented in the regeneration component, shows a very minor increase in importance to 0.36%. The importance of the ash-elm component increases roughly two-fold, and the importance of the hickory-maple component increases roughly four-fold.

**Table 3.** Number of trees per hectare by indicated diameter classes in the oak components, the yellow-poplar component, and for all tree species in Overton Park Forest.

Species component	0-4 cm	5-25 cm	26-50 cm	51-75 cm	76-100 cm	100+ cm
Red oak	573.6	1.5	3.0	8.7	8.3	3.3
White oak	252.9	0.3	1.2	2.2	0.5	0.3
Yellow-poplar	0.0	9.8	8.0	6.8	4.2	0.7
All tree species	9,429.9	461.8	31.4	23.7	14.9	5.0

The density of stems across the larger diameter classes of oak (Table 3) generally conforms to the mound-shaped or bell-shaped diameter distribution typical of an even-aged stand (Meyer, 1930). Conversely, yellow-poplar exhibits a gradual decrease in density from the 5-25 cm classes through the 100+ cm classes (Table 3). These data more closely approximate a typical uneven-aged structure rather than an even-aged one (Meyer and Stevenson, 1943).

Canopy gaps occupy almost one-sixth of the Overton Park Forest (Table 4). However, no gap was free of competition involving shrubs and woody vines, and only 10% of the gaps had less than 50% cover from grapevine, kudzu, honeysuckle and privet. The gaps exhibiting total suppression (13%) were aggressively invaded by kudzu to the

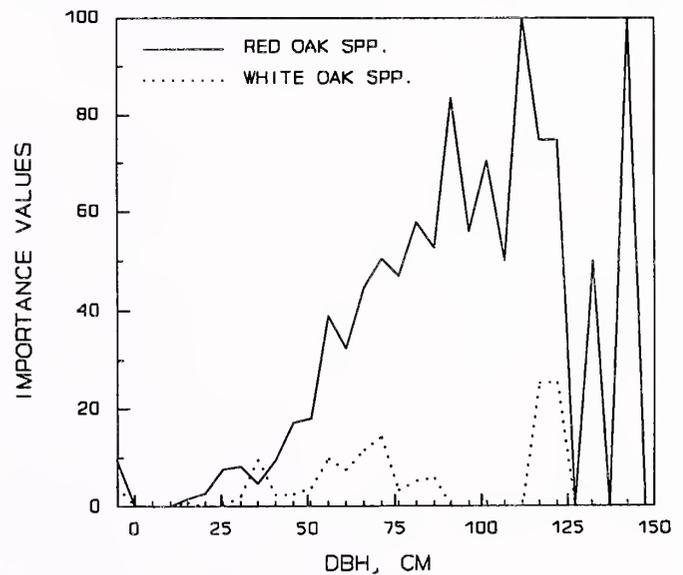
exclusion all tree regeneration and saplings within those gaps. Kudzu ascended poles and mature trees on the perimeter of the gaps as well.

**Table 4.** Extent of canopy gaps in Overton Park Forest occupied by native and non-native shrubs and woody vines.

Category	Per cent of Area, ha	Per cent of gap area	Per cent of total area
Canopy gaps	11.14		15.7 %
No competition (0-25% cover)	0.00	0.0 %	0.0 %
Minor competition (25-50% cover)	1.08	9.7 %	1.5 %
Major competition (50-75% cover)	8.60	77.2 %	12.1 %
Total suppression (75-100% cover)	1.46	13.1 %	2.1 %
Total forest area	70.82		100.0 %

#### Importance Values by Species Component:

Two thirds of the large trees in the forest (50 cm diameter class upward) are oaks, which dominate the site (Figure 1); the genus *Quercus* is thus the major contributor to the large-tree character in this forest. The red oak component is more prominent than the white oak component. This is not surprising, given that red oak is generally more prominent than white oak in this region. Red oaks constitute over half of the importance values of all tree species larger than 60 cm in diameter, and approximately three-quarters of all species larger than 100 cm. Yet, as important as the oak component is in the overstory, oaks are extremely poorly represented in the understory. This is a function of two conflicting factors. On the one hand, there are many woody species in the understory that are incapable of attaining the impressively large size of oak. Pawpaw (*Asimina triloba*), for example, exhibits extremely high density in the smallest size classes, and is more impor-



**Fig. 1.** Importance values, by 5.08-cm dbh classes, for the red oak group (*Quercus*, subgenus Erythrobalanus) and the white oak group (*Quercus*, subgenus Leucobalanus) in the Overton Park Forest.

tant than the oaks within those classes. On the other hand, the advance growth strategy of oak saplings requires that when gaps are created, saplings are in place, ready to respond to release (Carvell and Tryon, 1961). Visual assessment in the Overton Park Forest indicates that oak

saplings are absent from these openings.

Yellow poplar importance values exceed 20% from 25 cm through 100 cm (Figure 2), and this species also contributes significantly to the large-tree character of the forest. But, as in the oaks, the importance of the yellow poplar component is very low in dbh classes smaller than 25 cm, especially in the regeneration component, where it is absent. The sudden increase in importance of yellow-poplar in the 20-30 cm classes is possibly due to the declining prominence of the understory species.

The *Populus-Platanus* component has minor importance between 50 cm and 100 cm dbh (Figure 2). However, these species are largely restricted to the western edge of the Overton Park Forest adjacent to the local stream called Lick Creek. The presence of these early-successional species indicates that the park is probably not the old-growth relict stand that our imagination would hope for, but rather a stand approaching the transition stage or understory reinitiation stage

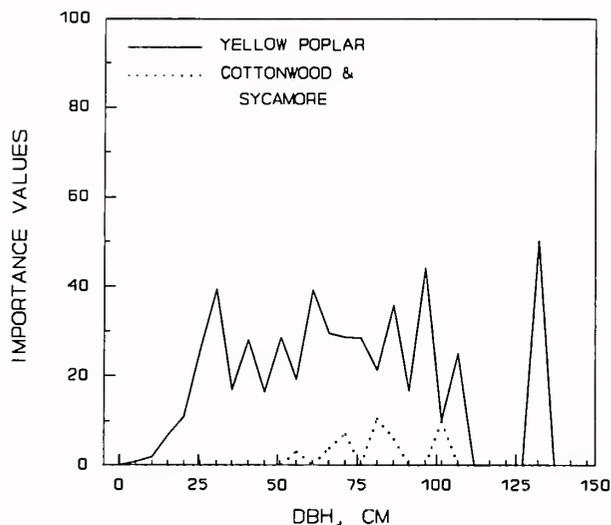


Fig. 2. Importance values, by 5.08-cm dbh classes, for yellow-poplar (*Liriodendron tulipifera*) and for cottonwood and sycamore (*Populus deltoides* and *Platanus occidentalis*) in the Overton Park Forest.

(Bormann and Likens, 1979; Oliver, 1981). Both stages are developmental transitions between an even-aged seral forest and the old-growth condition.

The *Fraxinus-Ulmus* component (Figure 3) is not a major constituent of the Overton Park Forest. American elm (*Ulmus americana* L.) was present in the forest at the time of the establishment of the park in 1901 (Hopkins, 1987); however, two disturbance events, Dutch elm disease and the extended drought in the region in the early 1950s (Fowells, 1975), were probably responsible for a drastic reduction of American elm from the forest. Ash is an important minor component in the park, and it will probably so remain for the foreseeable future.

Most species in the *Carya-Acer* component (Figure 4) are tolerant of shade, and their importance in the small size classes is therefore not surprising. They are poorly represented in the larger size classes, however, as they exist largely as scattered overstory representatives. Oliver (1978) speculated that hickories might ultimately replace decadent oaks as the dominant large trees in the overstory. Maples do not share in this projected possibility, and will probably remain minor but significant components of the Overton Park Forest; however, their tolerance to shade will contribute to their perpetuation as smaller trees in

the forest.

Minor overstory species (Fig. 5) showed greatest importance in the 15-cm to 60-cm dbh classes, where they exist in the transition from poles to the mature tree component. The nature of the species included in this component constrains the consistent progression of the group into the largest size classes, although the dbh of the largest sweetgum trees exceeds 75 cm.

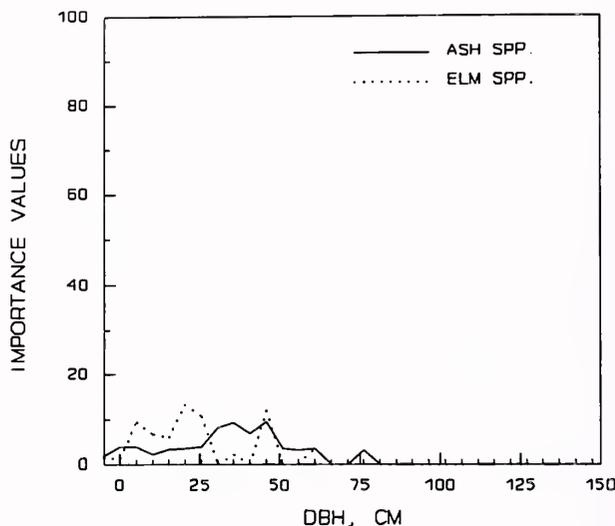


Fig. 3. Importance values, by 5.08-cm dbh classes, for ash (*Fraxinus*) and elm (*Ulmus*) species in the Overton Park Forest.

The major and minor understory tree species of the Overton Park Forest have importance values that exceed 60 per cent in the regeneration component, and 40 per cent up through the 15-cm dbh class. This prominent representation in the understory undoubtedly affects the regeneration of the large-tree overstory species through competition within the small size classes. The increase in importance of yellow

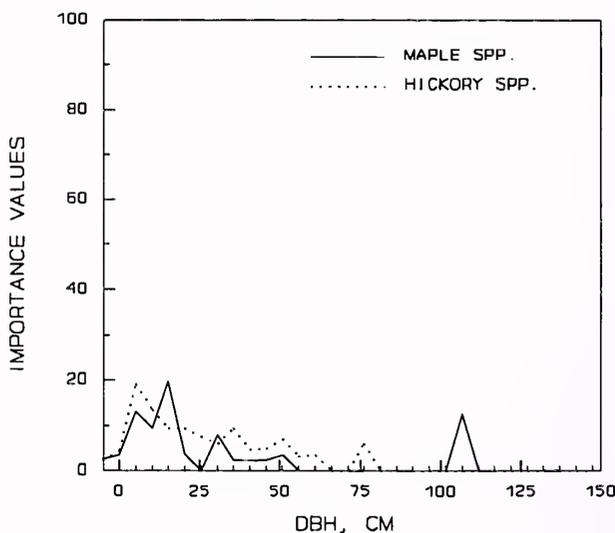


Fig. 4. Importance values, by 5.08-cm dbh classes, for maple (*Acer*) and hickory (*Carya*) species in the Overton Park Forest.

poplar and of the minor overstory species component in the 25-cm class (Figures 4 and 5, respectively) coincides directly with the decrease in importance of the understory species component.

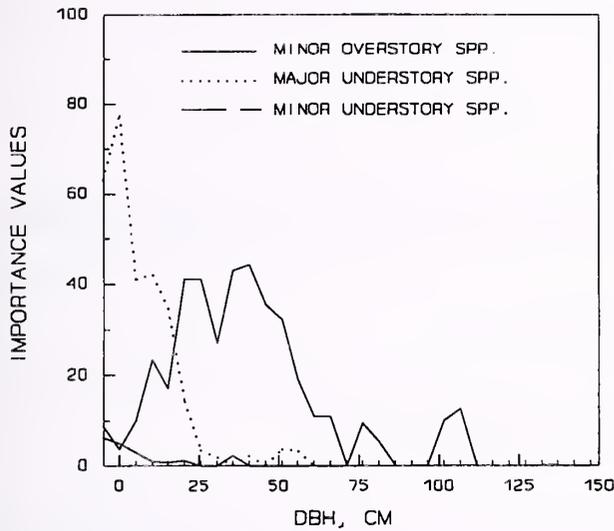


Fig. 5. Importance values, by 5.08-cm dbh classes, for minor overstory species, major understory species, and minor understory species in the Overton Park Forest.

## DISCUSSION AND CONCLUSIONS

The forest in Overton Park falls short of the classical definition of an old-growth or steady-state stage of forest development, partly because of the presence of early-successional trees in the overstory. The forest is currently in a transitional stage between the aggradation stage and the steady-state stage of Bormann and Likens (1979), or between the stem exclusion stage and the old-growth stage of Oliver (1981), but, in the public perception, it is an old-growth forest that clearly merits protection as a significant ecological resource.

Our observations indicate a pronounced under-representation of overstory tree species in the regeneration and sapling classes in the Overton Park Forest. This is consistent with other reports in the literature from both rural and urban old-growth upland hardwood forests. However, our results suggest that non-native plant species play at least some role in this under-representation, particularly for the oaks and yellow-poplar species that characteristically regenerate in canopy gaps. This is significant because the combined importance value of oaks and yellow-poplar is nearly 70% in the mature tree component.

The abundant advance regeneration of *Quercus* observed in this stand is consistent with the ecology of the species; if advance oak regeneration exists prior to the creation of an overstory gap, seedlings can respond and develop if the competition of associated species is not severe (Carvell and Tryon, 1961). However, the absence of oak saplings and poles in the Overton Park Forest suggests that recruitment of advance growth from regeneration to the larger classes is greatly under-represented. This observation is consistent with a hypothesis of gap occupancy by non-native species. If the 5-25 cm classes are relied upon to ensure the presence of oak in this forest, the numbers of oak in the large diameter classes will inexorably decrease, and the importance of oak in the forest as a whole will decline.

Similarly, the seeds of *Liriodendron tulipifera* remain viable on the

forest floor for extended periods of time, and germination commonly occurs upon creation of a gap in the canopy (Fowells, 1975). Putnam *et al.* (1961) noted that if yellow poplar is given one year of growth, it will outgrow most competing species. We think that the total absence of yellow-poplar regeneration in this forest is also consistent with a hypothesis of gap occupancy by aggressive non-native species. Under this hypothesis, yellow poplar germinates successfully, but fails to become established because of excessive competition from the non-native gap invaders. Over the next several decades, we suspect that yellow-poplar will remain an important component of this forest, especially in the large size classes, in view of stand structure coupled with the projected decline of oak. But, in the absence of intervention to promote regeneration, it will ultimately decline as well.

Thus, we conclude that in the Overton Park Forest, three non-native woody perennials: privet, honeysuckle and kudzu, act, along with native grape vines, to impede the gap-phase ecological processes by which intolerant and mid-tolerant components of old-growth forests typically regenerate. Over 90% of the gaps in this forest have greater than 50% cover values for these aggressive species. Kudzu only exists on 2% of the area, but where it occurs it has completely screened, matted and obliterated all other flora. The most widespread problem is with honeysuckle, which has been established via bird dissemination throughout the forest. A native of Japan, honeysuckle has become what Mohlenbrock and Voigt (1959) describe as "an obnoxious weed in some parts of our area." Correll and Johnston (1979) were even less gracious, referring to the species as "a rampant pernicious weed that has endangered natural vegetation from Florida to Texas, north to Massachusetts, New York, Ohio, Indiana, Missouri and Kansas". In canopy gaps within the Overton Park Forest, we observed instances where honeysuckle had bent saplings over with the weight of its foliage, killing them in the process. Native grape vines, with similar growth habit, also contribute to sapling mortality.

The abundance of native understory trees may also play a role in the suppression of overstory tree regeneration. If the 7,079 stems per hectare of understory trees in the regeneration and sapling components were uniformly spaced in a square grid, the intertree spacing would be 1.19 m. The overstory regeneration that can survive beneath this dense canopy must have a high shade tolerance, as indicated by the structural demographics of the species (especially the *Carya-Acer* component and some of the minor understory species) that show less variability of numbers with increasing size components. We speculate that if the two most apparent excluded disturbances in the Overton Park Forest — fire and white-tailed deer — were not excluded, they would probably act to reduce the numbers of stems in the regeneration and sapling classes. This would increase the importance of overstory species in those classes by reducing the density of the tolerant understory. But, in the absence of these natural disturbances, we suspect that the existing understory of tolerant species may be abnormally dense, thereby suppressing natural patterns of development that gave rise to the existing forest.

To establish the recruitment of regeneration into the sapling classes in a more natural fashion, managers should consider the elimination of non-native species. The degree to which intervention should be undertaken that specifically promotes the development of oak and yellow poplar in the gaps is a decision to be made with the input of the State Heritage Program. Restorative interventions that concentrate on physical removal (cutting), chemical removal (use of herbicides), or biological removal (allowing animals such as deer or goats to forage in fenced gaps, or conducting a small-scale burning program) should be considered. Chemical removal is most likely to be effective, but it also is

most likely to draw the greatest public objection.

We think that the oaks will gradually decrease in importance in this forest. The importance of yellow poplar will initially increase, but eventually decline as well. These decreases will be balanced by increases in importance of the mid-tolerant and tolerant species such as maples, hickories and possibly sweetgum. With the exception of sweetgum, it is unlikely that the tolerant species will grow as large as the current overstory. In addition, it is likely that the majority of overstory gaps that occur in the next few decades will be invaded by aggressive, non-native species, thereby accelerating projected declines. In the immediate future, the Overton Park Forest will retain its large-tree character; however, over the long term, the number of large-tree species in this forest will diminish, and they will not be replaced.

This pessimistic projection poses a dilemma for the resource managers of the Overton Park Forest. In the absence of intervention, many of the attributes for which this forest is renowned may ultimately be lost. Research on the enhancement of sapling recruitment through eliminating non-native vegetation, and perhaps also through control of the native understorey vegetation, should be undertaken. The goal should be the formulation of an ecologically-based silvicultural plan by which the large-tree character and functional developmental dynamics of the Overton Park Forest can be maintained in perpetuity.

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## **Chapter 3.**

### **MONITORING AND INVENTORY**



# Radio Telemetry Techniques Applied to the Bog Turtle (*Clemmys muhlenbergii* Schoepff 1801)

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**Abstract:** Simple and effective radio telemetry techniques are described which may be used to define seasonal movements and habitat use by the bog turtle (*Clemmys muhlenbergii* Schoepff 1801) in wetland environments. These techniques, refined from 45 radio tag applications to the bog turtle over a four year period, make use of readily available supplies and commercially prepared equipment. Discussed are: field techniques, types of equipment and an effective method of radio tag attachment that does not permanently mark or disfigure the animal or interfere with its movement.

Pages 69 - 70. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The adaptation of radio telemetry to ecological studies is a technological advance that has greatly affected field investigations of wildlife species. By facilitating multiple recaptures at predetermined intervals of free-roaming individuals, radio telemetry has expanded the potential for long-term observation of many elusive or secretive species. Since wildlife radio telemetry began in the early 1960's, researchers have made use of this technique to develop insights into movements and behaviors for most vertebrate species groups, including the turtles.

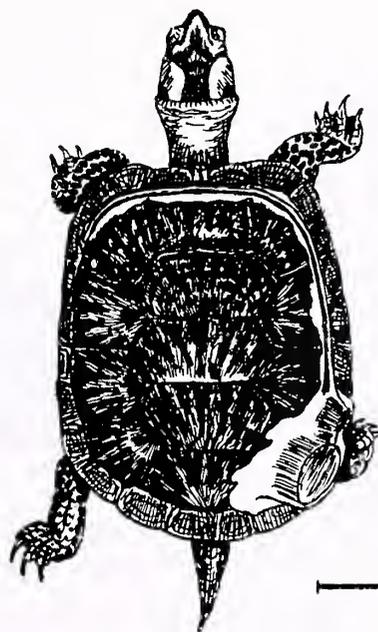
A variety of methods for turtle radio tag attachment have been reported in the literature. Harnesses of surgical rubber tubing have been used to attach radio tags to juvenile green turtles (Ireland, 1980). Radios have been attached to other aquatic turtles by drilling holes in the marginal scutes of the carapace, then inserting cable ties through these holes to secure the unit (Schubauer, 1981). Transmitters have been formed into radio "pills" and then fed to giant tortoises (Swingland and Frazier, 1980). We briefly utilized, with limited success, another method that has been recommended for bog turtles and other small turtles. With this method, the radio tag is directly glued to the carapace with silicone adhesive (Legler, 1979; Larson, 1984).

It has been widely held that in order for an attachment method to be effective, it must be secure, waterproof, non-restrictive, and light weight (Macdonald, 1978; Larson, 1984; Kenward, 1987). Our method meets the aforementioned criteria and, in addition, does not permanently mark or disfigure the specimen. It differs from established methods primarily in the type of adhesive that is used to attach the radio tag to the turtle.

## METHODS

Single-stage, 151 MHz radio tags with a three-month life expectancy were purchased from a commercial supplier (L. L. Electronics, Mahomet, IL). They were powered with lithium batteries and measured 13 mm x 21 mm x 10 mm. Each was fitted with a 15 cm whip antenna. The total weight of the radio tag, including the antenna, was 4.0 g. The radios were attached to bog turtles that ranged in size from 83-100 mm (carapace length) and in weight from 106-150 g. Prior to attachment, the surface of the carapace, beneath the radio tag, was thoroughly dried and lightly sanded. The radio tag was activated,

then dipped in epoxy to make a waterproof seal. The body of the radio tag was then temporarily secured with tape in the vicinity of the fourth costal scute, with the antenna placed toward the anterior of the specimen, between the marginal and the costal scutes (Figure 1). It was then secured along its entire length with five-minute waterproof epoxy (U. G. L., Scranton, PA). In some instances, silicone adhesive was used to fill in gaps around the body of the radio tag to prevent snags. The attachment process required less than an hour; however, turtles were held for an additional 10-12 hours to allow the epoxy to thoroughly cure. A radio attached in this manner could be easily removed by gently prying the unit from the carapace. Helpful supplies and equipment for attachment included a turtle "pedestal" (designed to support the turtle and render it immobile) and a timer. The timer was used to gauge the consistency of the epoxy as it began to set.



**Figure 1:** Dorsal view of bog turtle (*Clemmys muhlenbergii* Schoepff 1801) with radio tag attached. Scale bar = 2 cm.

Study animals were removed from the field when the radio tags were attached unless it appeared that removal might interfere with a stage of the turtle's annual cycle, such as egg laying. In those cases, turtles were either released without a radio tag or were temporarily radio tagged with an alternate method.

This alternate method involved preparation of the radios in advance of anticipated capture by dipping them in a flexible rubber coating (Plasti-Dip, PDI, Inc., St. Paul, MN) to ensure a waterproof seal. The encapsulated radio tag was then attached to the turtle with a strip of black plastic electrical tape wrapped around the carapace and plastron at the bridge. This secured the unit at the body of the transmitter and at the tip of the antenna in approximately the same position as with the epoxy mount. Because of the potential for snags along the anterior-most section of the antenna, with this temporary attachment method, turtles were re-sighted daily and tags were removed and re-attached with epoxy within a week.

In the field, multi-channel, portable radio receivers, and a variety of antennas were used to receive the signals produced by the radio tags. Five or three element yagi antennas were used for the initial bearings, and final capture was accomplished with a miniature loop antenna, valuable for its compact size (8 cm x 13 cm x 3cm).

The range for these radio tags varied depending on the topography and the position of the turtle, but a maximum of 200 m on level and open ground was observed. The maximum range for a properly operating unit dropped as low as 50 m in dense vegetation or when the animal was submerged.

## RESULTS AND DISCUSSION

If the 7% weight limit ratio (radio tag to turtle) suggested for aquatic turtles by Schubauer, *et al.* (1977) is applied to bog turtles, the maximum package weight for an average sized adult (130 g) would be 9.1 g. Brander and Cochran (1971), and Cochran (1980) suggest a 4-6% limit for animals above 50 g, limiting the radio tag to 5.2-7.8 g. Macdonald and Amlaner (1980) have suggested a more restrictive weight limit ratio of 3-5%; limiting the transmitter package to 3.9-6.5 g for a 130 g specimen.

We successfully met these more restrictive guidelines in all applications by using a radio tag with a three month life expectancy and by attaching these tags with epoxy. Radio tags, and the epoxy used to attach them, weighed an average of 5.7 g, and ranged from 4.5-7.5 g (4.2-5.0%).

The effectiveness of the epoxy mount in the field was outstanding. In 27 applications, there were no instances where the epoxy failed to keep the radio tag attached to the turtle within the three month period that the radio was activated. There were four cases where the unit remained attached throughout the winter (9 months). Furthermore, there were two instances where a predator destroyed the radio tag, but did not remove it from the turtle. We did encounter occasional failures of the radios themselves. At a savings of 75% of the cost of a new radio tag, recovered tags were refitted with a new battery. However, of the six instances (22%) where the units malfunctioned, all were rebuilt radios. The defective radios lasted an average of 60% of their projected life span, but all of the new radios lasted for their projected life span. We recommend that these units not be rebuilt if full life expectancy is desired.

With a temporary method of attachment, making use of black tape, we observed one failure in five applications. In this instance, the signal

was lost and the turtle was recovered without a radio tag, indicating that it was probably pulled off the turtle and destroyed.

There were no apparent behavioral changes or movement restrictions due to either method of attachment. Individuals were successfully re-fitted with radio tags as many as six times during the four year study period. Telemetered turtles were observed to mate, forage and nest, though overland migrations of as far as 750 m were observed.

## ACKNOWLEDGMENTS

Funding for this study was provided by voluntary donations through New York's Return-A-Gift-To-Wildlife Program and by contributions from the New York Zoological Society. The authors wish to thank Louis Luksander (L. L. Electronics, Urbana, IL) for constructing the radio tags. Thanks are also extended to Mike Stickney and Alan Andresen for their assistance. Use of brand names and references to equipment suppliers does not constitute endorsement by the New York State Department of Environmental Conservation or by the New York Zoological Society.

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# Response to Conspecific, Roadside Playback Recordings: An Index of Red-shouldered Hawk Breeding Density

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**Abstract:** Taped calls were broadcast along roadside routes in north central New York to aid in detection of red-shouldered hawks (*Buteo lineatus*). In 1986, three 8 km routes were established along woodland roads. Calls were broadcast every 0.8 km along each route and were replicated 12-15 times at 6-8 day intervals from 15 March through 15 July. The same procedure was repeated in 1987. There were 80 observations in 1986 and 63 in 1987. The mean number of observations per route, pooled for all routes and years, was 1.74. Phenological and behavioral differences in response were detected throughout the study periods. Conditional probabilities of detection and estimates of per cent area occupied (AO) are presented. The coefficient of correlation between %AO and observed nest density within the effective study area around survey routes is 0.981 ( $p < 0.001$ ). The technique has potential for state-wide application to establish an efficient index of red-shouldered hawk abundance that will aid in the development of a long term management strategy for this threatened species. Guidelines to implementing this technique are also presented.

Pages 71 - 76. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

An important topic of interest to wildlife managers is the question of whether particular areas and their associated habitats support species of special concern. Changes in occupancy or relative abundance may dictate the course of long-term management strategies. For breeding birds, many estimators of abundance are available either as counts or indices (Emlen, 1971; Burnham *et al.*, 1980; Ralph and Scott, 1981; Seber, 1982; Christman, 1984).

Territory or spot-mapping techniques (Williams, 1936) are frequently used to estimate avian population size, but they require extensive field efforts and are not suitable for large land tracts (Franzreb, 1981). A similar set of problems exists for mark-recapture models. Sample plot methods (Bond, 1957; Anderson and Shugart, 1974) record birds within a fixed distance, but species with low detectability are often underreported (Edwards *et al.*, 1981). Some transect methods (Emlen, 1977; see also Shields, 1979 for review) and point counts (Reynolds *et al.*, 1980) can correct for unequal detectability but these require that distances to detected individuals be known. This information is difficult to obtain for some species such as rails and woodland raptors.

If absolute population size is not the primary concern, a roadside census can be used to estimate abundance (Kendeigh, 1944; Howell, 1951; Bystrak, 1981). The drawback of this technique is that it requires that all individuals must be equally detectable (Dawson, 1981). Variation in singing activity during the breeding season is well known (von Haartman, 1956; Nolan, 1978). Best (1981) suggested that singing may dramatically decline after pairing in species where song primarily functions in mate attraction. Wilson and Bart (1985) demonstrated that random phenological differences may cause an error of up to 25% in estimating the relative density of house wrens (*Troglodytes aedon*). Even birds whose song functions largely as territory announcement show seasonal changes (Nowicki, 1974; Rosenfield *et al.*, 1988). Therefore a useful abundance estimator is lacking where: 1) distance cannot be accurately measured, 2) the probability of detection is low, 3) individuals cannot be distinguished, and 4) the sampling area is large.

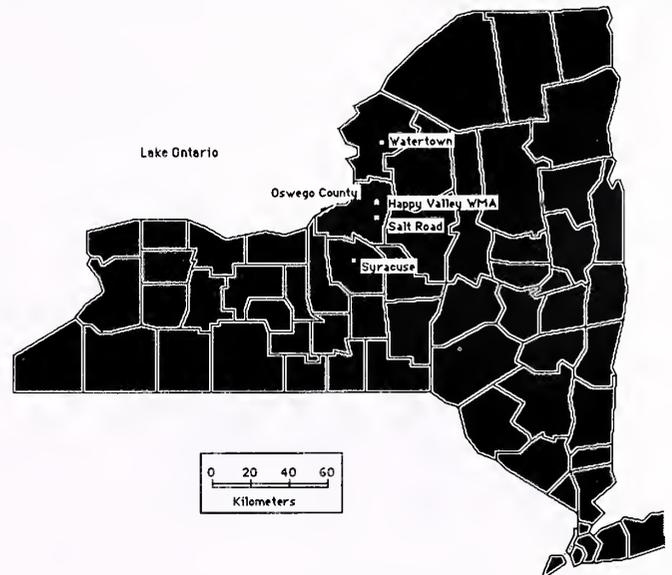
The area-occupied technique developed by Geissler and Fuller (1987) estimates the probability of detecting an animal, based upon repeated point transects, and it applies the Horvitz-Thompson

estimator to estimate abundance (Horvitz and Thompson, 1952; Cochran, 1977: 259-261). With many birds, especially raptors, detectability can be enhanced by broadcasting taped conspecific calls (Johnson *et al.*, 1981; Fuller and Mosher, 1981; Rosenfield *et al.*, 1985; Fuller and Mosher, 1987).

The present study evaluates the area-occupied technique's utility in estimating the abundance of the red-shouldered hawk (*Buteo lineatus*), a secretive woodland raptor that is currently listed as a threatened or special concern species in several northeastern states, including New York. As a measure of the area-occupied technique's effectiveness, we attempted to locate all active nesting territories within a defined area around the study transects.

## STUDY AREA

In 1985, three pilot survey routes were established in northern and eastern Oswego County, New York. Initial surveys were conducted



**Figure 1.** Locations of the two red-shouldered hawk study areas in Oswego County, New York.

from 15 April until 15 July 1985. Route locations were finalized in 1986, and results presented herein are from these routes only, for the years 1986 and 1987.

Two of the north-south-oriented survey routes (Churchill Road and Happy Valley Road) are located along unimproved roads entirely within the Happy Valley Wildlife Management Area (HVWMA), a state-owned tract in north central Oswego County, New York (Fig. 1). This glaciated, 3500 ha area is relatively flat, with elevations ranging from 200 to 300 m. There are 2.0 km/km<sup>2</sup> of drivable roads (April - October) in and around the area. HVWMA is characterized by second-growth forests of northern hardwoods and mixed hardwood-conifer stands. The dominant tree species is red maple (*Acer rubrum*) followed by American beech (*Fagus grandifolia*), black cherry (*Prunus serotina*), eastern hemlock (*Tsuga canadensis*), yellow birch (*Betula alleghaniensis*), and sugar maple (*A. saccharum*). Small (2-4 ha) stands of aspens (*Populus tremuloides* and *P. grandidentata*), conifer plantations, low-lying wooded swamps, alder (*Alnus rugosa*) stands, and old fields are interspersed throughout the area. A more complete description of HVWMA is given by Parris (1986).

A third north-south route (Salt Road) is located 10 km south of HVWMA. The southernmost terminus of this route is 2.2 km north of the village of Constantia. The area along the route contains sparse residential development and is mostly within two large private holdings. Vegetation and topography are similar to HVWMA.

## METHODS AND MATERIALS

Surveys were conducted from 15 March through 15 July along all three routes in 1986 and 1987. The routes were 8.0 km in length with 11 listening stations spaced at approximately 0.8 km intervals (Fig.2). Routes were surveyed every six to eight days between 0800 and 1100 hours. At each station, a conspecific tape recording (taken from Kellogg and Allen, 1959) was broadcast four times for 15 seconds at 45 second intervals. Observers would wait one minute prior to broadcast and four minutes post-broadcast. The number of responses, method of detection (aural or visual), direction, approximate distance when first detected, and behavioral observations were recorded for the entire listening period. A Sanyo M9970 "Boombox" cassette player was used for playback (minimum output: 100 dB at 1 m). Speakers were rotated 180° after the first two call series. Surveys were conducted under relatively still (0.0 - 8.0 km/hr wind velocity), under precipitation-free conditions. The direction of travel along each route was also alternated on each survey.

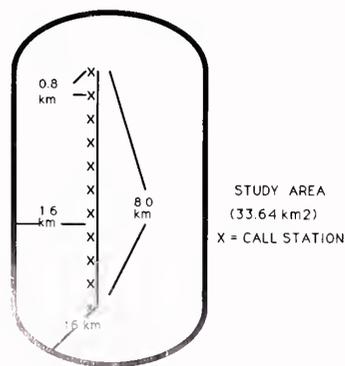


Figure 2. Diagram of one survey route and its associated study area.

An attempt was made during each survey year to locate all active nests within 1.6 km of either side of each route with a 1.6 km arc at either end (Fuller, 1984), defining an effective sampling area of 33.64 km<sup>2</sup> (Fig.2). All suitable habitat within these areas was searched during leafless periods for large stick nests. Nests were monitored during March and April for signs of use such as presence of birds, nest decoration (esp. eastern hemlock sprigs), and whitewash. Areas around survey stations with consistent responses were intensively searched.

Presence (X) or absence (O) of red-shouldered hawks at each listening station for each visit was tabulated for each route for both years. The probability of detecting a hawk at each station was calculated by dividing the number of X's at a station by the number of visits (m). To avoid biased estimates, only those detections made after it was established that a hawk occurs near a stop were used to calculate the estimate (i.e., if hawks were detected on three of ten visits to a station, only the last two were used to calculate probabilities). The probabilities from each stop were averaged to give the probability of detecting a hawk at any stop with one visit (PD1). To determine the probability of detection for m visits (PDM), the equation  $PDM = 1 - (1 - PD1)^m$  was used (Geissler and Fuller, 1987).

The proportion of each study area occupied by red-shouldered hawks (AO) was determined, using PDM to account for the variability of detection over the sampling period by:

$$AO = \frac{\text{number of stations with detections/PDM}}{\text{number of stations}}$$

Multiple replications of each survey allowed for adjusting AO estimates by using bootstrap estimates. Variance estimates and 95% confidence intervals were calculated with 1001 bootstrap samples. The MS-DOS computer program that was used to perform these calculations is available from the U.S. Fish and Wildlife Service (USFWS) in Patuxent, Maryland. This program provides unadjusted estimates and adjusted median and mean estimates of PD1 and AO. The relationship between actual nest density and AO was determined by simple linear regression.

## RESULTS AND DISCUSSION

We conducted 41 surveys between 15 March - 15 July in 1986 and 1987. The mean number of detections per route (all routes pooled) was 1.46 and 1.40 in 1986 and 1987 respectively (Table 1). More than one bird responded at 33.3% of the calling stations in 1986 and 14.5% of the stations in 1987. Iverson (1987) conducted similar surveys in Indiana and reported a pooled detection rate of 1.50 from five survey routes.

During both survey years, 41.3% of the responses occurred after the first call series began (Table 2). As the broadcast period progressed, the per cent initial response following each subsequent call series decreased. Nearly 10% of the initial detections occurred during the one minute pre-broadcast period and can be a measure of detection rate without the aid of broadcasting conspecific calls. This indicates a potential increase of up to 90.0% by employing taped calls. Fuller (1984) observed a similar (84.0%) increase in detection rate.

Combining data from both years, 70.6% of the responses were vocal only, while 5.6% were visual only (Table 3). The remainder of the responses (23.8%) included both a visual and aural component. These results are similar to those obtained by Balding and Dibble (1984), who reported that red-shouldered hawks in Wisconsin responded vocally to conspecific call broadcasts in 75% of their observations. Of the birds that responded vocally in our study, 68.8% vocalized during

or following two or more call series, while 28.2% called throughout the broadcast period. This may be related to the proximity of the call station to an important feature within a bird's territory, such as a nest site or prime foraging area.

**Table 1.** Summary of red-shouldered hawk survey route observations and detections in 1986 and 1987.

Route Name	No. Surveys	No. Obs.	Obs. Rate <sup>1</sup>	No. Det.	Detection Rate <sup>2</sup>
<u>1986</u>					
Churchill Rd.	13	22	1.69	18	1.22
Happy Valley Rd.	14	25	1.79	19	1.35
Salt Rd.	14	33	2.36	23	1.64
All Routes	41	80	1.95	60	1.46
<u>1987</u>					
Churchill Rd.	14	24	1.71	21	1.50
Happy Valley Rd.	12	14	1.17	12	1.00
Salt Rd.	15	25	1.67	22	1.46
All Routes	41	63	1.53	55	1.34
<i>Routes &amp; Years Pooled</i>					
	82	143	1.74	115	1.40

1. No. observations/No. times route was conducted. Observations are the total number of red-shouldered hawks seen or heard along a survey route.

2. No. detections/No. times route was conducted. Detections are the total number of stations along a survey route in which one or more hawks were seen or heard.

When hawks were seen following a broadcast, the typical behavioral pattern consisted of a low glide over or near the observer followed by a slow, circling climb. In most instances, birds disappeared from view by the end of the listening period. On two occasions, hawks made more than one low pass over the observer before vacating the immediate area. Balding and Dibble (1984) report similar observations.

Close proximity of an active nest to a call station may influence response behavior. Two nests in our study were located within 50 m of a call station. Although only a single response was recorded at the station closest to one of these nests (active both years), it is believed that the birds associated with this nest were those that responded at the station on either side of the "nest" station. At the other nest, responses were detected often prior to incubation and post-hatch. Other researchers (Rosenfield *et al.*, 1988; J. Mosher pers. comm.) have reported a decrease in response during incubation.

Phenological differences in response rate were observed throughout the breeding season (Fig. 3). Response rate was nearly twice as great during the period from arrival to immediately prior to egg-laying. Possible explanations include territorial boundary adjustments occurring between neighboring pairs, newly-mated pairs attempting to usurp portions of an established pair's territory, and the presence of unmated birds (floaters) within newly established territories of mated pairs. Overall, we observed no significant decrease in response frequency during incubation, although birds on or near their nest site responded less often during this time period.

In several instances, recent fledglings responded vocally to playback recordings. Their vocalizations might have confounded the detection rates during the later surveys. Consequently, the detection probabilities

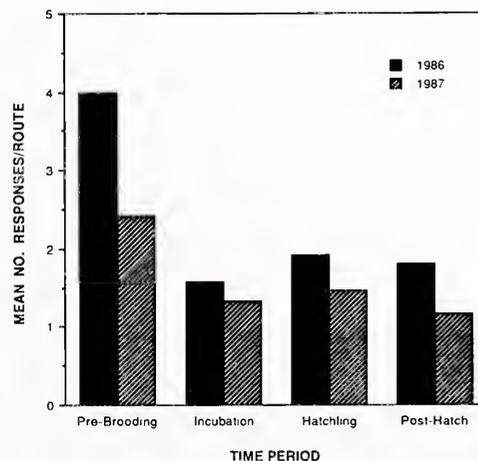
and %AO were calculated using only response data collected from the beginning of incubation until the earliest recorded date of fledging (1 July). Table 4 presents unadjusted PD1 and PDM values calculated with this reduced data set.

**Table 2.** Distribution of first detections of red-shouldered hawks over the entire listening period by survey route and year.

Route Name	Pre-Broadcast	<i>Call Series Prior To Detection</i>			
		1	2	3	4
<u>1986</u>					
Churchill Rd.	1	8	8	4	1
Happy Valley Rd.	2	6	8	4	5
Salt Rd.	8	16	4	4	1
Total	11	30	20	12	7
%	13.8	37.5	25.0	15.0	8.7
<u>1987</u>					
Churchill Rd.	2	12	4	2	4
Happy Valley Rd.	0	7	4	2	1
Salt Rd.	1	10	5	6	3
Total	3	29	13	10	8
%	4.8	46.0	20.6	15.9	12.7
<i>Years &amp; Routes Pooled</i>					
Total	14	59	33	22	15
%	9.8	41.3	23.1	15.4	10.5

Estimates of %AO for the three survey routes for both years is given in Table 5. The adjusted estimates were calculated using bootstrap samples. Confidence intervals were wide, indicating a large variance associated with bootstrap samples. Minimum nest density for all study areas is also given in Table 5. The coefficient of correlation between %AO and nest density, with nest density as the independent variable, is 0.981 ( $p < 0.001$ ). This near one-to-one correlation should not be taken to suggest that %AO can be directly equated with nest density; rather, %AO may be considered a useful index of nest density. It is possible that not all nests were located on the study areas and that other habitat types may produce different results. Further research is needed in different habitats and regions.

Table 6 presents the number of heterospecific vocal responses at the



**Figure 3.** Mean number of responses per red-shouldered hawk survey route by period of the nesting season in 1986 and 1987.

902 visits to calling stations during this study. Blue jays (*Cyanocitta cristata*) responded at 4.8% of the stations, typically with an imitation of red-shouldered hawk vocalizations. Some experience is necessary to determine audible differences between these two birds, but blue jays rarely imitate red-shouldered hawk calls for long periods and invariably switch to other vocalizations in their repertoire. Although Balding and Dibble (1984) report a high response rate of red-tailed hawks (*Buteo jamaicensis*) to red-shouldered hawk calls, the response rate was low in our study. No active nests of red-tailed hawks were found on the study areas, suggesting that the low response rate was due to the low density of this species. Competition between these two species has been suggested by Craighead and Craighead (1956) and Bednarz and Dinsmore (1981).

**Table 3.** Detection mode of red-shouldered hawks by survey route and year.

Route Name	Means of Detection			Total
	Aural Only	Aural & Visual	Visual Only	
<u>1986</u>				
Churchill Rd.	13	6	3	22
Happy Valley Rd.	17	6	2	25
Salt Rd.	26	6	1	33
Total	56	18	6	80
%	70.0	22.5	7.5	100.0
<u>1987</u>				
Churchill Rd.	17	6	1	24
Happy Valley Rd.	11	3	0	25
Salt Rd.	17	7	1	14
Total	45	16	2	63
%	71.4	25.4	3.2	100.0
<u>Routes &amp; Years Pooled</u>				
Total	101	34	8	143
%	70.6	23.8	5.6	100.0

## MANAGEMENT CONSIDERATIONS

Abundance of woodland raptors is often underestimated by most census techniques because of low detectability. By combining a series of roadside censuses with broadcasts of taped calls, it is possible to increase detectability and obtain more useful indices of abundance. Rogers and Dauber (1977) significantly increased detections of red-shouldered hawks along roadside survey routes using a commercially available hawk call. Iverson (1987) detected sharp-shinned hawks (*Accipiter striatus*) on four of twelve transects in Indiana while broadcasting calls of great horned owl (*Bubo virginianus*). Prior to his study, only five published nesting records existed in that state. Fuller (1984) detected broad-winged hawks (*Buteo platypterus*) only after broadcasting vocalizations.

Fuller and Mosher (1987) suggest that great horned owl vocalizations are as effective as conspecific calls for eliciting responses from Cooper's hawks (*Accipiter cooperii*) and red-shouldered hawks. We experienced limited success with owl tapes during a pilot study and found conspecific calls more effective; nevertheless, more rigorous testing with upgraded equipment is warranted. Owl tapes are most useful when a simultaneous survey of all woodland raptors is desired. For example, Iverson (1987) in

Indiana, and H. Devaul (pers. comm.) in Maine, used great horned owl calls to conduct general surveys of forest-dwelling hawks. Broadcasting a sequence of calls of a variety of target species is time-consuming and may be confounded by interactions between species.

Roadside censuses have the advantages of relative ease in use and applicability over large sampling areas. Conducting one or two surveys with an experienced person provides all the training time required for new observers. One must be able to recognize the target species by sight and by its call and to distinguish between blue jays and red-shouldered hawks by call, where the two species overlap in range.

Many factors may introduce bias into the counts. Dawson (1981) and Fuller and Mosher (1987) review various sources of variability and error and provide suggestions to reduce them. We recommend that surveys be limited to that portion of the breeding season following egg-laying and prior to fledging. Because of daily variation in activity in many raptors during the breeding season (Fuller, 1979), we

**Table 4.** Summary of red-shouldered hawk survey route statistics for 1986 and 1987.

Year/Route	m <sup>1</sup>	Detection Rate	p <sup>2</sup>	Pd <sup>13</sup>	Pdm <sup>3</sup>
<u>1986</u>					
Churchill Rd.	11	1.35	63.6	0.405	0.997
Happy Valley Rd.	11	1.40	50.0	0.159	0.852
Salt Rd.	10	1.64	81.8	0.159	0.823
<u>1987</u>					
Churchill Rd.	11	1.45	63.6	0.155	0.844
Happy Valley Rd.	10	1.00	40.0	0.183	0.868
Salt Road	11	1.55	72.7	0.126	0.773

1. Number of survey replications used for analysis.

2. Percentage of call stations with detections over survey period.

3. Unadjusted estimate.

**Table 5.** Per cent area occupied and nest densities of red-shouldered hawks at all study areas in 1986 and 1987.

Year/Route	% AO Unadjusted estimate (95% CI)	% AO Adjusted median est.	Nest Density nests/km <sup>2</sup>
<u>1986</u>			
Churchill Rd.	40.1 (0.0 - 87.5)	40.1	0.089
Happy Valley Rd.	74.7 (5.0 - 100.0)	77.6	0.119
Salt Rd.	99.4 (3.8 - 100.0)	103.1	0.148
<u>1987</u>			
Churchill Rd.	75.3 (22.3 - 100.0)	78.3	0.119
Happy Valley Rd.	52.4 (0.0 - 100.0)	55.8	0.089
Salt Rd.	94.1 (38.0 - 100.0)	98.1	0.148

recommend restricting surveys to one time period of the day. We suggest the period from sunrise until 1100 hours in Northeastern North America. Robbins (1981) determined that mornings were the best time to count diurnal raptors. The effect of weather variables on both birds and observers is difficult to assess. We suggest limiting counts to periods free of precipitation with wind speeds less than 8.0 km/hr. The Beaufort scale of wind speed and corresponding field indicators provide a useful guide (Erskine, 1978).

**Table 6.** Heterospecific vocal responses to broadcasts of red-shouldered hawk vocalizations.

Species	No. Responses
Blue jay	43
<i>Cyanocitta cristata</i>	
Broad-winged hawk	9
<i>Buteo platypterus</i>	
American crow	8
<i>Corvus brachyrhynchos</i>	
Red-tailed hawk	2
<i>Buteo jamaicensis</i>	
Yellow-bellied sapsucker	2
<i>Sphyrapicus varius</i>	
Sharp-shinned hawk	1
<i>Accipiter striatus</i>	
Northern goshawk	1
<i>Accipiter gentilis</i>	

Multiple replications allow the estimation of a conditional probability of detection (Pd1) for a species. Once Pd1 values have been generated for a study area, the number of replications necessary for an effective survey can be greatly reduced. The documentation included with the USFWS %AO computer program provides results of a series of simulations of surveys with various levels of effort and suggests the approximate number of visits to call stations required with different Pd1 values. One year of intensive field effort in several habitat types within a region will allow for extensive work in following years.

Statewide monitoring programs can be established using these techniques, and AO values can then be compared across years and habitats. Changes in relative abundance can be detected and used to formulate or modify management strategies and predict or document regional declines. For example, the New York State Breeding Bird Atlas project (Andrle and Carroll, 1988) documents regions in the state where woodland raptors occur at varying levels of abundance, and this book is useful for identifying areas where pilot studies should be conducted. Information obtained from these studies and continuing monitoring programs will be a valuable addition to our ability to predict population trends and to enhance effective management of the more secretive members of our avifauna.

#### ACKNOWLEDGMENTS

Partial funding was provided by the Research Foundation of the State University of New York, College of Environmental Science and Forestry. We thank the many individuals who participated in collecting field data, particularly B. Woodill, K. Clark, M. Kolozsvary, M. Ingraldi, A. Fisher, and J. Buck. H. Devaul, K. Titus, and J. Mosher provided helpful suggestions to this study. We thank P. Steblein, D.

Gefell and an anonymous reviewer for critical review of early drafts of the manuscript.

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# The Use of Departure Analysis to Protect Rare Species and Significant Habitats: An Example

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**Abstract:** Land management actions outside of units of the National Park System can affect park resource-related values. These effects become increasingly likely and consequential as demands on scarce resources become more intense. National Park Service (NPS) land managers must attempt to analyze possible impacts to park resources even though the action of concern may not be within their control. A methodology, called Departure Analysis, is used to evaluate external water development decisions that may affect natural resources within units of the NPS. The approach, described with an example, incorporates several studies which, in aggregate, examine the effects of modifications to surface and ground water hydrologic regimes upon a variety of natural values.

Pages 77-81. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Departure Analysis is a methodology for assessing impacts on water-related resource values (i.e., natural or cultural characteristics of a park, monument or other NPS unit, which are dependent upon or influenced by water) that are likely to result from on-site and off-site water development decisions. The methodology will be described first in overview and then through an example. The example is taken from an application in a National Park known for its rare species and significant habitats.

## LAND USE MANAGEMENT AND EFFECTS UPON RESOURCE VALUES

As demands upon limited resources increase, the balance between resource utilization and the protection of rare species and significant habitats becomes difficult to maintain. Resource scarcity limits a land manager's opportunities to achieve optimum resource allocations. Because of this scarcity and the inherent complexity of ecosystems, a decision in one resource area is likely to affect other resources which, in turn, affect future land management decisions. As a result, few land management decisions can any longer be made in isolation or from a single resource perspective.

One may view timber harvest and its effect upon the aesthetic values of a viewshed as a case in point. Decisions regarding timber harvest that could, at one time, be based almost entirely on silvicultural and engineering criteria, must now include many other resource considerations, not the least of which is visual impact. The once-common rectangular harvest units are not as frequently encountered nowadays. Hidden terrain, where such harvest units would not be easily seen, and, thus, pose few visual problems, has become scarce over time, necessitating greater consideration of visual resource values in timber harvest planning.

Not all the resource interactions associated with a land management decision are as obvious as that of timber harvest and scenic values. Those that relate to water development and use are particularly complex and difficult to assess. Complexities abound because water, so essential to man's well-being, is a "flowing resource" and one whose parameters are variable in time and space. Decisions regarding the development and use of water resources demand great concern and study of on-site and off-site effects. The numerous complex engineering and economic issues associated with water development have been made even more challenging by requirements demanding benefit/cost analyses and environmental protection regulations. For

example, the rapid growth experienced by our nation's population centers in recent decades has required additional facilities for water storage and transmission. When this need is met by reservoir construction, a series of interactions occurs that precipitates environmental effects having both long- and short-term consequences. When a reservoir is constructed inundating upland, aquatic and riparian ecosystems, a process of environmental change is initiated that may continue for years or decades.

On-site changes are obvious: standing water occupies a valley where once a stream flowed; streamside or river-based recreation is replaced by flat-water recreation; visual aesthetics are dominated by a reservoir and its shoreline rather than a river and its valley; local economic centers change internally or arise at new locations; changes in fluvial processes begin at the dam outlet (Petts and Lewin, 1979) and at the reservoir's influent points; and the fish species mix shifts to favor those that can best survive in the reservoir (Johnson and Carothers, 1987).

Other changes occur off-site which may be less obvious, but certainly no less consequential. The hydrologic regime of the watercourse downstream from a dam will suddenly change. Stream flows will either become less variable or, in the case of a peaking power facility, much more so due to flow regulation. Organic matter and other particulates that had been flowing through the system are likely to be trapped by the reservoir. Thus, the physical and chemical constituents of the water released below the dam may be quite different from those of the water entering the reservoir. For example, coarse organic matter may be trapped by a reservoir, broken down, then passed in finely divided or dissolved form. Benthic organisms downstream, that utilize coarse organic matter, may be put at a competitive disadvantage and replaced by others better adapted to utilize the finer organic forms.

Cummins (1979, p. 19-21) points out that "[a]lterations that would change the patterns of the annual hydrograph, such as low-flow augmentation or flood control, or change in light or thermal regimes and/or quantity and quality of POM-DOM [particulate organic matter - dissolved organic matter] inputs, can be predicted to produce major restructuring of running-water communities. In general, impoundment of smaller rivers and streams creates certain large-river characteristics. For example, a dam on a mid-sized river with hypolimnetic release might produce thermal conditions more typical of the headwaters, but in an autotrophic setting and with greatly altered POM fluxes."

In such situations, the new hydrologic regime may stress native fish or other higher organisms that are adapted to the existing regime. When customary food sources become unavailable or physical

## ASSESSMENT OF RESOURCE IMPACTS FROM MANAGEMENT

conditions become unfavorable, these organisms may become unsuccessful in reproduction or may fail in competition for food and living space against introduced (non-native) fish species. In other words, "...prior to alteration by man, many rivers were heterogeneous, but relatively predictable, environments. Organisms were able to survive and, in fact, maximize diversity, not in spite of, but because of the temporal and spatial heterogeneity of their environment. Not only do stream organisms respond in predictive ways but shoreline and estuarine communities are dependent upon these pulse events" (Ward and Stanford, 1979, p. 384).

Biotic impacts from reservoir construction were demonstrated by Williams and Winget (1979, p. 373) at Soldier Creek dam in Utah. With the closing of the dam they noted the elimination of "...those species [of macroinvertebrates] specialized to exist on clean rock surfaces." With regard to fish, Minckley and Mefee (1987, p. 104) observed that "[i]f floods are curtailed by damming, nonnative fishes typically increase in numbers to approach 100 per cent of the fauna."

Similarly, the balance between sediment and kinetic energy found in the fluvial system before construction of the dam begins to disequilibrate as sediment is trapped by the reservoir. As observed by Williams and Wolman (1984, p. 60) "...the construction of dams on alluvial channels, by altering the flow and sediment regimen, is likely to result in a number of hydrologic and morphologic changes downstream..." "Many large dams trap virtually all (about 99 percent) of the incoming sediment. The erosion of sediment immediately downstream from the dam, therefore is not accompanied by replacement..." "Hundreds of kilometers of river distance downstream from a dam may be required before a river regains, by boundary erosion and tributary sediment contributions, the same annual suspended load or sediment concentration that it transported at any given site prior to dam construction."

The dimensions, pattern and size of bed and bank material of an alluvial (self-formed and self-maintained) river are produced by the sediment and water flow regimes historically imposed by its watershed. Changes in the above characteristics are likely to result from alterations in the water and sediment discharge regimes produced by the existence of upstream water impoundment.

Reservoir operations, too, produce modifications of the physical characteristics of downstream channels as new equilibria, reflecting post-construction sediment and water flow regimes, are achieved. This effect has been demonstrated on a large scale in the Green River of Colorado and Utah (Andrews, 1986). Temperature and chemical quality of water may also change as a result of reservoir operations, making previously suitable stream reaches uninhabitable to fish and other aquatic organisms (Holden, et al., 1979).

Other land management actions may have impacts upon water-related resource values. The diversion of water away from normal water courses, as would occur for irrigation, may similarly alter a river's dynamic equilibrium. The result, a diminution in a river's competency to transport sediment, may produce accelerated bar formation, and this, in turn, may produce favorable habitat for vegetation encroachment (Northrop, 1965) and consequent increases in water consumption by emergent plants.

The reverse of this process can occur with urbanization and impervious eradication. There is a characteristic increase in impervious surfaces in urban areas (Park, 1977), and decreases in plantapotranspiration due to land clearing can increase surface water yield and decrease detention time (Troendle and King, 1987), thereby, increasing kinetic energy in downstream channels.

Cahn (1978), nine years after the passage of the Environmental Policy Act of 1969, pointed out the fact that thousands of environmental impact statements had been filed and many projects cancelled due to the requirements of the Act. While the need for adequate assessment of environmental impacts required by this and other acts may seem burdensome and costly, the importance of the process has been well demonstrated. It seems likely, therefore, that requirements for impact assessment and mitigation will remain with us.

As competition for scarce resources grows, management decisions increasingly carry a potential for producing economic or environmental costs that must be borne by disparate segments of society or even by future generations. Consequently, land managers need a means by which to examine the trade-offs that will occur as direct consequences of their decisions. Furthermore, methods employed should be statistically rigorous, technically valid and provide acceptable managerial certainty. As things stand, however, a manager would have to be truly omniscient to fully understand all the impacts of any given decision. The current state of the science does not offer a complete understanding of short-term ecosystem level effects, let alone long-term interactions.

## DEPARTURE ANALYSIS

The foregoing discussion notwithstanding, predictions can be made about the nature and magnitude of impacts on specific facets of an ecosystem after environmental variables have been altered. For example, the effects of alterations on hydrologic regimes, upon such things as channel morphology and visual setting, can, in some instances, be estimated within defined limits of acceptable error, using computer models or other algorithms.

The NPS has developed an approach for assessing the response of certain facets of the ecosystem, called water-related resource attributes, to man-imposed stress through alterations in hydrologic regimes. This approach, called Departure Analysis, is based upon examination of those water-related resource attributes that lend themselves to analysis and prediction, and which are affected by changes in surface or ground water regimes.

Departure Analysis describes the complex response of water-related resource attributes indirectly. The present condition of dependent variables associated with the selected water-related resource attribute is first characterized. Then, using simulation techniques, responses of the dependent variables to changes in one or more aspects of the hydrologic regime are simulated.

By way of explanation, consider a case in which it is felt that a water management decision might affect rare indigenous fish. The water-related resource attribute of interest in this example is the indigenous fish species itself. Responses of fish to changes in hydrologic regime are, however, difficult to predict directly. An alternative approach is to predict changes in those physical and/or biotic components of the aquatic environment most closely related to the wellbeing of fish populations. Such a component, or dependent variable, could be fish habitat as measured in units of square feet of channel bed area covered by a minimum depth of water during specified life stages of the fish (times of the year). This variable can be measured in an objective manner and reasonably predicted for hypothetical regimes of discharge.

For each hypothetical flow regime, Departure Analysis will estimate resultant habitat area. This value can then be compared to the habitat

area measured under existing hydrologic conditions. The comparison demonstrates departures, if any, from the existing condition of the dependent variable which, in turn, can be interpreted in terms of the water-related resource attribute (in this case, indigenous fish).

This approach has much in common with one used by the USDI Fish and Wildlife Service (FWS). Using a similar conceptual approach, the widely used FWS Instream Flow Incremental Methodology (IFIM) (Bovee, 1982) examines, among other things, potential impacts to fisheries from proposed water developments and/or operations. Departure Analysis, however, addresses a broader spectrum of resource values and impacts, attempting to consider all forms of potential water development. Furthermore, Departure Analysis attempts to identify impairment (human induced change, whether beneficial or harmful, is defined as impairment to natural processes) while IFIM strives to identify trade-offs and facilitate the negotiation of agreements concerning dam operations, minimum releases, etc.

A nomenclature loosely borrowed from Catastrophe Theory (Graf, 1979) is used in connection with Departure Analysis. An understanding of the terms may be gained through the fish example used above. The fish represents the "water-related resource attribute" of interest. The habitat area represents the "response variable" which can be estimated using existing scientific tools. Other response variables that might, of necessity, be used, are water temperature during critical time periods, water quality and available food sources for the fish. The flow regime, defined as quantity, quality or timing, represents the "control variable" of interest.

In summary, the Departure Analysis describes the changes (departures), from existing conditions, of certain response variables (those that can be associated with changes in selected water-related resource attributes) due to incremental changes in water-related control variables. The departures in response variables are, in turn, evaluated to estimate the probable effect upon the water-related resource attribute of interest.

## ZION NATIONAL PARK: AN EXAMPLE

### The Park Setting:

Zion National Park (Zion) is located in southwest Utah. It is characterized by wind and water eroded sedimentary deposits formed beneath and along the margins of an inland sea that once covered this area. Colorful mesas, pinnacles and narrow canyons, incised hundreds of feet into solid rock, typify Zion. The principal water resources of the park are springs and seeps, and the main stems and tributaries of the East and North Forks of the Virgin River. The rivers, their principal tributaries and many springs and seeps are perennial and of crucial importance to wildlife and visitors alike.

Because of its geographic location, the park is mostly semi-arid. The lush bottomlands maintained by runoff and spring flow are important islands of vegetation surrounded by virtual desert. Surprisingly, true swamps can be found in Zion along canyon walls where large springs flow out of water-bearing rock strata.

Unique assemblages of plants, called hanging gardens, cling to steep canyon walls at sites where ground water seeps along contact zones and flows through fractures in the rock strata. The plant associations of these gardens are unique both in location and in the plant species found there. These hanging gardens, were called "relictual refugia" by Welsh and Toft in 1976 "...because they provided sites for species from other southwest locations, boreal forests, and earlier epochs" (Malanson, 1980, p. 178). Some of the plants are not found elsewhere in Utah (Welsh and Toft, 1981). Further, the species-mix found in individual

gardens often shows dramatic variation from one hanging garden to another (Malanson, 1980).

The unique features of Zion also include a small, endemic snail that has been found only in a few isolated populations of no more than a couple of hundred individuals each. The 'Zion Canyon Snail,' *Physa zionsis* Pilsbry (Pilsbry, 1905), roughly a centimeter long, is listed as a category 2 endangered species because the data needed to assess "...biological vulnerability and threat(s) are not currently available..." (The Federal Register, 1984). These fingertip sized creatures are found only in specific portions of the wet areas of canyon walls below trickling springs, indicating that they probably have quite specific environmental needs, though little is known of the snail or its requirements.

Populations of non-game native fish inhabit the North and East Forks of the Virgin River; one of these is the Virgin spinedace, *Lepidomeda mollispinis mollispinis* (Cross, 1975). The FWS has given this fish a category 2 classification also (The Federal Register, 1985). Like the Zion Canyon Snail, the environmental requirements of the Virgin River spinedace have yet to be determined, but researchers generally feel that the fish will not successfully compete with certain introduced fish species in less variable stream environments (Holden, et al., 1974).

## DEPARTURE ANALYSIS STUDIES AT ZION NATIONAL PARK

Considering the critical importance of Zion's water-related resource values, the Departure Analysis approach appeared appropriate and viable for application to management decisions affecting Zion. The following is a discussion of the use of Departure Analysis at Zion.

### Inventory and Literature Review:

The Departure Analysis began with a field inventory designed to identify park resource attributes which are related to, or dependent upon, either surface or subsurface sources of water. Along with this effort, there was conducted an extensive literature search directed at system responses in general as well as site-specific investigations in Zion or in other unique ecosystems. The inventory included a review of administrative files, especially historic documentation of the role of water in park ecosystems and management actions associated with water-related resource values. Where available, historic documentation of circumstances surrounding establishment of the park was also researched.

### Selection of Variables for Study:

When the field inventory and literature search were completed, enough information had been compiled for the selection of response variables. This step was of pivotal importance, because it represented a distillation of all known information specific to the resources of Zion, as well as to the scientific disciplines to be employed in the assessment of impacts.

A variable was selected for study only if it met the following criteria:

1. It could be measured with acceptable practicality, precision, and accuracy;
2. It would be responsive to changes in surface and ground water hydrologic regimes;
3. Its responses could be separated from natural variation within the limits of a selected level of confidence;
4. Its responses could presage broader and more consequential system responses;

5. It could be measured at reasonable cost; and
6. Expertise would be available for its measurement.

Numerous water-related resource attributes were identified for study at Zion. The effects on these attributes of water diversion and/or storage are being estimated, using eleven individual studies which, in aggregate, form the basis of the Departure Analysis project. Each study will be used to predict the nature and magnitude of departures in selected response variables consequent to alterations in control variables associated with hypothetical flow regimes.

#### **Collection of Baseline Data:**

An understanding of existing conditions is required for the assessment of potential departures from existing conditions in water-related resource attributes. To develop this understanding, a baseline data collection effort was initiated and is nearing completion. This effort, it must be emphasized, intends to characterize the **existing condition** which, for purposes of Departure Analysis, is treated as baseline. Accomplishing this characterization at Zion requires the completion of several studies concurrently pursued.

1. **Water and Sediment Discharge.** This study is designed to evaluate the quantity of water and sediment flowing through Zion in the north and east forks of the Virgin River. The study examines the distribution, in time and space, of surface passage of water and sediment, and will attempt to segregate the relative amounts resultant from seasonal precipitation (e.g., snow melt) from sedimentation occurring after high-intensity, short-duration, storm events.

2. **Channel Processes.** The local transport and distribution of water, sediment and organic matter, through selected reaches of the east and north forks of the Virgin River, are examined in this study. From the relationships of river discharge to the quantity and size of sediment transported, the existing sediment load and transport capacity of the study river (at the sites examined) will be described. These relationships and transport capacity will be used to estimate the magnitude and nature of changes in channel cross section area and shape which are likely to result from alterations in the existing stream flow regime.

3. **Aquatic Organism and Physical Habitat.** The aquatic habitat needs of selected aquatic organisms (fish, insects or others) are being evaluated for selected reaches, and will be used to compute potential available habitat at various regimes of flow. Environmental variables affecting species survival and success are being identified, and these will be examined to determine the nature and degree to which they might limit population size or health.

4. **Ephemeral Stream (Dry Wash) Sediment Supply.** The potential for sediment storage in, and delivery from, ephemeral tributaries (dry washes) is being assessed in this study. An evaluation of the rate and quantity of sediment transport through ephemeral streams will be developed to facilitate the interpretation of data from the Channel Processes study.

5. **Riparian Physical Habitat and Associated Organisms.** In this study the organisms and physical habitat of the riparian corridor are being evaluated with respect to the effects of flow regime alteration. The responses of riparian areas will be assessed in order to postulate potential long-term effects from man-induced perturbations in flow regimes.

6. **Hanging Gardens.** The role of ground water flow regimes in the maintenance of plant species diversity in the hanging gardens is being assessed in this study. Plant community response to alterations in spring flow will be addressed in the context of long-term systematic change.

7. **Ground Water Contribution to Surface Flow.** This study assesses the relative contribution of ground water discharge to surface

flow regimes. The potential for surface flow expressions of ground water depletion will be addressed along with the potential for resource impacts from altered surface flow regimes in areas of little or no ground water contribution.

8. **Recreation and/or Aesthetics.** A determination of the recreational and aesthetic characteristics of water in Zion is being made along with an evaluation of probable impacts to these characteristics due to altered flow regimes.

9. **Archaeologic and/or Historic Resources.** The historic role of water in the lives and culture of early inhabitants of the Zion area is being studied along with the locations of archaeological sites which might be subject to impact from flow regime alterations. Results of other studies will be incorporated into the Archaeologic and/or Historic Resources study to assess the potential for impacts.

10. **River Continuum Concept.** Physical, chemical and biological variables are being characterized on the east and north forks of the Virgin River and their tributaries. These variables are being correlated with hydrologic data to estimate potential change in aquatic communities consequent to changes in carbon flux expected to result from alterations in hydrologic regimes.

11. **Habitat Requirements of Virgin River Spinedace.** Measurements are being made of the distribution and habitat requirements of the Virgin River spinedace, and evaluations made of physical and chemical limitations upon fish viability. These data will be used to describe existing and potential habitat quality and effects on of species survival. The interactions of hydrologic regimes, habitat, fish survival, and reproductive or other success will be inferred.

#### **Simulation of Hypothetical Hydrologic Regimes:**

After baseline data collection, the next step in Departure Analysis is the simulated perturbation of hypothetical hydrologic regimes, using standard hydrologic and engineering approaches. Such a simulation is already underway, and it will have been completed during the Summer of 1989. The procedure entails first calibrating and verifying with measured data, existing formulae and computer models characterizing water and sediment discharge. The formulae and models are then incrementally supplied simulated hydrologic input which is the product of hypothetical alterations of the historical regimes.

Examples of such alterations are: reduction in non-irrigation season peaks with associated increases in irrigation season flow due to upstream storage; decreased flows during all seasons with periodic peaks in discharge due to peaking hydropower flows; decreased flows during all seasons (or specific seasons) due to out-of-basin diversions; decreases in spring and seep flow due to heavy ground water withdrawals during periods of extended drought; and increased flows during all or certain specified seasons due to into-basin diversions.

#### **Changes in Response Variables and Comparisons With Existing Conditions of Water-related Resource Variables:**

The control variables (such as stream velocity) associated with hypothetical, altered hydrologic regimes are applied to the calibrated formulae and computer models to predict changes in selected response variables. The response variables, in turn, either directly or indirectly affect water-related resource attributes. By using formulae, computer models and expert opinion, the effects can often be predicted within identifiable levels of statistical confidence.

To elaborate, the formulae or models predict how the response variable will be affected. Prediction can be used directly to describe the effect upon the water-related resource attribute (e.g., altered low flow regime affects water temperature which then affects fish survival) or indirectly through expert opinion (e.g., a simulated increase in sand

bars, due to a simulated reduction in peak flows is interpreted by experts to result in an increase in exotic plants).

These simulated responses and their associated effects upon water-related resource attributes are next compared to existing conditions, as determined by field assessments and expert opinion. Simulated changes from the existing condition are, by definition, departures. These can be associated with the nature and magnitude of the simulated change in the hydrologic regime. Iterations of the procedure provide an indication of the magnitudes of the effects and of the perturbations which caused them. It also follows that a perturbation which produces no simulated departure could be taken as a hydrologic regime alteration which produces little or no impact.

Where hydrologic regime alterations have occurred, or will likely occur, barometer responses that presage system-level responses can be identified through simulations. These responses can then be used in monitoring programs instituted for the early detection of resource impact to initiate resource protection strategies.

It is important to note that, aside from the basic assumption that human-caused change constitutes impairment, the Departure Analysis approach makes no value judgments; a change from existing conditions is a departure, whether that change is positive or negative. The mandate of the NPS directs unimpairment of the areas under its administration, not just protection from degradation. If avoidance of degradation were the Service's objective, habitat improvement might be an acceptable result of a management decision; however, improvement in one aspect of an ecosystem might lead to degradation in another (i.e., an improvement in fish habitat could favor exotic fish at the expense of natives).

## SUMMARY

In an attempt to meet the intent of Congress as defined by the National Park Service's fundamental purpose, the NPS has developed, and is using Departure Analysis at Zion. This technique provides a means of demonstrating potential impacts and impairment of park resources due to alterations in hydrologic regimes.

The use of tools of this sort will probably become more common in those areas of decision-making that impinge upon natural ecosystems. This will arise because sound decisions will require that solid information and identification of trade-offs be coupled with innovative thinking to protect national interests, especially in locations where increasing demands are being placed upon already scarce resources.

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**Note:** The opinions expressed are those of the authors and may not necessarily reflect those of the USDI National Park Service or the U.S. Department of the Interior.

# Status of Threatened Spruce Grouse Populations in New York: A Historical Perspective

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**Abstract:** Spruce grouse populations were surveyed in 29 lowland conifer patches in the Adirondacks during May and early June, 1985-87. Data collected from populations in seven other conifer patches during 1976-80 were also included in the analysis. Spruce grouse abundance was shown to decline substantially from 1977 to 1979, and was apparently higher in 1977 than in any year since then. Of nine spruce grouse populations extirpated during 1977-86, only three have been recolonized. Persistent populations form the core of the current distribution range, however, and have shown greater recovery than formerly extirpated populations since their decline in the late 1970's. Breeding population size averaged 4.1 ( $R = 2-8$ ) for four persistent populations and 2.5 ( $R = 2-6$ ) for five extirpated populations, three of which were recolonized. Three other populations each included 14-20 spruce grouse. Conservative density estimates ranged from 1.0 to 9.6 birds/100 ha. The entire Adirondack breeding population is estimated at 175-315 spruce grouse. Population decline has been noted since the late 1800's and the geographic range is roughly half of what was earlier reported. Hypothesized reasons for the decline include reduction in habitat due to changes in forest composition, unwary behavior toward humans, and interspecific competition with ruffed grouse. Changes in forest composition resulted from selective logging and disease and probably contributed most to the decline of spruce grouse. Spruce-fir formerly accounted for perhaps 45% of the Adirondack forest, however, recent estimates range from about 10 to 25%. Although 60% of the land in the Adirondack Park is state-owned, 98% of spruce grouse numbers and 97% of the habitat area for 23 populations is under private ownership. Maintaining patch contiguity through preservation of corridors will enhance the viability of the complex of disjunct populations. Management efforts should focus on informing landowners of strategies for habitat protection and maintenance and educating hunters to avoid accidental shooting.

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## INTRODUCTION

Spruce grouse (*Dendragapus canadensis*) was added to the New York State threatened species list in 1983 after it was discovered that populations were small and declining (Chambers et al., 1982). Several populations studied in the 1970's appeared to be isolated with a high probability of extinction (Fritz, 1979). Application of extinction probabilities to management, however, is limited, because the fluctuation of small populations over a few years can result in radically variable estimates of those probabilities. Evaluation of historical population and habitat trends enables us to gain understanding of the ecology of small populations. Spruce grouse are particularly vulnerable in the Adirondacks because habitat specificity limits them to the patchy distribution of lowland conifer habitat dominated by black spruce (*Picea mariana*), red spruce (*P. rubens*), balsam fir (*Abies balsamea*) and eastern tamarack (*Larix laricina*). Needles of balsam fir, spruce and tamarack have been the major food items identified in seasonal analysis of spruce grouse fecal droppings collected in three Adirondack study sites (Gradoni, 1982).

Knowledge of the factors that contribute to the decline of populations is germane to the management of threatened species. Relatively few attempts have been made to explain the role of historical factors in shaping the current distribution and abundance of threatened species, since the common practice has been to focus efforts on endangered species only. Early synthesis of historic and geographic factors may facilitate development of management strategies that could help prevent a threatened species from becoming endangered. This paper examines the current population status of Adirondack spruce grouse and discusses the influence of historic factors and recent events on spruce grouse populations. We also discuss the implications of land use practices to the management and preservation of small spruce grouse populations.

## STUDY AREAS

This study was conducted in 29 coniferous habitat patches in Franklin and St. Lawrence Counties, New York, during 1985-87. Spruce grouse occupancy was recorded for 23 of these patches (Figure 1). Spruce grouse data collected during 1976-80 (Chambers et al., 1982) included parts of Essex and Hamilton Counties as well, and these data are also addressed in this report. Earlier spruce grouse data (Bull, 1974) encompass parts of Herkimer and Lewis Counties in addition to the current range.

Topography of the region ranges from moderately mountainous to relatively flat. Elevations in the region occupied by spruce grouse range from 460-490 m in valleys and 610-1340 m on ridge and mountain tops (Carlisle, 1958). Soils over most of the region have originated from glacial drift since the Late Wisconsin stage of Pleistocene glaciation.

The area has a humid continental climate. Temperatures range from -38<sup>o</sup>F to 98<sup>o</sup>F (-39 to 37<sup>o</sup>C) and average 17<sup>o</sup>F (-8<sup>o</sup>C) in winter and 63<sup>o</sup>F (17<sup>o</sup>C) in summer (Carlisle, 1958). Mean annual precipitation is 94 cm (37 in). The frost-free period normally lasts from early June until early September.

Spruce grouse habitat consists of conifer patches at lower elevations where sandy outwash soils are overlain with peat. Black spruce, red spruce, balsam fir, and eastern tamarack are the predominant lowland conifer species at these sites. Slightly higher areas of gravelly to loamy sand support some spruce grouse habitat (Carlisle, 1958), and white pine (*Pinus strobus*) is intermixed to varying degrees and often predominates on these well drained hills and eskers. Ericaceous shrubs are common in the understory. Ground cover is typified by sphagnum moss (*Sphagnum* spp.) on peaty soils, and sandy areas covered with needles are also common on some study areas. Conifer patch size ranges from 50 to 330 ha and the elevation seldom varies more than 20 m in any individual patch. Streams, bogs and ponds occur frequently in conjunction with conifer patches. These patches are nearly or

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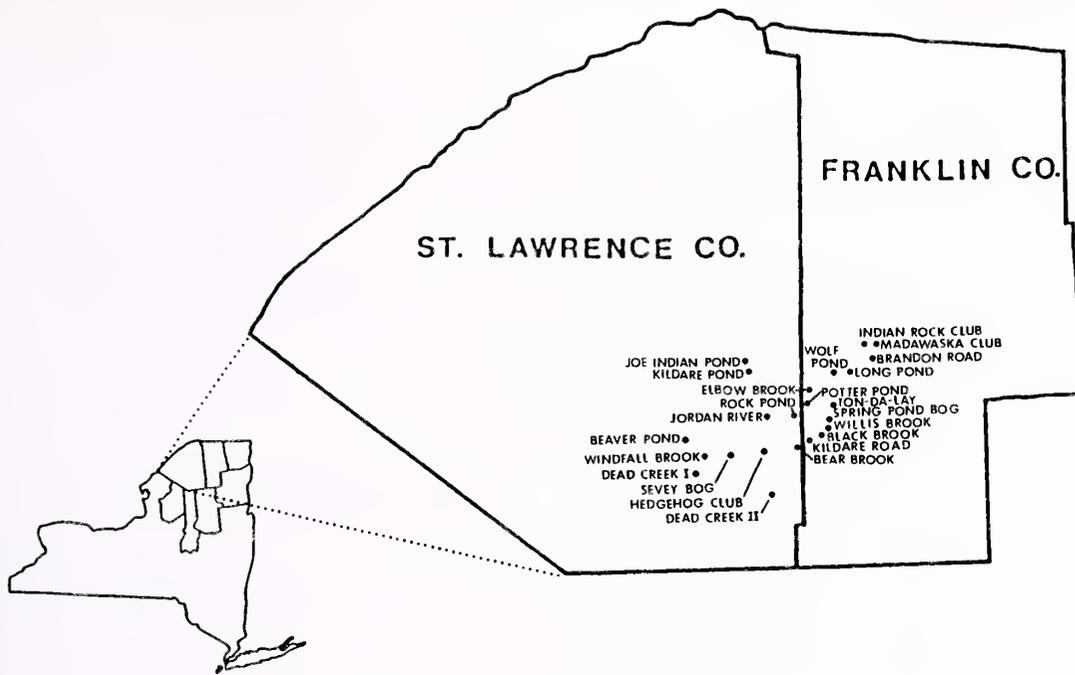


Figure 1. Location of 23 occupied spruce grouse study areas in New York checked during 1985-87.

completely surrounded by hardwoods that occur over most well drained uplands. A few patches are contiguous by virtue of marginal habitat where white pine dominates or connected by coniferous corridors along streams.

## METHODS

Study areas were identified utilizing reports of spruce grouse observations by local authorities and by perusal of aerial photographs and Landsat imagery to identify conifer patches. Patch size was determined from 1:17,000-scale black and white infrared aerial photographs with a compensating polar planimeter. The numbers of patches examined for spruce grouse occupancy were 12 in 1985, 21 in 1986 and 29 in 1987; 115 sites in all were examined for spruce grouse occupancy during 1976-1980 (Chambers et al., 1982).

Habitat patches were searched during May and early June 1976-80 and 1985-87 by one- to three-person teams broadcasting aggressive playback recordings of female spruce grouse (MacDonald, 1968, Fritz, 1977). Playback recordings are effective in locating breeding males throughout the day because males respond to the recording with the flutter flight display. Spruce grouse were captured with telescoping fiberglass poles fitted with a monofilament noose (Zwickel and Bendell, 1967). Grouse were individually marked with either radio transmitters (Amstrup, 1980) or color-coded ponchos (Pyrah, 1970).

Minimum breeding population estimates were calculated by assuming a 1:1 sex ratio and doubling the number of known males for each area. Several studies indicate spruce grouse sex ratios rarely differ significantly from 1:1 (Lumsden and Weeden, 1963, Zwickel and Brigham, 1970; Ellison, 1974; Robinson, 1980, p. 157). Some male spruce grouse, including nonbreeders, undoubtedly did not respond to the tape, so population estimates remain conservative.

Spruce grouse populations were classed as either persistent or "extirpated." Extirpated populations include those eliminated and recolonized unless otherwise indicated. Extirpation was only inferred when two or more visits to an area in years following occupancy revealed no sign of spruce grouse. Recolonization was inferred only

when spruce grouse were actually observed on an area at least a year following observed extirpation. Spruce grouse presence was indicated by fecal droppings in several areas and on several occasions. If grouse could not be located in such sites, they were considered transient, and their occupancy status was deemed unchanged.

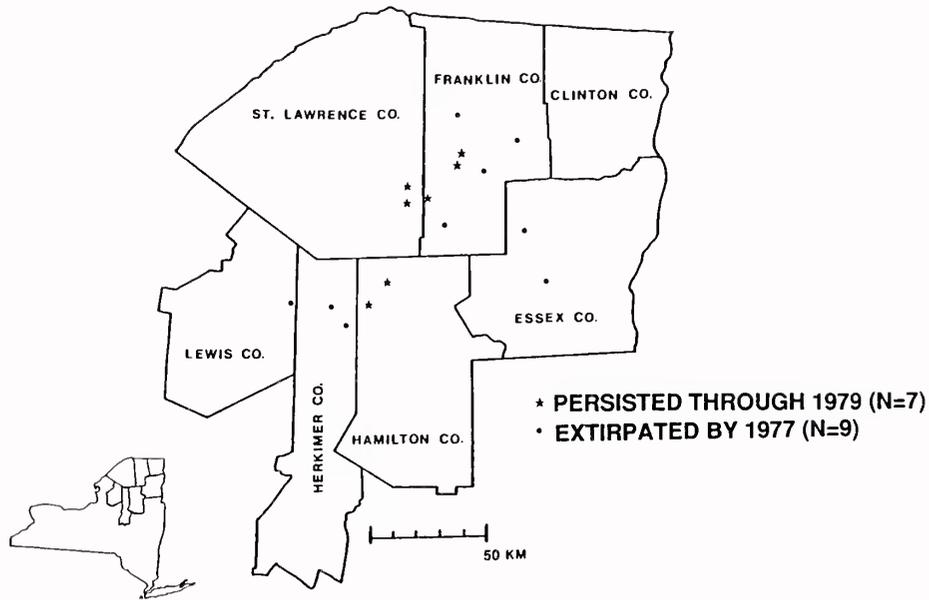
## RESULTS

### Population Characteristics:

The best documentation available for the historical range of spruce grouse in the Adirondacks was summarized by Bull (1974). After compiling historical records and casual observations, he identified 16 spruce grouse populations distributed over parts of six northern New York counties (Figure 2). Only seven of the 26 populations located during 1976-77 (Fritz, 1977, Figure 3) coincide with localities described by Bull (1974). Thus, it can be inferred that nine of the 16 populations identified previous to 1974 have been extirpated. The distribution determined by Fritz (1977) includes parts of four counties and encompasses only about half the area included by Bull (1974). Of nine extirpated sites, seven lie outside the current range, as defined in this report. This provides hard evidence that the range of spruce grouse in the Adirondacks has been shrinking.

Population decline of Adirondack spruce grouse occurred during the 1970's. From 1977-79, 14 of the 26 populations known to exist declined, and six became extirpated (Chambers, 1980, Fig. 3). The total estimated breeding population for comparable sites dropped from 68 to 28 spruce grouse, and by 1980 only 19 of the 26 study sites were still occupied. Nine cases of extirpation and three instances of recolonization were documented between 1977 and 1986, added to seven extirpations that occurred during 1977-80. Recolonization in extirpation sites was first documented in 1985.

Spruce grouse populations appear to have been higher in 1977 than in any year since then. This peak was followed by a sharp decline to 1979, and in 1980 it looked as if populations were beginning to recover. During 1985-86 population estimates were slightly higher



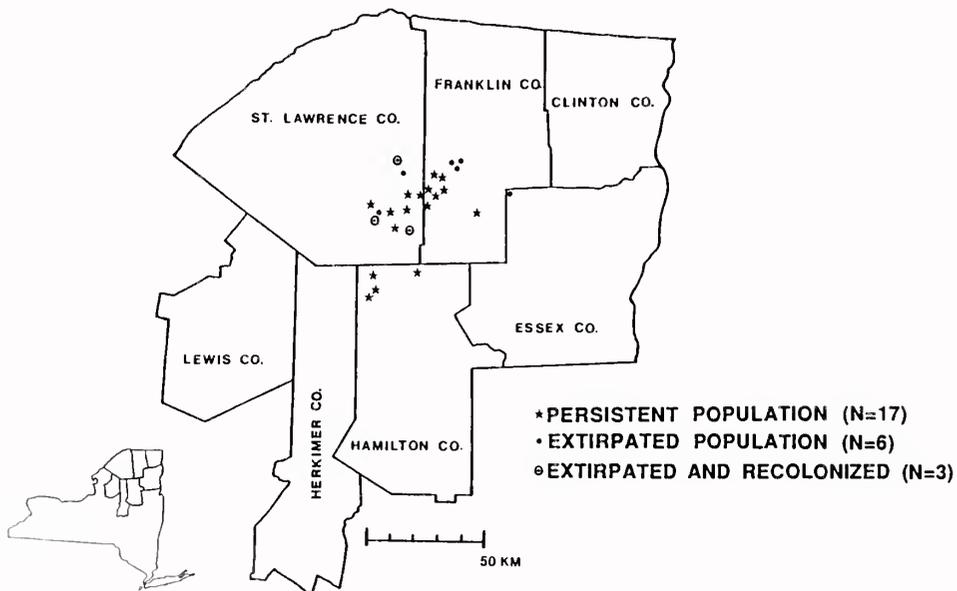
**Figure 2.** Distribution and fate of 16 spruce grouse populations in New York, updated from Bull (1974).

than in 1980. However, populations generally declined from 1986-87, and they remain lower than the level recorded previous to the decline in the late 1970's. Of the ten populations estimated during 1986-87, five had declined and five had remained stable. Populations in ten other areas were smaller and more difficult to estimate due to lack of male response to playback recordings.

Annual breeding population size for four persistent populations averaged  $4.1 \pm 2.0$  ( $N = 31$ ,  $R = 2-8$ , Table 1). Habitat patch size for these populations ranged from 83 to 162 and averaged  $117 \pm 32.9$  ha. The size of three larger, persistent populations, for which fewer years of data are available, ranged from 14 to 20. These are the largest known populations in the Adirondacks, occupying patches from 180

(Fritz, 1977) to 330 ha of coniferous habitat. Size of populations inhabiting five sites of extirpation averaged  $2.5 \pm 1.2$  ( $N = 13$ ,  $R = 2-6$ ). Patch size for these sites averaged slightly smaller than at persistent sites, at  $93 \pm 50.3$  ha, and ranged from 50 to 173 ha.

Breeding population densities for individual areas ranged from 1.0 to 9.6 birds/100 ha (Bouta, unpubl. data). These densities were lower than those previously reported for spruce grouse in New York (Chambers, 1980) because of slight changes in the delineation of study area boundaries. Density estimates were frequently conservative because spruce grouse did not appear to use the entire conifer patch. Alternative estimates of density based on nearest neighbor distances merit consideration.



**Figure 3.** Distribution and status of 26 spruce grouse populations in New York during 1976-87.

**Table 1.** Spruce grouse abundance in four persistent and five extirpated study areas in the northwest Adirondacks, New York, 1976-80 and 1985-87.

Study Area	Breeding Population Estimate <sup>a</sup>								
	1976	1977	1978	1979	1980	1985	1986	1987	
<b>Persistent</b>									
Beaver Pond	<sup>b</sup>	4	2	2	-	4	8	4	
Hedgehog Club	2	6	2	2	2	6	8	4	
Kildare Road	2	6	2	2	6	6	6	6	
Sevey Bog	4	8	4	4	4	4	2	2	
<b>Extirpated</b>									
Dead Creek I	-	2	0	0	0	2	4	2	
Dead Creek II	-	6	2	0	0	0	2	* <sub>c</sub>	
Joe Indian Pond	-	2	2	0	0	-	2	*	
Kildare Pond	-	2	2	0	0	-	0	0	
Windfall Brook	-	2	0	0	-	0	0	0	

<sup>a</sup> Minimum estimates based on assumed 1:1 sex ratio and derived by doubling the number of known different males.

<sup>b</sup> - = Not checked.

<sup>c</sup> \* = Droppings found but no birds observed.

Spruce grouse population densities in New York were lower than most reports in the literature (Table 2). Although maximum densities were comparable for New York and Michigan (Robinson, 1980), average density in New York was substantially lower. In Ontario, lowland black spruce forests supported lower densities of spruce grouse than other habitats (Szuba and Bendell, 1983). Spruce forests with some upland terrain supported denser populations, but the highest densities in Ontario were associated with young jack pine (*Pinus banksiana*) stands (Szuba and Bendell, 1983).

**Table 2.** Spruce grouse breeding population densities (grouse/100 ha) for New York and other localities.

Locality	Density Range	Mean Density	Source
Alaska	7.7-11.6	7.8	Ellison, 1974
Alberta	10.5-19.3	14.9	Boag et al., 1979
Michigan	4.9- 9.0	7.1	Robinson, 1980
Montana		3.3	Stoneberg, 1967
New Brunswick	9.8-21.9	14.9	Keppie, 1987
New York	1.0- 9.6	3.2	This Study
Ontario			
Jack Pine	12.0-80.0	39.8	Szuba and Bendell, 1983
Black Spruce <sup>a</sup>	2.0-30.0	6.6	Szuba and Bendell, 1983

<sup>a</sup> Based primarily on predicted levels rather than direct estimates.

Persistent populations showed greater recovery than extirpated populations since declining during 1977-79 (Table 1, Fig. 4). Most persistent populations form a "core" area within the geographic distribution of the 23 study sites (Figure 5). The area outside the core includes more isolated sites with a history of extirpation or intermittent

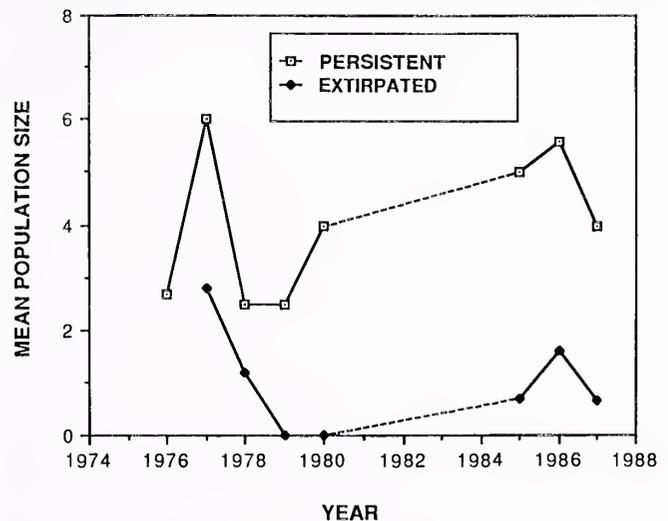
occupancy. The status of seven populations outside the core area has not been monitored since 1980. Because extirpated populations are generally located in the peripheral zone of the current range, continued extirpation there may result in future shrinking of the geographic range.

#### Total Adirondack Population:

The total number of spruce grouse in the Adirondacks was estimated in two ways. First, the population estimate for each area was averaged over all years surveyed. Average estimates were summed for all areas. If only droppings were found in an area, the population for that area and year was assumed to equal 1 for purposes of simplification. The total averaged population estimate for 23 areas surveyed during 1985-87 was 81.8 grouse. Adding the total averaged population for seven areas surveyed during 1976-80, for which current data are not available (Black Pond Swamp, Bloomingdale Bog, Bog Stream, Bog Lake, Grasse River, Lake Lila, and Single Shanty Brook), gives a total average, detected population of  $81.8 + 26.9 = 108.7$ .

The conservative bias of the population estimate was calculated from the per cent of males that were suspected to be present during the breeding census, but which were not detected due to nonresponse to playback recordings. Such males were located during the summer or the following spring. It was estimated that 22-38% of the males were probably missed during breeding season surveys (Bouta, unpubl. data). Virtually all populations within the region of study were believed to be detected. In 1987, we checked eight previously unchecked conifer patches located among the known occupied patches. Only two were found to be occupied, with no sign of spruce grouse found in the remaining six patches.

The total average breeding spruce grouse population was estimated at  $108.7 \times 1.613 = 175$  grouse, assuming that 38% of the population



**Figure 4.** Population trends of four persistent and five extirpated spruce grouse populations in the Adirondacks, 1976-87.

was undetected. A multiplicative correction factor was calculated as  $(\% \text{ undetected} / \% \text{ detected}) + 1$ . Thus, the total number suspected present is given by  $(N \text{ detected}) \{ [38/62] + 1 \} = (N \text{ detected})(1.613)$ .

The second scenario was hypothesized to provide a realistic upper bracket for the total number of spruce grouse. The maximum population estimate for each of 23 study areas surveyed during 1985-87 and the seven other areas surveyed only during 1976-80 was

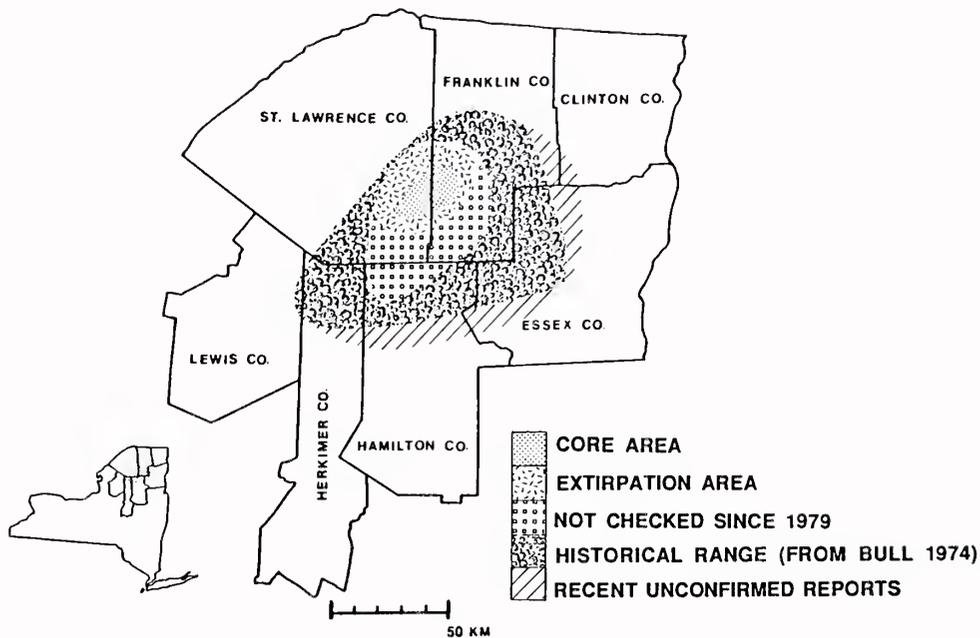


Figure 5. Present and historical geographic range of spruce grouse in New York.

summed, giving  $123 + 34 = 157$ . The correction factor gives  $157 \times 1.613 = 253$ .

Next, we considered three regions outside the survey area that may support populations. These included the High Peaks, Moose River Plains and the Osgood Pond-Saint Regis River region. A liberal estimate would be 20 birds per region, because this is the largest population ever estimated for a single study area and roughly four or five times the estimate for the average area. Assuming 20 birds per region gave  $253 + 60 = 313$ .

Early observations (eg. Eaton, 1910; Roosevelt and Minot, 1923) suggest that the current survey area has always been the region of greatest spruce grouse abundance, and recent reports of observations outside this area are incidental and somewhat suspect. Therefore, it is unlikely that these other regions support sizable spruce grouse populations, if any, and 315 birds would be a liberal total estimate for all breeding populations. Thus, the total Adirondack spruce grouse breeding population is in the neighborhood of 175-315 birds.

### HABITAT OWNERSHIP

Although land in the Adirondack Park is approximately 60% private and 40% state-owned, most spruce grouse populations occur in the northwest quadrant of the park, where private lands are more predominant. Land ownership in those portions of Franklin and St. Lawrence Counties that lie inside the park boundary is 73% private, as opposed to 53% private for the portions of ten other counties that are inside the park (Adirondack Park Agency, unpubl. data).

Nearly all of the areas that support spruce grouse populations are under private ownership. Of the 23 patches surveyed during this study, 22 were privately owned; 23 of 26 populations identified during 1976-77 were privately owned. The distribution of spruce grouse populations of habitat area by land ownership category was determined for the 23 areas recently surveyed (Figure 6). Three privately-owned parks supported 49% of the population surveyed. The Nature

Conservancy held the second largest ownership, supporting 24% of the total population. The Nature Conservancy has recently established the Boreal Heritage Reserve, which includes conservation easements on three private parks. Thus, spruce grouse habitat secured by The Nature Conservancy supports about 73% of the censused population. An additional 21% of the censused population occurs on industrial forest lands owned by four private corporations. The remaining 6% is split between nonindustrial private owners (4%) and state ownership (2%).

Percent habitat area was slightly lower than per cent population for private parks and The Nature Conservancy, while slightly higher for forest industries. Thus, population densities were higher in private parks and on land owned by The Nature Conservancy than on forest industry lands.

The Boreal Heritage Reserve contains the largest constellation of spruce grouse populations in the Adirondacks. Substantial populations within the Reserve include those at Bear Brook, Long Pond, Willis Brook, Potter Pond, Elbow Brook, Spring Pond Bog and Kildare Road, in decreasing order of population size. A lesser constellation, composed of smaller, more isolated populations, is located farther west, and includes Sevey Bog, Beaver Pond, Dead Creek I and the site of an extirpated population at Windfall Brook. Nearly all the habitat that supports this constellation is owned by forest industries, as is the Hedgehog Club, which contains a substantial population but lies between the two major constellations.

### DISCUSSION

#### Historical Abundance:

Spruce grouse were relatively abundant in certain areas of the Adirondacks as late as the 1870's (Roosevelt and Minot, 1923, p. 504). Eaton (1910, p. 365) noted that spruce grouse were "formerly common throughout the tamarack and spruce swamps of the north woods, but for many years it has become scarcer and scarcer, until it is now

threatened with extermination (in New York).” Spruce grouse populations apparently declined dramatically following 1877, and by 1923 they were said to be “very rare in all parts of the Adirondacks” (Silloway, 1923, p. 475). Shortly thereafter, Saunders (1929 p. 469) spent two summers in Essex and Franklin Counties, but reported he “did not find [spruce grouse], nor did inquiry among guides, hunters, and woodsmen in general reveal anyone who had seen this bird in recent years.” Today the range of spruce grouse in New York is an island to itself, isolated from the rest of the range by hardwood forest and agricultural areas associated with the St. Lawrence and Champlain Valleys.

The decline of spruce grouse in the Adirondacks was not an isolated incident. Between 1900 and 1950, various authors from Minnesota to Maine published descriptive accounts of the decline. Although abundant in north central Minnesota in 1870, a local population was extirpated by 1885 (Roberts, 1932 p. 367-368). Spruce grouse were “still common” in northeast Minnesota in 1901, but by 1928 they were “scarce” in northwest and “steadily decreasing” in northeast Minnesota (Roberts, 1932). In northern Wisconsin, spruce grouse were fairly common prior to 1890 (Scott, 1943), but by 1903 they were declining (Kumlien and Hollister, 1903 p. 56) and by mid-Century they were rare (Grange, 1948). Spruce grouse were common in northern

the eastern range to permit spruce grouse hunting since the 1960’s (Johnsgard, 1973).

#### Explanations For Decline:

Three hypotheses have been advanced to explain spruce grouse population declines: 1) Selective logging of softwoods results in the conversion of spruce-fir to hardwood habitat that is unsuitable for spruce grouse (Forbush, 1927, Roberts, 1932, Grange, 1948, Chambers, 1980). 2) The approachability or “tameness” of spruce grouse, combined with the similarity of their appearance to other tetraonids predisposes them to accidental shooting (Eaton, 1910, Bent, 1932, Roberts, 1932, Ammann, 1963, Chambers, 1980). 3) Interspecific competition with the relatively ubiquitous ruffed grouse (*Bonasa umbellus*) limits opportunities for spruce grouse in disjunct populations to establish breeding territories in marginal habitats (Brocke, 1979).

#### Forest Composition:

Historical forestry records support the hypothesis that spruce grouse populations declined concurrent with selective logging of conifers, which was sometimes severe enough to favor hardwood regeneration. In the Adirondacks, white pine was harvested almost exclusively until

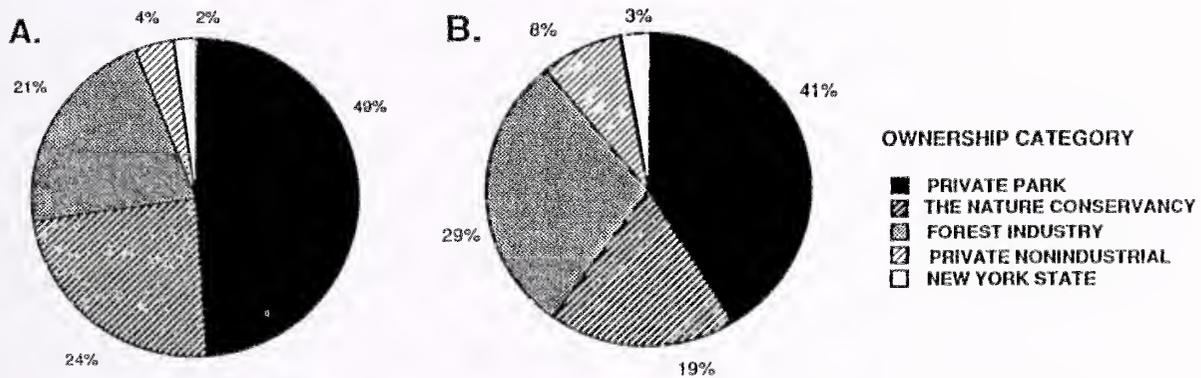


Figure 6. Per cent distribution of spruce grouse numbers (A.) and habitat area (B.) by land ownership category for 23 occupied habitat patches in the Adirondacks.

Michigan prior to 1912 (Barrows, 1912, p. 223, Wood, 1950, p. 133), but were uncommon there by 1963 (Ammann, 1963, p. 592).

The only two known cases of spruce grouse occurrence in Massachusetts were before 1870, and by 1927 they were “disappearing from the inhabited regions of northern New England” (Forbush, 1927, p. 24-25). In Maine spruce grouse were considered a “rare resident” in 1908 (Knight, 1908, p. 199) and populations were later noted to be declining (Palmer, 1949, p. 164). Spruce grouse have been extirpated from Prince Edward Island (Johnsgard, 1973).

The decline of spruce grouse populations has often been associated with widespread logging (Forbush, 1927; Roberts, 1932; Grange, 1948), and evidence of their recovery only exists where coniferous regeneration has been adequate. In the Upper Peninsula of Michigan, Robinson (1969, p. 113) noted increasing spruce grouse abundance since 1950 and stated that they were locally common by 1969. Minnesota spruce grouse populations recovered concurrent with the second growth of coniferous forests, despite shooting by mistaken hunters (Stenlund and Magnus, 1951). Minnesota is the only state in

about 1850 (Fox, 1976, p. 11). About that same time, New York’s lumber industry peaked (Hamilton et al., 1980, p. 15), and eventually the industry switched to emphasize pulpwood harvests, first utilizing “poplar” (*Populus* spp.), but soon switching to selective harvests of spruce and associated conifers (Fox, 1976, p. 77). It was estimated that spruce accounted for 59% of the lumber taken from northern New York during 1899, and 43% of the 533 million board feet harvested in 1900 was spruce pulp (Fox, 1976, p. 81). Also, in 1900, 61% of 39 million board feet of softwood produced in the vicinity of the core of the current spruce grouse distribution was spruce (Simmons, 1976, p. 109). Between 1890 and 1893, one mill produced 51 million board feet, of which 57% was spruce, 37% was pine and 6% was eastern hemlock (*Tsuga canadensis*) (Suprenant, 1982, p. 10). In 1905, 415 million board feet of lumber were milled in the Adirondacks and 49% of it was spruce; hardwoods, hemlock, and pine each comprised less than 20% of the total (Hochschild, 1962, pp. 69-70).

Hardwoods were seldom cut until the 1890’s and even then they comprised a small portion of the harvest (Hochschild, 1962). Unlike

softwoods, hardwoods floated poorly and could not be floated down rivers to mills. Therefore, hardwood logging was delayed until railroads were constructed to transport the logs.

By 1885, demand began to influence the proportion of spruce cut from a stand (New York State Forest Commission, 1886, p. 112). Although several logging firms reportedly left spruce and eastern hemlock timber under 30 cm (12 in) on the stump, the NYS Forest Commission (1886, p. 85) noted that one firm was "cutting everything, small and great alike." Fox (1967, p. 78) observed that "as the demand for wood pulp increased, all stumpage became more valuable for that purpose; all spruce timber, both large and small, was cut." Pulpwood production peaked in 1917 (Hamilton et al., 1980, p. 18) and pulpwood demand led to a rise in the price of spruce stumpage, making it worth more as pulp than lumber (Fox, 1976, p. 81).

As early as 1901, foresters recognized that logging the hardwoods first may have encouraged more softwood regeneration, but softwood lumbering was more profitable and thus it was highly promoted (Hosmer and Bruce, 1901, p. 38). Later it was suggested that logging hardwoods first would not have made a difference. McCarthy (1919, p. 393) noted that "either method of logging increased the proportion of hardwood." Diameter-limit cutting, a common practice at the time, promoted only the growth of low-grade hardwoods. Although softwoods grew well when released from competition by a hardwood cut, hardwoods were more vigorous than softwoods and a second hardwood cut was needed to release the spruce for "free growth" (McCarthy, 1919, p. 393). McCarthy (1919, p. 387) characterized the hardwood dominance on logged softwood stands as the "chief problem of management of the Adirondack forest."

Widespread mortality of spruce trees may have exacerbated the effects of selective logging on spruce grouse. The First Annual Report of the New York State Forest Commission (1886, pp. 50-51) described the spruce decline as "the sudden death of great blocks of black spruce." They also "noticed that as many trees have died in the swamps as upon dry slopes." Most large tamaracks in the Adirondacks also died out in the late 1800's, due to an infestation of the larch sawfly (*Pristiphora erichsonii*) (Graves, 1899, p. 56; Hosmer and Bruce, 1901, p. 16). The cause of spruce decline was never fully ascertained, but speculation pointed to drought, overmaturity and insect-spread diseases (NYS Forest Commission 1886). There was also contradiction as to whether or not mortality was restricted to mature trees (NYS Forest Commission, 1886, pp. 56, 102).

High mortality rates of spruce could have had serious ecological implications for spruce grouse and other boreal wildlife species. It was estimated that, in 1885, 33-55% of the mature spruce timber in the Adirondacks was dead (NYS Forest Commission, 1886, p. 52); all the spruce were dead in one 12,141-ha (30,000-acre) area (NYS Forest Commission, 1886, p. 54). Spruce mortality peaked in some areas between 1870 and 1875 (NYS Forest Commission, 1886, p. 54). Spruce decline has continued in the Adirondacks during recent years for reasons not fully understood; drought, acid deposition, and pollutants are suspected causes (Scott et al., 1984).

The exact effects of logging and spruce mortality on the composition of the Adirondack forest remain somewhat unclear, but significant reduction in the occurrence of spruce-dominated cover types undoubtedly occurred. Fox (1976, p. 12) described the primeval forest as predominantly mixed, with some ridges and slopes occupied by either hardwoods or conifers. Red spruce comprised 30% of the forest in a Hamilton County Township previous to logging (Hosmer and Bruce, 1901, p. 15). In a 423-ha park in Hamilton County, spruce accounted for 43% of the forest, whereas hardwoods

made up 47% of the forest, with sugar maple (*Acer saccharum*) comprising only 8.3% (Graves, 1899, p. 32). The Adirondack Museum at Blue Mountain Lake estimated that per cent composition of sugar maple in the forest increased from 15% to nearly 25%. Today, hardwoods cover from 57 to 88% of three central Adirondack watersheds, whereas lowland conifers cover only 5-26% (Cronan and DesMeules, 1985). Only 11.1% of the commercial forest land in the 11 counties of the Adirondack region is in spruce-fir types; pine types comprise 11.3% of the forest land, and deciduous types cover 77.6%, with northern hardwoods predominating at 61.5% (Considine and Frieswyk, 1982). Softwood harvests in the Adirondacks have continued, and evidence exists that logging has continued to impact spruce grouse since 1950. During the late 1940's and early 1950's, spruce grouse were observed in parts of Franklin County that lie within 5 km of the core of the current distribution (G. Chase, pers. comm., Saranac Lake, N.Y.). The forest in this area has since been logged. It is presently dominated by young hardwoods and devoid of spruce grouse.

The predominance of volunteer hardwood regeneration and absence of jack pine in the Adirondacks contrasts to habitat in Minnesota and Upper Michigan where spruce grouse have recovered somewhat since declining in the early 1900's. Spruce grouse populations in the northern Lakes States recovered as the forests regenerated to spruce, fir, and jack pine (Stenlund and Magnus, 1951). Jack pine is a fire sere species and spruce grouse rely on it for winter food in Upper Michigan (Robinson, 1980). Fires were frequent throughout the period of widespread logging in the Adirondacks. Over 243,000 ha (600,000 acres) burned in 1903 (Simmons, 1976, p. 127), and in 1934 fires raged over 3,200 ha (8,000 acres) of a 21,000 ha (52,000 acre) parcel (Suprenant, 1982, p. 38) that supports localized spruce grouse populations today. Fires may have encouraged early successional cover types, thus making habitat less suitable for spruce grouse. The early successional niche of jack pine is filled by hardwoods in the Adirondacks, especially by aspen (*Populus* spp.), yellow birch (*Betula alleghaniensis*), and red maple (*Acer rubrum*) on burned sites (Cutler, 1975, pp. 185, 212).

#### **Unwariness:**

Behavior of spruce grouse toward humans earned them the nickname "fool hen" and has often been cited as an explanation for their decline. Eaton (1910, p. 365) implied that the unwariness of spruce grouse, combined with their scattered distribution in the Adirondacks, precipitated the decline: "This grouse is so unsuspecting that when disturbed they alight in neighboring trees and the whole company may be shot down one after another without a single bird escaping." Others recognized that spruce grouse can be killed with clubs, stones or nooses, and that populations declined locally around human settlements (Bent, 1932, p. 166; Roberts, 1932, p. 368; Palmer, 1949; Travermer, 1974, p. 154). Although legally protected in Michigan since 1915, spruce grouse have been mistakenly shot for ruffed grouse (Ammann, 1963, p. 592; Johnston, 1969). Two spruce grouse were shot in the Adirondacks during a recent study (Fritz, 1977, p. 26). Brochures and posters directed to alert and educate hunters may provide a means of reducing accidental shooting (Chambers, 1980, p. 9).

#### **Interspecific Competition:**

Sympatric habitat use by contiguous ruffed grouse and insular spruce grouse populations during territorial periods may have negative effects on spruce grouse abundance. Spruce grouse populations declined as ruffed grouse populations increased and expanded in range

(Brocke, 1979). However, overlap in habitat use between ruffed and spruce grouse occurs primarily during nonterritorial periods. Sympatric habitat use by ruffed and spruce grouse in Michigan and Minnesota is restricted primarily to winter (Robinson, 1969; Pietz and Tester, 1982). Minnesota spruce grouse use jack pine in winter and black spruce bogs in summer, whereas ruffed grouse generally select mixed uplands and edge habitats, and their use of jack pine occurs most frequently during winter (Pietz and Tester, 1982). In Michigan, there was almost no overlap in summer habitat use; ruffed grouse used mixed stands and spruce grouse were found only in conifers (Robinson, 1969). Spruce grouse occupied the same habitat all year, but ruffed grouse sometimes moved into conifers for winter shelter (Robinson, 1969).

Ruffed grouse appear to use conifer cover more often in New York than in the Lakes States (see Bump et al., 1947 vs. Gullion and Marshall, 1968 and Gullion, 1972), thus increasing the potential for sympatric habitat use. In this study we found ruffed grouse infrequently during the breeding season, but winter ruffed grouse sign was found on most areas, and ruffed grouse broods were located more frequently than spruce grouse broods in some conifer patches; however, sympatric habitat use does not infer competitive exclusion unless grouse are territorial during the time of overlap. Exclusive use of habitats could be used to infer territoriality (Pitelka, 1959) if documented through simultaneous radio tracking of both species, but competition for display areas is probably not significant, because choice of habitat structure for displaying males differs distinctly between species. The wide movements and range overlap of grouse broods (Ellison, 1973; Herzog and Boag, 1978) concomitant with the winter flocking behavior of spruce grouse (Ellison, 1973), suggest that peaceful coexistence may occur during both summer and winter.

### Synthesis:

The three hypotheses advanced to explain the scarcity of spruce grouse may well interact to help explain the historical decline and current distribution of spruce grouse in the Adirondacks. Conifer logging fragmented and reduced the amount of spruce grouse habitat, displacing birds into more isolated conifer patches. Coniferous areas probably became more isolated as conifer patch size was reduced by logging patch perimeters and corridors. Logging increased habitat suitability for ruffed grouse and the potential for ruffed grouse populations to expand and interface with spruce grouse. Selective conifer logging undoubtedly increased the relative amount of hardwood-conifer ecotone and potential for interspecific habitat use.

The period of timber harvesting coincided with the time of human settlement; logging and subsequent tourism brought more people to the remote areas inhabited by spruce grouse. Unwary spruce grouse probably made easy meals for early settlers and continue to make easy targets for hunters. Increasing human settlement in the Adirondacks meant increasing numbers of hunters; remnant spruce grouse populations meant that fewer hunters may have learned to distinguish the two species. Hunters who did learn the distinction might not have known or recognized laws protecting spruce grouse and continued shooting, although accidental kills may have been a primary influence, exacerbating the decline of small, disjunct populations. Most spruce grouse populations in the Adirondacks currently occur on private lands that are inaccessible to most of the public, including hunters.

### CONCLUSIONS

Extirpation occurs relative to stochastic variation in the size of populations, and such stochastic perturbations are prone to increase

with decreasing population size (Shaffer, 1985). The most recent accounts of Adirondack spruce grouse extirpation are generally unrelated to recent logging activity. Rather, they are the consequence of small populations limited by patchy distribution of habitat (Fritz, 1979, 1985). When small populations fail to reproduce effectively, and are not replenished by immigrants from other areas, they may become extirpated. Small populations may fail to reproduce or survive even if the habitat is suitable, due to chance occurrences of reproductive failure or the loss of significant breeding numbers to severe weather or predation. Small populations may also represent spruce grouse of a single sex that might disperse into previously unoccupied habitats and then live out their lives in isolation.

The fragmentation of spruce grouse habitat in the Adirondacks may be limiting population densities in isolated patches. Fritz (1985) suggested that Adirondack spruce grouse occupancy of widely separated habitats is limited by dispersal. Wide spacing of suitable habitat has been shown to limit insect population density on host trees (Johnson, 1969, cited in Gadgil, 1971). Core populations may function as a source of immigrants for peripheral populations, and dispersal from core populations may result in a net loss of individuals there, if dispersal exceeds immigration. Presumably, survival of dispersers would be inversely related to the distance traveled and suitability of habitat along the dispersal route. Thus, habitat fragmentation may have a negative effect on survival of dispersers, reducing immigration and recruitment and keeping populations below carrying capacity.

The range of spruce grouse in the Adirondacks has decreased since historic times, and the low incidence of recolonization of populations extirpated since the 1970's suggests that range reduction may be continuing. Although the historical range, depicted by Bull (1974), covers approximately twice the current range, some recent reports suggest spruce grouse may occur outside the historical range (Andrie and Carroll, 1988). Such reports should be treated with caution, because distinguishing spruce and ruffed grouse females requires experience and attention to specific details. Furthermore, identification may have been influenced by "observer-expectancy bias" (Balph and Balph, 1983), particularly if coniferous habitat was within sight and observers desired the psychological reward of adding spruce grouse to their life lists.

In the Adirondacks spruce grouse are distributed as a network of small populations. The distribution fits the criteria of what Shaffer (1985) has termed a "metapopulation." Shaffer (1985) showed that each local population contributes to the probability of persistence for the metapopulation. Furthermore, any land-use practice that increases the insularity of populations may increase the probability of extirpation for those populations. Maintaining patch contiguity through preservation of currently unoccupied areas and potential travel corridors will enhance the viability of the metapopulation by enhancing habitat availability and survival for dispersing and transient grouse. With adequate survival of dispersers and recolonization, spruce grouse population dynamics in the Adirondacks may approximate an equilibrium system.

Management strategies for spruce grouse in the Adirondacks must address the issue of private ownership. Management and protection of small spruce grouse populations on some private forest lands will require diplomatic cooperation with forest industries that own them and sportsmen's clubs that lease them. The distribution of a progress report that noted the absence of spruce grouse in large, mature, even-aged stands helped improve our rapport with industrial forestry representatives. Although logging contributed significantly to spruce grouse declines in the past, the potential for improving spruce grouse

habitat through timber management should not be overlooked. A substantial proportion of the low-density, isolated populations may be found on industrial forest lands. New York State now imports more raw timber than it produces (Hamilton et al., 1980, p. 52) and forestry firms in the Adirondacks maintain a vested interest in softwood, forested sites. Management efforts should consider cooperative management with industrial forest corporations as an alternative or complement to land acquisition.

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#### ACKNOWLEDGMENTS

We appreciate the contributions of the 15 students from the State University of New York College of Environmental Science and Forestry who served as field assistants during the course of this study. Though their efforts were varied, each contributed significantly to the project. Field work was made possible with the cooperation of several private landowners who provided permission for access to study areas. Funding was provided by the Adirondack Wildlife Program of the College of Environmental Science and Forestry, The Nature Conservancy and the New York State Department of Environmental Conservation.

# The Raptor and Raven Community of the Los Medaños Area in Southeastern New Mexico: A Unique And Significant Resource

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**Abstract:** The Los Medaños Area of southeastern New Mexico provides habitat for 22 species of raptors including five species that are either candidates for, or listed as, threatened or endangered by state and federal governments. Nesting density of large avian predators varies between 3.1 and 6.8 nests/10 km<sup>2</sup>, among the highest reported in the literature from anywhere in the world. Reproductive performance and numbers of wintering raptors, except for the latter part of 1987, have generally shown a downward trend from 1985 to 1987, suggesting a decline in the numbers of available prey. Raptor diets include a variety of organisms ranging from insects to lagomorphs. In comparing the Los Medaños site to other areas, we find that the density of nesting birds is a potentially misleading criterion as an indicator of high raptor resource value. Alternatively, we recommend the use of multiple criteria to judge the merit of raptor use areas and that these areas should be placed in one of several alternative "importance" categories rather than individually ranked. Our data suggest that raptor population parameters respond both to direct (e.g., human disturbance) and indirect perturbations (e.g., declines in the availability of prey) to several levels of the ecosystem. Therefore, we believe the appraisal of several population parameters (e.g., density, productivity) of the raptor guild is probably the most efficient single approach by which to monitor the consolidated "health" of the Los Medaños community and possibly other ecosystems.

Pages 92 - 101. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Raptors, avian predators that commonly secure prey with their talons or feet, are relatively sensitive to several perturbations in the environment, including direct human disturbance (e.g. Stalmaster and Newman, 1978; White and Thurow, 1985), environmental contamination (e.g., Porter and Wiemeyer, 1969; Spitzer et al., 1978; Grier, 1982), and habitat modification (e.g., Bednarz and Dinsmore, 1982; Bryant, 1986). Other authors (Morrison, 1985; Temple and Wiens, in press) have questioned the value of sampling birds as a tool to monitor the health or dynamics of the environment. Several researchers (e.g., DeSante and Geupel, 1987; Cawthorne and Marchant, 1980), however, have reported measurable responses in avian populations that have resulted from an alteration in a different trophic component in the environment. These findings suggest that measurements of bird population parameters are useful indicators of environmental change.

Whether birds of prey provide an effective means of tracking the "health" of the ecosystem or not, these birds must be considered an important resource value in their own right. Raptors are held in high regard by wildlife enthusiasts and the general public. For example, in 1987 more than 46,500 people paid membership or admission fees to visit Hawk Mountain Sanctuary in rural Pennsylvania (Hawk Mountain Sanctuary Association Annual Report, 1987, unpubl. report), most with the intentions of viewing or learning more about raptors. In addition to the many people who put forth the necessary effort to observe these predators, many of the public regard hawks and eagles as important symbols of the existence of wildlands in North America. Such concern for the extinction of the continental raptor resources would not be acceptable to a large portion of the citizenry of the United States.

Our purpose here is to describe a unique concentration of raptors and ravens in the Los Medaños area of southeastern New Mexico that

warrants special conservation concern. We include Chihuahuan Ravens (*Corvus cryptoleucus*) in this paper because they function ecologically as raptors and interact with other members of the raptor guild (i.e., ravens are competitors, predators, and prey of the other species in the guild). Further, based on our preliminary research, we discuss the utility of sampling raptor populations as an effective approach to monitor the status of the Los Medaños ecosystem.

## STUDY AREA

The study site (32°20'N, 104°50'W) is located approximately 46 km east of the town of Carlsbad in southeastern New Mexico. Our research focused on an area of approximately 419.6 km<sup>2</sup> surrounding the project site of the Waste Isolation Pilot Plant (WIPP; a U.S. Department of Energy research and development facility to demonstrate nuclear waste disposal in a deep geologic repository). The site has been referred to as the Los Medaños Raptor Area and has been previously proposed as a Bureau of Land Management Area of Critical Environmental Concern (an area that requires special management to prevent irreparable damage to important biological, cultural, or natural resources; BLM, 1986). As of this writing, the Los Medaños area has been withdrawn from consideration as an Area of Critical Environmental Concern. Prior to the start of WIPP operations 41.4 km<sup>2</sup> of this area will be withdrawn from the Bureau of Land Management (BLM) and management authority will be granted to the Department of Energy for the operation of WIPP.

This site lies on a broad plain with an elevation range of 910 to 1150 m at the southwestern edge of the Llano Estacado. The sandy soils of this surface support a low shrubby vegetation with affinities to the southern Great Plains. Shinnery oak (*Quercus havardii*) and honey mesquite (*Prosopis glandulosa*) form stabilized coppice dunes in a rolling topography. Sand sage (*Artemisia filifolia*), dune yucca (*Yucca*

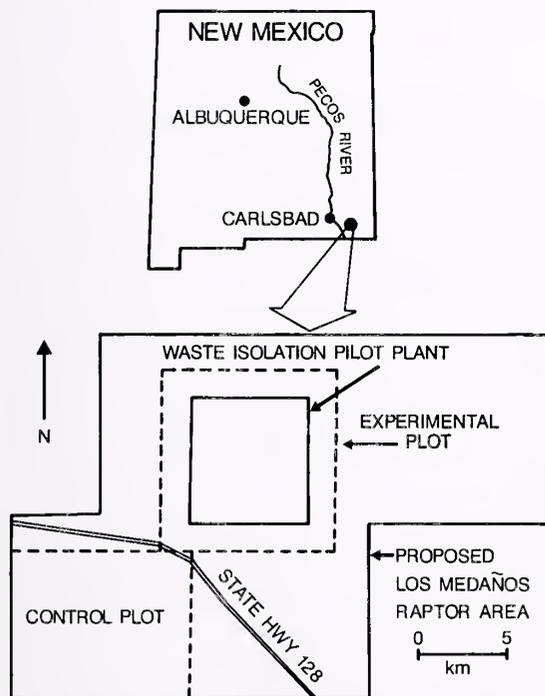
*campestris*), and a variety of perennial grasses are also important vegetational components. Scattered mesquites and western soapberries (*Sapindus drummondii*) provide the greatest vertical structure to the community, though they rarely exceed 4 m in height. These larger "trees" are used as perches and nesting locations by the raptors and ravens.

Interspersed in this area are patches of shallow, calcic or sometimes gypsiferous soils which support chihuahuan desert scrub vegetation dominated by creosote bush (*Larrea tridentata*), snakeweed (*Gutierrezia sarothrae*), whitethorn acacia (*Acacia constricta*), and others. There are also small areas of active dunes.

Temperature variation is extreme: summer temperatures regularly exceed 38° C, while winter readings below -12° C are not unusual. Violent convectional storms accompanied by high winds occur unpredictably throughout the summer. Snow and ice storms periodically transform the study site into a spectacular winterland at any time between October and April. In most years, winds surpassing speeds of 70 kph occur regularly in March and April.

## METHODS

Field work was initiated in April of 1985 and continued every month of the year until December 1987. Complete searches for all "large" raptor and raven nests were made on two intensive study plots: (1) the "experimental" plot (9,320 ha) centered on the WIPP site (an experimental geologic repository for nuclear waste managed by the Department of Energy), and (2) the "control" plot (7,770 ha), immediately southwest of the experimental area (Fig. 1). Searches consisted of walking a small parcel of the study area, often with two or more field workers, and attempting to examine all small trees of suitable stature to support a raptor or raven nest. If no nests were found, but hawks were observed in the area, the search was repeated



**Figure 1.** The location of the Los Medaños Raptor Area, the Waste Isolation Pilot Plant, the experimental study plot, and the control study plot.

two or three weeks later. Although every portion of the two intensive study plots was searched in this manner, it is probable that we missed an occasional nesting attempt. For this reason, our reported nesting densities must be considered as minimum estimates. Complete searches for nests of small raptors, primarily the Burrowing Owl (*Athene cunicularia*) were extremely time consuming, and thus were not done on the intensive study plots. Rather, nests of the smaller species were noted opportunistically. In addition to the primary study sites, all nests found immediately adjacent to or within the larger Los Medaños Raptor Area were monitored and the young banded.

Nests were visited at least three times to document reproductive success, including one visit during the incubation period, one visit during the middle of the brood-rearing period (nestlings 10-30 d old) at which time the young were banded, and a final visit immediately before or at the age of fledging (raptor nestlings 35-45 d old; raven nestlings 30-40 d old). We attempted to visit all nests during the fledging stage.

We assessed the impact of human activities on raptors by comparing reproductive success in the experimental plot with that in the control plot. Within the experimental plot numerous sampling and monitoring programs associated with the development of WIPP have been and are being conducted currently. Many of these activities, particularly groundwater and environmental monitoring, were carried out in remote locations and could disturb nesting birds of prey. In addition, traffic on the major roads in the experimental plot was heavy, including frequent deliveries of supplies and the daily movement of approximately 600 employees to and from the site. The control plot is immediately adjacent to the experimental area and contains similar habitat. Traffic in the control area is primarily limited to the activities of ranchers, recreationists, and occasional visits by workers maintaining gas and oil wells. All of these activities also occur in the experimental plot.

Raptors were trapped with bal-chatri (Berger and Mueller, 1959) and padded long-spring jaw traps. Since the summer of 1986 we discontinued using standard jaw traps and replaced them with Woodstream soft-catch traps (Woodstream Corporation, Lititz, PA 17543). Woodstream traps were placed on top of posts (2-2.5 m in length) and held in place by a single piece of masking tape over the base of the trap. A cord and weight (about 700 g) were attached to the trap and allowed to hang free. The masking tape stabilized the trap and held it in place during strong winds. Once hawks were trapped they were able to fly from the post, but the weight and its associated drag exhausted hawks, bringing the secured birds to the ground usually within 50 meters of the perch post. The Woodstream style trap was more successful in holding raptors after the trap was sprung (54%, N = 22) than for long-spring traps (trap success estimated to be less than 20%, but exact data were not recorded).

Observations of nesting activity were made from portable blinds located 100-150 m away from nests. Blinds were erected at least 36 hours prior to observations to allow hawks to become habituated to their presence. Observers were escorted to the blinds by a "diversion" person at about sunrise. With these procedures, adult hawks usually returned to nests and appeared to care normally for young within 20 minutes of the departure of the diversion person.

A twenty mile (32.18 km) raptor road census was run 31 times in the winter of 1985-1986 (5 November - 23 March), 37 times in the winter of 1986-1987 (23 October - 19 December), and 14 times in the autumn of 1987. All raptors and shrikes (*Lanius* spp.) were noted and their distance from the census route was estimated. A range finder was used to estimate distances less than 150 m, and raptors at greater distances were placed in 100 m distance intervals (150-200 m, 200-300 m, 300-

**Table 1.** Species observed using in the Los Medaños Area in southeastern New Mexico. Relative abundance designations are defined as follows: Abundant = species often observed in large numbers, Breeding = nesting documented, Common = species commonly observed in its appropriate habitat, Occasional = species seen on one or more dates in successive years, Rare = species has been recorded less than four times on the study area during the season.

Species	Season of use <sup>a</sup>			
	Spring	Summer	Fall	Winter
Turkey Vulture ( <i>Cathartes aura</i> )	Common	Abundant	Common	-
Osprey ( <i>Pandion haliaetus</i> )	Rare	-	Rare	-
Mississippi Kite ( <i>Icthinia mississippiensis</i> )	Rare	-	-	-
Northern Harrier ( <i>Circus cyaneus</i> )	Abundant	Occasional	Abundant	Abundant L <sup>b</sup>
Sharp-shinned Hawk ( <i>Accipiter striatus</i> )	Common	Rare	Common	Occasional S <sup>c</sup>
Cooper's Hawk ( <i>Accipiter cooperii</i> )	Common	-	Common	Occasional L
Harris' Hawk ( <i>Parabuteo unicinctus</i> )	Breeding	Breeding	Abundant <sup>d</sup>	Abundant L
Swainson's Hawk ( <i>Buteo swainsoni</i> )	Common	Breeding	Common	-
Red-tailed Hawk ( <i>Buteo jamaicensis</i> )	Abundant	Occasional	Abundant	Abundant L
Ferruginous Hawk ( <i>Buteo regalis</i> )	Common	-	Common	Common L
Rough-legged Hawk ( <i>Buteo lagopus</i> )	-	-	Rare	Rare L
Golden Eagle ( <i>Aquila chryseatos</i> )	Occasional	Occasional	Occasional	Occasional L
American Kestrel ( <i>Falco sparverius</i> )	Common	Breeding	Common	Abundant S
Merlin ( <i>Falco columbarius</i> )	-	-	Occasional	-
Apomado Falcon ( <i>Falco femoralis</i> )	Rare	-	-	-
Prairie Falcon ( <i>Falco mexicanus</i> )	Common	-	Common	Common L
Peregrine Falcon ( <i>Falco peregrinus</i> )	-	Rare	-	-
Common Barn-Owl ( <i>Tyto alba</i> )	Common	Breeding	Common	Common
Great Horned Owl ( <i>Bubo virginianus</i> )	Breeding	Breeding	Abundant	Abundant
Burrowing Owl ( <i>Athene cunicularia</i> )	Abundant	Breeding	Abundant	Rare
Short-eared Owl ( <i>Asio flammeus</i> )	Occasional	-	Occasional	Occasional
Western Screech Owl ( <i>Otus kennicottii</i> )	Breeding	Occasional	Rare	-
Northern Shrike ( <i>Lanius excubitor</i> )	-	-	-	Rare S
Loggerhead Shrike ( <i>Lanius ludovicianus</i> )	Breeding	Breeding	Abundant	Abundant S

<sup>a</sup> Seasons are as follows: Spring = March - May, Summer = June - August, Autumn = September - November, Winter = December - February.

<sup>b</sup> L = classified as large raptor during winter census.

<sup>c</sup> S = classified as small raptor during winter census.

<sup>d</sup> Harris Hawks occasionally breed in autumn (see Bednarz, 1987).

400 m, etc.). All hawks were identified to species. Age, sex, and color phase (i.e., dark, light, or intermediate) were noted when these could be determined. The Fourier series estimator was used to calculate density and variance as suggested by Burnham et al. (1980). Comparison of 95% confidence intervals to point estimates was used to determine significant differences. For analysis, species were lumped into two groups: "large" raptors and "small" raptors (see Table 1).

Reproductive success of the experimental and control study plots was statistically compared with the Wilcoxon two-sample test (Sokal and Rohlf, 1981).

**Table 2.** Density and number of large raptor and raven nests in the WIPP experimental and control plots.

	Number of nests					
	Experimental area (93.2 km <sup>2</sup> )			Control area (77.7 km <sup>2</sup> )		
	1985	1986	1987	1985	1986	1987
Harris' Hawk	16	10	7	18	17	14
Swainson's Hawk	<u>8</u>	<u>10</u>	<u>7</u>	<u>13</u>	<u>9</u>	<u>7</u>
Great Horned Owl	7	7	6	8	6	6
Barn Owl	1	0	0	0	2	5
Chihuahuan Raven	31	32	9	11	11	10
Totals	63	59	29	50	45	42
Density/10 km <sup>2</sup>	6.8	6.3	3.1	6.4	5.8	5.4

<sup>a</sup>Searches for this species were not complete. Search effort was most intensive in 1987.

## RESULTS

### Species Composition:

We have confirmed the presence of 22 species of raptors in the Los Medaños area (Table 1). Three species classified as threatened or

endangered are the Peregrine Falcon (*Falco peregrinus*), Mississippi Kite (*Ictinia mississippiensis*), and Aplomado Falcon (*F. femoralis*) (New Mexico Department of Game and Fish, No date; U.S. Fish and Wildlife Service, 1985). Thus, the Los Medaños habitat may provide resources that aid in the maintenance of the currently limited populations of these species. Thirteen species commonly use the study area in large numbers and 7 species have been documented to nest. Breeding may also occur on rare occasions by the Turkey Vulture (*Cathartes aura*) and the Northern Harrier (*Circus cyaneus*), which are present in the summer. Two species, Golden Eagle (*Aquila chryseos*) and the Short-eared Owl (*Asio flammeus*), cannot be classified as common or rare on the study site. Golden Eagles may be relatively common in the winter in certain years, but for the most part are only occasionally seen in the Los Medaños area. Short-eared Owls were commonly observed on the study site in the early 1980s by J. C. Bednarz, but have not been seen in either 1986 or 1987.

### Nesting Density:

The density of large avian predator nests decreased from a high of 6.8/10 km<sup>2</sup> in the experimental plot in 1985 to a low of 3.1/10 km<sup>2</sup> in 1987. A smaller drop in nesting density was also recorded in the control site (6.4 to 5.4/10 km<sup>2</sup>; Table 2). The average density of both the experimental and control study plots, (5.4 nests/10 km<sup>2</sup> and 5.9 nests/10 km<sup>2</sup>, respectively) are among the highest reported in the world.

Overall raptor nesting densities are not reflected in the figures presented above since suspected Burrowing Owl nests were not considered in the estimates. In 1987 an incidental effort resulted in the location of 17 cavities used by Burrowing Owls, and fledglings were observed near six of those sites. These casual observations suggest that this species is a significant component of the Los Medaños raptor community.

### Reproductive Success:

Reproductive performance of the four primary study species (Harris' Hawk [*Parabuteo unicinctus*], Swainson's Hawk [*Buteo swainsoni*], Great Horned Owl [*Bubo virginianus*], and Chihuahuan Raven [*Corvus cryptoleucus*]) generally showed a decline in several measures

**Table 3.** Mean reproductive success of raptor and raven nests monitored in the Los Medaños raptor area, 1985-1987.

Species	Clutch size			Number Fledged			Nest success	
	mean	SD	N	mean	SD	N	%	N
Harris' Hawk								
1985	2.85	0.86	27	1.56	1.31	39	69.2	39
1986	2.81	0.51	21	1.78	1.26	32	72.7	33
1987	2.68	0.47	28	1.53	1.13	32	71.9	32
Swainson's Hawk								
1985	2.55	0.78	29	1.33	1.17	36	66.7	36
1986	2.21	0.62	29	0.94	0.95	36	55.5	36
1987	2.04	0.70	23	0.87	0.86	30	56.7	30
Great Horned Owl								
1985	2.25	0.50	4	2.13	0.81	16	94.1	17
1986	2.38	0.52	8	1.44	1.21	16	75.0	20
1987	2.80	0.84	5	1.67	1.19	18	72.2	18
Chihuahuan Raven								
1985	5.00	1.06	107	1.43	1.73	117	47.9	111
1986	4.80	0.96	95	1.46	1.78	121	48.4	122
1987	4.40	1.23	47	0.74	1.30	73	31.1	74

of success (clutch size, number of young fledged, nest success, and number of nests found) during this study (Tables 2 and 3). Although there were some exceptions, we recorded the greatest reproductive success in 1985 and the lowest in 1987. The Harris' Hawk was the only species to show an increase in the mean number of young fledged (Table 3) between 1985 (mean = 1.56,  $n = 39$ ) and 1986 (mean = 1.78,  $n = 32$ ). This is contrary to other measures of reproductive output that indicate relatively poor conditions for breeding in 1986 (smaller clutches, Table 3; fewer nests, Table 2; and no re-nesting).

Swainson's Hawks demonstrated a progressive decline in breeding success from 1985 to 1987 (Table 3). For Great Horned Owls, breeding success was noticeably low in 1986 and 1987, while relatively poor reproduction by Chihuahuan Ravens was recorded only in 1987 (Table 3). The general trend of declining breeding performance in all species suggests a biologically important decrease in productivity through time in the Los Medaños raptor population.

Comparison of measures of reproductive output in the experimental area surrounding WIPP and the control area indicate no obvious differences (Tables 4 and 5). Clutch size of Harris' Hawks was the only 1985 reproductive component that was significantly lower ( $P < 0.02$ ) in the experimental area (mean = 2.36,  $n = 11$ ) than in the control area (mean = 3.23,  $n = 13$ ). In addition, the mean number of young fledged by Harris' Hawks and Great Horned Owls was noticeably lower in the disturbed area than in the control plot with differences approaching significance ( $P \leq 0.1$ ). Reproductive performance of Swainson's Hawks and Chihuahuan Ravens revealed no such trend (Tables 4 and 5) suggesting that these species were generally not disturbed by the types of human activities that occur within the WIPP experimental plot.

Thus, we assume that Swainson's Hawks and Chihuahuan Ravens are not sensitive to WIPP-related disturbances and consider the Great Horned Owl and the Harris' Hawk comparisons as independent tests of

the effects of human activities on raptors susceptible to being disturbed in 1985. This analysis was done using Fisher's combined probability test (Sokal and Rohlf, 1981:779-782) and revealed that fledging success of Great Horned Owls and Harris' Hawks in the WIPP experimental area was significantly (Chi-square = 9.65,  $df = 4$ ,  $P < 0.025$ ) less than in the control area. This pattern did not hold true for the years 1986 and 1987; and in fact, for the Great Horned Owl and Chihuahuan Raven there was a nonsignificant tendency for birds in the experimental area to produce more young than in the control plot (Table 5). This is a shift in pattern from a significant adverse effect on two species of raptors to one in which no consistent trend is evident, and coincides with a major modification in management policy by WIPP project personnel. In 1985 work activities proceeded around the WIPP site with no regard for the presence of active raptor nests, and it seems clear that the reproductive output of the affected populations of Harris' Hawks and Great Horned Owls suffered (Tables 4 and 5). The following year, our study team provided information on the location of nests potentially vulnerable to specific sampling or work projects sponsored by WIPP and recommended changes in work protocol or scheduling that would minimize the disturbance of active nests during the critical incubation and early brood-rearing periods. These recommendations were implemented, when possible, by WIPP personnel. This management was in effect for both 1986 and 1987, the years in which no consistent differences were seen between reproductive measures recorded in the experimental and control plots for any of the species monitored.

A substantial decline in raptor nest numbers has occurred in the experimental plot over the three years of the study (Table 2), but this trend was at least in part related to a decline in the availability of food resources (Bednarz and Hayden, unpublished data). If our assessment is correct, then we expect the number of nests in the experimental plot

**Table 4.** Comparison of the clutch size of raptors and ravens nesting in the WIPP control and experimental study areas.

	Clutch Size				$P^a$
	Control area		Experimental area		
	mean	N	mean	N	
Harris' Hawk					
1985	3.23	13	2.36	11	0.02*
1986	2.73	11	3.00	6	0.20
1987	2.57	14	2.50	6	0.81
Swainson's Hawk					
1985	2.50	10	2.14	7	0.41
1986	2.28	7	2.25	8	0.84
1987	1.60	5	2.20	5	0.26
Great Horned Owl					
1985	3.00	1	2.00	3	—
1986	2.33	3	2.67	3	0.62
1987	2.50	2	4.00	1	—
Chihuahuan Raven					
1985	4.75	8	4.79	29	0.98
1986	4.78	9	4.84	25	0.95
1987	4.37	8	4.00	5	0.70

<sup>a</sup>The probability that means are equal; Wilcoxon test.

\*Means significantly different,  $P < 0.05$ .

**Table 5.** Comparison of the fledging success of raptors and ravens nesting in the WIPP control and experimental study areas.

	Number of young fledged per nest				$P^a$
	Control area		Experimental area		
	mean	N	mean	N	
Harris' Hawk					
1985	1.89	18	1.06	16	0.10
1986	1.50	16	1.82	10	0.45
1987	1.86	14	1.14	7	0.18
Swainson's Hawk					
1985	1.15	13	1.50	8	0.47
1986	1.33	9	0.70	10	0.20
1987	0.71	7	1.14	7	0.41
Great Horned Owl					
1985	2.50	8	1.71	7	0.08
1986	1.00	6	2.20	5	0.11
1987	1.33	6	2.00	6	0.44
Chihuahuan Raven					
1985	1.36	11	0.90	31	0.15
1986	0.82	11	2.00	31	0.18
1987	0.70	10	1.22	9	0.48

<sup>a</sup>The probability that means are equal; Wilcoxon test.

to recover as the availability of prey improves.

### Raptor Diets:

During 338 hours of observations from blinds, 84 prey items were brought to nests by adult Harris' Hawks and fed to nestlings. The most common prey were lagomorphs comprising at least 65.1% of the biomass consumed by nestlings (Table 6). Wood rats (*Neotoma* spp.) were second in importance contributing 14% biomass to the nestling diet. Although lizards make up 22.6% of the prey deliveries, they contributed only a small fraction of the dietary biomass (1.5%).

**Table 6.** Prey brought to Harris' Hawk nests during 338 hours of observations from blinds, 1985 - 1987.

Prey	Number	Per cent frequency	Per cent biomass <sup>a</sup>
Desert Cottontail ( <i>Sylvilagus auduboni</i> )	15	17.8	38.8
Black-tailed jackrabbit ( <i>Lepus californicus</i> )	2	2.4	9.3
Unidentified lagomorph	6	7.1	17.0
Total lagomorphs	23	27.4	65.1
Large mammal <sup>b</sup>	6	7.1	10.8
Woodrat ( <i>Neotoma</i> )	14	16.7	14.0
Kangaroo rat ( <i>Dipodomys</i> )	3	3.6	0.8
White-footed mouse ( <i>Peromyscus</i> )	1	1.2	0.1
Unidentified small mammal	9	10.7	1.5
Unidentified mammals	3	3.6	4.5
Total mammals	59	70.2	96.8
Whiptail lizard ( <i>Cnemidophorus</i> )	5	5.9	0.4
Horned lizard ( <i>Phrynosoma</i> )	1	1.8	0.1
Unidentified lizard	13	15.5	1.0
Total reptiles	19	22.6	1.5
Scaled Quail ( <i>Callipepla squamata</i> )	2	2.4	1.7
Total birds	2	2.4	1.7
Unidentified	4	4.8	. <sup>a</sup>

<sup>a</sup>Biomass expansion factors used were described in Bednarz 1986; unidentified prey (N = 3) were excluded.

<sup>b</sup>Lagomorph or woodrat.

Swainson's Hawks also relied heavily on lagomorphs to nourish offspring (Table 7; Gerstell, 1988). Because most rabbits taken by Swainson's Hawks were small (Bednarz and Hoffman, 1988), the biomass expansion coefficient used to determine the importance of rabbits to this species (567 g, the same used for the Harris' Hawk data in Table 6) probably was biased high. The importance of both reptiles and insects in the Swainson's Hawk diet, therefore, is probably greater than indicated by the biomass estimates in Table 7.

### Winter Raptor Populations:

Road counts indicated a substantial decline in winter raptor populations from Oct-Dec 1985 through Jan-Mar 1987 (Tables 8 and 9). The overall density estimate for large raptors for Jan-Mar 1987 (2.2/10 km<sup>2</sup>) was significantly below the estimate for Oct-Dec 1985 (6.8/10 km<sup>2</sup>). All species of large raptors declined, including the

Northern Harrier (67 to 29 estimated hawks), Harris' Hawk (93 to 47), Red-tailed Hawk (*B. jamaicensis*) (85 to 11), and Ferruginous Hawk (*B. regalis*) (18 to 0). Only the American Kestrel (*F. sparverius*) showed no decline in winter numbers during the period of this study. The decrease in hawk numbers parallels the observed decline in raptor productivity from 1985 through 1987 (Table 3) and probably was due to decreased numbers of available mammalian prey (Bednarz and Hayden, unpubl. data). However, raptor numbers rebounded dramatically in the latter part of 1987 (Oct-Dec). The estimated density for this period (8.2/10 km<sup>2</sup>) was higher than for any other winter census period, and was significantly higher than the Oct-Dec 1986 (3.5/10 km<sup>2</sup>) and Jan-Mar 1987 (2.2/10 km<sup>2</sup>) census periods.

We suggest that road counts may underestimate raptor densities in the study area. Accipiters (Cooper's [*Accipiter cooperi*] and Sharp-shinned hawks [*A. striatus*]) were infrequently recorded during the raptor counts, but were commonly seen during other field work. For this reason, we subjectively estimate that wintering diurnal raptor numbers inhabiting the Los Medaños area (419.6 km<sup>2</sup>) have ranged from at least 200 to 450 individuals during 1985-1987. In addition, we believe that more than 150 owls (e.g., Great Horned and Common Barn Owls) probably used the area. Our subjective estimate of wintering raptors on the Los Medaños area is between 350 and 700 birds; the actual numbers supported probably depends upon the availability of prey.

**Table 7.** Prey brought to Swainson's Hawk nests during 115 hours of observations from blinds, 1986 - 1987 (Data from Gerstell 1988).

Prey	Number	Per cent frequency	Per cent biomass <sup>a</sup>
Total lagomorphs	11 <sup>b</sup>	20	79.9
Woodrat ( <i>Neotoma</i> )	2	3.6	5.6
Ground squirrel ( <i>Spermophilus</i> )	1	1.8	1.5
Kangaroo rat ( <i>Dipodomys</i> )	2	3.6	1.3
Unidentified small mammal	3	5.5	1.9
Unidentified mammals	5	9.1	4.8
Total mammals	24	43.6	95.1
Horned lizard ( <i>Phrynosoma</i> )	4	7.3	0.9
Unidentified lizard	12	21.8	2.6
Total reptiles	17	30.9	4.5
Large insects <sup>c</sup>	14	25.5	0.4

<sup>a</sup>Biomass expansion factors used were described in Bednarz 1986.

<sup>b</sup>Five lagomorphs were noted to be "small."

<sup>c</sup>Seven insects were identified as "grasshoppers."

## DISCUSSION

An impressive feature of the Los Medaños area is the extremely dense concentration of nesting avian predators. During the three years of study, density estimates varied from 3.1 nests to 6.8 nests per 10 km<sup>2</sup> (Table 2). The mean densities calculated for the two intensive study plots were 5.4 nests/10 km<sup>2</sup> and 5.9 nests/10 km<sup>2</sup> which is substantially greater than the estimate of 1.9 nests/10 km<sup>2</sup> (based on an estimate of 646 nesting pairs in an area of 3,387.8 km<sup>2</sup>) reported for the Snake River Birds of Prey Area, publicized as having the highest density of nesting raptors in the world (BLM 1979). The combination

of this high density of breeding birds and the easy accessibility to nests (nests are placed in very small trees) has enabled us to monitor the success of 576 raptor and raven nests in a 3-yr period.

**Table 8.** The average numbers of raptors observed per 100 km of road counts in the Los Medaños raptor area between November 1985 and December 1987. Number of censuses are in parentheses.

	Average number observed per 100 km				
	Nov-Dec	Jan-Mar	Oct-Dec	Jan-Mar	Oct-Dec
	1985	1986	1986	1987	1987
	(11)	(20)	(17)	(20)	(14)
Northern Harrier	11.9	13.0	9.2	8.1	12.3
Harris' Hawk	16.6	18.9	11.0	13.0	9.8
Red-tailed Hawk	15.0	8.4	8.1	3.1	6.9
Ferruginous Hawk	3.1	1.1	0.7	0.0	0.0
American Kestrel	0.9	3.6	2.8	1.1	2.2
Other raptors <sup>a</sup>	4.1	3.6	2.8	1.7	2.0

<sup>a</sup> Includes unidentified raptors.

The highest concentration of nesting falconiform birds in the world occurs in some Asian cities where scavenging species exploit huge amounts of garbage and animal carcasses. Newton (1979) reported that the mean density of Black Kite (*Milvus migrans*) and vulture nests in Delhi was 19.3 territories per km<sup>2</sup>. This extraordinary concentration is certainly unique but involves species that cannot be classified as predators and that are well-adapted to coexist with man in an urban environment. From a trophic viewpoint, this group of falconiform birds may be better classified as a "decomposer guild," which in part explains the high densities observed. This concentration of scavengers certainly is of scientific interest, but is not ecologically comparable to communities of avian predators.

The highest density of large avian predators we have found described in the literature was 9.3 nesting pairs/10 km<sup>2</sup> (this estimate excludes small raptor species: kestrels, screech owls, and Sharp-shinned Hawks) in Jackson Hole, Wyoming reported by Craighead and Craighead, (1969). Their study plot, however, was relatively small (31 km<sup>2</sup>) and apparently purposely placed to encompass various canyon and floodplain features that attracted the highest density of birds of prey in that region of Wyoming (Craighead and Craighead, 1969). Densities determined from strategically-placed small plots (see Bednarz and Hoffman, 1988 for a discussion of the biases of using small plots) are not representative of a larger sample area and hence are not directly comparable to the information we report for the Los Medaños area. Nesting density (4.4 nests to 5.7 nests/10 km<sup>2</sup>) reported by Craighead and Craighead (1969:220) for the Superior Township, on the other hand, should reasonably reflect the numbers of breeding birds using the heterogenous mix of the agricultural lands that they sampled and are near what we found in the Los Medaños area. Unfortunately, these relatively high densities of nesting raptors were measured in the 1940's, and recent reports suggest raptor populations in both Wyoming and Michigan have declined substantially (S. Postupalsky, pers comm. [unpublished], 1987). A current site reputed to support a relatively high density of breeding birds of prey is the Zumwalt Prairie in Oregon which had a mean of 2.3 nests/10 km<sup>2</sup> in 1979 and 1980 (Cottrell, 1981). No doubt other areas that support relatively high numbers of raptors exist, but the sampling of literature described above

suggests the Los Medaños area supports one of the densest concentrations of nesting avian predators currently existing in the world.

How do the Snake River Birds of Prey Area and The Los Medaños Area compare in terms of their significance as resources of international importance? As mentioned above, the overall raptor density of the Los Medaños Area is greater than the Snake River Birds of Prey Area. The Los Medaños Area is a relatively homogenous shrubland habitat (several different shrub and grass associations occur) that is used by at least 22 species of raptors and supports an estimated 239 nests (based on a mean estimated density of 5.7 nests/10 km<sup>2</sup>) of large avian predators. The Snake River Birds of Prey Area is eight times larger in area, used by 25 species of raptors, and provides habitat for about 646 nesting pairs of hawks and owls (BLM, 1979). Most of these nests are densely packed along the Snake River Canyon, and the density in this smaller area is 10.9 nests/10 km<sup>2</sup>, which probably represents the greatest density of breeding raptors in the world (BLM, 1979). Many of the birds contributing to the cluster of nests in the canyon, however, rely on the surrounding sagebrush habitat for the procurement of prey. Without these foraging areas the concentration of nests in the Snake River Canyon would not exist. At Los Medaños, nesting substrates are not confined to a limited area; hence, active nests are generally placed in small trees scattered throughout the foraging habitat. Therefore, the Los Medaños Area exceeds the Snake River Birds of Prey Area in its potential to produce young raptors and ravens per unit of land area. From another perspective, the degree of packing of nests within the Snake River Canyon is unique. The clumping of hawk nests in isolated canyon or mountain areas occurs elsewhere, and these sites (e.g., the Owyhee Canyon Area along the Idaho-Nevada boundary; T. Sopher, pers. comm.), for the same reasons as the Snake River Area, harbor important concentrations of raptors and certainly merit conservation protection.

**Table 9.** Winter estimates of the number of diurnal raptors inhabiting the Los Medaños raptor area (419.6 km<sup>2</sup>). Density determined by transect analysis (Burnham et al. 1980) and expanded to size of the raptor study area. Number of censuses are in parentheses.

	Nov-Dec	Jan-Mar	Oct-Dec	Jan-Mar	Oct-Dec
	1985	1986	1986	1987	1987
	(11)	(20)	(17)	(20)	(14)
Northern Harrier	67	57	42	29	136
Harris' Hawk	93	83	51	47	109
Red-tailed Hawk	85	37	37	11	76
Ferruginous Hawk	18	5	3	0 <sup>b</sup>	0 <sup>b</sup>
American Kestrel	22	29	67	30	52
Other raptor species <sup>a</sup>	9	2	8	14	4
Unidentified raptors	19	14	10	7	17
Total diurnal raptors	313	227	218	138	394

<sup>a</sup> Cooper's Hawks, Sharp-shinned Hawks, Prairie Falcons, and Merlins have been observed on counts.

<sup>b</sup> Ferruginous Hawks were observed on the study area, but not recorded on censuses. This value is an underestimate.

The point of this discussion is to illustrate how difficult it is to compare directly the "resource values" of specific raptor use areas. Depending on how you choose to compare densities (to include or not include foraging areas), the rankings of sites will vary. Therefore, we propose that simply using density to evaluate the significance of an area is misleading. More appropriately, biologists should use several criteria to appraise the merit of any natural resource area, including (1) species composition, (2) density, (3) duration of use, (4) vulnerability, (5) the potential for management to affect long-term protection or maintenance of the resource, (6) how the resource area contributes to the global populations, and (7) the "keystone" importance of the area to the life cycle of the birds in question. The last two criteria are probably among the most important, but are also the most difficult to assess. Understanding how an area contributes to the global population in terms of breeding recruits and favoring survival, requires long-term population data that essentially do not exist. Also, it is very difficult to determine the keystone value of an area to a particular population or species. An example of a site with keystone importance would be an area used by raptors for staging prior to migration. For instance, Millsap (1987) described a staging area used by at least 26% of the total population of the American Swallow-tailed Kite (*Elanoides forficatus*). Even though this area is used for a short period, loss of this habitat could result in severe consequences in the future of the species. If the Swallow-tailed Kite staging area were destroyed, could the birds simply use an alternative area or would their ability to procure the resources needed for successful migration be seriously impaired? Since it is not feasible to evaluate with exact precision any habitat or geographical area on all these criteria, we suggest it is most prudent to place each raptor habitat in one of several alternative classes of priority (e.g., high value, moderate value) that would indicate its probable resource value or need for protection.

The Los Medaños habitat supports an astounding array of birds of prey (22 species), especially considering its relatively small size and the homogenous nature of the habitat (almost no cliff habitat). Four species (Ferruginous Hawk, Swainson's Hawk, Peregrine Falcon, and Aplomado Falcon) using the Los Medaños site are either candidates for classification as federally endangered or threatened, or are already listed (U.S. Fish and Wildlife Service, 1985). The density of nesting avian predators per unit area, including foraging habitat, ranks among the highest values reported in the scientific literature (see earlier discussion). The Los Medaños area receives high use by raptors year-round (Tables 1, 2, and 9), but the composition of this predatory guild varies with season. Three of the four primary breeding species (Harris' Hawk, Swainson's Hawk, and Chihuahuan Raven) either do not nest or occur only incidentally at the Snake River Birds of Prey Area. Conservation of the raptor resources of the Los Medaños site thus would complement the Snake River Birds of Prey Area. The raptors breeding in southeastern New Mexico are forced to place their nests in very flimsy, small trees that are extremely vulnerable to human and other types of disturbance. Various types of management approaches, including adjustments of WIPP work activities, have improved reproductive performance. Therefore, our observations indicate the raptors of Los Medaños are both vulnerable to disturbance and may be protected by implementing practical management prescriptions.

How much this site contributes to the global raptor resource is difficult to ascertain and will require many years of gathering additional data, both from the site and from other areas. Recoveries from banded birds indicate that many Harris' Hawks and Great Horned Owls hatched on the study area become residents near their natal nest site, and that many disperse and apparently colonize areas within a 160

km radius of the study area (Bednarz and Hayden, unpubl. data). On the other hand, all four band recoveries of Swainson's Hawks were more than 200 km from the study area, including locations as far away as Ibague, Columbia (4,000 km). The limited but growing body of data suggests that raptors reared in the Los Medaños area contribute recruits to the local, regional, and Western Hemisphere populations of raptors. Whether the Los Medaños site is of "keystone" importance to a substantial segment of any raptor population is unknown, but we suggest this is unlikely. We believe habitat features that play such a pivotal role in the life cycle of a population of raptors are probably a rare occurrence in nature. In summary, the Los Medaños site ranks as an extremely valuable raptor habitat for all criteria that we evaluated, except for the "keystone importance" category. Based on the data presented, we conclude the Los Medaños area is a truly unique habitat that supports an exceptional diversity and density of large avian predators. On any scale, the Los Medaños area is worthy of international recognition and should be placed in the highest priority category as a site that merits special consideration for conservation and system-level management.

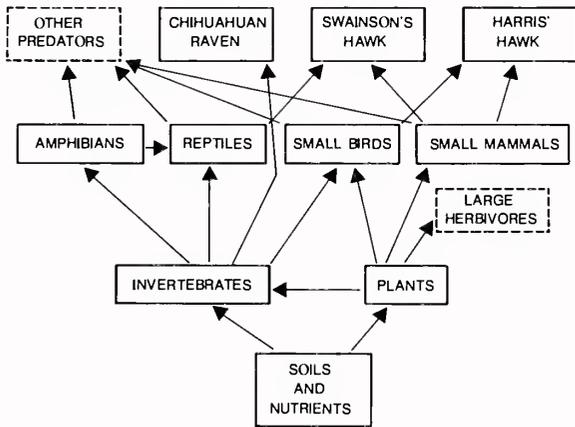
We have shown that WIPP-related activities (e.g., construction, drilling, road construction, environmental sampling) adversely affected raptor productivity in the experimental plot in 1985. Conversely, the data do not support this conclusion for 1986 and 1987 (Table 4). We attribute this change to a shift in management policy that provided for protective management of active raptor nest sites within the experimental plot beginning in 1986. This manipulation provides strong evidence that unmanaged human activities have adverse population effects on birds of prey. A corollary conclusion to this experiment is that careful human management may substantially reduce these impacts and allow for the coexistence of both raptor and human use in many areas.

The use of birds to monitor the larger scale ecosystem has become an area of vigorous debate (e.g., Morrison, 1985; Temple and Wiens, in press). At least for the Los Medaños site, we suggest that raptor populations do provide the index variables that most likely reflect major changes that could occur within almost any component of the ecosystem. We base this argument on the evidence that (1) raptor populations are sensitive to direct human or other disturbance, (2) the raptor guild subsists directly on most primary and secondary consumers present in the ecosystem, and (3) both primary (e.g., reproductive success) and secondary population (e.g., density of nesting pairs) parameters of the raptors are relatively easy to monitor at our study site. The evidence for direct effects of human activity on primary population measures (reproductive success) is presented above. Also, our data suggest that measures of primary population parameters reflect the availability of prey, as has been documented by other workers (Adamcik et al., 1979; Phelan and Robertson, 1978; Newton, 1979, 1986; Bednarz, 1987). Therefore, any major change in the population of a primary prey species should be eventually detected by monitoring the population of the appropriate avian predator.

We suggest that, by closely monitoring three species in the avian predator guild, researchers could roughly monitor the entire Los Medaños ecosystem (Fig. 2). Harris' Hawks would provide a means of indirectly indexing mammal populations, particularly rabbits (Table 6). The lagomorph populations in turn rely on food resources provided by vegetative productivity. Secondly, Harris' Hawks feed on other species of small mammals and birds, and fluctuations in the hawk's productivity or density may reflect substantial changes in these prey elements of the system. Besides the data presented in this paper (Table 6), analysis of prey remains in regurgitated pellets from the Los

Medaños study area indicate that Harris' Hawks feed on a variety of small mammals and often take a fair proportion of birds (Bednarz, 1988). A different sampling of the ecosystem is provided by monitoring the Swainson's Hawk which depends more heavily on reptiles, small mammals, and insects than does the Harris' Hawk (Table 7; Bednarz, 1988; Fig. 2). Finally, the Chihuahuan Raven feeds heavily on insects (Ligon and Cole, 1978), and population parameters of this species should be more sensitive to changes in a different niche of the ecosystem than either of the two hawks (Fig. 2). For indicator species, sampling of primary parameters (reproductive success) would alert resource managers to short-term natural and man-caused perturbations, and secondary parameters (nesting density) would provide information on long-term changes.

We do not advocate that this three-species system is the ideal monitoring approach for the Los Medaños ecosystem. To the contrary, the monitoring scheme is far from perfect. For instance, bird populations will only signal a change in a more elementary trophic level after a suitable lag period. This delay in reaction makes it difficult both to establish cause and effect linkages (Temple and Wiens, in press) and for managers to respond with timely corrective actions. In addition, we find that three major components of the Los Medaños system would simply be missed by our proposed monitoring scheme (Fig. 2; other predators, amphibians, and large herbivores). The raptor populations, however, are linked to all components (primary producers and consumers) that would provide support for these skipped entities (Fig. 2). In this sense the raptor populations would indirectly reflect major changes in the system that also might affect the populations of other higher-level vertebrates.



**Figure 2.** Generalized linkages of components of the Los Medaños ecosystem to three proposed indicator species (Chihuahuan Raven, Harris' Hawk, and Swainson's Hawk) in the avian-predator guild. The boxes with dashed lines denote elements with no direct or indirect links to the indicator species.

A major advantage of monitoring raptors in the Los Medaños area is that a skilled biologist can monitor very efficiently both population density and productivity in a reasonable amount of time. We estimate that by following established standardized procedures, monitoring could be accomplished with approximately three months of full-time effort by one biologist. The population parameters of mammalian predators would be much more difficult to measure, and small sample sizes would result in low precision estimates. Monitoring alternative lower-trophic-level components (e.g., soils or plants) would directly provide information on the fundamental productivity of the system, but could fail to track other

major perturbations affecting the complex animal network of the ecosystem. Conversely, any significant change in vegetative or rodent productivity would eventually be echoed by the raptor populations. Therefore, we propose that the best single system component that reflects the consolidated health of the Los Medaños ecosystem is the large avian predator guild. If resources for monitoring the environment necessitate sampling to be limited to one entity of the ecosystem, then the prudent single focus would be to sample the raptors and ravens. However, this does not address the problems mentioned earlier, that lag-periods in response may make identification of the specific disturbances in systems difficult to elucidate, and the appropriate management might be implemented too late. More appropriately, a comprehensive monitoring system would include sampling of the raptors and other selected key elements (e.g., vegetation and rodent communities) of the Los Medaños ecosystem, which would allow managers, in a timely fashion, to recognize disturbances and to determine which components of the system require management attention.

Although we basically concur with Morrison (1985) that birds often are not particularly sensitive indicators of the environment, we emphasize, for the Los Medaños system at least, that the raptor populations should act as delayed indicators of environmental change. We can find no better single alternative for monitoring the myriad processes that make up the Los Medaños ecosystem.

The data summarized in this report suggest that the Los Medaños area sustains a natural resource of global significance. This is the exceptional convergence of large numbers of avian predators in the sandy plains of southeastern New Mexico. Protection of the life-support system of these predators should be an important objective of land stewardship. We suggest that this can be accomplished most effectively by monitoring and managing the conspicuous avian predators which seem to act as indicators of the dynamics and the stability of the local ecosystem.

## ACKNOWLEDGMENTS

We thank the following individuals for their invaluable contributions to this research. Bill Howe, Jack Barnitz, William Iko, and Dirk Freeman helped collect much of the data presented in this report. David Ligon assisted in the field and has continually offered useful suggestions. Jesse Juen, Stan Van Velsor, Charles Reith, George Anne Thibodeau, Bob Kehrman, Don Byers and many others have provided logistical support, constructive suggestions, and much encouragement. Tay Gerstell, Greg Jillson, Bob Grey, Tom Grey, Stewart Jones, Annabelle Rodriguez, Joneen Cockman, and Stephen Daniel assisted in the field. Christine Lambert produced the figures. This work was funded by the Department of Energy (Westinghouse Contract No. 59-WRK-90469-SD). The Bureau of Land Management provided a field vehicle and other assistance. The University of New Mexico provided animal care facilities and office support and supplies. This is contribution No. 8 of the Los Medaños Cooperative Raptor Research and Management Program.

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# Techniques Used for the Inventory of Rare Ecological Communities in New York State

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**Abstract:** New York Natural Heritage Program ecologists have used a variety of inventory techniques to locate or verify occurrences of rare ecological communities in New York State. Five techniques that have been fruitful are: 1) interpretation of literature records, 2) interpretation of soil surveys, geological maps, and aerial photographs, 3) interviews with knowledgeable people, 4) aerial surveys, and 5) compiling herbarium records of indicator species. Techniques for finding occurrences of communities vary depending on the community; the most productive techniques for several different types of communities are distinguished here. These techniques are also described and illustrated along with brief case studies.

Pages 102 - 105. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

One of the goals of the New York Natural Heritage Program (NYNHP) is to establish and maintain an up-to-date inventory on the location and status of the highest quality examples of the ecological communities in New York State. This goal is common to all heritage inventories established under the guidance of The Nature Conservancy. Priorities for the community inventory are determined by the status of each community within the State, based upon the rarity of the community as well as its vulnerability to destruction or degradation. The focus of NYNHP inventory efforts to date has been on the rarest and most vulnerable communities. These include communities with the fewest occurrences in the state as well as those with the least remaining acreage relative to estimated presettlement acreage.

Techniques for locating and verifying occurrences of natural communities are often more complex than those used to find individual species, partly because of the scarcity of historical inventories of communities, and partly because of the lack of standardized collections of community data (which would be analogous to museum or herbarium collections of specimens that document species occurrences). This paper describes five inventory techniques that have been productive for the NYNHP inventory of rare and vulnerable ecological communities. The simplest and fastest techniques are described first, followed by increasingly complex and time-consuming methods.

## INVENTORY TECHNIQUES

### Literature Records:

Interpretation of literature records is a standard procedure in the initial stages of heritage inventory work (Radford *et al.*, 1981; The Nature Conservancy, 1982). A variety of publications and theses are available that describe communities in New York at a scale useful for the NYNHP inventory. Most of these focus on one community type or a small group of closely related communities within one region of the state. For example, several old journal articles and one thesis describe the Hempstead Plains grassland in western Long Island (Harper, 1918; Ferguson, 1925; Cain *et al.*, 1937; Seyfert, 1973). Likewise, several publications and two theses describe the alpine communities in the Adirondacks (Adams *et al.*, 1920; Woodin, 1959; Holway and Scott, 1969; Kerchledge and Leonard, 1970; DiNunzio, 1972; LeBlanc, 1981). These papers usually give precise localities that are relatively easy to relocate in the field. Such sources are often the most productive for rare communities that are well known in the state.

A few regional studies are available that describe a variety of

community types within a given region of the state. Examples from New York include publications describing forest types or vegetation types in the Adirondacks (Heimburger, 1934), Catskills (McIntosh, 1972), Monroe County (Shanks, 1966) and southwestern New York (Gordon, 1937). This type of publication usually provides at least a few specific localities to illustrate a community type; however, the complete list of localities studied may only be available in data tables or in unpublished data forms that can be difficult to locate and interpret. These sources are useful for common communities, but they are less likely to include detailed descriptions of rare communities.

County floras with descriptions of habitats or vegetation types can be another useful source of community locations, but they may not describe the composition and structure of the communities at specific sites. General lists of plants characteristic of a community type are usually provided, often with a photograph illustrating the community type. In these cases either the text or the photo caption usually provides enough locality information to allow one to find the site.

Photographs illustrating communities are especially useful for finding community occurrences, because they provide a search image for subsequent field work. Developing a good search image for a community is an important step in the inventory process. Field visits to high quality, well-documented examples of a community type are probably the best way to train a field biologist to recognize a good example of that community (i.e. one that is relatively pristine, lacks unnatural disturbances or exotic species) in a new locality.

One example of a rare community relocated by NYNHP based on a description in a literature source is a rich graminoid fen in Mendon Ponds County Park, Monroe County (south of Rochester, NY). In a publication treating the flora of the park, Goodwin (1943) described a marsh meadow underlain by a deep marl deposit. He presented a table showing characteristic species of marly habitats found in the Quaker Pond meadows, and two other wetlands well known for their marly, rich fens. Based on the locality provided by Goodwin, NYNHP ecologists were able to visit the fen and verify the presence of several rare species.

Even if descriptions of vegetation or habitat types are not included in the text, county floras and other biological inventories may indirectly provide enough information in annotated lists of flora and fauna to infer the presence of a particular community type. For example, W.C. Muenscher's contributions to the biological survey conducted in New York in the 1930's have brief references to the habitat of each aquatic plant identified during the inventory. In the survey of the Delaware and Susquehanna watersheds (Muenscher, 1936), *Carex cryptolepis* Mack. is reported from a "marl bog bordering

Summit Lake.” Since a bog is a peatland, and a peatland associated with marl is likely to be strongly minerotrophic, the reference implies the presence of a rich fen. A field survey of the wetlands surrounding Summit Lake confirmed that interpretation.

### Maps and Aerial Photographs:

Soil maps, geological maps, and aerial photos, used in combination with U.S. Geological Survey topographic quadrangles, may provide clues to the location of certain types of communities (White, 1978; Radford *et al.*, 1981). Soil maps, especially those drawn on aerial photographs, are useful for finding communities associated with a particular soil type. For example, several rich fens in New York occur on soils identified as Carlisle muck. By reviewing county soil surveys and selecting open canopy wetlands with soils mapped as Carlisle muck, an NYNHP botanist located a rich fen known as Hidden Lake. The name “Hidden Lake” does not appear on the topographic map, and herbarium labels for several rare species from this site did not provide specific locality information; therefore, the fen could not be relocated using only a topographic map. However, during a field survey to one of the Carlisle muck wetlands, the NYNHP botanist discovered that it is locally known as Hidden Lake.

Geological maps are useful for finding communities associated with, or restricted to, a particular type of bedrock outcropping or surficial deposit. One example of this has been described by Andreas (1985) in her work on the distribution of Ohio’s peatlands in relation to buried river valleys.

If a community also has a distinctive pattern visible in aerial photographs, then geological maps and aerial photos may be compared, and areas with the desired features in both images can be selected for field verification.



**Figure 1.** The striated pattern of the calcareous pavement barrens (see arrow) that corresponds to an outcrop of Chaumont Limestone.

I used this technique to locate additional occurrences of two rare communities known to be associated with limestone pavements in Jefferson County, northeast of Lake Ontario. Detailed maps of the

limestones of Jefferson County (Johnsen, 1971) show areas with outcrops of Chaumont Limestone. These areas were then compared with black-and-white aerial photographs, and any large outcrops showing a striated pattern in the aerial photograph (Fig. 1) were selected for field surveys. The striated pattern indicates vegetation patterns associated with glacially scoured limestone outcrops. In field surveys to these sites, I found a few areas disturbed by grazing (since these could not be distinguished in the aerial photos from ungrazed areas) as well as several sites with good to excellent occurrences of the calcareous pavement barrens community we sought. Two of the newly found sites have alvar grasslands (an even rarer community) and numerous rare species.

Unusual features visible on aerial photographs may also be worth a field investigation. In the process of looking for calcareous pavement barrens in Jefferson County, I noticed an unusual feature in one of the aerial photographs that looked somewhat like the pattern of a large zipper stretching across the landscape (Fig. 2). I visited the site out of curiosity, and found a series of sinkhole wetlands. The vegetation of these wetlands is distinctive and had not been described. In a meadow within this area I also found *Lilium michiganense* Farw., which was new to Jefferson County, and at the time was thought to represent a new state record for the plant (subsequently an unpublished account led to the discovery of a 1911 specimen from Monroe County).



**Figure 2.** The unusual pattern that corresponds to a series of sinkhole wetlands (see arrow).

### Interviews:

Interviews with biologists who are well acquainted with a particular area or with certain communities can be very productive. Interviewing biologists is a routine procedure for heritage inventories (Radford *et al.*, 1981; The Nature Conservancy, 1982). In the NYNHP search for old-growth forests in the Adirondacks, conducted during the 1987 field season, interviews with local foresters and botanists were the best source for locating old-growth stands. Biology professors who take their classes on field trips to local natural areas can also be a good source of leads for community occurrences.

Interviews with landowners are another useful source of tips on local community types. During a survey for oak openings in Monroe County, I located one new remnant (disturbed, but with native species still dominant) by asking the owner of the best quality remnant if he knew of any areas nearby that looked like his property: areas with scattered trees and a tall, grassy groundcover. The key is simply to describe the community in terms meaningful to the landowner.

### Aerial Surveys:

Aerial surveys conducted from small aircraft flying at low altitudes (500 to 1000 feet) can efficiently screen sites selected from maps and aerial photographs and verify community boundaries based on interpretation of aerial photos. NYNHP utilized aerial survey techniques developed by the Illinois Natural Areas Inventory (White, 1978) to verify community boundaries; other inventory programs have used these techniques to locate new sites (White, 1981).

LOCALITY COUNTY	SUMMIT LAKE MARSH, SPRINGFIELD OTSEGO
QUADNAMES	EAST SPRINGFIELD
QUADCODES	4207477
BETULAFUM	
CAREXBUXB	
CAREXKFW	
CAREXDIAN	Y
CAREXGAB	
CAREXFFAI	Y
CAREXSART	
CAREXSCHW	
CAMPYLSTEL	Y
CYPRILET	Y
CYPCANDID	Y
ELEOCHFOS	
PLATANITHE	
POSAALD	
SPHASTERES	
SPHAGNARNS	
TOPFIELDIA	
TRIGLOCHA	
TROLLIUS	Y
VALERIANA	

Figure 3. An example of one locality record from the computer file. "Y" means the species is present.

The procedure begins with identifying sites on topographic quadrangles to be screened or verified. The sites are then circled on a small-scale map that covers a large area (we used county highway maps), the sites are given identification numbers, and a flight plan is developed that connects the sites in an efficient pattern. The flight plan is given to the pilot, and specific tasks are assigned to each of two passengers in the plane. The passenger in the front seat serves as the primary observer and describes the vegetation patterns observed from the plane. The second passenger, on the same side of the plane in the back seat, serves as the note-taker responsible for writing brief notes on maps or separate sheets. In one flight the airplane's built-in cassette recorder was used to record discussions during the flight. A tape recording is a useful back-up to written notes, however, written notes are less vulnerable to mechanical malfunctions and therefore may be the most reliable way to record observations. A videotape recording of the view from the aircraft is another option to consider during an aerial survey (J. White, pers. comm.).

NYNHP used this technique to survey two types of communities, pine barrens and riverine meadows. In the valleys of the Hudson and Mohawk Rivers from Albany to Glens Falls, sandplains that historically supported pitch pine-scrub oak barrens were surveyed. Surveyors were unable to identify any new localities, because most of the area had been developed; however, community boundaries drawn from aerial photographs and local field surveys were verified.

In the upper Hudson River valley, from Glens Falls to Newcomb in the central Adirondacks, the river was surveyed in late spring to search for late-melting ice jams that remain on certain cobble shores of the river. These ice jams are known from one area along the river that

supports an unusual meadow community with several rare species. No new areas with late-melting ice jams were located, but the boundaries of the meadows that had been mapped were found to correspond to the only large areas with late-melting ice jams along the river shore.

### Indicator Species:

NYNHP developed procedures to compile lists of sites based upon herbarium records of species that serve as good indicators for specific community types. One example is the procedure I used to identify potential sites for calcareous wetlands. First I selected a group of vascular plants and bryophytes that are considered good indicator species for calcareous wetlands. I selected species that were reported from a relatively small number of sites in the state (ranging from approximately five to 50), based on the master file of plant distribution records at the New York State Museum. Selecting species with a relatively small number of sites limited the total number of sites reviewed to a manageable quantity. I prepared a computer file in dBASE-III, organized by locality. Each locality had one record with fields for entering the site name, county and each of the indicator species. I entered data by reviewing the herbarium records for each indicator species and entering the locality, county and a code denoting the presence of the appropriate indicator species (Fig. 3). This process transformed the species records (list of sites where a species occurs) into site records (lists of species that occur at given sites).

187	ONONDAGA	2	ONONDAGA LAKE
106	ONONDAGA	1	OTISCO
107	ONONDAGA	1	POMPEY
194	ONONDAGA	2	RAMS GULCH, JAMESVILLE
218	ONONDAGA	1	SALINA
108	ONONDAGA	1	SOUTH OF ONATIVIA (wooded swamp)
210	ONONDAGA	2	WHITE LAKE SWAMP, DEWITT
109	ONONDAGA	1	EAST GREENLAKE, JAMESVILLE
110	ONTARIO	1	GORHAM
244	ONTARIO	1	GULICK SWAMP, SOUTH BRISTOL: Trill
245	ONTARIO	1	HONEYE LAKE, RICHMOND Trill
111	ONTARIO	1	MERTENSIA
112	ONTARIO	1	RAILROAD MILLS
192	ONTARIO	1	SULLIVANS
220	ORANGE	1	FORT JERVIS SWAMPS, DEER PARK, GREENVILLE
214	ORANGE	1	SFRUCE POND, SMITHFIELD
322	ORANGE	1	LITTLE FALLS, BOGE & SWAMPS NEAR CLOSTER NJ AND...
112	ORLEANS	1	ALBION Trill
272	ORLEANS	1	WEST BARRE, 1.5 MI N OF
320	OSWEGO	0	CLARK CORNERS FEN (CARDAMINE FEN of TJA), SCRIBA
114	OSWEGO	1	HANNIBAL
192	OSWEGO	1	LILY MARSH, SOUTH NEW HAVEN
318	OSWEGO	3	MUD CREEK SE OF HANNIBAL, HEADWATERS OF
112	OSWEGO	1	MUD LAKE, N HANNIBAL
206	OSWEGO	1	OSWEGO
190	OSWEGO	1	FORT ONTARIO: SANDY SLOPES NEAR FEAT BGS
317	OSWEGO	1	ST MARYS FEND
182	OTSEGO	1	COPESTOWN
116	OTSEGO	5	SUMMIT LAKE MARSH, SPRINGFIELD Cyp. w/ Trill
117	QUEENS	1	AGUEDUCT
118	QUEENS	1	N OF QUEENS
274	RENSSELAER	1	CHERRY FLAIN, W END OF
328	RENSSELAER	1	DUTCH CHURCH, TYFHA FEN 0.5 MI S OF, NEAR RD
290	RENSSELAER	1	DUTCH CREEK, 0.5 MI S OF
119	RENSSELAER	1	LANSINGBURG

Figure 4. An example of a sorted list of localities, with handwritten tally of the number of indicator species reported from each locality.

Once I had entered all the localities for one species, I began to enter localities for the next species by first searching the existing records for the same locality. If the locality was already in the data base, I simply added the presence-code for the second species to the existing record. If the locality was new, a new record was added to the data base. The result was a set of locality records showing which indicator species had been collected or reported from each locality.

The locality records were then sorted by county and locality to check for duplicate localities with slightly different names. Finally, the total number of indicator species was tallied for each locality (Fig. 4). Localities with the greatest numbers of indicator species, or with the greatest number of rare species, were selected for field surveys. Other criteria may be useful for selecting sites, depending on the goals of the

survey. The same procedure can be used with manual card files; the computer simply speeds the process by allowing an automatic search of a large number of records.

## CONCLUSIONS

Successful application of these techniques depends partly on the ability of the field biologist to recognize a good example of a particular community type. The development of a good search image is therefore an important step in the community inventory process. Another important aspect of using these techniques is selecting the best technique for each community type. The best technique for locating one type of community may be essentially useless for another; therefore, a thoughtful and creative approach, taking into account all the kinds of features that might distinguish a particular community, will produce the best results.

## ACKNOWLEDGMENTS

The techniques presented in this paper are derived from the collective experience of many field biologists working in Heritage programs and natural area inventories throughout the U.S. Some of these techniques are briefly discussed in The Nature Conservancy's Heritage Operations Manual (1982), as well as by Radford et al. (1981). John White has been particularly helpful in sharing his technical knowledge and experience with natural area inventories.

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# Use of a County Soil Survey to Locate Remnants of Native Grassland in the Willamette Valley, Oregon

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**Abstract:** The upland grasslands and oak savannas that occupied much of the Willamette Valley prior to Euroamerican settlement have been almost completely eliminated by the combined agencies of intensive agriculture, domestic livestock grazing, and fire suppression. An inventory of native grassland remnants in Marion County, Oregon, utilized the county soil survey to pinpoint potential sites for field survey. Stayton silt loam, a shallow, excessively drained soil type generally unsuited for agriculture, was the primary map unit targeted for field work, because it was less likely to have had its native vegetation destroyed by cultivation. In five days of field work during the 1987 field season, 23 different tracts were visited, resulting in the discovery of five new occurrences of special plant species, a high quality native grassland remnant, and areas of unusual, vernal moist seepage channel habitats. Under the proper circumstances, this soil survey method provides a useful and efficient means of locating biologically significant tracts in an otherwise unpromising landscape.

Pages 107 - 112. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

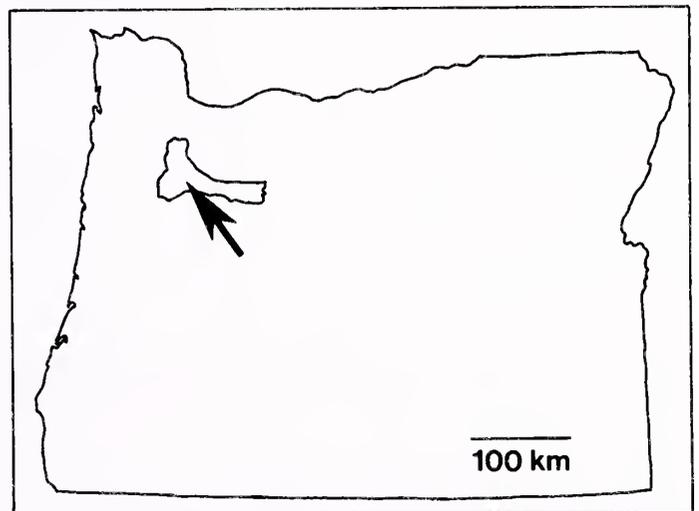
Field inventories of potential natural areas and habitats of threatened and endangered species are an essential component of any natural areas program. Protection of Natural Heritage "elements" on the basis of incomplete inventories may result in political or administrative headaches, if better examples are subsequently discovered. The execution of a thorough inventory can be a challenging task, particularly when only limited funding and personnel are available to survey a large geographic area; consequently, it is valuable to have techniques that can focus survey efforts on areas within the landscape that are most likely to support targeted taxa. This paper describes the use of a county soil survey to aid in locating remnants of native grassland in the Willamette Valley region of western Oregon, an area in which nearly all natural vegetation has been destroyed or strongly modified in the 150 years since the arrival of Euroamerican settlers.

The pre-settlement vegetation of Oregon's Willamette Valley is thought to have been dominated by grassland and oak savanna (Habeck, 1961; Johannessen *et al.*, 1971). The grasslands appear to have been species-rich communities dominated by C-3 ("cool-season") grasses and perennial herbs. Red fescue (*Festuca rubra*) is thought to have been the characteristic bunchgrass of well-drained soils of the valley and adjacent foothills (Habeck, 1961), with tufted hairgrass (*Deschampsia cespitosa*) dominant in the extensive, poorly drained soils of the valley floor (Moir and Mika, 1972). Common or characteristic grassland herbs were species of *Achillea*, *Aquilegia*, *Balsamorhiza*, *Brodiaea*, *Calochortus*, *Camassia*, *Delphinium*, *Dodecatheon*, *Eriophyllum*, *Fragaria*, *Geranium*, *Lomatium*, *Lupinus*, *Potentilla*, *Prunella*, *Ranunculus*, *Sidalcea*, *Vicia*, *Wyethia*, and *Zigadenus*. The historical understory flora of the oak (*Quercus garryana*) savannas is largely undocumented, but it may have included species of *Elymus*, *Erythronium*, *Iris*, and *Ligusticum*, and many grassland species.

Since they occurred in a region potentially capable of supporting forest vegetation, these grasslands and savannas are thought to have been maintained in prehistoric times by fires set by the aboriginal inhabitants (Boyd, 1985). With the settlement of the Willamette Valley in the mid 1800's, such burning practices were halted. Grasslands were converted to grain fields and orchards, or were heavily used by domestic livestock. In a very short period of time, native species were

largely replaced by an exotic flora of ruderal and grazing-resistant species of Eurasian origin (Johannessen *et al.*, 1971). In addition, woody vegetation quickly colonized areas protected from fire and grazing, ensuring that any grassland habitats that managed to escape outright destruction were gradually converted to woodland (Habeck, 1962). Today, native grassland species survive primarily in small patches in the corners of fields and along fencerows, particularly bordering county road rights-of-way. Very few larger tracts of native grassland have been identified or studied, and, as a consequence, little is known of the structure and composition of pre-settlement Willamette Valley grasslands.

An inventory of the Willamette Valley's foothill grasslands was conducted by The Nature Conservancy in the early 1980's (Seyer, 1982). These surveys relied upon existing, known sites and visual checks from highways to identify grasslands of potential interest. While this work was successful in identifying grassland remnants in some parts of the Willamette Valley, it was entirely unproductive in other areas. One such region, where no grassland remnants were located, was in Marion County, in the rolling foothills east of Salem (Figure 1.).



**Figure 1.** Location of Marion County within the State of Oregon. The arrow indicates the general vicinity of the survey area.

A chance encounter with a small tract of interesting grassland habitat in Marion County, along the West Fork of Drift Creek, was the genesis of the present paper. This site, surrounded by cultivated land, was occupied by a mosaic of vernal moist seepage channels, vernal pools, and, on slightly deeper soils, habitat of native grassland species. A check of the Marion County Soil Survey (Williams, 1972) indicated that the soils of the site were mapped as Stayton silt loam. Since this soil map unit occurs throughout the low foothills of Marion and northern Linn Counties, I hypothesized that it could be used as an indicator of interesting native grassland habitats, sites that had escaped the destructive effects of agriculture and settlement by virtue of their dry, shallow soils and rocky substrates.

At the beginning of the 1987 field season, field surveys were proposed to search for previously undetected remnants of native grassland vegetation, as well as to identify the kinds of habitats serving as refugia for native grassland taxa. About 15 vascular plant taxa are endemic to Willamette Valley grasslands, and, because of extensive habitat loss, eight of these taxa are considered to be Threatened or Endangered (Oregon Natural Heritage Data Base, 1989). Conversations with local botanists and ecologists concerning the possibilities for success were not encouraging; a common perception was that, given the land use history of the past 150 years (summarized by Johannessen *et al.*, 1971), the destruction or alteration of the pre-settlement grassland communities had been more or less complete.

## METHODS

Stayton silt loam was chosen as the primary target soil type for this study. Stayton silt loam is described by Williams (1972) as comprising well drained soils that have formed in alluvia underlain by basalt. The shallow depth to bedrock (20 inches or less) results in soils that are poorly suited to most types of agriculture. Stayton silt loam is a Class V1 soil, a soil that has severe limitations that make it generally unsuited to cultivation (Williams, 1972).

In Marion County, Stayton soils are mapped as scattered occurrences in the low foothills between Stayton and Silverton. The upland soils of this region are predominantly described as belonging to the Nekia-Jory association, which appears to represent, in large part, the areas that were originally vegetated by grassland and oak savanna.

One additional soil map unit of the Nekia-Jory association was considered as a suitable focus for this survey. This was Witzel, very stony silt loam, which represents well-drained, very stony soils formed partly in loess and mainly in colluvium. These are also shallow soils, with basalt bedrock lying at an average depth of 19 inches (Williams, 1972).

Prior to the field surveys, soil survey documents were studied and the boundaries of the targeted map units were highlighted with colored pen. The aerial photographs on which the soil maps were overlain were then examined in detail to select specific sites to visit during the field work. At this stage, sites were eliminated from further consideration if they showed evidence of cultivation, if completely forested, or if disturbed by urban development or quarrying.

In 1987, visits were made to a total of 23 sites during five days of field work from April through July. Some of the more interesting sites were revisited during the 1988 field season, and further inventories of grassland tracts occupying Stayton silt loam were made in adjacent Linn County in 1988.

Written notes were taken on the vegetation and environment, and on current and inferred past land-use regimes. Native plants were identified and vouchers made for difficult taxa; then lists were compiled of the native grassland taxa found at each site. Priority was

given to sites where native vegetation appeared to be relatively intact. Special attention was also given to locating populations of species listed as threatened or endangered by the Oregon Natural Heritage Program (1989).

On returning from the field, I prepared short written descriptions of the sites that had been examined, including documentation of sites where nothing of interest was found.

## RESULTS

One great strength of this method is its systematic nature, which gets the surveyor away from roads and other easily accessible areas. The majority of the sites of interest were not visible from any public road, and most would have gone undetected using a visual ground survey.

Several examples of sites visited will serve to illustrate the useful features of this approach. The Sublimity grassland occupies an area mapped as Stayton silt loam along a broad valley bottom. Upon examination, much of the mapped area was found to consist of bedrock-lined seepage channels, a habitat that is very wet in the spring but excessively dry in the summer. A very showy spring flora was found in this habitat, reminiscent of the well known vernal pools in the Central Valley of California. Habitats such as this had not previously been documented in the Willamette Valley, but in this survey they were found to occur fairly predictably on sites mapped as Stayton silt loam. Interestingly, these shallow rocky soils were invariably dominated by native plants, even in sites that had received considerable grazing disturbance. It is possible that the weedy, exotic species dominant in disturbed sites on deeper soils are unable to compete in these stressful, vernal moist habitats, enabling the native flora to persist.

At the Sublimity grassland, this habitat supports a sizable population of Bradshaw's desert-parsley (*Lomatium bradshawii*), a Willamette Valley endemic that is a federally listed Endangered species. No extant populations of this species were known from Marion County before this survey, and the most recent historical collection was made in 1916. Also at the Sublimity grassland, in deeper soils adjacent to the seepage channel habitat, a high quality example of red fescue grassland was located in a small tract that had been fenced to exclude entry by livestock. In this area was found a small population of the Willamette daisy (*Erigeron decumbens* var. *decumbens*), another Willamette Valley endemic and federal candidate (Category 2) species. This was the first sighting of this species in Marion County since 1924. The combined occurrence of two special plant species, a high quality remnant of native grassland, and the rare seepage channel habitat has motivated the Oregon Field Office of The Nature Conservancy to begin efforts to protect the site as a Natural Area.

Another site provides a second example of how the survey process worked. The Hidden Oaks site is an area of grassland, oak savanna and woodland that occurs on soils mapped as Stayton silt loam. It is completely surrounded by extensive cultivated fields, a half-mile walk from the nearest road. Here several extensive colonies of the white-topped aster (*Aster curtus*) were found. This is another grassland species considered Endangered in Oregon; it had not been collected in Marion County since 1918.

Though this tract had been the goal of that visit, the route to Hidden Oaks passed a low hilltop rising above the cultivated fields that had managed to escape plowing, and which harbored a diverse assemblage of native grassland species. This remnant was mapped as Nekia silty clay loam, the predominant soil type in the area, rather than an atypical soil such as Stayton silt loam.

A third remnant, along Starlight Road east of Silverton, provides a

more typical example of the fate of many valley grassland remnants. The soil survey, with its aerial photographs taken in the 1960's, showed an extensive remnant occupying an area mapped as Stayton silt loam. In the intervening years, the area has been subdivided, with new roads providing access to several recently constructed homes. One parcel of approximately 3 ha has remained in relatively undisturbed condition, and harbors a number of native grassland taxa, including a population of the Willamette daisy. In addition, Idaho fescue (*Festuca idahoensis*), a species more typical of Palouse grasslands to the east and Puget Trough grasslands to the north (Franklin and Dyrness, 1973), occurred as a co-dominant with red fescue. Idaho fescue was found in several other small grassland remnants during the survey, generally in shallower soils in the higher foothills.

Observations such as these have proved of particular importance in attempting to develop a more thorough understanding of the diversity of pre-settlement Willamette Valley grassland communities. These examples illustrate the value of a county soil survey as a convenient and productive means of locating relict grasslands. Of the 23 sites visited in 1987, at least some semblance of native grassland was found at 16, and several sites were particularly noteworthy. Five new populations of Endangered or Threatened plants were located during the survey, none of which had been collected in Marion County for over 50 years. During the 1988 field season, another high quality grassland remnant, with both *Lomatium bradshawii* and *Erigeron decumbens*, was located on Stayton soils in nearby Linn County. The repeated occurrence of a distinctive community of vernal pools and bedrock-lined seepage channels that had not previously known to occur in the Willamette Valley was documented as well. Baseline descriptive data were collected regarding the dominant native grasses, and a checklist of the native grassland flora of Marion County, comprising 115 species, was compiled (Appendix I).

## DISCUSSION

Soil surveys are just one of a variety of available tools for performing inventories of rare species and potential natural areas. In recent years, soil surveys have been utilized in a variety of contexts for inventory work, such as an inventory of sand prairies in Newaygo County, Michigan (Chapman and Crispin, 1984), where isolated grasslands are characterized by soils of Sparta loamy sand. In New York, Carlisle muck soils have been found to be indicators of fen habitats (Carol Reschke, pers. comm.), and in Austin, Texas, soil surveys have aided in the identification of relict blackland prairie remnants (Chuck Sexton, pers. comm.). Soil surveys have also been used to identify potential habitat of the small whorled pogonia (*Isotria medeoloides*) in New Hampshire, (Frank Brackley, pers. comm.), the grassland habitats of *Aster curtus* in western Washington (Alverson, 1983), and assemblages of rare plants in South Carolina that are associated with soils of the Iredell soil series (Nelson, 1987).

In this study, the usefulness of the soil survey in the effort to locate native grassland remnants was due, in part, to some unique attributes of the primary targeted map unit, Stayton silt loam, and the grassland-seepage channel mosaics that often were found. Still, many aspects of this study may be applicable to the use of soil surveys for natural area inventory work in any context.

Several strengths of the soil survey method emerge from this analysis. Readily available from the Soil Conservation Service, often at no cost, soil surveys provide simple and inexpensive access to several types of data, including soil-vegetation relationships, land uses, and successional patterns. Soil surveys provide a systematic means of

inventory, since aerial photographs provide a complete overview of the region under examination. By focusing on the specific soil map units most likely to produce results, efforts are concentrated and efficiency of labor and use of funds is increased. Field work based on soil surveys is more likely to be comprehensive, since it leads to tracts located away from easily surveyed roadsides.

The inherent drawbacks of soil surveys must also be recognized, however. Overall usefulness of the soil survey method may vary with the quality and accuracy of published maps. Out-of-date aerial photographs may not reflect current land uses, as in the case of the Starlight Road site described above. Another problem is that the resolution of the aerial photographs used in soil surveys is generally not high, so that fine details may not be evident. As a result, certain types of land use may not be distinguishable on aerial photos. In this study, for example, it was often not possible to use soil maps to distinguish between native grasslands and ungrazed pastures dominated by exotic species. This problem may be remedied in some cases by using higher resolution aerial photographs in conjunction with soil surveys in the future.

When focusing on anomalous soil types, such as the shallow and droughty Stayton silt loam, there may be concern that the natural vegetation occurring on these sites is also anomalous, and not representative of typical pre-settlement vegetation of the region. To some extent, this may be true, but it should be pointed out that boundaries delineating soil map units are abstractions, generalizations of soil mosaics that exist in the field. Land uses, by contrast, tend to be relatively homogeneous over a given tract of land. Because of this, portions of tracts typified by the anomalous soil type may be relatively typical, but protected from destruction because the predominance of poor soils precluded the application of intensive agricultural practices. In essence, anomalous soils can provide protection for pockets of more typical soils and the vegetation they support.

## CONCLUSIONS

Inventory work is the cornerstone of any natural areas program. By employing an existing data base that provides insights into soil-vegetation relationships, information can be extracted about a variety of categories of natural heritage elements ranging from habitats of rare species to remnant occurrences of once extensive vegetation types. This particular study was successful in targeting specific sites having a high potential for harboring significant remnants of native vegetation. This was accomplished in a landscape largely altered from its natural state by the activities and needs of the dominant human culture. Our specific approach may be particularly useful in other areas where agriculture has greatly reduced the extent of pre-settlement vegetation types, but the general principles should be applicable everywhere.

## ACKNOWLEDGMENTS

I thank Kenton Chambers, Robert Frenkel, Jimmy Kagan, John Nelson, and John White for assistance during the course of this study. Funding for field work was provided by the Oregon State University Herbarium, and by a grant from the Mazamas Research Committee.

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# Appendix 1.

## A Partial Checklist of the Native Grassland Flora of Marion County, Oregon.

### ISOETACEAE

*Isoetes nuttallii* A. Br.

### OPHIOGLOSSACEAE

*Botrychium multifidum* (Gmel.) Trevis

### POLYPODIACEAE

*Pteridium aquilinum* (L.) Kuhn var. *pubescens* Underw.

### PINACEAE

*Pseudotsuga menziesii* (Mirb.) Franco var. *menziesii*

### FAGACEAE

*Quercus garryana* Dougl.

### SANTALACEAE

*Comandra umbellata* (L.) Nutt. ssp. *californica* (Eastw.) Hitchc.

### PORTULACACEAE

*Claytonia perfoliata* Willd.

*Montia fontana* L. var. *tenerrima* (Gray) Fern. & Wieg.

*Montia linearis* (Dougl.) Greene

*Montia parvifolia* (Moc.) Greene var. *parvifolia*

### RANUNCULACEAE

*Aquilegia formosa* Fisch.

\**Delphinium oregonum* Howell

*Delphinium trolliifolium* Gray

*Ranunculus flammula* L.

*Ranunculus occidentalis* Nutt.

*Ranunculus orthorhynchus* Hook. var. *orthorhynchus*

*Ranunculus uncinatus* D. Don var. *parviflorus* (Torr.) Benson

### BRASSICACEAE

\**Cardamine penduliflora* Schulz

### SAXIFRAGACEAE

*Lithophragma parviflora* (Hook.) Nutt.

*Saxifraga integrifolia* Hook. var. *integrifolia*

*Saxifraga oregana* Howell var. *oregana*

### ROSACEAE

*Crataegus douglasii* Lindl.

*Fragaria vesca* L. var. *crinita* (Rydb.) Hitchc.

*Fragaria virginiana* Duchesne var. *platypetala* (Rydb.) Hall

*Potentilla gracilis* Dougl. var. *gracilis*

*Sanguisorba occidentalis* Nutt.

### FABACEAE

\**Lathyrus holochlorus* (Piper) Hitchc.

*Lotus micranthus* Benth.

*Lotus pinnatus* Hook.

*Lotus purshianus* (Benth.) Clements & Clements

*Lupinus micranthus* Dougl.

*Lupinus polyphyllus* Lindl.

*Psoralea physodes* Dougl.

*Thermopsis montana* Nutt.

*Trifolium tridentatum* Lindl.

*Trifolium variegatum* Nutt.

*Vicia americana* Muhl.

### GERANIACEAE

*Geranium oregonum* Howell

### LINACEAE

*Linum digynum* Gray

### MALVACEAE

\**Sidalcea campestris* Greene

*Sidalcea virgata* Howell

### HYPERICACEAE

*Hypericum anagalloides* C. & S.

*Hypericum formosum* Kunth var. *scouleri* (Hook.) Hitchc.

### VIOLACEAE

*Viola adunca* Sm.

*Viola nuttallii* Pursh var. *praemorsa* (Dougl.) Wats.

### ONAGRACEAE

*Boisduvalia densiflora* (Lindl.) Wats. var. *densiflora*

### APIACEAE

*Ligusticum apiifolium* (Nutt.) Gray

\**Lomatium bradshawii* (Rose) Math. & Const.

*Lomatium dissectum* (Nutt.) Math. & Const. var. *dissectum*

*Lomatium utriculatum* (Nutt.) Coult. & Rose

*Sanicula bipinnatifida* Dougl.

*Sanicula crassicaulis* Poepp. var. *crassicaulis*

### ERICACEAE

*Vaccinium caespitosum* Michx.

### PRIMULACEAE

*Centunculus minimus* L.

*Dodecatheon hendersonii* Gray

*Dodecatheon pulchellum* (Raf.) Merrill

### GENTIANACEAE

*Centaurium muhlenbergii* (Griseb.) Wight

*Gentiana sceptrum* Griseb.

### APOCYNACEAE

*Apocynum androsaemifolium* L.

### POLEMONIACEAE

*Gilia capitata* Sims

*Linanthus bicolor* (Nutt.) Greene var. *bicolor*

*Microsteris gracilis* (Hook.) Greene

*Navarretia intertexta* (Benth.) Hook. var. *intertexta*

### BORAGINACEAE

*Myosotis laxa* Lehm.

*Plagiobothrys figuratus* (Piper) Peck

### LAMIACEAE

*Prunella vulgaris* L. var. *lanceolata* (Barton) Fern.

*Trichostema oblongum* Benth.

### SCROPHULARIACEAE

*Collinsia grandiflora* Lindl.

*Gratiola ebracteata* Benth.

*Linaria canadensis* (L.) Dumont var. *texana* (Scheele) Pennell

*Mimulus guttatus* DC.

*Orthocarpus pusillus* Benth.

*Veronica peregrina* L. var. *xalapensis* (H.B.K.) St. John & War.

### OROBANCHACEAE

*Orobanche uniflora* L. var. *minuta* (Suksd.) Beck

### RUBIACEAE

*Galium aparine* L.

*Galium boreale* L.

### CAPRIFOLIACEAE

*Symphoricarpos albus* (L.) Blake var. *laevigatus* Fern.

### VALERIANACEAE

*Plectritis congesta* (Lindl.) DC.

CUCURBITACEAE

*Marah oreganus* (T. & G.) Howell

CAMPANULACEAE

*Downingia yina* Appleg.

*Heterocodon rariflorum* Nutt.

ASTERACEAE

*Achillea millefolium* L.

*Aster curtus* Cronq.

\**Erigeron decumbens* Nutt. var. *decumbens*

*Eriophyllum lanatum* (Pursh) Forbes var. *lanatum*

*Gnaphalium palustre* Nutt.

*Hieracium cynoglossoides* Arv.-Touv.

*Madia exigua* (J.E. Smith) Gray

*Psilocarphus elatior* Gray

*Solidago canadensis* L. var. *salebrosa* (Piper) Jones

JUNCACEAE

*Juncus kelloggii* Engelm.

*Juncus patens* E. Meyer

*Juncus tenuis* Willd.

*Luzula campestris* (L.) DC. var. *multiflora* (Ehrh.) Celak.

CYPERACEAE

*Carex tumulicola* Mack.

*Eleocharis ovata* (Roth) R. & S.

POACEAE

*Agropyron caninum* (L.) Beauv. ssp. *majus* (Vasey) Hitchc.

*Agrostis microphylla* Steud.

*Agrostis exarata* Trin.

*Agrostis exarata* Trin.

*Bromus horridus* L. & A. var. *carinatus*

*Danthonia californica* Boland.

*Deschampsia cespitosa* (L.) Beauv. var. *cespitosa*

*Deschampsia danthonioides* (Trin) Munro

*Elymus glaucus* Buckl. var. *glaucus*

*Festuca idahoensis* Elmer var. *idahoensis*

*Festuca rubra* L. var. *rubra*

*Koeleria cristata* Pers.

*Panicum occidentale* Scribn.

*Poa scabrella* (Thurb.) Benth. ex Vasey

LILIACEAE

*Allium amplexens* Torr.

*Brodiaea congesta* Smith

*Brodiaea coronaria* (Salisb.) Engl.

*Brodiaea hyacinthina* (Lindl.) Baker

*Calochortus tolmiei* H. & A.

*Camassia leichlinii* (Baker) Wats. var. *suksdorfii* (Greenm.) Hitchc.

*Camassia quamash* (Pursh) Greene var. *maxima* (Gould) Hitchc.

*Erythronium oregonum* Appleg.

*Fritillaria lanceolata* Pursh

*Veratrum californicum* Durand var. *caudatum* (Heller) Hitchc.

*Zigadenus venenosus* Wats. var. *venenosus*

IRIDACEAE

*Iris tenax* Dougl.

*Sisyrinchium idahoense* Bicknell

\*An asterisk denotes a Willamette Valley endemic.

# Preliminary Results of a Program to Monitor Plant Species for Management Purposes

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**Abstract:** The Illinois Department of Conservation has implemented a program of monitoring and data analysis aimed at guiding conservation and management of endangered plants. The system unites annual census and demographic data with information on climate, disturbances, management activities and other factors important in developing guidelines. The system is implemented by District Biologists, many of whom are not botanists. This paper summarizes methods and presents findings of significance concerning the management of 15 plant species.

Pages 113 - 116. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The Illinois Department of Conservation's Natural Heritage Division has been monitoring and managing special plant species, mostly endangered or threatened ones, since the early 1980's. In 1985 a standardized monitoring program, including demographic plots and census of whole populations, was initiated.

This system is designed to meet the needs of managers to document the status of special plant species and to understand in broad terms the probable causes for changes in their numbers or vigor. This information guides management and may reveal areas where detailed scientific studies are needed to solve specific management problems.

The purpose of this paper is to give managers a brief overview of our methods and the kind of management-related information that has been obtained to date for 15 species. The system provides good management information at a much lower cost than detailed life history and ecology studies.

## METHODS

For demographic monitoring, two standard plots (Schwegman, 1986) are used. Each is permanently marked in the field by two half-inch diameter pipes. Individual plant markers of plastic or aluminum are usually placed near each plant. Age class, reproductive class and easily noted characters that reflect vigor changes in individual plants are established for the species. Initial observations are made on the plot's soil and substrate, soil moisture, slope, aspect, natural community, plant community, crown cover and associated plants. Annual observations are recorded on the condition of individual plants, climate, disturbances (including fire), grazing/trampling, succession/competition, disease, apparent mortality factors, flowering activity, pollinator activity, seedling establishment, seed production and parasitism/predation (plants and seeds).

Population census normally consists of total counts by age and reproductive class. Where populations exceed 200 individuals, total counts can be replaced by one or more permanently marked circular sample plots at the option of the worker. In these cases, the total dimensions of the population are measured and the total population is estimated. If no demographic plot is being monitored in conjunction with a censused population, the climatic and other annual observations usually associated with a demographic plot are recorded.

The demographic monitoring and/or census are conducted annually for a given species or population, usually at the peak of flowering. For some species, such as sedges, the sampling is carried out at the time of peak fruiting. For some plants, monitored during the flowering season,

a second monitoring at fruiting time is also conducted.

Population data are analyzed annually and compared with observations on climate, disturbances (including fire), management, disease and reproduction to determine probable causes of shifts in population size and vigor. This analysis, together with a review of available literature on the species, is the basis for any active manipulation or management of the population on a year to year basis.

In Illinois, the program is implemented by eleven District Heritage Biologists in coordination with the Staff Botanist. These biologists are the field managers for the Division of Natural Heritage and are mostly trained either as botanists or wildlife biologists. Monitoring one species averages two person days of field time and two days of office time annually. Persons interested in implementing a similar program should write to the author for a copy of the procedures and forms.

## RESULTS

This paper presents findings to date for 15 species. While some of these plants have had considerable research conducted on them, I have not included these findings in the species accounts and no literature is cited. While utilization of all available literature is part of the procedure for developing management guidelines, the purpose of this paper is to present only the information that has been derived from this monitoring program. The intent is to show the manager what can be expected if a similar program is initiated. The information should also be helpful to persons involved in managing these particular species. The taxonomy follows Kartesz & Kartesz (1980), except for *Boltonia decurrens* (Torr. & Gray) Wood, which follows Schwegman & Nyboer (1985).

In addition to species for which accounts are included in this report, eight other species are being monitored, but only the first year's data have been gathered. These species are *Cirsium hillii* (Canby) Fern., *Cladrastis kentuckea* (Dum.-Cours.) Rudd, *Cypripedium reginae* Walt., *Hudsonia tomentosa* Nutt., *Platanthera leucophaea* (Nutt.) Lindl., *Silene regia* Sims, *Styrax grandifolia* Ait. and *Veratrum woodii* J. W. Robbins.

## SPECIES ACCOUNTS

Mead's milkweed (*Asclepias meadii* Torr. is a federally Threatened tallgrass prairie herb that is being monitored on the Shawnee National Forest in Southern Illinois. Its vigor and flowering are apparently related to moisture availability, and more plants seem to emerge following a spring burning than in unburned habitats. No seed has been

produced in seven years of observation. Hand pollination has failed, indicating, with other vigor observations, that plant condition (related to available light, moisture and grazing) is an important factor in seed production failure. Pollination is accomplished by bumblebees. Plants appear long-lived, with one plant having been known for 28 years. The habitat has been undergoing succession from prairie to xeric forest in the absence of fire, but this trend has been reversed by recent prescribed burning and selective tree removal. One small population is in sharp decline, probably due to shading. As many as 18 stems in three populations have been monitored and censused annually from 1983 to 1987.

Forked aster (*Aster furcatus* Burgess) is a midwest endemic of moist, lightly shaded habitats that is Threatened in Illinois. It is a rhizomatous perennial herb that sends up aerial shoots mostly from different active growth points on its rhizome each year. Plants monitored in northern Illinois exhibited a slight reduction in vigor in response to a winter, spring and early summer drought. One population, at a site drier than is typical, failed to flower at all under drought conditions. The disturbance caused by trails favors vigor and flowering of the species, probably due to increased light and reduced competition. The population is stable. Approximately 35 stems were monitored, and 270 were censused in the late summers of 1986 and 1987.

Tennessee milk vetch (*Astragalus tennesseensis* Gray) is a state endangered perennial herb of gravel prairies. It is monitored at its only remaining midwestern population in central Illinois. It has a mortality rate of about 25% annually, with drought in summer killing seedlings and stem borers killing mature plants. Vigor (size) is not a reliable indicator of age as large plants may be small in following years. A spring fire does not injure plants, but aids seed germination by scarifying some seeds. However, fire also destroys some seeds in the capsules that burn. Seedlings tend to survive better in the partial shade at the edges of prairies; however, plants growing in these ecotones are generally poor seed producers because they are in shade at mid-morning and apparently are largely overlooked by pollinators. The primary constraint on population growth is adequate spring and summer moisture for seedling germination and survival. The monitored population increased after the moist spring and summer of 1986. The population is healthy but dynamic due to rapid turnover of individuals. Active management in the form of prescribed burning and physical brush removal appears effective, and will be continued. The effect of fall fires needs to be determined. Approximately 30 individuals have been monitored and up to 120 censused annually in the late springs of 1985, 1986 and 1987.

Kitten-tails [*Besseyia bullii* (Eat.) Rydb.] is a midwestern, endemic, perennial herb of open to lightly shaded sand and gravel soils. A population of 142 plants declined 35% in one year at a site monitored in central Illinois. While the overall population declined, the number of flowering plants increased. Of six marked plants flowering in one year, none flowered the next. Seventeen new flowering plants in the plot the following year had been marked as sterile plants during the previous year. Sterile plants were not all juveniles, since some had flowered the first year and were sterile the next. One excavated plant showed no evidence of asexual reproduction by rhizomes. The relatively high mortality occurred in a severe drought year, and losses were higher for plants in competition with invading shrubs. A spring fire in part of the plot had no detrimental effect. In fact, mortality was much higher in the unburned part of the plot.

Decurrent false aster [*Boltonia decurrens* (Torr. & Gray) Wood] is a federally Threatened plant endemic to the Illinois River and Middle

... it is a large perennial herb of moist open ... It is taxonomically distinct from *B. asterionoides* L. Her. based on leaf characters and lack of rhizomes. Plants are perennial from basal offsets, such that second and third-year plants have a clumped appearance. Asexual (offset) reproduction may persist for only two or three generations, making populations dependent on frequent recurrence of conditions suitable for seedling establishment. Drought conditions that prevented flooding and silt deposition in the spring of 1987 resulted in a healthy population on one sandy bottomland lake shore. All other surviving populations of the species are in man-disturbed habitats. The center of distribution for the species is the region of greatest Wisconsin glacial outwash deposition along the Illinois River. Outliers occur downstream, especially around St. Louis. In most years plants exist only in managed or man-disturbed (plowed) habitats. Survival of this species can be assured only through active management, such as periodic plowing and fallowing of bottomland fields. Twelve plants were monitored in the late summer of 1982 and 1983. An aerial survey was conducted in 1984, and annual census of all known populations has been conducted since 1984.

Bluehearts (*Buchnera americana* L.) is a sporadically distributed, dry prairie herb. It appears sometimes to be a perennial, but most often plants are annual. Twenty flowering plants were marked, but have never reappeared in three subsequent growing seasons. In this period, only a single flowering Blueheart appeared in the plot, and it was 20 cm from the nearest previously known plant. Of three excavated flowering plants outside of our plot, two had fine root systems typical of annuals, but one had a thickened root with remnants of 2 previous years' stems remaining on the root crown. It was not possible to excavate the entire perennial root to see if it had a parasitic attachment. Nodules are prominent on the fine roots of the plants that appear to be annual. Fire may stimulate seedling establishment, but we have not been able to burn the central Illinois plot to assess this. Twenty-one plants have been monitored in the early summers of 1985 through 1988.

Mohlenbrock's umbrella sedge (*Cyperus grayioides* Mohlenbr.) is a perennial sedge of loose blow sand, restricted to dunes in Illinois, Louisiana and Texas. It has declined in numbers and vigor over a two year period in a central Illinois plot. Some mortality resulted from burial of plants by oak leaves that collected around a black oak sprout near the plot, and one plant may have died of leaf blight. A decline in vigor of individual plants in 1987 was correlated with severe drought. This species reproduces asexually by offset bulbs and by runners up to 10 cm long. Seedling establishment is in bare sand. This species and *Carex pensylvanica* Lam., which grows with it, are severely infected with a leaf disease that appears to be a blight. This population is expected to continue to decline, and it will be used to monitor the effects of vegetation management and soil disturbance before it is extirpated due to succession. Twelve plants were monitored in the summers of 1985, 1986 and 1987.

Leafy prairie-clover [*Dalea foliosa* (Gray) Barneby] is a state Endangered species. It grows in dolomite prairies in northern Illinois and in the limestone glades of Tennessee and Alabama. It is a short-lived perennial with about 50% annual mortality for juveniles and flowering plants, but with abundant seedling recruitment. No asexual reproduction has been noted. Vigor of individual plants is not a good indicator of age, since the number of flowering heads may decline from one year to next. Plants in the wild usually begin flowering on about their third year, but in cultivation they can flower abundantly (32 flower heads) in the first year. Rabbit grazing destroyed 19% of 96 wild flowering stems by the peak of anthesis and also devastated some cultivated populations. Spring

burning did not harm the species. Seeds germinate abundantly without scarification, and the species is easily propagated. Frost heave and drought seem to be the principal cause of seedling and juvenile mortality. Density in our demographic plot was 12.75 juveniles and 1.77 flowering plants per m<sup>2</sup> in 1987. The monitored population is increasing at present. One hundred fifteen plants were monitored (others censused) in mid-summer during 1986 and 1987.

Prairie white trout-lily [*Erythronium albidum* Nutt. var. *mesochoreum* (Knerr.) Rickett] is a state Endangered species that is monitored in central Illinois at its only known population east of the Mississippi River. This early-blooming, perennial herb of mesic loam prairie has more than doubled its number of flowering individuals after two spring, prescribed burns. Juvenile plants have declined slightly by gaining fewer seedlings through recruitment than grow into flowering individuals or are lost to mortality. No flowering plants have died or gone dormant in the plot. Juvenile mortality is 15% annually. Some seedlings germinate early in the spring, but they are very difficult to see at monitoring time, even on burned soil. Mortality factors for juvenile and seedling plants are unknown. There is no evidence of asexual reproduction in this plot. Fifty plants were monitored (others censused) in spring (1986 and 1987).

Kankakee mallow [*Iliamna remota* Greene] is a state Endangered species that is endemic to a single Kankakee River island in northern Illinois. It is a perennial herb of well drained soils in thin woods and on open river banks. Individual plants live at least up to five years (possibly to a much greater age) in the wild and can spread asexually by root sprouts. Seeds can survive for ten years, and possibly much longer, in the soil. The heat from burning brush piles and logs (not herbaceous vegetation) scarifies buried seed and stimulates germination. Mature plants are typically not injured by fire, but they can be killed by hot, log fires or bush fires if they occur very close to them. The plants increase in numbers with annual, early spring burning. They may be subject to severe selective browsing by deer just prior to, and during flowering, which can greatly reduce seed production in a given year. Shading and crowding by the exotic shrub Amur honeysuckle [*Lonicera maackii* (Rupr.) Maxim.] and other woody and herbaceous vegetation can stunt and possibly kill plants. Up to eleven separate stands have been censused in the early summer, annually from 1983 to 1987.

Prairie bush-clover [*Lespedeza leptostachya* Engelm.] is a federally listed, Threatened species. It is a prairie forb of well-drained, loamy to sandy soils in northern Illinois and adjacent Iowa, Minnesota and Wisconsin. No evidence of asexual reproduction by roots or rhizomes has been found. Vigorous flowering plants can return as smaller sterile plants the following year. Plants can disappear for a growing season and return the following year. Spring fire seems to stimulate multiple stem production from some single root crowns, although exposed (elevated) root crowns can also be killed by spring fires. Drought stress reduces average plant height and can cause abortion of flowers and failure of seed set. Over 90% of the plants studied have cleistogamous flowers only, and no seeds have been observed developing from chasmogamous flowers. Seedlings have been observed in bare soil areas created by fire. Plants are easily grown from seed. Natural populations most frequently occupy thin, loam soils with rapid internal drainage into underlying gravel or sand. Plants usually grow below the crests of hills or in slightly lower areas in level terrain, where they receive some moisture run-off. Some plants suffer from chlorosis, which may be disease or nutrient related, but does not appear fatal. Fifteen plants were monitored in mid-summer 1985, 1986 and 1987 and an additional 32 plants were monitored in 1986 and 1987.

Bladderpod [*Lesquerella ludoviciana* (Nutt.) S. Wats.] is a state Endangered species that is disjunct at one central Illinois location from the sandhills of Nebraska. Nine of 39 monitored plants died in one year (five flowering and four sterile plants). Three were killed by fire, two by burrowing pocket gophers, one was trampled by deer and three died of undetermined causes. The plants may have more than one vegetative crown from one root, apparently due to their response to burial by shifting sand. The crowns remain close, and since new root systems do not develop this is not considered asexual reproduction. Seeds germinate and seedlings develop in a mulch of leaves and old *Cassia fasciculata* Michx. pods which accumulate on flat open areas, in depressions and in the lee of grass clumps. The mulch apparently provides the moisture necessary for seed germination and protects seedlings from being uprooted by shifting sand. Fire may stimulate seed germination, but it does kill some larger plants. A 100 m<sup>2</sup> census plot that was not burned increased 66% during the same season in which a similar, but burned, plot declined 10%. Vigorous flowering plants can return as less vigorous individuals the following year. One plant that had four inflorescences and had lost its leaves (to grazing?) returned as a small sterile plant the next year. Thirty nine plants were monitored and 160 were censused in the springs of 1986 and 1987.

Heart-leaved plantain (*Plantago cordata* Lam.) is a state Endangered species that occurs on dynamic, gravelly streambanks in small forested watersheds. This perennial herb also grows in moist soil of seep-springs, where it has been monitored by census in central Illinois. Major flood events can produce annual mortalities of 56% of adult and juvenile plants in stream habitats. This mortality is mainly due to burial of plants by shifting gravel and uprooting of plants. At the same time, populations in seep-springs can increase by 40%. The species can be extirpated from stream habitats by floods, especially at the upstream end of occupied territory. Stress from having leaves shredded in a flood can prevent mature plants from flowering the following year and compound mortality through reduced seed production. While the species survives occasional severe flood mortality, human activities that increase runoff volume and speed in a watershed theoretically might lead to extirpation of populations. This is supported by greater mortality in larger, more disturbed watersheds as opposed to adjacent smaller, more natural ones. A population with a maximum of 1,320 plants was censused in 1986 and 1987.

Buffalo clover (*Trifolium reflexum* L.) is a state Endangered species with a single known population in Illinois. It is an herb of steep slopes and thin soil in open upland woods at its single western Illinois locality. This species appears to be a biennial or monocarpic perennial, but no plants have flowered, nor have any seedlings survived from one year to the next since establishment of the plots. All plots included flowering plants when established, but all flowering plants were dead the following year. In some years no flowering plants are visible in the entire population. Seedlings are visible in bare, steep clay soil and on mossy, flat, thin soil areas in late spring each year. Seedling mortality might be due to drought, frost heave or erosion. Forty-four plants have been monitored in the late spring during 1985, 1986 and 1987.

A trillium (*Trillium cuneatum* Raf.) is a state Endangered perennial herb of mesic forests in southern Illinois. It has a single-leaf seedling stage that lasts at least two growing seasons; plants then undergo a sterile three-leaf juvenile stage of unknown duration. Seedling establishment is greatest where bare soil is exposed by gaps in the leaf litter. Two of eight flowering plants and two of six large, juvenile plants disappeared in one year. This apparent mortality seemed due to competition from Japanese honeysuckle. One plant may have been killed by a falling log. Thirteen per cent of 29 seedlings in the plot died

in one year. A total of 50 plants were monitored and 60 during the early spring in 1986 and 1987.

#### ACKNOWLEDGMENTS

I am pleased to acknowledge Heritage Biologists Ed Anderson, Margaret Cole, William Glass, Randy Heidorn, James Heim, Robert Lindsay, William McClain, Randy Nyboer, Lawrence Stritch, Robert Szafoni and Andrew West for their participation in the monitoring program. I would also like to thank Marcella DeMauro, Jean Karnes, Julius Swayne and Renee Thomas for assistance in the field.

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# Monitoring *Gentiana saponaria* L. (Gentianaceae), an Endangered Species in Ohio.

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**Abstract:** *Gentiana saponaria* is a herbaceous perennial that is widespread throughout the southeastern United States and northward to the southern Great Lakes. In Ohio, the only extant populations of this endangered species are reported from Lucas County. The largest of the three known populations occurs at Lou Campbell State Nature Preserve and is being monitored by the Ohio Division of Natural Areas and Preserves to learn more about its life history and implications for management. Baseline demographic data have been collected for this population since 1984. Information, such as reproductive status, stem height, flowers per stem, and seed capsules per stem were recorded for a sample of the population. It was discovered that only a small percentage of the capsules successfully disperse seed. Most seed capsules were severely damaged by seed predation activity of a moth larva, *Endothenia hebesana* (Walker), that apparently causes the seed dispersal mechanism to become nonfunctional.

Pages 117 - 118. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

*Gentiana saponaria* L. (Section Pheumonanthae), soapwort gentian, is an herbaceous perennial that flowers from late September to early October in Ohio, where it is rare. It has 1-7 erect or decumbent stems, 10-60 cm in height that arise from a short rootstock (Pringle, 1967). Its dark green leaves are borne in pairs along a glabrous stem. The corolla lobes in *G. saponaria* are wider than the fringed corolla plaits, and the lobes are usually incurved. The seeds are winged and are dispersed as the capsule extends upward beyond the corolla tip and splits lengthwise. These plants are similar to those of other species in the section and may hybridize with them. *Gentiana saponaria* is a species distributed widely from the Coastal Plain of Long Island south into the Piedmont region of North Carolina and from northern Florida sporadically west to Texas then north to southern Michigan. It is known from a wide range of habitats including wet or seasonally wet swamps, thickets, prairies, meadows and open woods. It is most common in fairly open sites with filtered sunlight.

*Gentiana saponaria* is an endangered species in Ohio, presently recorded from only three locations in the Whitehouse 7 1/2" topographic quadrangle in Lucas County. Although there are pre-1940 records for the species in Clermont, Jackson, and Ross counties in southern Ohio, it is now apparently extant only in Lucas County in northwestern Ohio. The known locations in Ohio are in the Oak Openings area near Toledo, a region of sandy deposits laid down by post-glacial Lake Warren. This sand belt, approximately 25 miles long and seven miles wide, lies on an impervious, clay base resulting in poor drainage. The Oak Openings region was originally surrounded by swamp forest with scattered oaks on the dunes and sandy plateaus (Campbell, 1979). The fine quartz sand is organically enriched and generally acidic. Between the raised sand ridges lie wet meadows or prairies where the water table is near the surface.

The largest of the three known populations of *G. saponaria* in Ohio occurs at Lou Campbell State Nature Preserve. We have been monitoring this population of 700-750 plants since 1984 to record changes in the population in an attempt to determine variables in the species' survival potential and determine satisfactory management strategies. The habitat of the species at Campbell Preserve is a seasonally wet sedge meadow with invading woody species such as *Acer rubrum* L., *Alnus rugosa* (DuRoi) Spreng., *Populus deltoides* Marsh. and *Salix exigua* Nutt. Other species associated with *Gentiana saponaria* at this site include *Aletris farinosa* L., *Calopogon tuberosus*

(L.) BSP., *Drosera intermedia* Hayne., *Eleocharis elliptica* Kunth, *Liatris spicata* (L.) Willd., *Rhynchospora capitellata* (Mich.) Vahl., *Solidago remota* (Greene) Fries., *Sphagnum subsecundum* Nees ex Sturm., *Spiraea tomentosa* L., *Vaccinium augustifolium* Ait. and *Viola lanceolata* L.

Demographic data collection on *G. saponaria* at Campbell Preserve was begun in 1984. The population was divided into three subpopulations on the basis of plant distribution and minor habitat differences. Plants were tagged with aluminum, numbered tags and mapped. Since that time, data have been collected in October, when plants are in flower, and again in November, after they have set seed. Data were collected for all stems in 1984 and 1985, and, in 1986, we began sampling random meter-square plots within the subpopulations and thus collected data on approximately 40% of the population. For the purposes of this study it was impractical to distinguish between ramets and genets, therefore we have collected data for individual stems since 1986. Data include reproductive condition (vegetative or flowering), number of flowers per stem and position on the stem (terminal or axillary), number of seed capsules per stem, and height (for a smaller sample).

Approximately 300 tagged stems were sampled each year during 1984-1987 (Table 1). Between 65-79% of these stems flowered, while 21-35% were vegetative only or grazed, perhaps by deer and rabbits. The number of flowers per stem ranged from one to 15, with an average of three flowers. Most flowering stems (94-97%) had only terminal flowers, but larger plants may have flowers in the upper 3-4 axils. We discovered in 1984 that one out of every 16 (6%) seed capsules was functional while the other 15 (94%) were nonfunctional and appeared decayed. With the help of lepidopterists, Eric Metzler from ODNR and J.F. Gates Clarke from the Smithsonian Institution, it was determined in 1985 that the seed capsules were damaged by a moth larva, *Endothenia hebesana* (Walker). The larva of this widely-distributed species feeds on seeds of many plants (pers. comm., J.F. Gates Clarke). Predation by this larva on the seeds of *G. saponaria* apparently causes the seed dispersal mechanism to become nonfunctional; most of the seeds are destroyed by the larva or decay in response to the activity of the larva. When the capsule is infested by the larva, seeds often do not mature or the dispersal mechanism may not function to allow seed dispersal by wind. The flowers of many gentians are subject to insect infestation (pers. comm., B.K. Andreas). This is particularly significant in Ohio, since the percentage of

**Table 1.** *Gentiana saponaria* monitoring project, Lou Campbell State Nature Preserve.

	# OF STEMS SAMPLED	% OF STEMS FLOWERING	1984-1987		% OF SEED CAPSULES NONFUNCTIONAL	AVERAGE* HT. (cm)
			% OF STEMS NONFLOWERING OR GRAZED	% OF SEED CAPSULES FUNCTIONAL		
1984	307	79%	21%	6%	94%	NR
1985	329	70%	30%	8%	92%	NR
1986	301	65%	35%	10%	90%	19
1987	301	75%	25%	7%	93%	25

\*height data were not recorded (NR) in 1984-85; 40-50 stems were sampled for height in 1986-87.

nonfunctional seed capsules in *Gentiana saponaria* is consistently high at our site each year (90-94%).

The effects of predispersal seed predation on plant recruitment are poorly understood, although this type of seed predation by insects is common (Louda 1982). Louda (1983) found that insect seed predation can be the critical factor limiting population recruitment for certain *Haplopappus* species. It has also been suggested that recruitment may be limited by a lack of suitable sites for germination and juvenile growth, rather than by seed supply (Parker 1985). It is unclear which explanation is applicable to *Gentiana saponaria*, but since its seeds are difficult to germinate (pers. comm., B. Parsons) and the habitat at Campbell Preserve appears to be succeeding to woody species, Parker's explanation may be more relevant.

Monitoring projects on rare species often raise more questions than answers. Despite four years of data collection, there are still questions to consider regarding the long-term survival of *Gentiana saponaria* in Ohio. Some issues for further investigation include seedling recruitment and effects of insecticide on a sample of plants. We began manual removal of woody species, as well as controlled burning, in 1987 at Campbell Preserve. Continued monitoring and management of *Gentiana saponaria* at this preserve should ensure its perpetuation as a part of Ohio's flora.

#### ACKNOWLEDGMENTS

I want to thank Allison W. Cusick of the Ohio Division of Natural Areas & Preserves for his help with this paper. My research is supported by the Ohio Division of Natural Areas & Preserves, Ohio Department of Natural Resources.

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# Progress of Rare Plant Monitoring, Assessment and Recovery in Alberta, Canada

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**Abstract:** Although there is no federal legislation in Canada comparable to the Endangered Species Act of the United States, the identification and protection of rare plant species in Canada have been ongoing since 1973. Plants are generally under provincial jurisdiction, but only two provinces have endangered plant legislation that protects a total of seven species. In Alberta, a quarter of the flora (360 species) is included on the rare listings. Since 1986, work has been ongoing to determine the status of these species in the province and to raise the profile of rare and endangered plants, since conservation efforts to date have largely centered on protection of ecosystems through establishment of natural areas and ecological reserves.

Pages 119 - 121. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

### Rare Plant Conservation in Canada:

While the passage of the Endangered Species Act in 1973 served as impetus for the determination and monitoring of rare species, including plants, in the United States, there has been no comparable legal action in Canada (Butler, 1986). Nevertheless, work has been ongoing to determine rare plant species in Canada.

The Canadian National Museum of Natural Sciences began a rare and endangered plants project in 1973 to first develop lists of the rare plants in the provinces and territories and to then use these to compile a national listing of rare plants for Canada. Although lists for the provinces and territories were not yet completed, work on the national list was initiated in 1986 (Argus and Pryer, 1986).

The initial lists defined a rare plant as "one that has a small population within the province or territory. It may be restricted to a small geographical area or it may occur sparsely over a wide area" (Argus and White, 1978). No attempt was made to categorize species as endangered or threatened, because information on current, extant population sizes, and on factors such as those controlling distribution, that would be required to make that type of evaluation generally had not been available.

In 1977, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) was founded. A subcommittee for plants was formed in 1979 and the task initiated of writing status reports and assigning species to one of five categories: rare, threatened, endangered, extirpated or extinct (Argus and Pryer, 1986). To date, 53 plant species have been categorized (COSEWIC 1988). COSEWIC, however, has no legislative power.

Plant species are generally under provincial rather than federal jurisdiction in Canada (Tingley, 1987). To date, one species (*Pedicularis furbishiae* Watson) is protected under provincial endangered species legislation in New Brunswick and six are protected in Ontario: *Cypripedium candidum* Muhl. ex Willd., *Isotria medeoloides* (Pursh) Raf., *I. verticillata* (Muhl. ex Willd.) Raf., *Magnolia acuminata* L., *Opuntia humifusa* (Raf.) Raf. and *Plantago cordata* Lam.

### Rare Plant Conservation in Alberta:

In 1986, two events served to increase the profile of rare and endangered plants in Alberta. First, a workshop was held on rare and endangered species in the Prairie Provinces. The plant session of this

workshop looked critically at the status of rare plant protection in Alberta, identified many problems and recommended required action (Lee, 1987). In addition, the World Wildlife Fund of Canada initiated the "Wild West Project", a conservation action program for the prairies. These two events provided the impetus and focus for the work on rare plants that has recently taken place in Alberta.

The most recent list of rare plants for Alberta includes 360 species or 24 per cent of the Alberta flora (Packer and Bradley 1984). An evaluation of the species listed (Kershaw 1987) shows that:

1. Ten per cent occur in Alberta as populations disjunct from their main range;
2. Five per cent are limited to a local area or are restricted geographically although they may occur locally in large numbers; and
3. The remainder (85 per cent) are generally widespread but with only small populations in Alberta because the province is on the periphery of their range.

One of the first projects, initiated in 1986, was a detailed literature search on each species included on the list of rare plants for Alberta. The purpose of this project was to look at the status of each species throughout its range in order to develop priorities for further species studies (Fairbairns et al., 1987; Wallis, 1987; Wallis et al., 1987). Generally, endemic species were considered highest priority, then disjunct species, then peripheral species. This was the first step in compiling the information required to assess and manage rare plants properly in Alberta. The next phase was to locate populations of rare species, concentrating on those of highest priority, and to determine their status.

Alberta includes six natural physiographic regions. Areas with known concentrations of rare plant species include the Cordillera, the Canadian Shield, the southern grasslands and a diverse area in the southwest corner of the province where several natural regions converge.

A study area in the diverse southwest corner of Alberta was chosen that included previously-recorded locations for 28 high priority, rare species. Known locations and areas of similar habitat were identified through analysis of bedrock and surficial geology maps and visited. Detailed information on population size, location, habitat, threats and phenology was recorded for each population of a rare species located (Wallis et al., 1986).

Of the species reviewed, this study found four to be endangered in Alberta (*Allium geyeri* S. Wats., *Castilleja cusickii* Mutis ex L.f., *Cypripedium montanum* Dougl. ex Lindl. and *Iris missouriensis* Nutt.),

one to be threatened (*Astragalus lotiflorus* Hook.), 15 species to be rare but not threatened, and three species were recommended for removal from the rare listing.

In 1987, a second similar, broad study took place to locate rare plant populations concentrated on sand dunes of the prairies and Parklands that provide habitat for 30 species of rare plants. Sand dune areas were identified from air photos and field checked. Again, detailed observations on populations of rare species were made and status of species recommended. Two species studied were considered endangered: *Cyperus schweinitzii* Torr. and *Tradescantia occidentalis* (Britt.) Smyth. Three species were considered threatened: *Abronia micrantha* Torr.; *Chenopodium subglabrum* (S. Wats.) A. Nels. and *Lygodesmia rostrata* A. Gray. Most species studied were considered rare, and five were recommended to be deleted from the rare species list (Wallis and Wershler, 1988).

This study also compared the present extent of active dunes to the historical dunes and concluded that extensive stabilization has occurred. Since many of the rare plant species found in sand dune habitats depend on active or only partially stabilized dunes, it was concluded that dune stabilization constitutes a major threat to those species.

The broad studies described here have proven to be extremely important in helping to sift through the large number of potentially "rare" species and to focus in on those species and habitats most in need of protection. Work has subsequently been initiated on some of the species identified as endangered.

*Cypripedium montanum*, known from only a few locations in Alberta, occurs in small numbers here and has a low reproductive rate. Since most Alberta populations occur on public land, a report including management recommendations could be developed with some surety of implementation. This report was distributed to land managers and appropriate government personnel (Wilkinson, 1987). Now, monitoring of populations on provincial lands is to be initiated.

*Iris missouriensis* is found in small populations, restricted in Canada to southwestern Alberta. It is considered endangered because populations occur in seepage areas that are threatened by drought, cultivation, heavy grazing (although moderate grazing appears beneficial) and invasion of introduced species. One population is within a provincial park, but most are found on private lands. For this species, landowners were contacted, populations censused and a monitoring program initiated (Wallis, 1988).

To increase the level of public awareness of the need for rare plant protection, a series of public information sheets are being developed on Alberta's rare plants. Two have been printed, and eventually one on each of the endangered and threatened species will be available.

### FUTURE PROJECTS

In addition to preparation of recommendations concerning individual species, several sites with populations of rare species requiring protection have been identified through the broad studies. Protection of representative and special habitats has been the main focus of conservation activity in Alberta. There is no legislation specific to the protection of endangered plant species, whereas there is provision under the Wilderness Areas, Ecological Reserves and Natural Areas Act to protect habitats of rare plant species.

Ecological reserves are established to protect, among other things, "rare and endangered plants or animals that should be preserved" (Wilderness Areas, Ecological Reserves and Natural Areas Act, 1980). Under the same act, natural areas can be established to "protect sensitive...public land from disturbance" and the act has also been

used to protect known rare plant populations. Sites identified through the broad studies will be evaluated for possible protection under this act.

Additional broad studies are ongoing in hope of documenting a number of rare species populations to help determine their status. These are focusing primarily on the Grassland and Parkland natural regions. Considering such factors as human population densities, rate of habitat destruction and recent extinctions, among others, these natural regions have been identified as having the highest priority for protection (Cottonwood Consultants Ltd., 1983).

There is, at present, no central data base for information on rare plant species. The information collected to date will be computerized in a form that is compatible with the government of Alberta's Recreation and Conservation Information System. A micro-computer system was developed to identify recreation potential and conservation resources on public land. By incorporating the rare plant location information into this system, it will become a part of the information base used to make land management decisions.

### CONCLUSIONS

Recent studies on rare plants in Alberta will help to focus future conservation activities at the community level and at the individual species level. While the passage of endangered plant species legislation would be beneficial, Coggins and Harris (1987) suggest that one of the major problems with federal plant protection law in the United States is its emphasis on saving single species rather than broadening the focus to include habitat maintenance, enhancement and protection. It is perhaps the lack of similar legislation in Alberta that has stimulated much of the conservation work to be focused on protecting habitats and communities rather than on individual species. So, despite the lack of federal and provincial legislation to protect rare plants, work has been ongoing in Alberta to identify species requiring conservation action and in some cases such action has been initiated.

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# Attaching the New Jersey Natural Heritage Program Database to a Geographic Information System

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**Abstract:** The New Jersey Natural Heritage Program has been making use of a Geographic Information System (GIS) to produce maps of endangered species and natural community locations, so that regulatory and planning agencies can learn of these environmentally sensitive areas. Maps depicting generalized locations are designed to meet three objectives: 1) they must adequately red-flag sensitive areas so that users will contact the Natural Heritage Program if potential conflicts arise, 2) they must maintain data security to protect them from unscrupulous plant and animal collectors, and 3) they must be easily updated and reproduced. With the assistance of the GIS staff, the Natural Heritage Program has transferred data from its PC to the GIS minicomputer and produced the maps. An overview of the data transfer procedure is presented including some of the difficulties encountered.

Pages 122 - 124. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Natural Heritage Programs are often called upon to reproduce maps of endangered species and natural community locations so that regulatory agencies and planning agencies can learn of environmentally sensitive areas. A Geographic Information System (GIS) can allow a program to produce and update these maps efficiently. The New Jersey Natural Heritage Program has been making use of a GIS to produce maps of endangered species and natural community locations. The New Jersey Department of Environmental Protection operates an ARC/INFO GIS on a Prime 9955 super minicomputer with a full time staff to develop and operate the system, as well as to train users in other sections of the Department. With the assistance of the GIS staff, the Natural Heritage Program has transferred data from its PC to the GIS minicomputer and produced maps of the data for use by planning and regulatory agencies.

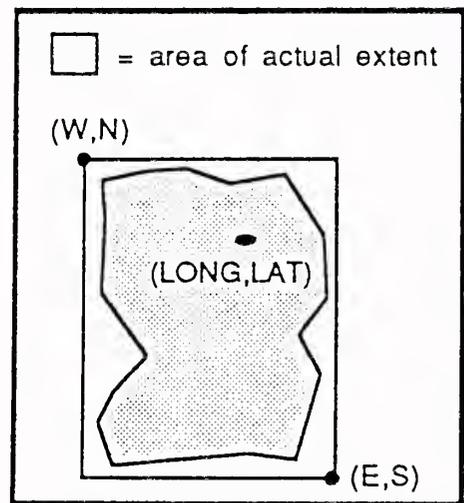
There are three requirements that this mapping system must fulfill: 1) it must adequately red-flag sensitive areas so that users contact the Natural Heritage Program if potential conflicts arise; 2) data security must be maintained, so that unscrupulous plant and animal collectors can't use them as treasure maps; 3) maps must be easily updated and reproduced.

This paper provides an overview of the data transfer procedure from PC computers to a GIS minicomputer, including some of the problems encountered. Maps currently being provided to planning and regulatory agencies are described, and the required investment in terms of staff is discussed.

## TRANSFORMATION AND TRANSFER OF DATA

Natural Heritage Programs maintain computerized map and manual files on occurrences of rare and endangered species and natural communities in accordance with methodology developed by The Nature Conservancy (1988). These occurrences of species and communities are hereafter referred to as "element occurrences" or EOs. The location of each EO is designated by the latitude and

longitude of its centrum point. When boundaries have been determined and mapped for an EO, a least rectangle is placed around the boundaries with the latitude and longitude assigned to the north, south, east and west points. This rectangle then has perimeter lines oriented north-south and east-west (see Fig. 1). Least rectangles are used in geographic searches of the database performed on the PC. Therefore, the areal extent of each EO can be approximated using the latitude and longitude coordinates for the least rectangle.



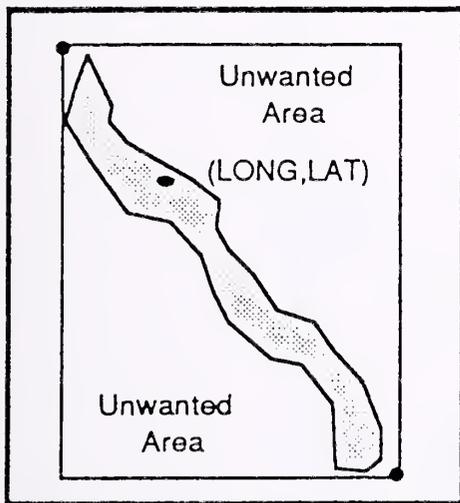
**Figure 1.** Centrum, boundaries and least rectangle of an element occurrence.

Rather than using point data, it is necessary to transfer areal boundaries of occurrences to the GIS. Many animal and natural community EOs require several acres of habitat. If only point data were used, important habitat might not be flagged.

Unfortunately, many EOs do not have known boundaries and least rectangles. In those instances, a centrum dot is mapped with a latitude

and longitude and precision-value assigned to it. The precision indicates how accurately located the centrum dot is on the map. Occurrences with seconds precision are accurately documented to be within one second latitude or longitude from where the centrum dot occurs on the map. Those with minutes precision are mapped within one minute latitude or longitude of their probable location on the map. In order to transform these point occurrences into polygons, a computer program was developed which generates default least rectangles based on precision value. With this program, unbounded EOs with seconds precision are given least rectangles which are 4 seconds on each side, and EOs with minutes precision are given least rectangles which are one minute on a side. In this way, the location for the seconds precision EO is contained within the least rectangle together with a small amount of buffer land. This small amount of buffer land would likely be included if the critical habitat for the EO were delineated. The actual location for the minutes precision EO is also contained somewhere within the minute sized least rectangle, but a much larger amount of buffer land is included. Because, in many instances, the minute sized least rectangles contain significantly more land than would be included if critical habitat for the EO were delineated, it is important to distinguish these areas from the seconds precision least rectangles in any graphic presentation of data.

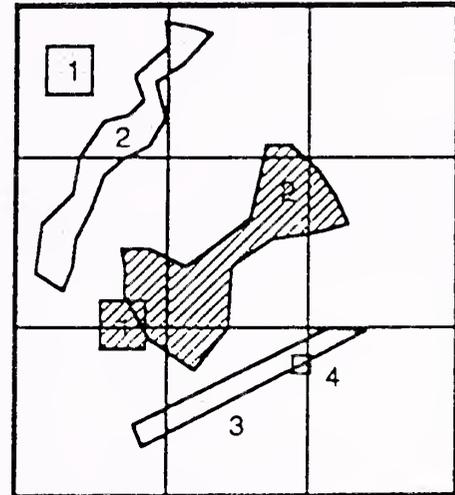
An additional problem occurs involving least rectangles for EOs which have boundaries. Some of the boundaries have irregular shapes which make it difficult to contain them efficiently within the least rectangle (see Figure 2). This problem occurs most often with animals or communities that follow linear features such as rivers, mountain ridges, and railroad beds. Thus, a least rectangle can include many square miles of unwanted area. For such occurrences the least rectangle is not used, and the actual boundaries are manually digitized directly into the GIS database. From our database of 3,500 EOs, approximately 70 required manual digitizing, involving ten staff days of time.



**Figure 2.** Least rectangle for irregularly shaped element occurrence.

Prior to transfer into the GIS computer, latitudes and longitudes are checked for errors. A computer program checks the latitude and longitude of each EO centrum to ensure it is located within its least rectangle, as well as within the appropriate USGS quadrangle map. The verified data is then sent to the GIS computer. The data consist of

several files for polygons, points and attribute data including: identifiers, such as species name and element occurrence code; location data, such as county code, municipality code and site name; data on the rarity and quality of the occurrence, such as element rank and element occurrence rank; data on protection status, such as a species' status on state or federal endangered species lists, information on whether the occurrence is contained within a protected public or private managed area, and additional data caveats such as last observed date, EO precision, delineation of habitat boundaries and last update for the computer record.



**Figure 3.** Element occurrences with grid overlay. In this example the only grid cell which would not be shaded is the upper right cell.

### GENERALIZATION OF LOCATION DATA

One of the concerns in making maps for distribution outside the program is that, once they are available to the general public, unscrupulous plant or animal collectors may try to use them to locate populations of rarities. To deal with this, a method was developed using the GIS to generalize the locations with an overlaid grid. This grid divides up a typical 7 1/2 minute quad into 100 cells each approximately 340 acres in size. Any grid cell that is touched by a polygon is shaded (see Fig. 3). The final map only displays shaded grid cells, not the original boundaries of the polygons. Although all of the sensitive areas are flagged, exact locations cannot be determined from the final map. A more detailed discussion of the GIS mapping procedure is presented in Schooley and Breden (1988).

### FINAL MAPS

The resulting maps, to be provided to regulatory and planning agencies, consist of a photo-reduced USGS quadrangle map with a locator grid superimposed, and a matching GIS generated map (Fig. 4). A user of the maps simply finds the area to be searched on the reduced quadrangle map, then refers to the corresponding grid cells on the GIS produced map. A shaded cell alerts the user to contact the Natural Heritage Program for more specific information on the significance of the area. These maps are easily reproduced by photocopying.

Many additional map formats are possible, utilizing GIS for other purposes. Maps can be prepared identifying locations for globally rare EOs vs. state rare EOs, to identify regions of global

significance. Recently documented EOs can be contrasted with historic EOs to identify regions that have not been surveyed recently. Particular species or communities that are officially recognized by a regulatory statute can be mapped separately for use by the regulatory agency.

### REQUIRED INVESTMENT

The required investment for a Natural Heritage Program to attach its database to a GIS is not great if the program has access to a well supported State GIS. The New Jersey Department of Environmental Protection has a GIS unit staffed with six full time people to maintain and develop the GIS and to train other users in the department. The authors have invested several months of time in developing procedures and programs to facilitate the transfer of data and production of maps. These computer programs will be shared with other Natural Heritage Programs upon request. Each staff person using the system is trained for one to two weeks in using the ARC/INFO software. Initial digitizing of irregularly shaped EO's required ten staff days. The maps are updated

twice a year requiring between two and five days for each update.

### CONCLUSION

The goal of producing maps that alert planners and regulatory agencies to sensitive areas of natural diversity has been achieved in New Jersey. These maps only provide generalized locations and therefore cannot be used as treasure maps by unscrupulous plant and animal collectors. Such maps can be produced rather efficiently, if one has access to a well supported GIS.

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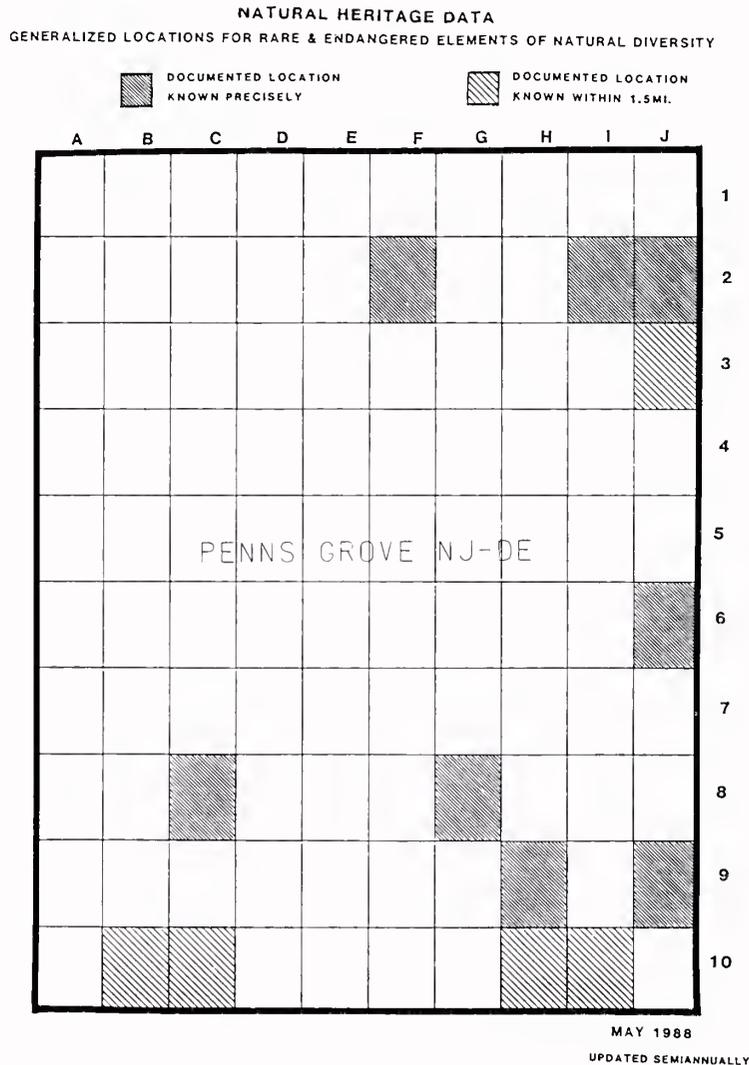


Figure 4. GIS-Produced Natural Heritage data map for Penns Grove USGS quadrangle.

# A Modern, Computer-accessed Flora of Pennsylvania: A Tool for Resource Managers

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**Abstract:** We have established a computer-accessed data base for the flora of Pennsylvania at the Morris Arboretum. The system, which presently contains several interrelated files, includes up-to-date information on the taxonomy, distribution, and legal status of the 3,346 native and naturalized species in the Pennsylvania flora. It provides an important context for evaluating the status of rare plants as well as a useful tool for guiding fieldwork. Publication of an *Annotated Checklist of the Flora of Pennsylvania* is planned for publication the near future, and, eventually, the data base will be expanded to encompass the components of a comprehensive flora.

Pages 125 - 126. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Current involvement of the Morris Arboretum in the Pennsylvania flora began shortly after the establishment of the Arboretum as part of the University of Pennsylvania in the early 1930's. At that time, the job of compiling a list of herbarium specimens from the state was begun. This project eventually led to the creation of a master file of herbarium records and the preparation of dot maps showing species distributions. The *Atlas of the Flora of Pennsylvania* (Wherry *et al.*, 1979), published by the Morris Arboretum, represented approximately 40 years work by Edgar Wherry and his collaborators. The *Atlas* is unusual among state flora atlases in that each dot represents an actual collection site documented by a specimen located in one of the five major herbaria in the state (Fogg, 1947).

## CREATION OF A COMPUTERIZED FLORA

After the *Atlas* was published, we began the task of creating a computerized data base of the Pennsylvania flora. Since the initiation of this phase of the project in the early 1980's, a system of interrelated files has been developed. The central file is a taxonomic list of the 3,446 species of native and naturalized plants found growing wild in the state. Based originally on the species included in the *Atlas*, the list has undergone extensive editing and review to update both the taxonomic treatments and nomenclature. Included taxa have been reviewed by specialists in systematics and floristics. Additions have been made, misidentifications uncovered, and unusual occurrences verified. Editing is scheduled for completion in late 1989.

The Pennsylvania flora data base system was developed using dBase III Plus software; we have subsequently upgraded to dBase IV. The system runs on an IBM AT personal computer equipped with a 140 megabyte internal hard disk. The central taxonomic file includes the following data fields in addition to basic taxonomic information: common names, native vs introduced, distribution by physiographic province and glacial boundary (Wherry *et al.*, 1979), federal (Federal Register, 1980) and state (PA DER, 1987) conservation status, status under the Federal Noxious Weed Act and the Pennsylvania Noxious Weed Control Act (Act 1982-74), wetland indicator status (Reed, 1986), poisonous or toxic status, and growth form. In addition, a comment field is used to include a brief statement of habitat and frequency. A related file contains synonyms. At this stage, we have not included an exhaustive list of synonyms, but only those used in the standard regional manuals and earlier works on the Pennsylvania flora.

Additional files contain location and distribution data. The location file includes a gazetteer of approximately 8,000 collection sites with their latitude and longitude coordinates. This file had its origin on a

card file of collection sites compiled by Edgar Wherry in connection with the preparation of the atlas. Site names and coordinates are now being brought into conformance with the US Geological Survey Geographic Names Information System (US Geological Survey, 1987). The Pennsylvania flora distribution file lists the sites where each taxon has been collected. These two files, when coordinated with the main taxonomic file, provide a high resolution, specimen-based data base on the state flora.

The information can be accessed either by taxon, to obtain site specific distribution information for each plant species, or by location, to obtain a list of taxa collected at a given site or within a rectangle described by latitude-longitude coordinates. For example, a list of native tree species of a given watershed area could be generated by entering the coordinates for the smallest rectangle that can be drawn around the study area. Regional, county, or local area checklists can also be generated in this way. The inclusion of latitude-longitude coordinates also allows for preparation of updated distribution maps for all taxa.

Initially the distribution file contained the 250,000 records that made up the original atlas file. Additional records which were not included in the atlas are now being added. To this end, we have sought the cooperation of the 21 herbaria in Pennsylvania to locate and enter new herbarium specimen data. This process has involved the development of a herbarium inventory program as part of the overall computerized flora system. The program can be used to add herbarium label data to our data base as well as to develop a herbarium inventory or to generate herbarium labels.

We estimate that there are approximately 125,000 Pennsylvania specimens that should be added to our system. We are presently making the herbarium inventory software available to several Pennsylvania herbaria. Several other herbaria, including the Pennsylvania State University Herbarium, the Pennsylvania Department of Agriculture Herbarium, the Millersville University Herbarium, the Shippensburg University Herbarium and the Wilkes College Herbarium have already sent us data from specimens not included in the atlas. In addition to searching Pennsylvania collections for unrecorded specimens, we plan to canvass other important regional collections to locate Pennsylvania material.

## FUTURE DEVELOPMENT OF THE SYSTEM

The Pennsylvania flora data base is an evolving system. A bibliographic file with links to the central taxonomic file and synonym file will be added next, using the format developed by the Hunt Institute for Botanical Documentation for the Flora of North America.

We also anticipate the expansion of poisonous plant information in cooperation with the Center for Food Safety and Applied Nutrition (U.S. Food and Drug Administration). Under consideration for addition are data on categories of toxicity to humans and grazing animals and related literature references.

An annotated checklist of the Pennsylvania flora is planned as the first published product of the data base. The checklist will include updated species distribution maps which will be computer generated.

We are firmly committed to maintaining high resolution, specimen-based distribution data as a key element of our state flora project. After the publication of the annotated checklist, we plan to develop a computer mapping-capability which will allow us to correlate our floristic data with other spatial data of a physical and biological nature in a geographic information system (GIS). Our long term plans call for expanding the Pennsylvania flora data base to include descriptions, keys, and other elements of a comprehensive flora.

The system has the potential for serving as an analytical tool for understanding plant distributions, as well as a predictive tool for guiding field surveys and managing rare and endangered species. The record of specimens collected at a given site includes information on historic as well as current species occurrence, providing insights into the persistence of individual populations and plant communities. The presence of introduced or exotic species can similarly be traced on a site by site basis.

The data base can also be used to identify additional habitat areas for targeting field surveys through the use of habitat indicator species. For example, records for the common bog-inhabiting species *Chamaedaphne calyculata* could be checked to locate possible sites for less conspicuous, endangered or threatened bog species. The gazetteer of collection sites could also be used to determine whether an area has been botanized in the past (and by whom) by revealing the date and collector of all specimens from a given site.

We have established good communications with the Flora North

America project and will continue to coordinate our efforts as the continent-wide flora project develops. Because of the detailed nature of the Pennsylvania flora data base, we feel that it could serve as a model for the development of specimen-based inventories for other states or regions.

#### ACKNOWLEDGMENTS

Dr. Robert Wirtshafter and Robert Noland, of the Department of City and Regional Planning, University of Pennsylvania, have been of invaluable assistance in translating our ideas into computer programs.

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## **Chapter 4.**

### **DEMOGRAPHY AND SPECIES BIOLOGY: RARE ANIMALS**



# The Dangers of Being Few: Demographic Risk Analysis for Rare Species Extinction

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**Abstract:** Rare species are susceptible to extinction under unpredictable yet ubiquitous environmental fluctuations and demographic stochasticity, both of which become more important as a population gets smaller. To design effective conservation strategies, we need to be able to estimate the consequences of these influences in terms of potential risk that the population might fall below a given abundance. Environmental fluctuations (which we cannot predict but may be able to characterize statistically) induce variation in demographic vital rates. A population dynamic model incorporating this variation and accounting for demographic stochasticity produces a bundle of different population trajectories, each of which represents a possible future of the population. From this bundle of trajectories, one can compute the probability of the population's falling below threshold levels. Information from this kind of risk assessment can answer questions about the design of management regimes and the capabilities of natural populations to recover from disturbance in the face of natural variability.

Pages 129 - 132. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The population sizes of all living species fluctuate over time. They change in response to periodic or random changes in the environment that affect the ability of individuals to survive and reproduce. Population size, even if it is drawn to some equilibrium carrying capacity, constitutes a random walk, rising in good years and declining in bad. When this random walk wanders to zero, we call this extinction and recognize it as a special state from which the population cannot return without introductions from the outside. The randomness that influences population trajectories is especially significant in rare species that by definition have locally small populations (Brown, 1984) and thus little insulation from episodic insults or the happenstance of a few bad years in a row.

Evolutionarily speaking, nearly all the species that have ever lived are now extinct. While extinction may be the eventual fate of all forms of life (no matter what efforts humans make to preserve them) the current rate of extinction is several orders of magnitude greater than the background rate, in geological time, of about one species per year (van Valen, 1985). The existing rate will result in the decimation of biological diversity and it is incumbent on conservation biology to develop methods to deal with the problem. But questions about the demise of rare species should be addressed in full light of the realities of biological variation and randomness. This paper outlines a methodology by which one can estimate the probability of extinction, or of dropping below a specified population size, which we call quasi-extinction. Making this estimation under several different scenarios allows environmental scientists or managers to design an appropriate management regime, maximizing the chances that species of concern survive and thrive.

## STOCHASTICITY

Even excluding sampling error, there are several kinds of uncertainty that models of population dynamics must contend with. Shaffer (1987) classified four forms of uncertainty that impinge on minimum viable population analyses: demographic (resulting from

random events in the survival and reproduction of individuals), environmental (resulting from changes in the environment), catastrophic and genetic uncertainty. We restrict ourselves here to the first two.

Environmental uncertainties are intrinsic to population dynamic systems, and they result in a fundamental unpredictability that constrains our practical use of the models and estimations derived from them. For example, the weather is a source of unremovable error because it is not possible to predict exactly what it will do next year. Because weather has impacts on the vital rates for most species, no matter how much we know about their population dynamics, we cannot predict next year's population sizes exactly. Such unremovable error reflects true stochasticity, that is, variation that is fundamental and intrinsic to the system. For all practical purposes, there will always be a component of randomness in any realistic model of population dynamics that reflects the natural stochasticity in the world. Sometimes the stochasticity in population dynamics is quite large, *e.g.*, first-year fish survival rates can vary widely, occasionally by many fold. Andrewartha and Birch (1954) give many examples of the sometimes overwhelming importance of stochasticity in the dynamics of natural populations.

Demographic stochasticity (Shaffer and Samson, 1985; Shaffer, 1987) arises, even when vital rates are constant, from chance events of survival and reproduction in populations composed of a finite, integral number of individuals. It has the most severe consequences for small populations. For instance, the rarer the species, the more likely its small populations will suffer a strongly biased sex ratio, generational or distributional imbalances. If we assume the deaths of individuals in any age class are independent of one another, then the demographic variance of the number of survivors is approximated by the binomial distribution (H.R. Akçakaya, pers. comm.). For example, given a mean survival rate, if we take a number of individuals and ask of each one whether it survives, the variance after repeated samples in the number of individuals that we observe surviving will be the variance of the binomial. Demographic stochasticity is contrasted with environmental stochasticity which is the random variation in population parameters, such as survival and fecundity due to change

in the environment, *i.e.* the direct effect of the environment on population vital rates.

The usual effect of the inclusion of stochasticity in deterministic models is to reduce the expected persistence times as some function of the magnitude of the variance (*e.g.*, Ginzburg *et al.*, 1982). Demographic stochasticity is important only if the population is small (less than about twenty individuals) but the effects of environmental variation make the inclusion of environmental stochasticity essential for the estimation of risks of extinction or quasi-extinction (Boyce, 1977; Goodman, 1987a; Shaffer, 1987; *cf.* Burgman *et al.*, 1988).

## ESTIMATING QUASI-EXTINCTION RISK

There are two things needed to estimate the chances that a particular population will go extinct or suffer some radical decrease in abundance. First, one needs a model of population growth and decline that is appropriate for the life history of the species of concern, and second, one needs data to evaluate the model's parameters. By this, we mean to say that estimating quasi-extinction risks is neither a purely theoretical nor purely empirical problem. The modeler or manager must make decisions and, thus assumptions, both about whether certain functions appropriately characterize population dynamics and about whether available measurements adequately assess population sizes and vital rates.

As extinction is a population's falling to zero, quasi-extinction is the phenomenon of falling to any specified abundance level. The chance of extinction or quasi-extinction at some level is just the probability that the population's trajectory will cross a threshold abundance. This chance depends on the current population size, the trend of population growth, the time allotted to cross and the variability of population size under the influence of stochasticity. The time horizon might be infinity, or it might be limited to a few generations or a fixed number of years, depending on the particular question we pose. Clearly, this problem is quite similar to the computation of the expected time to quasi-extinction, often called persistence time or crossing time, and typically one can interconvert these estimations. One might prefer quasi-extinction risk to crossing time simply because it is already phrased in a way more relevant to planning conservation strategies.

For a population growing exponentially, given that a population starts at the value  $N_0$ , what is the probability that it will fall to a level  $N_c$ ? The solution depends on two parameters: the rate of the exponential increase (or decrease) and the uncertainty in that rate, that is, the amplitude of fluctuations in the rate induced by stochasticity. Ginzburg *et al.* (1982) reviewed the case of simple Malthusian population growth and gave an expression for the probability of quasi-extinction. With this expression one can compute either the chance that the population will eventually cross, or the chance that it will cross within some fixed time period. Goodman (1987a), using a comparable formulation (in terms of persistence time) to model environmental stochasticity, found vast differences compared to a model considering only random differences between individuals. If a species is rare because it has been artificially reduced in abundance, and is therefore much below carrying capacity, then this kind of model may be appropriate for such a species if it is in fact growing nearly exponentially.

But Malthusian growth is really far too much of a simplification for it to be reasonable for general application to actual biological populations. There are some grossly unrealistic assumptions that should be relaxed. Real populations do not experience exponential growth for very long, and most presumably have some kind of density

dependence at some life stage or under some conditions (Strong, 1986). Thus, there have been attempts to incorporate nonlinearity into the problem. Ginzburg *et al.* (1982) considered the case of logistic growth which exhibits a very simple density dependence. Braumann (1988) suggested a method by which to compute quasi-extinction risks when the growth rate is practically any function, but his method is probably flawed because it assumes that noise decreases as the population approaches equilibrium (L.R. Ginzburg, pers. comm.). Goodman's (1987a, 1987b) general formulation allows nonconstant, possibly discontinuous dependence of population growth rate on population size. Unless a model includes such nonlinearity, it might underestimate the risk of extinction if the population undergoes regular abundance cycles independently of the effects of environmental stochasticity. These cycles will bring the population regularly closer to the extinction boundary, increasing its chances of crossing it due to the effects of demographic and environmental stochasticity. Such cycles may have a number of causes. First, cycles may be induced by the structure of the life-history matrix, in particular by the pattern of fecundity values (Sykes, 1969; Cull and Vogt, 1974). Second, strongly nonlinear density dependence (Ricker, 1975) may induce abundance cycles (see below). Last, short-term abundance cycles can occur, even in a population with no long-term cyclical behavior, when the initial distribution is unlike the stable age distribution. Commonly, these fluctuations dampen quickly, within about five generations, and their magnitude may be explicitly calculated in a deterministic model (Charlesworth, 1980). Nevertheless they may have important consequences for small populations in variable environments.

Another unrealistic assumption in simple models is that populations consist of identical individuals. Incorporating age structure into the model is a crude but perhaps sufficient way to account for significant differences among individuals in terms of their fecundity and survivorship. In many cases, details of age structure and age-specific vital rates are not available, but, when they are, it is often very important to take them into account when estimating quasi-extinction risks. Ginzburg (1984) suggested an approach to use in this case that assumed stochasticity in a single variable. Lande and Orzack (1988) generalized the results of Ginzburg *et al.* (1982) to age-structured populations with moderate levels of environmental stochasticity (up to 30%) by simplifying the problem to a one-dimensional approximation. This approach properly takes into account age structure in a stochastic environment, but it uses a purely linear model that cannot capture density dependence. Currently, the only known way to incorporate age-structure, density dependence and stochasticity into a model and estimate quasi-extinction risks is to use a computer simulation.

RAMAS (Ferson *et al.*, 1988) is a Monte Carlo computer simulator for age-structured population dynamics that incorporates density dependence and stochasticity. In spirit, the software is not itself a model, but instead it facilitates the design and employment of models by the user. Thus, there are several kinds of density dependence that can be invoked with RAMAS; the user can specify how many age classes there are and what the vital rates and current abundances are in each, and fairly completely describe the variance-covariance structure of environmental stochasticity that influences variation in demographic parameters. RAMAS was designed to be extremely user friendly and to generate curves that characterize quasi-extinction risk as a function of population size. It does, however, ignore genetic complications such as inbreeding depression and the possibility of catastrophic events. Simulations we have performed using RAMAS have confirmed some of the conclusions reached by Goodman (1987b). In particular, if there is no possibility of migration between reserves, then a single

population will have a smaller chance of extinction than a population of the same size, subdivided into spatially separated reserves. This result is independent of the degree of correlation of environmental variation between reserves.

## DENSITY DEPENDENCE

Rate of fecundity is usually a nonconstant function of population size. It is often the case that when organisms are extremely crowded few offspring are produced, while at lower densities reproduction is comparatively accelerated. Among several phenomena that share the name, such nonconstancy in the relationship between reproductive effort and reproductive success is called density dependence. Although it is apparently a component in the dynamics of many species, density dependence is notoriously difficult to estimate in natural populations, and its significance has been debated in ecology for decades. Model population trajectories exhibit strong dependence, especially in terms of asymptotic behavior, on whether density dependence is incorporated into the simulation. We find in particular that the values estimated for quasi-extinction risks depend quite sensitively on whether and how density dependence appears in the model (*cf.* Goodman, 1987a, on persistence time).

When, as may often be the case, density dependence tends to draw the population size toward a carrying capacity (because it decreases growth for large populations and increases growth for small populations), it acts as an equilibrating influence on the trajectories. Population dynamics with this kind of density dependence will exhibit less fluctuation than if no density dependence existed. Since quasi-extinction probabilities are related to the magnitude of fluctuations in population size, models that do not represent density dependence will overestimate these probabilities. This means that ignoring such density dependence tends to yield conservative quasi-extinction risk estimates.

There is, however, a complication that may be especially important for rare species. The Allee effect (Allee, 1949) is also a form of density dependence, but in this case, when the population goes below some critical level, growth rate becomes negative. Allee effects are observed, for example, when reproduction opportunities are diminished in populations so sparse that individuals cannot easily find mates. This may be the fate of some whale species. Allee effects can also happen when the group ameliorates the environment for population growth in some significant way. For instance, the disjunct populations of hemlock in western Indiana seem to acidify the soil and sequester water in the upper horizons, both of which processes facilitate hemlock reproduction. If the hemlock density were greatly reduced, it is very doubtful that the populations could maintain themselves. Since such a phenomenon requires locally high densities, it presumably only occurs in species that are geographically rare, existing in clumped distributions. In general, the consequence of Allee effects is that, when the impact made by the group is removed, the population experiences a downward slide toward extinction.

Ignoring Allee effects may cause a population dynamic model to greatly underestimate risks of extinction by failing to take into account the cascade caused by the population's slipping past some threshold of abundance. Clearly, then, it is incumbent upon the modeler to know enough about the biology of the species of interest to recognize likely Allee effects and include them in the model. This is no easy task unless there is natural history evidence that may have arisen when the species was first recognized to be in need of conservation management.

## CONSERVATION STRATEGIES

Population growth is determined by the vital rates of fecundity and survivorship and, for small populations, demographic stochasticity. For almost all natural populations we cannot hope to make precise predictions about future population size because we do not know how all the things that influence it will vary. From past observations it may be possible, however, to estimate how the parameters range, that is, to know something about their statistical distributions (even though we do not know what particular value each will take). With this information and an appropriate model of the population dynamics, it is possible to extrapolate the variation that is likely for the parameters into the variation we could therefore expect in the population trajectories. The quasi-extinction risk curve discussed above is an especially relevant summary of this variation for conservation planning. Every natural population experiences fluctuations, so there is a curve that represents the background risk against which management strategies can be compared. Effective conservation efforts should minimize the risk that the population experiences low abundances; therefore, a better strategy can be recognized as one that shifts the quasi-extinction risk curve down or to the right. Conversely, adverse impacts that lower population size, or make it more likely that population size will drop, are manifested in a shifting of the quasi-extinction risk curve up or to the left.

Compare the criterion of minimizing quasi-extinction risk to one that seeks to maximize mean population growth. It may not be prudent to foster a rare species in a way that, on average, will yield the largest population increases possible if there is a substantial danger that the actual population suffers extinction. In general, the strategies developed with reference to these two criteria will be different, and one cannot say *a priori* just how they will disagree.

Sensitivity analysis, exploration of how the position of the risk curve depends on various parameters in the model, can determine where conservation efforts should be focused. For instance, simulations that compare the effectiveness of stocking with reduction in mortality allow informed decisions about investment in rearing programs versus poaching control. Sensitivity studies sometimes lead to surprising suggestions, in that the complexity of demography makes the estimation of quasi-extinction risks seem counterintuitive under several sets of circumstances. Some conditions yield a masking of impacts while other seem to amplify them. As might be expected, quasi-extinction risk is often insensitive to mortality of post-reproductive individuals, while small insults to critical life stages can result in dramatic effects, although what stages are critical is often by no means apparent before the analysis. More surprisingly, under strong, highly nonlinear density dependence it is possible for additional mortality to actually lessen quasi-extinction risk because it dampens oscillatory behavior induced by the density dependence. As a rule, it is imperative that one make a comprehensive risk analysis for quasi-extinction using the demographic information available on vital rates and their variation to plan an effective strategy for conservation.

There are, of course, other considerations besides those of demography that must concern the manager of rare species. Among these are questions of habitat use and spatial distributions (Gilpin, 1987), genetics (Lande and Barrowclough, 1987; Crow and Denniston, 1988) and other species in the biological community, including competitors, predators, prey or forage species and humans. The interactions with other species, space, and even genetics can, when they are known, be incorporated into the population dynamics model

to better represent the circumstance of the managed species and to produce more accurate quasi-extinction risk estimations (see Starfield and Bleloch, 1986, for examples).

## CONCLUSIONS

For a rare species, extinction is always nearby, and the policies and management regimes that best insulate the species from this fate are not always evident. In part, this is because demographic complexity can be very important in the population biology of rare species and, often leads to surprising or counterintuitive results. Much of the uncertainty in designing conservation efforts arises from the intrinsic stochasticity of the species' population dynamics, resulting from both demographic stochasticity and environmental variation.

The demographic complexities of density dependence and age structure introduce nonlinearity and time lags to the problem of planning strategies for rare species. Thus, it is by no means a simple matter to extrapolate, say, a 10% additional mortality on juveniles to the consequences in terms of how often the population is likely to decline to a given level. This mortality may be absorbed and result in little impact; it may lower the population by a proportionate amount; it may initiate a cascade into extinction, or, in some unusual circumstances, it may even lessen the risks of extinction.

Also obscuring the picture is the environmental noise and, in small populations, demographic stochasticity. The straightforward way to account for these forms of stochasticity in demography is to propagate the observed variability in vital rates through a model of population dynamics and observe the implied range of possible population trajectories. From this output one can compute a curve characterizing the risk of quasi-extinction (falling below a specified abundance). Sensitivity analysis on such a simulation can direct or help in the design of effective conservation strategies. Although engineers building a bridge cannot predict exactly how severe winds will be over the planned life of the bridge, they may have a fair statistical characterization of the windspeeds to expect and can estimate the probability the bridge will collapse. In the cases of both the bridge and rare species, society decides how much investment to make to lower risks and what levels of risk are acceptable. It is one of the tasks of conservation biology to develop and apply techniques that allow the quantitative estimation of risk of extinction for rare species.

## ACKNOWLEDGMENTS

We thank Lev Ginzburg and H. Resit Akçakaya for sharing their ideas with us. Geoffrey Jacquez read the manuscript and offered invaluable criticisms.

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# Viable Populations of Spotted Owls for Management of Old Growth Forests in the Pacific Northwest

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**Abstract:** The Northern Spotted Owl (*Strix occidentalis caurina* Xantus 1859) has become a surrogate for efforts to preserve remaining old growth forests in the Pacific Northwest. We review here a viability analysis prepared by the U. S. Forest Service which evaluates risks of extinction resulting separately from (a) stochastic demographic processes, (b) inbreeding and loss of genetic variability, and (c) habitat loss. Stochastic demography must include density dependence to be realistic, particularly for a territorial species. Because of high sampling variance, it is unlikely that adequate data can be accumulated for most rare species to provide demographic projections that differ significantly from a constant population. A model that integrates all components of the Forest Service viability analysis predicts a low probability that Spotted Owls will go extinct under the Forest Service's preferred management alternative. However, we emphasize that such a model is unrealistic because it does not incorporate spatial distribution of owls, and population fragmentation imposes the greatest risk of their extinction. A metapopulation model by Lande (1988) estimates acreage of old-growth forest necessary to preserve the Spotted Owl on Spotted Owl Habitat Areas (SOHAs). However, such estimates are dependent upon accurate determinations of the proportion of SOHAs and other habitats that are truly suitable for Spotted Owls. Habitat requirements of the owls must be carefully documented to justify management based upon the metapopulation model.

Pages 133 - 135. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The Northern Spotted Owl is currently the focus of the most heated conservation debate in the United States (Simberloff, 1987; Wilcove, 1987). The Spotted Owl is being used as a surrogate species for the preservation of remaining old-growth forests in the Pacific Northwest, where the timber industry depends largely on forests older than 200 years for its wood supply. Because most remaining old-growth habitats are on National Forest lands, the U.S.D.A. Forest Service is in the uncomfortable position of trying to compromise timber interests with those favoring preservation of old-growth forests and Spotted Owls.

The U.S. Forest Service (USDA, 1986; 1988) has recently attempted a viability analysis for Spotted Owls, using the three distinct approaches of (1) stochastic demography, (2) population genetics, and (3) habitat characterization, to forecast the probability of extinction under various management alternatives. We will review this viability analysis and suggest different approaches for future efforts.

## VIALE POPULATIONS OF SPOTTED OWLS

In its Draft Supplemental Environmental Impact Statement (USDA, 1986), the Forest Service concluded that stochastic demography is the modelling approach of greatest significance to Spotted Owls, because it yields the lowest prospects for their survival. In general, the greater the environmental and demographic variance the lower the long-run growth rate and the higher the probability of extinction for the population (Tuljapurkar and Orzack, 1980). The Forest Service's demographic projections were based on the Leslie matrix, however, which is an exponential growth model. A stochastic exponential model has essentially two possible outcomes: the population will increase to infinity or it will decline to extinction (Boyce, 1977). For a territorial species such as the Spotted Owl, we find such a model structure totally

unrealistic, and insist that some form of density-dependence be incorporated into it.

Another common difficulty with demographic analysis is the inability of the user to estimate reliably for all of the parameters in the model. For example, a 15x15 Leslie Matrix for Spotted Owls requires that at least 28 parameters be estimated. As a consequence, projections based upon such models are statistically unreliable. A smaller model, e.g. a 3x3 stage projection matrix (Lefkovich, 1965), will have almost identical dynamics; however, the smaller model requires that only three or four parameters be estimated, and will thus provide projections with substantially smaller confidence intervals, since the degrees of freedom associated with each parameter estimate will be larger. Bootstrapping and jackknifing procedures exist to evaluate the uncertainty in such population projections (Meyer *et al.*, 1986), but these have not been applied to Spotted Owl data.

Based upon meager data existing on the demography of Spotted Owls, Lande (1988) could not show that the population growth rate differed significantly from that of a constant population. Lande (1988) came to this conclusion despite the fact that he seriously underestimated the magnitude of variance in growth rate by calculating only the binomial component of sampling variance. Although the Final SEIS (USDA, 1988) claims that Spotted Owl populations are declining at 0.5% to 1.1% per annum, it offers no confidence intervals around these rates of decline. Although nest sites have been destroyed at a rate of 1.5% per year (Forsman *et al.*, 1984), the ultimate fate of the occupants is unknown. Because of inadequate population surveys and small sample sizes for demographic parameter estimates, there does not appear to be any reliable evidence that Spotted Owl populations are indeed declining in the Pacific Northwest.

The second approach used by the Forest Service in their viability analysis was to estimate the degree of inbreeding expected to occur in Spotted Owl populations. In general, inbreeding is not likely to be an important consideration for viability analysis of Spotted Owls, because

inbreeding is unlikely to affect the birds until populations decline to approximately 120 individuals (USDA, 1986; 1988). This may present a very real risk for isolated populations, *e.g.*, those on the Olympic Peninsula, but, for the core distribution of Spotted Owls it is unlikely that inbreeding depression is an immediate threat.

The disjunct approaches employed by the Forest Service can be integrated into a single density-dependent age-structured model where the genetic inbreeding effects are incorporated as an Allee effect (Boyce, 1987). Our studies suggest that the critical parameter determining the probability of extinction in such an integrated model is the habitat's carrying capacity for Spotted Owls. Therefore, perhaps the most useful calculations performed by the Forest Service (USDA, 1988) were the estimates of a habitat capability index (HCI) and projected capacity of various areas to support owls under various management alternatives.

Estimates of the habitat's carrying capacity for Spotted Owls were assumed to represent equilibrium population sizes in a density-dependent model. As was true for the exponential growth model: as environmental and demographic stochasticity increases, the population growth rate declines, along with the average population size (Boyce and Daley, 1980). Nevertheless, even with stochastic variation, the probability of extinction is very low for Spotted Owls at carrying capacities projected for the Forest Service's preferred alternative (Boyce, 1987). This is because density dependence dampens population fluctuations and greatly reduces the probability of extinction (see Ludwig, 1975). There is a serious weakness in this approach, however, because it ignores the metapopulation structure, *i.e.*, spatial variation, of the population (Lande, 1987).

Ultimately, we suspect the fate of the Spotted Owl will depend upon geometry of distribution of subpopulations and isolated mating pairs in the Pacific Northwest. Lande (1987, 1988) made a first attempt to model the spatial structure of the owls and estimated that at least 21% of the Spotted Owl habitat (in old-growth forests) in the Pacific Northwest must be preserved for persistence of Spotted Owls. This is in line with the preferred alternative of the Forest Service (except on the Olympic Peninsula), but there are a host of assumptions in Lande's modelling effort that require careful evaluation before accepting the 21% value proposed. The assumptions include (1) random spacing of habitat patches, (2) density dependence only based on juvenile dispersal and prospects for establishment in an unoccupied patch of habitat, (3) accurate characterization of Spotted Owl habitat, and (4) that the baseline population was in demographic equilibrium. Also, note that if old-growth habitat drops below 21%, Lande's model guarantees extinction. But, if old growth is maintained above 21% one can only conclude that the probability of extinction is something less than 1.0, *i.e.*, the owls may still have a high probability of extinction.

It is perhaps too easy to be critical of these pioneering efforts at viability analysis, and it is even more difficult to offer alternatives. No clear guidelines exist for performing a viability analysis (Soule, 1987); therefore, one must view the Forest Service efforts as a commendable first attempt. Nevertheless, there is still much to be resolved both theoretically and empirically before much confidence can be placed in the management alternative for Spotted Owls promoted by the Forest Service.

#### PROPOSED ACTION BY THE FOREST SERVICE

The preferred alternative (Alternative F) offered by the U.S. Forest Service is to maintain habitat for 2,180 pairs of Spotted Owls over the 15-year planning period (USDA, 1988). This constitutes 80% of the

existing Spotted Owl habitat. The no-action alternative (Alternative A) with no restraint on old-growth forest harvest over current trends will allow habitat for 2,140 pairs or 79% of existing habitat. The 50-year projection predicts that habitat for only 60% of the pairs will remain under the preferred alternative whereas 62.5% of the pairs will be supported if the Forest Service were to invoke Alternative L (no further harvest of Spotted Owl habitats).

Although this appears at first to be only a token effort for Spotted Owl conservation by the Forest Service, the perceived costs to the timber industry are still a billion dollars, assuming that old-growth forests can be valued at \$10,000 per hectare (Simberloff, 1987). If a moratorium on old-growth mining were instituted, the ultimate cost perceived by the timber industry may be as high as \$10.4 billion.

To a conservationist, perhaps the most distressing part of the preferred alternative is the extent of risk accepted relative to persistence of the Spotted Owl population on the Olympic Peninsula of Washington. Even by the Forest Service's projections, there is a high probability that the small, apparently isolated population on the Olympic Peninsula will be extirpated as a consequence of habitat alteration within the planning period. If, in fact, this population is isolated, the proposed action would constitute a violation of the mandate given by the National Forest Management Act of 1976 that a viable population be maintained "which has the estimated numbers and distribution of reproductive individuals to insure its continued existence is *well distributed* in the planning area" (USDA, 1988).

On the other hand, the validity of the assumption that the Olympic Peninsula population is isolated has not been established. Indeed, several pairs and scattered individuals have been found in developing forests south of the Olympic Peninsula (Irwin *et al.*, 1987). The value of the establishment of these individuals to a well-distributed population will only be significant if it is shown that they breed successfully and contribute to gene flow between subpopulations.

We question the proposal in the Final SEIS (USDA, 1988) to allocate smaller areas of old growth to SOHAs in core portions of the Spotted Owl range in central and southwestern Oregon. The justification for large SOHAs in Washington is that home ranges of pairs of birds are substantially larger than those in Oregon (Allen and Brewer, 1985; USDA, 1988). However, populations in the central Cascades of Washington have also been much less productive than those in central and southwestern Oregon. To maximize the chances of persistence for the Northern Spotted Owl, we suggest that SOHAs in Oregon should be at least as large as those set for Washington.

Finally, the U.S. Fish and Wildlife Service has recently recommended listing for the Northern Spotted Owl as a threatened species, affording it protection under the Endangered Species Act. At this time, there are a number of active and pending court cases involving Spotted Owls. It seems likely that federal courts and U.S. Congress will determine the ultimate fate of the Spotted Owl and old-growth forests in the Pacific Northwest.

#### CONCLUSIONS

1. It is essential that statistical reliability be evaluated in viability analyses for rare species.
2. Reliable projections of the probability of extinction using demographic models require enormous sample sizes; therefore, demographic projections cannot be justified in most viability analyses.
3. Exponential growth models such as the Euler equation or the Leslie matrix are inappropriate for projecting the probability of extinction in territorial species where density dependence stabilizes numbers.

4. Habitat fragmentation is an essential component of any realistic model of Spotted Owl viability. It appears that the greatest threats to extinction for Spotted Owls exist because of the potential for population fragmentation.

5. The Forest Service's preferred alternative in the Final SEIS (USDA, 1988) accepts considerable risk of extinction for the Spotted Owl population on the Olympic Peninsula. Furthermore we are concerned that the Forest Service is making the greatest efforts to preserve populations in Washington while allocating less habitat for productive populations in core areas of central and southwestern Oregon.

#### ACKNOWLEDGMENTS

We benefited from discussions and correspondence with Rollie Lamberson, Charles Biles, Barry Noon, David Wilcove, Gordon Orians, Andy Stahl, Harriet Allen, Bruce Marcot, H. Holthausen, K. Johnson, David Ligon, Jim Nichols, K. O'Halloran, Mark Shaffer, and members of the Northwest Forest Resource Council. Thanks to E. H. Merrill for reviewing the manuscript. We received support from the National Council of the Paper Industry for Air and Stream Improvement and the Wyoming Water Research Center.

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# The Red Wolf: Extinction, Captive Propagation and Reintroduction

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**Abstract:** The red wolf (*Canis rufus*) is a large and unique wild canid that once ranged over the southeastern United States. Extinction of this canid outside captivity had occurred by 1980, due to 100 years of human encroachment, destruction of bottomland forest and coastal marsh habitat, persecution and predator control activities. In 1973, a Red Wolf Captive Breeding Program was established by the U.S. Fish and Wildlife Service and the Point Defiance Zoo in Tacoma, Washington. Today, there are fewer than 100 red wolves in captivity in about a half-dozen institutions. The American Association of Zoological Parks and Aquariums (AAZPA) recently established a Species Survival Plan (SSP) for the red wolf. Masterplans for SSP species are designed to prevent extinction of species, carry out eventual reintroduction of rare species into protected natural areas, and augment these wild populations with captive-born animals. The U.S. Fish and Wildlife Service, in cooperation with the AAZPA SSP Propagation Group, recently released four captive-born pairs of red wolves into the Alligator River National Wildlife Refuge, North Carolina. The majority of this Refuge property was donated through the efforts of The Nature Conservancy in 1984; a large private "in-holding" was purchased by The Nature Conservancy in 1988. This paper addresses the role of multidisciplinary conservation strategies for preserving endangered species.

Pages 136 - 139. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The red wolf (*Canis rufus*, Audubon and Bachman 1851) is a large canid that ranged over the southeastern United States, from the Atlantic Ocean to central Texas and from the Gulf of Mexico to Central Missouri and southern Illinois (Thornback & Jenkins, 1982; Carley, 1979a). Carley (1979a) examined hundreds of red wolves from throughout the last remnants of their range and emphasized that, in size and proportion, they were more similar to eastern gray wolves (*Canis lupus lycaon*) than to coyotes (*Canis latrans*); weight of females in captivity is in the high-40 to mid-50 pound class, and weight of males is in the mid-50 to mid-60 pound class (Carley, pers. comm.). The most common color phase of this species is generally more reddish than in *C. latrans* or *C. lupus*, but this is not a valid diagnostic character, because individual specimens of all three species may be virtually indistinguishable due to the variability inherent in North American *Canis* (Young & Goldman, 1944; Paradiso & Nowak, 1972; Carley 1979a). Red wolves shipped to the U.S. Fish & Wildlife Service captive breeding site, managed by Point Defiance Zoo in Tacoma, Washington, changed color after a period of time, put on a heavier coat, and became somewhat grayer, and white coloration on the chest, legs, feet and muzzle became more pronounced (Carley 1979a).

Historically, red wolves and coyotes probably coexisted along the western edge of the red wolf range, where deciduous cover gives way to open prairie (Texas and Oklahoma). Gray wolves may also have been sympatric with red wolves in the Appalachians south to Georgia and Alabama at one time (Poulos *et al.*, 1986). Despite early arguments among taxonomists over "pure" versus possible "hybrid" status of the red wolf, it is now "considered a true species beyond question" (Parker, 1986). The red wolf may even be a unique surviving descendent of primitive wolves that ranged over North America one million years ago (Nowak, 1972).

## SPECIES DECLINE AND EXTINCTION

The red wolf, like other wild canids, declined in numbers after the arrival of European man on the North American continent. Initial population declines were caused by increases in human population and subsequent forest clearing, human settlement and changes in land use through the early 20th century, and by persecution and predator control activities (Paradiso & Nowak, 1972; Carley, 1979a). Some sketchy

ecological information about the species is available. Historic range of the species was mostly within the humid division of the Lower Austral Life Zone; the animals apparently preferred warm, moist and densely vegetated habitat. They were equally at home in pine forest, bottomland hardwood forest, and the coastal prairies and marshes which in the 1970's formed a very restricted, yet critical, refuge for survival of the species (Paradiso & Nowak, 1972; Carley, 1979a).

Alarming declines in red wolf populations were first noted in 1962 (McCarley, 1962). A limited Red Wolf Recovery Program was begun in 1967, and the Endangered Species Act of 1973, and resulting funding for endangered species programs, resulted in renewed and increased efforts to save this critically endangered species (Carley, 1979a). Early findings confirmed that remnant populations of the red wolf were threatened by loss of habitat, loss of young to parasites, persecution by man, and possible irreversible dilution of the gene pool by invading coyotes (Carley, 1979a; McCarley and Carley, 1979). By late 1975, the Fish and Wildlife Service concluded that it was no longer feasible to preserve the species in its steadily shrinking range in Texas and Louisiana; the prime objectives then were to capture and place in captivity as many red wolves as possible, and to explore the feasibility of reestablishing red wolves within their historic range. After several animals were captured and placed safely in a captive breeding program, wild populations continued to decline in numbers. The species was declared "biologically" extinct in the wild in July, 1980 (Thornback & Jenkins, 1982).

## CAPTIVE MANAGEMENT AND PROPAGATION

A Red Wolf Captive Breeding Program had been established in the 1970's in conjunction with the Point Defiance Zoo, Tacoma, Washington, which established twelve, 100' x 100' breeding pens at an isolated, off-exhibit site. Presently, there are fewer than 100 individual red wolves in captivity in about a half dozen institutions. The American Association of Zoological Parks and Aquariums (AAZPA) recently established a Species Survival Plan (SSP) for this species, with Roland Smith, Assistant Director of Point Defiance Zoo, as Species Coordinator. All captive red wolves are maintained under the stewardship of the U.S. Government; management and breeding are controlled by the SSP Propagation Group.

AAZPA Species Survival Plans are designed to make zoos and aquaria viable as a collective "ark" to carry selected species safely

through the modern flood of habitat destruction, animal persecution and subsequent extinction. Captive populations maintained by zoos should reinforce, not replace wild populations; that is, they can revitalize small wild populations through active management and interchange between, captive and wild populations. Zoo biologists are studying the theoretical and practical aspects of small populations, including stochastic problems of demographic, genetic and environmental uncertainty. Species Survival Plans define demographic and genetic objectives for captive population management over long periods of time, and establish a zoo-wide carrying capacity for the target species. Minimum viable population size is usually on the order of a few hundred animals. Survival of the species in the wild will be dependent on the goals and execution of the reestablishment program as well as the biological characteristics of the population, including generation time, effective number of founder animals, and intrinsic growth rate of the population. Ideally, evolutionary processes should be allowed to continue, and this does not happen in small, captive populations. Hence, it is most desirable to have a large, effective population size and to interact with wild populations so that natural selection may occur (Foose, pers. comm.). In the case of red wolves, the SSP protocol states that pups "are not to be handraised nor are extraordinary measures beyond the Red Wolf Captive Breeding Program Guidelines to be employed to increase litter survival rates." In large outdoor enclosures around the United States, some pups die naturally and some survive, and this ensures at least some natural selection in the captive population.

Despite the SSP strategies, we might ask: are zoos and captive propagation facilities undesirable allies in the conservation movement? This question has been posed by some conservationists using strong arguments: the presence of zero animals in the wild (as with condors) does not motivate habitat protection; creatures in cages produce offspring that will not survive in the wild, even if habitat for them is established later. In terms of small population biology, though, natural sanctuaries may be thought of as large, natural habitat zoos; small and fragmented or isolated populations in the wild require the same type of management as captive populations. A metapopulation of isolated, captive populations and wild populations provides a degree of biological security to protect endangered animals against stochastic disasters, like contagious disease outbreaks or physical plant destruction by fire or earthquake.

The theory of minimum viable population size, combined with a species' biological needs and the need to protect small populations against catastrophic disasters, dictate a minimum viable reserve or zoo sanctuary size. Although this suggests a combined zoo and reserve carrying capacity for the species, carrying capacity will ultimately be determined by cost-benefit analyses. Triage will occur; species to be saved will be selected, based upon available species-specific space in captivity and in the wild, and on the best use of available financial resources for maximizing preserved biological diversity (Foose, pers. comm.; Foose, 1983).

Minimum viable population size for the red wolf, given the number of founder animals and reproductive constraints, is about 200 captive and 300 wild wolves. Currently, the red wolf numbers fewer than 100 captive and wild individuals in less than 10 sites. The SSP Propagation Group that manages these populations, consists of wolf biologists, U.S. Fish and Wildlife Service administrators, population biologists, and zoo curators.

The Burnet Park Zoo (Syracuse, NY) red wolf exhibit is fairly typical of other exhibits around the country. It is 60' x 140' and is

heavily planted for the wolves' benefit. An "airlock" or double-door security entry is the Keepers' means of access into the enclosure. The enclosure fence is medium-gauge chain link, buried 18" in the ground and rising minimally 10' above wolf grade. Vegetation in the exhibit includes pine (*Pinus*), buckthorn (*Rhamnus*), grape (*Vitis*) and a large fir (*Abies*). We have resisted pressure to cut large amounts of vegetation in the exhibit to provide full and easy public viewing of wolves, even though it is often difficult to catch a glimpse of our animals. We have cut some shrubs to give views of their preferred resting sites, but we feel that the public should not be given an easily-gained, full and complete view of every single species in the zoo. The density of red wolf exhibit vegetation was justified in local media by the reintroduction objective of the SSP. It was emphasized that if adults remain skittish and wild, even in captivity, perhaps their pups will survive better when reintroduced into the wild. The observation time required to get only fleeting glimpses of wolves darting in and out of the vegetation, seems to have been accepted by the public as a valuable part of a wilderness-like experience in the midst of an urban park.

Burnet Park Zoo wolves each receive two pounds of Nebraska Brand Feline Diet and two cups of Blue Seal Natural 26 dog food daily, along with deer legs, cow bones and rabbit heads as treats. Megage acetate ("The Pill") is fed in the diet from January through March to restrict reproduction in those pairs selected out of the SSP breeding program in any given year.

Given current exhibit needs and expenses, it will be expensive to increase the number of captive wolves to reach the SSP population goal. Fortunately, zoos around North America are undergoing a "renaissance" of sorts, and many are now capable of successfully raising funds for new exhibits and off-exhibit breeding areas. Also, SSP species are being given high priority for exhibition and breeding in most zoos. Presentation of charismatic megavertebrates, like red wolves, to the large numbers of zoo-going public develops a sense of appreciation and concern for the living animals that people can meet "up close and personal." Conservation education, especially about the need for habitat and species protection, is another primary mission of SSP-participating zoos. Exhibit graphics, zoo volunteers and special programming all contribute to the goal of increasing public awareness of the plight of endangered SSP species. Media attention to SSP species and their newborn offspring also increases public awareness of program objectives such as increasing reproduction and eventual reintroduction of the animals into the wild. Our assumption is that awareness and concern for the species will stimulate the public to contribute to the total conservation effort, including the building and renovation of captive habitats, lobbying for legislation, and acquisition and protection of reserve natural areas.

## REINTRODUCTION AND SPECIES SURVIVAL

The primary habitat requirement of the red wolf in its final range was heavy vegetative cover along bayous and fallow fields that constituted the primary resting and denning areas for the species. Access roads, dikes and canal levees provided canid travel routes through the area. Still, coastal marshes did not appear suitable for year 'round habitation by wolves, and their lifespans there appeared to be very short (Carley, 1979a). Some information suggests that the species was most numerous in the once-extensive, densely-vegetated bottomland forests of the southeast; hence, this habitat type, in preference to coastal marsh, is considered more typical, preferred habitat (Parker, 1986).

Unlike gray wolves, red wolves are believed to subsist more on small animals than on "big game". A recent (1978) experimental release of a pair of red wolves onto Bulls Island of the Cape Romain National Wildlife Refuge, South Carolina, confirmed this: analysis of red wolf scats collected during the translocation showed that the animals preferred marsh rabbits, and ate squirrels, muskrats, nutria, small rodents, fish, insects and plant materials (Parker 1986). The animals also fed on road-killed deer (Carley 1979b). Natural behavior of the red wolf is not well known, and only study of such released animals will provide information needed to preserve the species (Carley, 1979a; Parker, 1986).

The primary objective presently established for the species by the U.S. Fish and Wildlife Service and the AAZPA/SSP is to produce enough red wolves in captivity to effect recovery of the species in the wild. There is not a lot of suitable habitat available for reintroduction of the species into the southeastern United States, but the program goal remains reintroduction into the historic range and subsequent establishment of self-sustaining wild populations (Parker, 1986; R. Smith, pers. comm.).

Conservation should mean more than just saving a species' genome; it should involve maintaining the ecosystem in which the species lives and evolves (Western, 1986). As noted above, self-sustaining, captive populations can help conserve nature by interchanging genes with larger, self-sustaining, natural populations. Since evolutionary forces in captivity can never mimic the totality of selective forces acting upon the parent genome, large captive populations relative to natural populations can conceivably swamp natural evolutionary processes (Western, 1986). Because the red wolf has been breeding in captivity for only a short time, however, we expect this to be a minor issue for population management at this point.

Alligator River National Wildlife Refuge, in Dare and Tyrrell Counties, North Carolina, is over 135,000 acres of protected red wolf habitat. The Prudential Insurance Company bought thousands of acres of Dare County land in 1980 and 1983. When environmental groups sued to stop Prudential's plans to drain and develop the property for farmland, the Nature Conservancy orchestrated a 1984 donation by Prudential of 118,000 acres of pocosin wetland, forest and coastal marsh to the U.S. Fish and Wildlife Service, and a new federal wildlife refuge for migratory waterfowl, raptors and other wildlife was created. The peninsula is relatively isolated from the neighboring Outer Banks and surrounding tourist areas, and it is sparsely inhabited by humans. On the south side of the Peninsula is a 47,000 acre bombing range; the Department of Defense is managing about 19,000 acres for conservation, through an agreement with the North Carolina Natural Heritage Program. Recently, the North Carolina office of The Nature Conservancy acquired an additional 6,000 acres of land along the Alligator River south of the refuge (Bass, 1987). Still, success of the wolf release was jeopardized by a 10,000 acre Prudential inholding that bisected the refuge. The property was half-drained farmland that provided important forage for deer and other wildlife; lack of USFWS control over the tract meant that it could not regulate public uses such as hunting, so purchasing the property was the Service's top acquisition priority for the Refuge. The Nature Conservancy bought the land for \$3.5 million in January, 1988 after the USFWS signed a letter of intent to buy the tract from the Conservancy (Henderson, 1988). Because it takes federal agencies years to acquire money for equipment through the budget process, The Nature Conservancy provides a quicker-acting intermediary means of negotiating and obtaining desirable critical properties when they are offered for sale. The Alligator River NWR is now a large, isolated tract of protected

land, perfect for a red wolf release.

Public antagonism had killed a previous reintroduction effort in the south, and Fish and Wildlife Service biologists were careful to plan a more acceptable proposal for Alligator River. Key arguments were that red wolves generally prey on small mammals, not deer; the wolves would have "experimental population" status to authorize more active management of the animals; and a special rule allowing managers to "take" released wolves to replace their high-tech remote recapture radio-collars, to provide veterinary care, to return to the Refuge animals that stray outside its boundaries, or to return to captivity animals that threaten human safety or property (Poulos *et al.*, 1986). A major goal was that red wolf release would not have significant impact on public use of the area, including hunting and trapping. In addition, it is significant that the FWS personnel involved are extremely outgoing and personable; they have apparently swayed local public opinion in favor of the wolf through numerous informal contacts with friends and neighbors.

Eight captive-born animals arrived at Alligator River in November, 1986, and were placed, as pairs, in widely separated, remote acclimation pens. They were switched from captive diets to road-kills and live prey, and prepared for "slow release," a process similar to raptor hacking. In 1987, high-tech recapture collars were finally produced, and the wolves were released. An additional pair of animals was placed on Bulls Island, South Carolina in November 1987 for 1988 release; this pair had two pups as of June 1988. The four pairs released on Alligator River were monitored daily by radio receivers. One released female died in December, 1987; her mate was recovered in January 1988, re-paired with another female, and that pair was released in April 1988; the male of the pair was hit by a car in mid-June 1988, but the female remained free-roaming as of 20 June 1988. The male of a second pair was hit by a car in May 1988, but his mate remained free and produced a female pup which was seen with its mother as of 20 June 1988. The third pair was alive as of 20 June 1988, and had a female pup. The male of the fourth pair created a small problem: he was frequenting a local community in November 1987, so was recovered and replaced in his pen; his mate was euthanized in December 1987 due to injuries received in the wild from another red wolf, so he was re-paired in winter 1987, and released again in April 1988; he was observed again in the local community in May 1988, and again was recovered and replaced in captivity in June 1988 (M. Phillips, pers. comm.). Although six of the original ten released wolves either died or were recovered, the remaining released animals generally are killing food animals, eating well and staying healthy. Other captive wolves successfully augmented deaths or removals, and the two pups wild-born to the only two females which could possibly have pups, survived their first summer. The wild red wolf pups are an important preliminary indication of the success of the current propagation and reintroduction program.

In summary, the Fish and Wildlife Service provided field biologists, refuge management, and biological protection that allowed establishment of the red wolf in captivity and its eventual reintroduction. A group of individual zoos across North America, united by the AAZPA Species Survival Plan for red wolves, is providing captive habitat, propagation and expertise on the biology of small populations. The Nature Conservancy, with its corporate and political contacts and private sector expertise, has provided refuge lands through contracts and purchases. Together, these diverse organizations have rescued the red wolf from the brink of extinction, and have moved it closer to the self-sustaining population status which it so desperately needs.

## ACKNOWLEDGMENTS

Thanks to Tom Foose for Species Survival Plan information, to Warren Parker, Roland Smith and especially Mike Phillips for technical information and review of this paper, and to Dick Coles for critical review of the draft manuscript. Special thanks are due to Friends of the Burnet Park Zoo, Inc. for monetary support for my participation in the Red Wolf SSP, and to that organization's staff and volunteers for typing this manuscript and for moral support.

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# Management Oriented Research: Marine Turtles in the Southeastern United States

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**Abstract:** The National Marine Fisheries Service of the National Oceanic and Atmospheric Administration is the lead federal agency for the protection, management and conservation of marine turtles in the water. The Endangered Species Research Program at the Southeast Fisheries Center in Miami, Florida is a multi-laboratory program which focuses on the five marine turtle species that occur in southeastern U.S. waters. These species are the loggerhead, green, leatherback, hawksbill and Kemp's ridley turtles. All are currently listed as either endangered or threatened throughout their entire ranges or portions of their respective ranges. The Kemp's ridley, which decreased from at least 40,000 females nesting in 1940 to about 600 in 1987, is considered the most endangered marine turtle species in the world. The decline was largely caused by continued exploitation of turtles and eggs on the nesting beach. From 1978 to 1985, the number of females nesting decreased at an average annual rate of 3%. We have determined that one of the most significant causes of this recent decline is the incidental capture and drowning of turtles in shrimp trawls. To eliminate this cause of mortality, a Turtle Excluder Device (TED) was developed by the Southeast Fisheries Center that is capable of releasing the living turtles from shrimp trawls. Regulations that require the use of TEDs were finalized in June, 1987 and implemented October 1, 1987. To improve recruitment into the breeding population a "headstart" program was initiated in 1978. Up to 2000 turtles are reared in captivity and released at about one year of age. The working hypothesis for the program is that turtles of about one year of age have reached a size that makes them less likely prey for birds or fish. Thus, these turtles receive a "headstart" on survivorship.

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## BACKGROUND INFORMATION

The National Marine Fisheries Service (NMFS) of the National Oceanic and Atmospheric Administration (NOAA) is mandated to protect, conserve and manage marine turtles in the water, under the Endangered Species Act, 1973 (PL 93-205 and amendments, heretofore referred to as the ESA). The U.S. Fish and Wildlife Service (USFWS) retains the lead agency position for marine turtles while on land. This arrangement was formalized in a Memorandum of Understanding (MOU) between the U.S. Departments of Commerce and Interior. The Southeast Fisheries Center (SEFC) is one of six research centers within NMFS. The SEFC Protected Species Program is divided into an Endangered Species Program and a Marine Mammal Program. The Endangered Species Program is essentially devoted entirely to marine turtles. Shortnose sturgeon are managed currently by the USFWS and various states while NMFS maintains an advisory role in the management of this anadromous species.

## PROTECTED SPECIES PROGRAM

### Program Overview:

There are five species of marine turtles that predictably occur in southeastern U.S. coastal waters. The loggerhead turtle (*Caretta caretta*) is listed as threatened, though it is the most abundant and conspicuous species found in the U.S. In fact, what may be the world's largest nesting aggregation occurs on beaches from North Carolina to Florida. The green turtle (*Chelonia mydas*), which continues to be exploited, is listed as endangered in Florida, and threatened elsewhere in U.S. waters. The leatherback turtle (*Dermochelys coriacea*), the species largest in size, is listed as endangered. The hawksbill turtle (*Eretmochelys imbricata*), the source for tortoise-shell is also listed as endangered. The Kemp's ridley (*Lepidochelys kempi*) is the focus of much of our current research effort because of its extremely precarious state.

The objectives of our program are: 1) to obtain information on distribution, abundance and life history of marine turtle stocks; 2) to obtain estimates of mortality and to identify patterns and causes of mortality; 3) to evaluate the status of stocks over time, and 4) to

determine the impact of each source of mortality at a given population level. From these assessments, management options will be developed. It is our responsibility to evaluate the effectiveness of our management measures relative to species recovery.

Our research activities are specifically directed toward meeting the objectives outlined above. We have completed aerial pelagic surveys, aerial and ground-nesting beach searches and small-scale vessel surveys to describe the distributions and abundance by species in the southeastern U.S. From our pelagic-aerial and nesting-beach surveys we have established baseline estimates for the abundance of loggerhead and leatherback turtles in the waters of the southeastern U.S. To describe habitat utilization and residency in local areas, we have radio tagged turtles and we continue to tag turtles with conventional flipper tags.

Empirical estimation of mortality requires the placement of observers on fishing vessels, oil platforms and at power stations. All data collected by observers are included in our assessments of the status of turtle species in U.S. waters. The assessments are then included in the "status reviews" required of the agency by the ESA at least once every five years, and these reviews are used by the Secretary of Commerce to determine the need for listing or de-listing endangered and threatened turtle species.

### Management Oriented Research:

Our research program is management oriented with the major goal of promoting species recovery. As an example I will highlight research activities that focus on the recovery of Kemp's ridley turtle, which, as stated previously, is in the most precarious state of all marine turtle species worldwide.

The biology of this turtle differs significantly from other species. It is the only turtle that demonstrates a type of social behavior, though this is limited to nesting habits. It nests in large aggregations called "arribadas" (Spanish for arrived groups) during daylight hours. Other species nest during the night. Nesting is seasonal, beginning in May and continuing through August. So far, we have only found one nesting beach, a 17 km long strip on the east coast of Mexico near Tampico. We have yet to identify additional nesting beaches, though

**Table 1.** Estimated annual capture and mortality of sea turtles by offshore commercial shrimping in the southeastern U.S. (Henwood and Stuntz, 1987)

Area	Estimated Capture by Species					
	Loggerhead	Kemp's	Green	Hawksbill	Leatherback	Total
N.C. to Fl. Keys	32,120	1,268	493	70	211	34,162
Gulf of Mexico	10,789	1,726	432	432	432	13,811
Totals	42,909	2,994	925	502	643	47,973

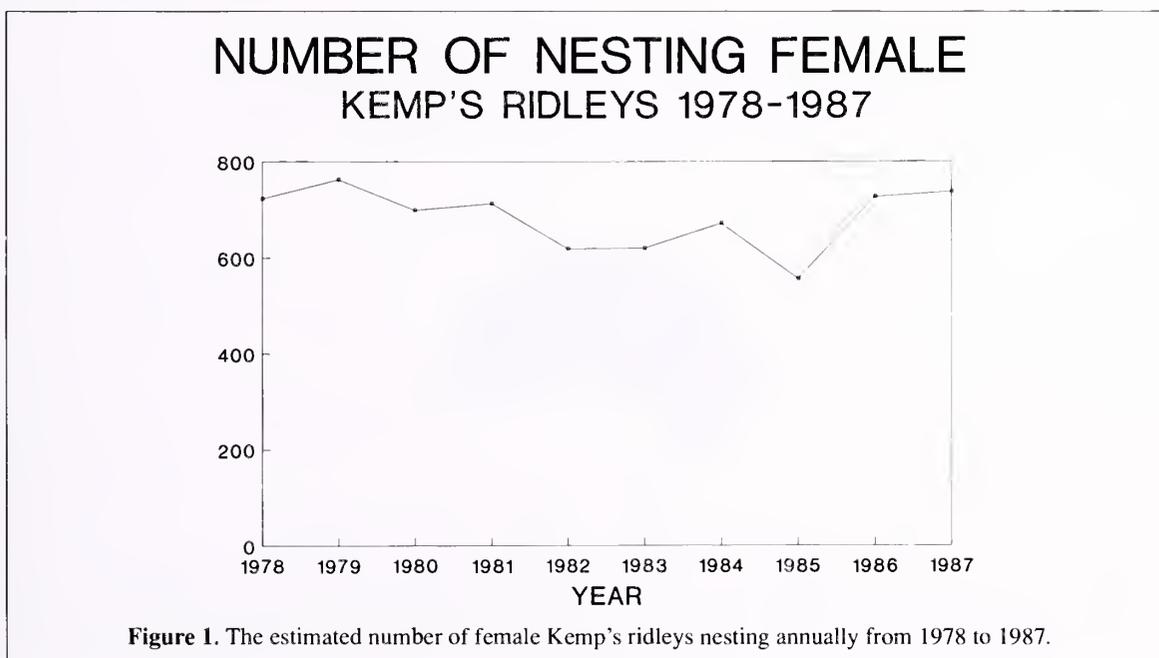
  

Area	Estimated Mortality by Species					
	Loggerhead	Kemp's	Green	Hawksbill	Leatherback	Total
N.C. to Fl. Keys	6,745	266	104	15	44	7,174
Gulf of Mexico	3,129	501	125	125	125	4,005
Totals	9,874	767	229	140	169	11,179

sporadic nesting is known at other Mexican locations. The only information we have available to assess the status of the species is an estimate of the number of females nesting every year. This is obtained by counting the number of nests excavated on the beach and dividing the total count by the number of nests excavated per female. A per capita nest production of 1.3 nests was derived from Marquez et al (1981) as an average, utilizing data collected over fourteen nesting seasons. Based on the estimated number of females nesting every year, we have determined that from 1978 to 1985 this population component decreased at an annual average rate of 3%. We interpret this to mean that, over this period, recruitment into the nesting stock was less than mortality suffered by nesting females in subsequent nesting years. While the estimated number of nesting females appears to have fluctuated annually, it appears to have remained stable from 1986 to 1987 (Fig 1).

Research efforts over the past ten years were focused on determining reasons for this population decline. The initial estimate for an index of nesting females dates back to 1947, from a home movie showing an "arribada" of at least 40,000 females (Rebel, 1974). This figure represented one day during a three month nesting season, so the value likely represents a minimum estimate of females nesting for that year. Our current estimate of nesting females is about 600. The major cause of the precipitous decline was undoubtedly the uncontrolled removal of turtles and eggs from that major nesting beach from the 1940's through the 1960's. In 1973 Mexico began protecting this beach and adjacent waters, and the government currently deploys marines to patrol the beaches and deal with poachers. Guarding of the beaches protects both adult females and eggs.

Additional mortality of juvenile and sub-adult turtles is caused by the inadvertent capture and drowning of turtles in shrimp trawls. We



first needed to determine the extent of mortality from shrimping, so observers were placed on shrimp vessels. From data collected by these observers, we determined that almost 800 primarily juvenile and sub-adult turtles were being killed by shrimpers and that this most likely represented the major in-water impact on this species (Table 1). Our approach for species recovery has been two-fold:

- 1) to reduce mortality caused by shrimping and increase recruitment into the breeding population;
- 2) to increase recruitment through captive rearing of no more than 5% of the total hatchlings produced each year.

Our initial strategy was to develop a device that shrimpers could use to eliminate turtles as by-catch. In addition, we would attempt to increase the chances of hatchlings surviving to adulthood by raising them and releasing them at about one year of age, when they are appreciably larger than hatchlings and less likely to be preyed upon by fish and birds.

### TURTLE EXCLUDER DEVICE DEVELOPMENT AND REGULATIONS

After 27,000 hours of observation on shrimp vessels from 1978 to 1983 it was estimated that about 48,000 turtles are caught each year with about 11,000 of these turtles drowning accidentally in shrimp nets (Henwood and Stuntz, 1987). About 90% of the turtles caught are loggerhead turtles, with six per cent identified as Kemp's ridleys and three per cent as green turtles. About 1% were identified as hawksbill and leatherback turtles (Table 1).

Reduction of the incidental capture and drowning of turtles during shrimping operations requires the use of a device to exclude turtles from the nets. In 1978 NMFS initially tested the placement of webbing in the opening of the net. While this reduced turtle catch by 75% it also resulted in shrimp losses of 15-35% which was an unacceptable level. Research was then directed at developing a method to release turtles once they were in the trawl. By 1981 a Turtle Excluder Device or TED was developed that released 97% of turtles captured and prevented any statistically significant loss in shrimp catch (Final Supplement to the Final Environmental Impact Statement Listing and Protecting the

Green Sea Turtle, Loggerhead Sea Turtle and Pacific Ridley Sea Turtle Under the Endangered Species Act of 1973, U.S. Dept. of Commerce, NOAA/NMFS. June 1987). Regulations requiring the use of TEDs went into effect June 29, 1987 (Federal Register, Part II Dept. of Commerce NOAA 50 CFR Parts 217,222 and 227 Sea Turtle Conservation; Shrimp Trawling Requirements; Final Rule).

The final regulations require the use of several conservation measures while shrimping. In inshore waters, tow times are limited to a 90 minute maximum. In offshore waters, TEDs are required in all trawls. There are also seasonal constraints that optimize conservation measures during those times of year when turtles are found coincidental to shrimping. Only one area requires the use of TEDs year round. This is the area off of Cape Canaveral on Florida's east coast, known to harbor unusually large numbers of sea turtles (Schroeder and Thompson, 1987). Regulations first went into effect on October 1, 1987 in the Cape Canaveral area. The required use of TEDs is to be completely phased in by May 1, 1990. In addition to the NMFS TED there are five other TEDs developed by fishermen and certified by NMFS to exclude at least 97% of turtles captured.

### HEADSTARTING

A captive rearing program was initiated in 1978 as a cooperative project with the U.S. Fish and Wildlife Service, National Park Service (NPS), the Mexican National Institute of Fisheries (PESCA) and NMFS. Details on the incubation and captive rearing of Kemp's ridley have been published (Caillouet et al, 1987). All the eggs laid, about 60,000 per year are removed and incubated at a local hatchery. Of these about 2,000 are shipped to the NMFS Galveston Laboratory (Texas) where they are hatched and the young reared to 8 to 12 months of age. Hatchlings are "imprinted" onto Padre Island National Seashore beaches (NPS) and released into the Gulf of Mexico from shipboard. All the turtles are tagged before release. There has never been a verified return of a "headstarted" turtle to the nesting beach; however, tagged, headstarted turtles have been reported in the wild in non-nesting areas. The age of sexual maturity for females is probably about 7 to 10 years. If headstarted turtles are "recruited" into the breeding population, the first recruits should have nested in 1986,

## ALLOCATION OF FUNDS ENDANGERED SPECIES PROGRAM

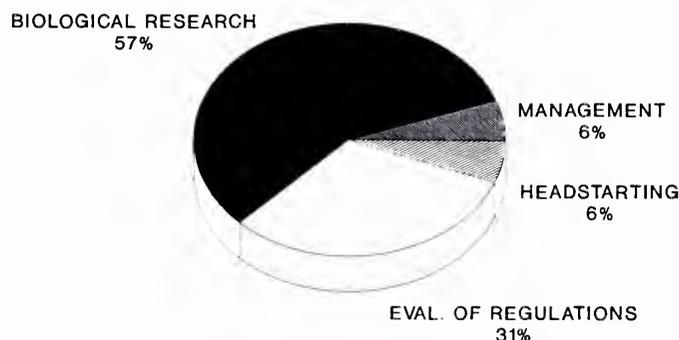


Figure 2. The allocation of Endangered Species Program funds by research objective for fiscal year 1988.

which does coincide with the first recorded increase in nesting females since 1978.

This program is considered experimental, and, because we have no verified return of a sexually mature animal, the existing program has not been expanded. At this time, we plan to take no more than 2000 eggs or hatchlings for the purposes of headstarting, a level of removal that we believe poses no risk to the population.

### CONCLUSIONS AND PROJECTIONS

With regulations that require the use of TEDs having been developed, we are now involved in research activities that will allow us to evaluate the impact of those regulations on both turtles and the shrimp fishery. We will continue headstarting activities and long-term biological research. The biological information we accrue will allow us to assess trends in distributions and abundance of turtles over time, at least at a local level. These assessments will allow us to identify potential sources of mortality and develop further methods to mitigate this mortality process.

Programmatic funding is allocated by research objective. Fifty seven per cent of our 1987 budget was allocated to developing long and short term data bases that allow us to assess the status of stocks. Thirty one per cent was allocated to evaluating the impact of our management methods. Six per cent of our total budget was dedicated to headstarting Kemp's ridleys. Six per cent was dedicated to the management of a multi-laboratory and multi-agency program. The evaluation of TED regulations will take several years. In 1987, to evaluate the impact of TEDs on mortality, we initiated systematic sampling for dead turtles that strand on the coast. Continuation of sampling will allow us to estimate total mortality for the first time. Presumably, mortality will decrease when TEDs are used. To evaluate the impact of the TED on turtle stocks requires continued monitoring of the number of females nesting each year. Again, there should be increased recruitment over time to the nesting beaches, but we likely will not be able to measure

this for at least another three years because of the protracted age to sexual maturity. We must also evaluate the economic impact of these regulations on the shrimp fishery, which in total value is one of the more important fisheries in the U.S. The jury is still out on headstarting. While we have yet to place all of our "eggs in one basket" so to speak, we will continue headstarting at a level of about 2000 turtles each year. We hope that the last two nesting seasons bode well for our management approach and that Kemp's ridley may be on the road to recovery.

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# Assessing Cumulative Impacts to Wintering Bald Eagles in Western Washington

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**Abstract:** The Bald Eagles (*Haliaeetus leucocephalus*) of Washington, the largest wintering population in the lower 48 states, are subject to numerous pressures and impacts from human activities. An evaluative method was developed to estimate potential cumulative impacts of multiple hydroelectric development and logging activities on known and potential eagle use areas. Four resource components were assessed: food supply, roost sites, mature riparian forest, and disturbance. In addition to actual estimates of losses in food supply (fish biomass in kg) and habitat (km<sup>2</sup>) in one river basin, impact levels from 0 (none) to 4 (high) were assigned for each development and for each component based on the impacts anticipated and the estimated value of the site to eagles. Midwinter eagle surveys, aerial photography, topographic and forest stand maps, and site visits were used in the analysis. Impacts were considered additive for all but the disturbance component, which was adjusted for potential synergism between developments. Adjustments were made for mitigation before the impacts were aggregated into a single, dimensionless cumulative impact score. The evaluative method and another, more quantitative, approach were applied to two river basins and helped distinguish potentially harmful from relatively benign development scenarios. Ways of improving the evaluation method are also discussed.

Pages 144 - 150. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

A Bald Eagle, perched on a large tree along a river or estuary, is a common winter sight in western Washington. Washington is a stronghold for this threatened species (endangered in most states). About 1,400 Bald Eagles winter in Washington, nearly 10 per cent of the wintering population of the entire lower 48 states (Department of the Interior, 1986; Millsap, 1986). Although Bald Eagle populations appear to have stabilized and are even increasing in some areas, there are still numerous threats to their full recovery (Green, 1985; Gerrard, 1983). Although the impacts from pesticides and shooting have declined, Bald Eagles are still subject to impacts to their food supplies, disturbances from human activities, and habitat losses from logging, energy development (especially hydroelectric), transmission lines, and commercial, residential and recreational expansion (Fraser, 1985; Department of the Interior, 1986; Fielder and Starkey, 1986). Furthermore, the cumulative, long-term effect of individual and small actions poses the single most important threat to the recovery of Bald Eagles in the Pacific states (Department of the Interior, 1986). Few cumulative impact assessment methods have been developed or used, despite the requirement to do so under several federal and state laws, regulations and court rulings (Paquet and Witmer, 1985).

The identification and management of Bald Eagle wintering habitat has received less attention than nesting habitat, despite the fact that survival rates may currently be more important than reproductive rates in determining Bald Eagle population size (Grier, 1980; Green, 1985). A Habitat Suitability Index (HSI) model has been published for Bald Eagle nesting habitat (Peterson, 1986), but not for wintering habitat. A draft HSI model for Bald Eagles in Alaska emphasizes nesting habitat and only includes one aspect of wintering habitat—food supply (Department of Interior, 1980).

Our objectives are (1) to describe the main components of Bald Eagle wintering habitat that should be included in the assessment or management of wintering Bald Eagle populations in the Pacific Northwest, and (2) to discuss the results of efforts to assess the potential for cumulative impacts to wintering Bald Eagle populations in the Hamma Hamma and Snohomish River Basins of western Washington.

## STUDY AREAS AND METHODS

### Hamma Hamma River Basin:

The Hamma Hamma River Basin is west of Seattle, Washington, and lies within Jefferson and Mason Counties. The 218-km<sup>2</sup> basin is relatively undeveloped with 80 per cent in public ownership (Olympic National Park and Olympic National Forest). The main land uses are forestry and recreation. Three hydroelectric projects are proposed for development in the basin.

The maritime climate is characterized by mild, wet winters and cool, dry summers. Average annual precipitation in the basin ranges from about 380 cm at the western boundary to about 150 cm at the eastern boundary. The average annual discharge of the 29-km-long river into Hood Canal exceeds 16 cubic meters per second, but varies dramatically on a seasonal basis.

The river supports one of the last remaining natural (non-hatchery) runs of anadromous fish in the Puget Sound area. Runs of chum salmon and pink salmon (odd-numbered years only), occur in the lower river, along with small runs of chinook salmon, coho salmon, and steelhead trout. In 1983, the Washington Department of Fisheries began releasing juvenile chinook salmon in the Hamma Hamma River above the anadromous fish barrier (a waterfall) that occurs 4.3 km upstream from the river's mouth.

The basin is mountainous and heavily vegetated, primarily with coniferous forest. Six to 23 Bald Eagles winter in the basin, and they are observed primarily near the mouth of river where they congregate to feed on spawned-out salmon.

### Snohomish River Basin:

The Snohomish River Basin is north and east of Seattle, Washington, and lies within King and Snohomish Counties. The approximately 4,662-km<sup>2</sup> basin is relatively undeveloped, but it is being influenced by the growing Seattle-Bellevue areas. While much of the lowland area is available for pasture and cultivation, forestry and wood products account for the leading land uses and industries in the basin. Seven hydroelectric projects are proposed for development in the basin, and the basin is also used for year-round recreation. Land

ownership in the basin is about 50 per cent public, primarily under the management of the Mt. Baker-Snoqualmie National Forest.

The climate is maritime-influenced and varies dramatically along a gradient from the sea-level estuary to the high (2,134 m) Cascade Mountains ridges that form the eastern rim of the basin. The lowlands receive about 90 cm of precipitation per year, while the mountainous eastern portion of the basin often receives 460 cm. The lowland area has hot, dry summers and mild winters, while the mountainous areas have cool summers and cold winters with a deep snowpack.

The 33-km-long Snohomish River begins at the confluence of its two major tributaries, the Skykomish River to the northeast and the Snoqualmie River to the southeast. These subbasins drain about 2,202 and 1,813 square km, respectively. The Snohomish River Basin has more than 4,344 km of streams. The average annual discharge into Puget Sound is about 280 cubic meters per second, but this varies dramatically on a seasonal basis from 32 in late summer to 3,808 in late winter. The basin's large, natural populations of anadromous fish (primarily coho and pink salmon, but also chum and chinook salmon) contribute substantially to commercial and sport fisheries important to the economy of western Washington. These natural populations are also augmented by hatchery-reared fish.

The basin contains coniferous forest, although a large portion of the lowlands in the western third of the basin is an open, grassy, floodplain area. As many as 70 Bald Eagles winter in the basin, with heaviest use along portions of the mainstem Skykomish River and South Fork Skykomish River (Department of the Interior, 1986; Hansen, 1978).

#### Resource Components:

To assess potential impacts to wintering Bald Eagles, it was necessary to identify specific resource components that were (1) important to Bald Eagles, (2) reasonably well understood in terms of their relationships to Bald Eagles and to each other, (3) likely to be affected by proposed land use activities, and (4) reasonably practical to measure (Department of the Interior, 1981a). We reviewed the literature, visited the target basins, and used our knowledge of Bald Eagles to select four resource components for use in the assessments: food supply (primarily salmon), mature riparian forest (perch trees), roost sites, and disturbance (from human activities). The importance of these four resource components has also been noted by Steenhof (1978), Department of the Interior (1986), and Stalmaster *et al.* (1985).

In the Pacific Northwest, wintering Bald Eagles prefer streams that provide abundant spawned-out salmon carcasses. Several studies have documented a linear relationship between the number (and availability) of spawning salmon and the number of Bald Eagles wintering along a stream (Servheen, 1975; Knight and Knight, 1983; Stalmaster *et al.*, 1979; Taylor, 1986). We acknowledge that interior populations of Bald Eagles often rely upon waterfowl in addition to fish (Fielder, 1982; Isaacs and Anthony, 1987; Keister *et al.*, 1987); however, we assume that only salmon abundance is important along western Washington rivers.

Wintering Bald Eagles perch on large trees along streams, where they spend up to 90 per cent of the daylight hours resting, preening, feeding, and looking for food (Stalmaster *et al.*, 1985). Preferred perch trees are large, often dead, bigleaf maple, red alder, cottonwood, or sitka spruce trees that provide a good view of the surrounding area (Stalmaster and Newman, 1979; Fielder and Starkey, 1986). Most perching occurs along or within 50 m of the river, and Bald Eagles tend to use the highest perch sites available (Servheen, 1975; Stalmaster, 1976). We defined mature riparian forest as forest stands that are within 91 m of a stream, contain some trees at least 30.4 m tall,

and are generally more than 100 years old.

Roost sites are used communally by Bald Eagles primarily at night, but also during the day during severe, inclement weather. Roost sites are usually uneven-aged, old-growth conifer stands in a protective landform setting on the lower portion of slopes (Knight *et al.*, 1983; Keister, 1981; Anthony *et al.*, 1982; Allen and Young, 1982; Hansen *et al.*, 1980; Stalmaster *et al.*, 1985; Department of the Interior, 1986). Because prevailing winter winds are generally from the south or west, roost sites are commonly located on northeastern or eastern aspects (Hansen *et al.*, 1980; Keister and Anthony, 1983). Roost sites may be several km from feeding areas. They vary in size from 0.02 to 0.09 km<sup>2</sup> in western Washington, but may be much larger where large numbers of Bald Eagles congregate. Roost sites usually have a clear line of sight to the surrounding area, a favorable microclimate, stout perches high off the ground, and freedom from human activities (Hansen *et al.*, 1980). Roost sites are usually located in remote areas, and they may be abandoned if human activity increases in the area (Hansen *et al.*, 1980). Nonetheless, some roost sites have been used by Bald Eagles even after the adjacent areas have been logged (Knight *et al.*, 1983).

Perching, roosting, and feeding Bald Eagles are sensitive to disturbance by humans (Fraser, 1985; Knight and Knight, 1986). Bald Eagles are more sensitive to human activities on the river (*e.g.*, boating or fishing) or to humans walking along the river's edge than to vehicle traffic or airplane flights (Stalmaster and Newman, 1978; Knight and Knight, 1984; Department of the Interior, 1986; Servheen, 1975; Skagen, 1980; Stalmaster, 1980). Bald Eagles can also adjust to some human activities, especially where habitat is optimum and food is abundant (Russell, 1980; Stalmaster and Newman, 1978; Biosystems Analysis, Inc., 1980; Steenhof, 1978). Human activities beyond 500 m from Bald Eagle use areas seldom disturb the birds (Stalmaster and Newman, 1978).

#### Data Sources:

Annual counts of wintering Bald Eagles in each basin from 1982 through 1985 were obtained from the Natural Heritage Data Base, Washington Department of Game, Olympia, Washington. Additionally, Bald Eagle numbers, habitats, and habitat-use patterns for the Skykomish River area were available from the Forest Service (Hansen, 1978, 1979; Hansen and Bartelme, 1980; Hansen *et al.*, 1980). Fish species composition and run-size estimates for the two basins were obtained from the data files of the Washington Department of Fisheries, Olympia, Washington. Potential Bald Eagle habitats in the basins were mapped using site visits as well as maps and aerial photos on file at Forest Service offices in Seattle and Hoodport, Washington. Information on past, current, and proposed logging in the basins was obtained from the same Forest Service offices. Information on proposed hydroelectric developments in the basins was obtained from license applications submitted to the Federal Energy Regulatory Commission.

#### Cumulative Impact Analyses:

The Hamma Hamma River Basin analysis concentrated on two resource components: food supply and potential roosting habitat (Federal Energy Regulatory Commission, 1986). Mature riparian forest was not included; it is presently abundant and not threatened by logging, because the Forest Service leaves buffer zones along major streams. The human disturbance component was also not evaluated, because there is relatively little development or human activity during the winter. The fish biomass requirement of wintering Bald Eagles was

estimated using parameters from published literature (Table 1). Although there are no known roost sites within the basin, one or more are suspected to exist (K. McAllister, pers. comm.). Consequently, we mapped potential roost habitat, defined as mature or old-growth forest stands containing trees with  $\geq 0.53$  m diameter at breast height within 0.8 km of the Hamma Hamma River and its main tributaries. We then assessed the per cent changes that would occur in potential roost habitat with proposed logging activities and hydroelectric development within the 99 km<sup>2</sup> mapped area.

**Table 1:** Method of determining the theoretical Bald Eagle population supported by available fish biomass.

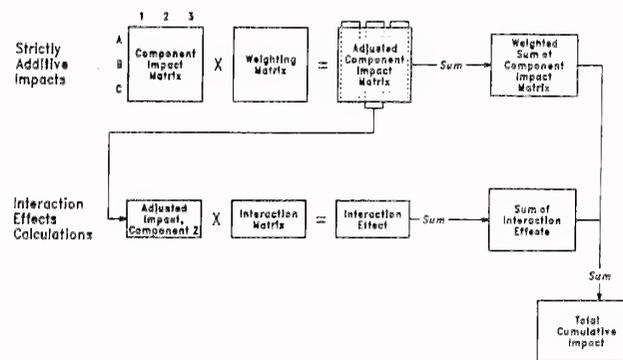
Parameter or adjustment	Value or calculation	Reference/source
A. Number of fish in run	From data files	Washington Department of Fisheries, Olympia, WA
B. Number of fish in run adjusted for avail ability	14% of (A)	Stalmaster, 1983
C. Biomass of fish available (kg)	8.9 kg times (B) for chinook 2.9 kg times (B) for chum 1.1 kg times (B) for coho or pink	Stalmaster, 1983 Williams <i>et al.</i> , 1975
D. Biomass of fish available adjusted for decomposition (kg)	15% of (C)	Stalmaster, 1983
E. 4.5 kg Bald Eagle food requirement per day	0.5 kg per day	Stalmaster and Gessman, 1984
F. Length of wintering period (15 Nov - 15 Mar)	120 days	Steenhof <i>et al.</i> , 1984
G. Fish biomass requirement per Bald Eagle per wintering period (kg)	(E) times (F) = 60 kg	
H. Theoretical Bald Eagle population	(D) divided by (G)	

The Snohomish River Basin analysis covered all four resource components (Federal Energy Regulatory Commission, 1987). Because of the size of the basin and an incomplete data set, however, the analysis was kept relatively more simple and qualitative than for the

Hamma Hamma River Basin. An evaluative method using multiple matrices was used to assess the potential effects of multiple hydroelectric development on wintering Bald Eagles (Bain *et al.*, 1986). The method consisted of three phases: analysis, evaluation and documentation. In the analysis phase, matrices were used to organize data and compute relative levels of cumulative impact on resource components. The criteria used for the four resource components are presented in Table 2. The total impact of a combination of hydroelectric projects was computed as the sum of all project-specific impacts, adjusted for the effects of interactions among projects' impacts (Figure 1). We considered interactions (or synergism) only in assessing the effects of the human disturbance component. This is because the Bald Eagle population in the Snohomish River Basin is considered a single population, and because Bald Eagles respond strongly and unpredictably to human disturbance, especially in combination with other types of impacts (Fraser, 1985). The analysis phase resulted in a list of values representing the relative cumulative impact of every possible combination of the proposed projects. In the evaluation phase, a computer screening program was developed and used to identify project combinations that met a set of maximum cumulative impact criteria for Bald Eagles (Bain *et al.*, 1986). Also, projects were flagged that exceeded an allowable level of impact for one or more resource components, based on biological thresholds, limiting factors, or management objectives. Project combinations with one or more flagged projects were screened out and eliminated from further consideration. The documentation phase included a concise summary of the anticipated environmental impacts of each project combination remaining after the screening and verification that the combination would indeed not cause significant cumulative impacts to Bald Eagles.

Quantification of the Bald Eagle's food supply within the Snohomish River Basin was limited because of the size of the basin, the widespread distribution of Bald Eagles and anadromous fish in the basin, and the limited data on fish runs within specific portions of the basin. We were, however, able to evaluate the overall amount of fish biomass that is potentially available to wintering Bald Eagles, based on estimates of fish run sizes for the entire basin. We also compared the biomass of adult fish transported above Sunset Falls on the South Fork Skykomish River with Bald Eagle use of that area.

In some cases, the degree of impact that could potentially be felt by Bald Eagles was adjusted according to the amount of use the area received by them (for example, see Table 2). The levels of use were defined as high (10 or more Eagles usually seen in the area each winter), moderate (3-9 Eagles usually seen in the area each winter), and low (1-2 Eagles seen in the area, but not necessarily every winter).



**Figure 1:** Illustration of cumulative impact computations, using matrices, for three projects (A,B,C) and three resource components (1,2,3).

**Table 2:** Description of criteria for determining the level of impact on Bald Eagle resource components.

Impact level	Resource Components			
	Salmon food supply and availability <sup>1</sup>	Roost sites	Mature riparian forest (km <sup>2</sup> )	Disturbance factor
4 (high)	A severe or complete loss of anadromous fish.	Project area includes or physically disturbs a current use roost site.	Loss of >0.125	Some project features in moderate or high Bald Eagle use area; or winter access by developer may be a problem; or river activity by developer expected.
3 (mod)	A readily measurable impact to anadromous fish.	Project area includes or physically disturbs a past use or potential roost site.	Loss of 0.09-0.125	Some project features in low or moderate Bald Eagle use area; but no river activity by developer anticipated, and winter access by developer not expected to be a problem.
2 (low)	A low but measurable impact to anadromous fish.	Project area does not include or physically disturb, but is within 100 m of, a roost site.	Loss of 0.05-0.08	Some project features within 300 m of Bald Eagle use area; or area is already developed for other human uses.
1 (very low)	A negligible impact to anadromous fish.	Project areas does not include or physically disturb, but is within 200 m of, a roost site.	Loss of <0.05	No project features within 300 m of Bald Eagle use areas, but access to site goes through Bald Eagle use areas via routes not normally used in winter, <i>i.e.</i> , not thoroughfares.
0 (none)	No projected impact to anadromous fish.	No project features within 200 m of a current or past use roost site.	No loss of mature riparian forest.	No project features within 300 m of Bald Eagle use areas, and access to site does not pass through Bald Eagle use areas except along existing, thoroughfare routes.

<sup>1</sup>For further discussion of fisheries criteria and potential impacts, see Federal Energy Regulatory Commission, 1987.

## RESULTS AND DISCUSSION

Any of the four resource components (food supply, mature riparian forest, roost sites, or human disturbance) can play an important role in limiting Bald Eagle populations wintering in western Washington (Stalmaster and Gessaman, 1984; Hansen, 1978; Hansen *et al.*, 1980; Knight *et al.*, 1983; Stalmaster and Newman, 1978). Our analysis of two western Washington river basins suggest that one or more of these components may be contributing to significant cumulative impacts to wintering Bald Eagles in a given basin, and that different components may be involved in different basins within the same region.

### Hamma Hamma River Basin:

Food supply (fish biomass) may be a key factor in determining the wintering Bald Eagle population size of the Hamma Hamma River Basin. A maximum wintering population size of 23 was recorded in the basin in 1984. We estimated, however, that only 17 Bald Eagles can be supported in even-numbered years (when pink salmon runs do not occur) by existing fish run sizes (Table 3). This may explain why the average number of wintering Bald Eagles in the basin (13.5 for 1980-1984) is much lower than the maximum count of 23. The average number of Bald Eagles, however, can be supported in any year with existing fish run sizes. We have estimated that a reduction of about 25 per cent in available fish biomass would have to occur before the average-sized wintering Bald Eagle population could not be supported. It seems probable, however, that any reduction in existing fish run

sizes in even-numbered years would probably make it less likely that wintering populations as large as that recorded in 1984 (23 Bald Eagles) would occur again. This is especially important because three proposed hydroelectric developments in the basin could cause an 83 per cent increase in stream sedimentation during construction and an eleven per cent increase during operational years, and because other ongoing land uses (primarily road building and logging) have already caused more than a 100 per cent increase in stream sedimentation over natural (background) levels (Federal Energy Regulatory Commission, 1986). The detrimental effects of stream sedimentation on salmonids have been well documented (Everest *et al.*, 1985). Consequently, there appears to be a high potential for cumulative impacts on the Bald Eagles' food supply in the basin.

About 31.3 km<sup>2</sup> (32% of the 99-km<sup>2</sup> mapped area) of the Hamma Hamma River Basin met the criteria for potential roosting habitat. Consequently, there appears to be an adequate amount of potential roosting habitat in the basin. We estimated, however, that, at current rates of logging (about 0.27 km<sup>2</sup> are harvested each year), allowing for some growth of younger stands into the potential roosting habitat category (about 0.09 km<sup>2</sup> per year), there would only be 22.3 km<sup>2</sup> of potential roosting habitat remaining after 50 years.

The three proposed hydroelectric projects would have very little impact on potential roosting habitat, removing a total of about 0.06 km<sup>2</sup> (0.2% of that existing). The impacts of a single hydroelectric project or clearcut harvest unit could be very significant, however, if the area involved contained an actual roost site. Although no roost sites

**Table 3:** Comparison of observed wintering Bald Eagle populations and the theoretical Bald Eagle population supported by available fish biomass.

River basin/situation	Available fish biomass (kg)	Theoretical Bald Eagle population	Average annual Bald Eagle population <sup>1</sup>
Hamma Hamma River Basin			
Existing, even-years	1,046	17	13.5 (1980-84)
Existing, odd-years	2,659	44	13.5 (1980-84)
25% fewer fish, even years	785	13	13.5 (1980-84)
25% fewer fish, odd years	1,987	33	13.5 (1980-84)
Snohomish River Basin			
Existing, even-years	35,368	589	51 (1982-85)
Existing, odd-years	19,455	324	51 (1982-85)
Skykomish River, adult coho transported above Sunset Falls, by year			
1980	3,165	53	52 (1981)
1981	2,049	34	24 (1982)
1982	882	15	12 (1983)
1983	787	13	12 (1984)
1984	638	11	9 (1985)

<sup>1</sup>Skykomish River Bald Eagle population sizes above Sunset Falls are not averages, but maximum number of Bald Eagles counted each winter.

have been identified in the basin, there has not been a concerted effort to locate roost sites. Because of the significance of this potential impact, a survey for roost sites should precede any mature or old-growth forest harvest in the basin.

**Snohomish River Basin:**

Theoretically, there is an abundant food supply for the approximately 70 Bald Eagles that winter in the Snohomish River Basin; in fact, it appears that many more Bald Eagles could be supported, based on food supply (Table 3). This suggests that other factors may be limiting the population. Apparently, food may limit Bald Eagle use of some portions of the basin, for example, above the natural fish barrier, Sunset Falls. For a number of years, the Washington Department of Fisheries transported adult coho salmon above Sunset Falls, and there is a relationship between the number of fish transported and the number of wintering Bald Eagles observed above Sunset Falls (Cascades Environmental Services, 1985). We calculated a correction coefficient ( $r$ ) of 0.98 for this positive correlation. Furthermore, the method used to determine and compare the theoretical eagle carrying capacity, based on available fish biomass, and the observed number of Bald Eagles was verified using this data set (Table 3). The theoretical Bald Eagle carrying capacity and the observed number of Bald Eagles were not significantly different (Chi-square = 1.91,  $p = 0.75$ ) for the period 1981-1985.

The application of the impact level criteria (Table 2) to the proposed hydroelectric projects individually and in concert (all seven projects developed), and to logging and other activities in the basin revealed some interesting patterns (Table 4). Two projects (E and F) are expected to cause no impact to Bald Eagles because the associated impacts would occur far from areas used by Bald Eagles. Development of these projects would be consistent with policies that avoid the occurrence of impacts on the birds (Department of the Interior, 1981b). Other projects would cause various levels of impacts on Bald Eagle

resource components (Table 4). Although very low levels of impact on food supply, roost sites, and mature riparian forest might be felt, the main impacts would result from human disturbance. Human disturbance from logging, recreation activities (such as rafting, kayaking, and fishing), traffic, and activities around towns and other buildings already cause moderate to high levels of disturbance, especially along or near the privately-owned lands of the mainstem Skykomish River which is the most important area for wintering Bald

**Table 4:** Potential impacts to Bald Eagles from seven proposed hydroelectric projects (A-G, individually and combined) and from other Snohomish River Basin activities, including logging. See Table 2 for a description of impact level (0-4) criteria.

Project(s)/ activity	Impacts to resource component			
	Food supply	Roost sites	Mature riparian forest	Human disturbance
A	1	1	1	1
B	1	0	1	3
C	2	0	1	3
D	1	2	1	2
E	0	0	0	0
F	0	0	0	0
G	1	1	1	1
A-G <sup>1</sup>	2	1-2	2	3
Other basin activities	2	3-4	4	3-4

<sup>1</sup>Estimated level of impact if all seven hydroelectric projects were developed.

Eagles, based on midwinter surveys. Nonetheless, most Bald Eagles are observed along this stretch of river, which suggests that they have adjusted somewhat to current levels of human activity.

Other impacts have occurred, and continue to occur, to roosting habitat and mature riparian forest. Although the hydroelectric development considered here will not contribute substantially to changes in these resource components (e.g., less than 0.08 km<sup>2</sup> of mature riparian forest would be removed if all seven projects were developed), logging and other ongoing land use activities will continue to result in a cumulative impact. We believe that the existing, potential roosting habitat (over 800 km<sup>2</sup>) and mature riparian forest (at least 7.3 km<sup>2</sup>) are adequate to provide for existing populations of wintering Bald Eagles; however, we note that this statement applies primarily to the eastern, upper portion of the basin which is, to a large extent, in public ownership. The western, lower portion of the basin is primarily in private ownership, and deficiencies in roosting habitat and riparian perches may limit Bald Eagle use of that area (Washington Department of Ecology, 1981; Anderson, 1983). Consequently, because the main use area of wintering Bald Eagles is along the mainstem Skykomish River and in private ownership, the wintering Bald Eagles population may be affected and limited by cumulative impacts to roosting habitat, mature riparian forest, and human disturbance.

The approaches to cumulative impact assessment described in this article can be improved in several ways: (1) incorporation of more site-specific Bald Eagle occurrence and habitat use data; (2) examination of certain assumptions, for example, that only spawned-out salmon are providing the Bald Eagle food source along western Washington rivers; (3) incorporation of the impacts of more types of land use activities; (4) quantification of response curves and synergism, especially as related to human disturbance of Bald Eagles; and (4) incorporation of Bald Eagle energetics models that include more than just food supply. Hopefully this method, especially with further improvement and validation, will aid resource managers and regulatory agencies in their decision-making and compliance with the cumulative assessment requirement of the National Environmental Policy Act.

#### ACKNOWLEDGMENTS

The work reported here was conducted while the authors were at Argonne National Laboratory's Portland Office. We acknowledge the support of the U.S. Department of Energy, Bonneville Power Administration and the Federal Energy Regulatory Commission under Contract W-31-109-Eng-38 in the completion of the analyses and literature review contained in this article. Opinions and viewpoints presented in this paper do not necessarily represent official policy of the funding agencies.

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# Western New York Bog Turtles: Relicts of Ephemeral Islands or Simply Elusive?

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**Abstract:** The bog turtle, *Clemmys mühlenbergii*, reaches the north-western periphery of its range in western New York and is listed as endangered by New York State. Surveys over the past decade have identified several viable populations in the Southern Tier region of New York; however, the status of populations in the disjunct western New York range in eight historic sites, remains poorly known. No demographic studies of these populations have yet been conducted. During 1987 and 1988, the six extant western New York bog turtle sites were surveyed six or more times. A viable bog turtle population was identified at only one site, and bog turtles were not observed at any other site. An assessment of the availability and quality of suitable habitat at the western New York sites is problematic for several reasons. Exact capture localities of some of the historic specimen records are ambiguous, and the early to mid-successional communities in which the bog turtle occurs are highly dynamic over time. The zoogeography and recent history of the bog turtle in western New York are explored, and characteristics of habitat components of western New York sites are compared with those of bog turtle habitat elsewhere in its range. Methods of bog turtle habitat identification and assessment are discussed, and preliminary results of 1988 field surveys are presented.

Pages 151 - 153. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The bog turtle (*Clemmys mühlenbergii* Shoenff 1801) has probably elicited more speculation as to its rarity and natural history than any other North American turtle. Bury (1979) lists 90 citations in a special report on the ecology and conservation of the bog turtle for the U.S. Fish and Wildlife Service. Landry (1979) cited 158 references in her bibliography of the bog turtle. Nonetheless, knowledge of the ecology of the species throughout its range is incomplete, and a real understanding of the zoogeography of its distribution is lacking.

The bog turtle is the smallest of three species of *Clemmys* native to eastern North America. Members of this genus, including the spotted turtle (*C. guttata*) and the wood turtle (*C. insculpta*), exhibit tremendous versatility in exploiting both aquatic and terrestrial environments.

Throughout most of its range, the preferred habitat of the bog turtle is an open canopy, wet meadow or bog meadow with soft, muck substrate. It is generally associated with shallow water one to five cm deep, frequently as cool, running rivulets or shallow pools that persist throughout the year. Emergent tussock or hummock-forming vegetation provides low elevation and seasonal dense cover. The low herbaceous canopy creates a microclimate of high relative humidity throughout the summer. Sedge tussocks or sphagnaceous hummocks also offer important nesting and basking sites.

Bog turtles usually emerge from hibernation from mid-April to early May. They are most visible early in the season (April through early June) when they tend to bask, and when herbaceous vegetation is relatively sparse. By early June, the daytime, ambient air temperatures are sufficiently high to stimulate foraging behavior without extended basking, and turtles are able to thermoregulate by subtle movements within the herb layer and aquatic environment. The bog turtle is an extremely adept burrower, using burrowing for both escape and thermoregulation. The animals are relatively inactive or possibly aestivating during periods of high temperature (July and August), but they show increased levels of activity (usually in September), prior to hibernation. The relatively short annual activity period may be significant in influencing the northern extremes of the species range.

The bog turtle occurs in a discontinuous range from northeastern Georgia to New York. Large gaps exist between the species' core range (southeastern Pennsylvania, Maryland, New Jersey and southeastern New York) and the southern Appalachian and western New York portions of the range. The species has long been considered

one of the rarest North American turtles; however, surveys conducted independently in Maryland (Taylor *et al.*, 1984), Delaware (Arndt, 1977, 1978), and North Carolina (Zappalorti 1975; Herman and George, 1985) suggest that within portions of its range it is probably more cryptic than rare. Maryland removed the species from its endangered species list following surveys during 1976-1978 that documented 173 previously unknown bog turtle sites (Taylor *et al.*, 1984). Although listed as "threatened" in the USFWS Red Book (USD1, 1973) it is not federally protected under the federal Endangered Species Act. It is listed in Appendix II of CITES and is protected in New York, New Jersey, Pennsylvania and Connecticut. It is listed as endangered by New York State.

In 1977, New York State, in connection with the New York Zoological Society, initiated a statewide survey of the bog turtle. All historical literature citations, museum records and miscellaneous reports were compiled, and field surveys of each known site were made to assess the status of the bog turtle and suitable habitat throughout the state. While several viable populations were located in the Southern Tier, no bog turtles were found in any of the Western Tier or North Hudson Valley regions (Anon., 1980). Over the following ten years, the Endangered Species Unit of the NYS Department of Environmental Conservation continued to monitor the occurrence of bog turtles and received two, apparently legitimate, reports from the Western Tier region. One was from an historic site in Seneca County, and the second from a new site in Oswego County. The status of these sites, as well as the remaining historical sites, remains ambiguous.

In 1986, the Burnet Park Zoo initiated a Bog Turtle Conservation Program that addresses a number of aspects of the ecology and conservation of this species (Collins, 1987a). In 1987, surveys of all western New York historical sites were conducted under the support of the Zoo and the Friends of the Burnet Park Zoo, and these continued in 1988 under the additional support of the New York State D.E.C. Gift to Wildlife Program. The primary objective of these surveys was to locate and identify existing bog turtle populations and to locate and identify existing or altered bog turtle habitat that may have supported the populations in the past.

## RESULTS AND DISCUSSION

Between 21 April 1987 and 22 August 1988, 51 field surveys were

conducted at six historical bog turtle sites in western New York. Two additional historic sites were excluded from these surveys: a Tompkins County site was destroyed some years ago by a sand and gravel operation, and a Wayne County site reported in 1919 could not be located. The number of visits to each study site ranged from six to seventeen. A minimum of eight visits to each site was projected by the end of the 1988 field season.

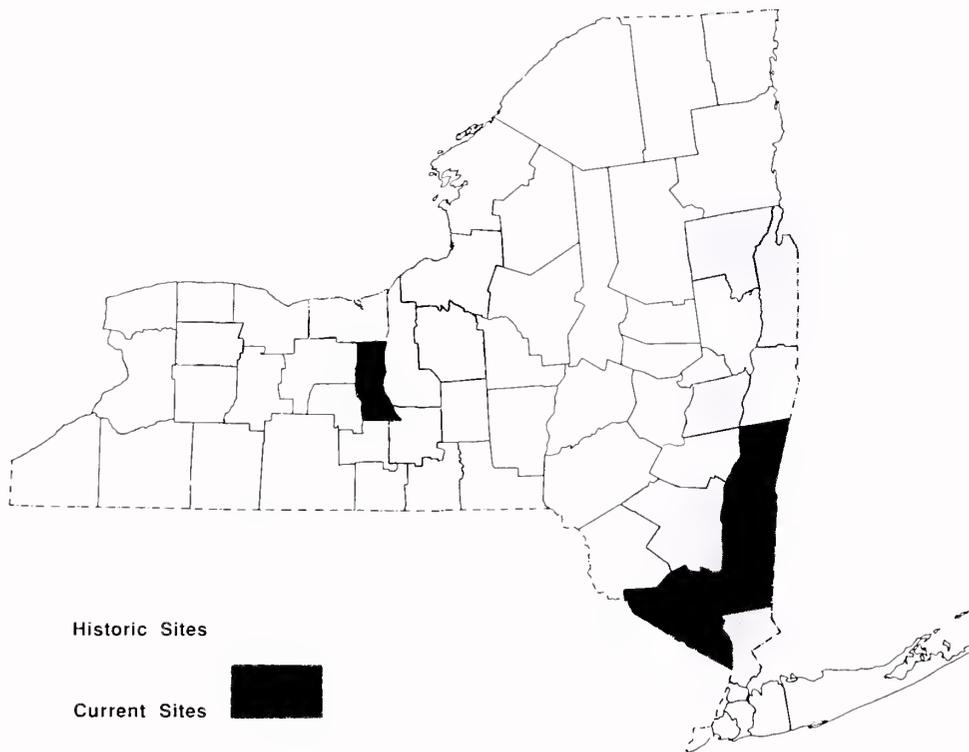
Bog turtles were captured at only one of the six historic sites (Figure 1). The age and sex distribution of the turtles captured suggests that this is a viable population. The habitat characteristics of the site conform closely to the general habitat description given earlier. The site is under open canopy; annually persistent springs maintain a surface water table, and elevated hummocks are present.

The remaining five sites are perplexing in that they appear to offer suitable habitat; however, each, with the possible exception of Bergen Swamp in Genesee County, seems to be lacking in the critical association of open canopy, persistent rivulets, dense herbaceous layer and elevated hummocks or tussocks. The author has examined each site to identify specific aspects that would render the habitat unsuitable for the bog turtle.

ground water. These "pseudo-rivulets" are highly dependent on rainfall as well as long term changes in hydrology that can affect the stability of the water table.

A third site, in Oswego County, also offers apparently suitable habitat, but the water is supplied by "pseudo-rivulets". The habitat, however, is a lake shore fen with the water table contiguous with pond level and further augmented by flow from two large watersheds. Water availability here appears to be relatively stable; however, the thermodynamics of the surface water on the fen are unknown at this time. This site offers somewhat atypical bog turtle habitat; however, it raises an important question. Will the species shift its habitat use to adapt to environmental constraints at the extremes of its range? Studies of the spotted turtle in eastern Ontario and Quebec (Cook, *et al.*, 1980) suggest that it is almost entirely a species of pond-fen communities in that region. The wet meadow-shallow marsh habitat in which the spotted turtle commonly occurs farther south apparently freezes too deeply to support populations at the northern extreme of its range. It may be significant that the spotted turtle is abundant on the Oswego County fen.

Two turtles are documented from Zurich Bog, and one animal each



**Figure 1.** Distribution, by county, of historic and current (1988) bog turtle sites in New York State.

Two sites, Cicero Swamp (Onondaga County) and Zurich Bog (a Bergen Swamp Preservation Society preserve in Wayne County) are very similar in several respects. Both sites are characterized by open canopy and mature, grounded, sphagnum bog mats. Both sites are bordered by mature, second growth, wet woods that support pools of standing water through most of the year. At both sites vegetational characteristics appear suitable. Hummocks and a dense herb layer are present, but both lack flowing springs or rivulets. Instead, narrow channels and pools between hummocks are filled seasonally by high

from Cicero and Oswego sites. What exactly do the records represent? Do these animals represent a resident or relict population at that specific locality, or are they transients?

Ecologically, the wet meadow habitat of the bog turtle is ephemeral. Pioneering woody plant species such as red maples, alders, willows and dogwoods are frequent components of bog turtle habitats. Maintenance or creation of an open canopy has been strongly reliant on changes in hydrology, such as beaver activity, fire or wind. More recently, agricultural and construction activities have had a more

profound effect on changes in wetlands. Bog turtles tend to occur in small, discrete populations, occupying suitable habitat dispersed along a watershed. The Maryland surveys concluded that most of the populations occurred in sedge meadows of less than two acres. If the habitat inhabited by a population becomes unsuitable through succession, modification by agricultural practices or other causes, the turtles can disperse into the adjacent suitable habitat. If an adjacent population exists, it may absorb some individuals and may itself expand into sub-optimum habitat and serve as a reservoir for colonization of other nearby sites. However, if there is no suitable habitat nearby, the probable result of disturbance is local extinction of the population. As such local extinctions occur the intervening distance between viable population increases, thus increasing the chances of regional extirpations. At the northern extremes of the bog turtle range, increasingly closed canopy situations probably effect reproduction rates and survivorship much more than in milder climates further south. It is probably much more critical for a transient New York population to locate open canopy habitat quickly than it is for a North Carolina population.

Zurich Bog offers an excellent example of the probable effects of succession compounded by wetland alteration from field crop agriculture. The locality record there is from an open-canopy bog "room" that lies to the south of the present quaking mat. The watershed to the east is a second-growth closed-canopy wet woods that may have provided open canopy habitat earlier in the century. Earlier in the succession of this woods, and on the grounded mat surface, streams may have been present. Soil type correlations were used successfully in Maryland and North Carolina surveys to identify bog turtle habitat. I have used them in this survey to help identify potential bog turtle habitat near recorded sites. An examination of the muck soil distribution surrounding Zurich Bog clearly illustrates the devastating effect that muck farming has had on potential neighboring bog turtle habitat. This form of agriculture is prevalent in Western New York, and is probably a significant factor in the reduction of bog turtle habitat. In New Jersey, Maryland and Pennsylvania, many bog turtle populations are located in active dairy pasture. Pasture agriculture, as opposed to field cropping, may actually enhance bog turtle habitat by maintaining an open-canopy, wet meadow.

Bergen Swamp in Genesee County is an area of special concern. Bergen is a unique area, probably best known for its rare plants. Bog turtles were reported regularly in Bergen Swamp throughout this century until the mid-seventies. The vegetational community at Bergen is highly dynamic, probably, to a large extent, in response to changes in hydrology. One of the primary pioneering species is northern white cedar. Despite increases in canopy cover in many areas of apparently otherwise suitable habitat, many large areas of remain open, and it is difficult to speculate on the cause of the decline of the bog turtle population. This site is extensive, and hopefully turtles will be found; however, one activity that may have seriously reduced the population is overcollecting. The effects of even minimal collecting can be devastating to small populations with low reproductive potential. As an example, a spotted turtle population that I have been studying as part of a bog turtle surrogate headstart program (Collins, 1987b) is comprised of an estimated 29 individuals (Lincoln Index,  $r=25-33$ ,  $P<.05$ ). Of 27 turtles actually marked, ten are mature females; if just five of these females were removed, a tremendous loss of reproductive

potential would result. For a bog turtle population with already depressed reproduction, caused by an expanding canopy, such a loss could be irreparable. Between 1929 and 1947 seven known Bergen bog turtles went into museum collections; undoubtedly, others were removed for other purposes.

Are Western New York bog turtles relicts of ephemeral islands or simply elusive? The predominant, field crop agricultural practices of the Western Tier, combined with the more critical climatic constraints of the northern-most bog turtle population apparently has exacerbated the island effect and impacted on local populations. The species is cryptic and easily overlooked, so surveys that focus on the identification of new populations should be initiated, but, until newpopulations have been identified, this species should be managed on the assumption that it is seriously declining along its northern periphery.

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# Criteria for the Introduction of the St. Croix Ground Lizard

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**Abstract:** The St. Croix ground lizard (*Ameiva polops* Cope) is a small *Ameiva* (snout-vent length = 30-79 mm) that was once common on St. Croix, U.S. Virgin Islands and outlying islands and cays. Today it is restricted to Green Cay and Protestant Cay off the northern shore of St. Croix. The species was listed as endangered and critical habitat was declared in 1977. The establishment of additional populations is desirable for the protection of the species. Major criteria that will influence the choice of introduction sites are: 1) presence of the mongoose (*Herpestes auropunctatus* Hodgson), 2) presence of congeners, 3) availability of preferred habitats, 4) land ownership, and 5) vulnerability to storm overwash. Preferred habitats are described. Most suitable sites for introduction of the lizard belong to the National Park Service, and most of these sites are outside the natural range of *A. polops*. The Park Service is opposed to the introduction of non-native species onto Park Service lands, and this problem is addressed.

Pages 154 - 156. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The endangered St. Croix ground lizard (*Ameiva polops* Cope) is a small *Ameiva* (snout-vent length 30-79 mm) that was once common and widespread in coastal St. Croix (Cope 1862). The lizard was last seen on St. Croix in 1968 when several were observed along the Frederiksted waterfront (Philibosian and Yntema, 1976). Today the St. Croix Ground Lizard is restricted to Green Cay (5.7 ha) and Protestant Cay (1.2 ha) near the northern shore of St. Croix. The species was listed as endangered by the U.S. Fish and Wildlife Service and critical habitat (Green Cay) was declared in 1977.

The primary cause of the lizard's decline appears to be predation by the small Indian mongoose, *Herpestes auropunctatus* Hodgson (Seaman 1952, Heatwole and Torres, 1967). The mongoose was introduced in 1884 to control roof rats, *Rattus rattus* L. (Seaman and Randall, 1962).

## METHODS

St. Croix ground lizard was observed on Green Cay and Protestant Cay to obtain information on habitat preferences, populations, recruitment, food habits and presence of congeners and predators. We also examined a number of sites not occupied by St. Croix ground lizards to determine if they were feasible sites for introduction of this species. We spent 32 days on Green Cay between 25 March 1987 and 5 October 1987, and seven days on Protestant Cay between 3 April 1987 and 3 August 1987.

We examined Ruth Cay (also called Harvey Island) and Buck Island, both near St. Croix; Henley Cay, less than a km west of St. John; and Hassel Island, 100 m south of St. Thomas, as possible sites for introduction of the St. Croix Ground Lizard. We also searched for possible sites on St. Thomas, St. John, and St. Croix.

## RESULTS AND DISCUSSION

It was decided that the following factors will influence the choice of introduction sites:

1. The lizard must be protected from mongoose populations at the

- site. Absence of mongooses, removal of mongooses, or mongoose exclosures would meet this criterion.
2. The presence of congeners. *Ameiva exsul* Cope and other *Ameiva* species are found on the Puerto Rican Bank. It would be preferable to avoid placing the St. Croix ground lizard in competition with these larger members of the genus.
3. Availability of preferred habitats. The preferred habitat of *A. polops* supports stands of trees and has nearby beach habitat that receives an influx of food organisms such as *Orchestia*, annelids, and dermapterans. There must be sufficient habitat available to support a large enough population to avoid inbreeding depression. We propose that this should be a population at least as large as that now occurring on Green Cay (see below).
4. The land should be under the protection of a public agency, and the public agency must support the introduction and necessary habitat management activities.
5. The site must not be highly vulnerable to storm overwash or other natural disasters.

It would be nearly impossible to extirpate mongooses from any of the larger Virgin Islands, and it would be difficult to exclude them permanently from even a portion of St. Croix, St. John, or St. Thomas. Buck Island, Ruth Cay, Green Cay, and Protestant Cay are the only outlying islands of St. Croix. Buck Island and Ruth Cay are not currently occupied by the St. Croix ground lizard. The 71.4-ha Buck Island (McGuire, 1925) contains much preferred lizard habitat. A littoral woodland occupies much of the western end of the island, and a broad coastal plain stands about 3 m above mean sea level. The Park Service is also involved in a long-term effort to eradicate the mongoose from Buck Island where it was introduced about 1912 (Philibosian and Ruibal, 1971). Results of the eradication effort remain uncertain, though Park Service employees did not see a mongoose on the island between December 1987 and May, 1988. However, no really concerted effort was made to detect mongooses during that period (pers. comm., Zandy Hillis, U.S. Park Service, Biotechnician, Christiansted, St. Croix, U.S.V.I., 24 May 1988). Philibosian and Ruibal (1971) introduced 16 St. Croix ground lizards from Protestant Cay to Buck Island in 1968. At the time, the mongoose population seemed under control. The lizards showed initial reproductive success

and release-site tenacity, but the population declined after the Park Service discontinued predator control in 1970. No St. Croix ground lizards have been seen on Buck Island since about that time (Wiley, 1984, unpubl. report)

Ruth Cay is a 19-ha dredge-spoil island 175 m south of St. Croix. It was built in the 1960's of sand, coral and shell (Yntema and Sladen, 1987, unpubl. report). Much of the island is low and lies less than a meter above high tide. Vegetation is primarily restricted to the finer spoil deposits at elevations up to 2 m above mean sea level. Mangroves, particularly *Conocarpus erectus* L., are the dominant trees and shrubs on the island. They often attain heights of 3-4 m in dense stands. A low, grass-covered plain occupies the eastern end of the island. During our visit in August 1987, there was a small quantity of *Thalassia* and *Halodule* wrack containing *Orchestia* on the north side of the island. The "forested" areas and the beach area may provide mediocre to good St. Croix ground lizard habitat, but the better habitat is barely above high tide, and may be too vulnerable to storm overwash. A coral reef surrounds the island, and it may afford some protection from tidal surges. The higher elevations of the island, including a coral rubble ridge 10 m high, are largely barren of vegetation and would probably not sustain populations of the lizard.

We found no evidence of mongoose on Ruth Cay; however, we did observe anoles (*Anolis acutus* Hallowell) on the cay, and their presence indicates some ability for the island to support lizards. However, observations on Green Cay indicate that *Anolis acutus* has a broader niche than *Ameiva polops*. Furthermore, we observed much greater densities of *Anolis acutus* on Green Cay and Protestant Cay than were observed on Ruth Cay. *Anolis acutus* feeds on insects, which also make up a significant portion of the diet of *Ameiva polops*. Whatever is limiting the abundance of *Anolis acutus* on Ruth Cay might also limit St. Croix ground lizards if they are introduced. We predict that, with ongoing succession, the quality of *Ameiva polops* habitat on Ruth Cay will improve.

Hassel Island south of St. Thomas is about 60 ha in area. It offers large quantities of good habitat for *Ameiva polops*, but the island is inhabited by mongooses and *Ameiva exsul*; furthermore, the land belongs to the U.S. Park Service, and the Park Service is opposed to the introduction of non-indigenous species on Park Service land, even if the species is endangered, unless there is no other possible relocation site. *Ameiva polops* is not native to the Puerto Rican Bank, which includes St. John, St. Thomas, and their outlying cays.

Henley Cay is a 5-ha cay near St. John with excellent coastal forest habitat, no mongooses, and no *Ameiva exsul*. There is virtually no seagrass wrack on the shores. *Anolis cristatellus* Dumeril and Bibron is abundant. This cay is also outside the natural range of *A. polops*, however, and belongs to the Park Service.

Immediately after the 1987 reproductive period (June-September), there were about 530 St. Croix ground lizards on Green Cay and perhaps 60 on Protestant Cay. Juveniles constituted a minimum of 18% of the post-hatching population on Green Cay. Therefore, "carrying capacity" of St. Croix Ground Lizards on Green Cay is at least 435 with an annual "recruitment" of at least 21.8%.

## CONCLUSIONS AND RECOMMENDATIONS

The small population of St. Croix Ground Lizards on Protestant Cay may soon be reduced or eliminated by habitat destruction, since the owners of the hotel on the cay plan to expand the facility. The populations on Green Cay and Protestant Cay could be extirpated by the introduction of the mongoose or by a natural disaster. Therefore,

we strongly recommend the establishment of additional populations to increase the probability of survival of the species.

The St. Croix ground lizard is vulnerable to the mongoose. If the mongoose gains access to Green Cay and Protestant Cay, *A. polops* is almost surely doomed. With regard to our recommendation that immediate steps be taken to introduce *A. polops* to additional lands, the habitat on Ruth Cay is less than ideal, but a release at that site stands a fair chance of providing a third population. Much superior habitat also exists at Buck Island National Monument, and, due to greater elevations on Buck Island, populations of *A. polops* would certainly be more secure from natural disasters on this island than they would be on Ruth Cay. As soon as Buck Island is cleared of the mongoose, we strongly recommend a release of *A. polops* on the island. The best months for such a transfer are November and December, well after the breeding season, so that the young will have had time to grow in size and resilience.

We feel that a minimum of 50-95 St. Croix Ground Lizards could be removed annually from Green Cay for restocking purposes. Twenty animals would provide a sufficient number to insure statistically that not all individuals are of the same sex. The safest means of trapping *A. polops* is to install drift fences attached to tunnel traps in the beach forest. Animals should be transported to release sites in cool, well ventilated containers as soon as possible after capture. Members of a group should also be released together to facilitate their finding one another. Releases should occur early in the day to permit the lizards to become acquainted with new habitats before they seek burrows in mid-afternoon to facilitate thermoregulation.

Protestant Cay does not currently carry a sufficient number of *A. polops* for the population to remain viable, nor can it be expected to do so. However, so long as a population survives on the island, it can serve as a reserve population and as a source of additional genetic diversity. We recommend that the population on Protestant Cay not be removed in its entirety (as has been suggested by some authorities) to start a new population elsewhere unless a clear and eminent danger arises that would insure their immediate extirpation on Protestant Cay. We do recommend that at least six individuals from Protestant Cay be added to each introduction to increase genetic heterozygosity in the new population.

## ACKNOWLEDGMENTS

Funding was provided by the Caribbean Islands National Wildlife refuges via the Louisiana Cooperative Fish and Wildlife Research Unit, Louisiana State University with the Louisiana Department of Wildlife and Fisheries, U.S. Fish and Wildlife Service, and the Wildlife Management Institute cooperating. We thank the U.S. Virgin Islands Division of Fish and Wildlife, the U.S. Park Service, and the U.S. Fish and Wildlife Service. Without their assistance this work could not have been conducted. We are especially indebted to Jaime Collazo, Marcia Taylor, Caroline Rogers, Joel Tutein, Marilyn Parris, and Zandy Hillis. This paper was approved for publication by the Director of the Louisiana Agricultural Experiment Station as manuscript number 88-22-2353.

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## **Chapter 5.**

### **SPECIES BIOLOGY: RARE PLANTS**



# The Management of Rare Plants: Suggestions Derived From Paleocological Studies of Late-Pleistocene Floras<sup>1</sup>

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**Abstract:** At the height of late Wisconsinan glaciation the now heavily populated New York-New England region lay deeply buried by ice, which began to recede at ca. 18,000 yr B.P. The present flora and vegetation of this region was derived through immigration followed by changes in population sizes under the effect of climatic change, soil development, competition and other ecological processes. Between 15,000 and 12,500 to 10,500 yr B.P. the late-Pleistocene vegetation consisted largely of herbs, shrubs, and bryophytes that now have geographic affinities with arctic, boreal, or northern temperate floras. The fossil record of these plants offers insights on the management of existing botanical resources. The species represented by fossils are pioneers in contemporary habitats subject to recurring disturbances and/or lacking tree cover. Plants of calcareous, mineral soils or base-rich wetlands dominate the fossil floras. These facts indicate that climate was not the sole factor controlling species occurrence, but rather that the frequency of disturbance, absence of competition from trees, and soil and groundwater chemistry were important factors allowing plant establishment and persistence. Some species represented by late-Pleistocene fossils are great rarities in the contemporary flora. The survival of these species in a region climatically suited to forests is dependent upon managing habitats to reduce competition, as well as to permit periodic natural or artificial disturbances. Examples of rare plants occurring as late-Pleistocene fossils are: *Harrimanella hypnoides*, *Elaeagnus commutata*, *Saxifraga aizoides*, *S. paniculata*, and species of *Dryas* and *Gentianopsis*.

Pages 159 - 162. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Studies of late-Pleistocene plant fossils in the New York-New England region have revealed a rich flora of bryophytes, herbs, and shrubs that grew on deglaciated terrain before the arrival and spread of forest trees. Although much is still to be learned about this ancient flora, what its constituent species indicate about late-Pleistocene environments is pertinent to questions of species rarity in the contemporary flora of this large and botanically heterogeneous region.

Rare and uncommon native plant species occur in several distinct habitat-types in New York-New England. I am concerned here with rare plants that grow at open sites and other locations suited to early successional (pioneer) species. Except for limited areas of alpine vegetation, naturally occurring habitats of this sort were of restricted distribution in both the pre- and early post-settlement vegetation of the region, which, based on analyses of the original land surveys and other historical records (e.g., Gordon, 1940; Shanks, 1966), was largely deciduous or mixed deciduous-conifer forest. Documented occurrences of late-Pleistocene fossils of early successional species at places where they do not now occur, but in ecological settings inferred to be similar to those of extant populations, suggest that certain rare elements in the flora of New York-New England may be relicts of more common late-Pleistocene populations. Successful management of plant resources of this type depends on knowing which elements of the contemporary flora have a late-Pleistocene history and selecting for preservation certain appropriate habitats in which the requirements of early successional species can be met, either by natural or artificial means.

## SOME PALEOECOLOGICAL BACKGROUND

The late-Quaternary vegetational history of the northeastern United States has been summarized by Davis (1983) and Watts (1983), and Davis and Jacobson (1985) have interpreted the paleobotanical record

of northern New England for the period 14,000-9000 yr B.P. These summaries are based on a synthesis of numerous pollen diagrams and radiocarbon dates obtained through the study of superimposed layers of lake and peatland sediment, supplemented by investigations of macrofossils (fruits, seeds, leaves, and other plant fragments).

Across much of New York-New England, following deglaciation, an initial treeless vegetation of bryophytes, herbs, and shrubs was replaced by spruce-jack pine communities, which during the late-Pleistocene-Holocene transition gave way to pines and associated species of deciduous trees. Later Holocene forests were dominated by deciduous trees, hemlock and sometimes other conifers, as species migrated to and spread across the region, then changed in abundance. While many details remain to be resolved, it is evident that the immigration of forest species took place individualistically and in time-transgressive patterns.

Knowledge of postglacial vegetational changes is based mostly on studies of fossil pollen from wind-pollinated trees and other plants. As a result, information is available about the Quaternary history of only a small percentage of the flora, and indeed for few species that would today be considered rare. We have, for example, scant knowledge of the past distributions of insect-pollinated herbs and shrubs, the pollen of which rarely or never reaches a suitable environment for preservation.

The vagaries of pollen dispersal and preservation, as these affect the reconstruction of floras, are partly overcome by studies of plant macrofossils, which in general are from plants that grew near the study site. Well-preserved plant macrofossils can usually be identified to species, sometimes through the application of special techniques such as scanning electron microscopy or anatomical sections. Late-Pleistocene plant macrofossil assemblages include parts of terrestrial and aquatic plants, the former washed or blown in from surfaces above a depositional basin, the latter from plants growing in the basin. The paleofloristic data I summarize here are based mostly on identifications of plant macrofossils, although these data in no way contradict conclusions drawn from studies of pollen occurring dispersed in lake and peatland sediments.

<sup>1</sup> Published as contribution number 594 of the New York State Science Service.

Assemblages of late-Pleistocene plant macrofossils, while differing greatly in species composition from the contemporary flora of the area near the site of recovery, are not known to contain fossils of extinct species, insofar as this can be judged on the basis of morphology. It is justifiable, therefore, to infer the physical and biological character of past environments, under the assumption that late-Pleistocene and contemporary populations of the same species are likely to share a similar ecology. The indicator value of macrofossil assemblages is enhanced when identifications are made of both tracheophyte and bryophyte fossils that often occur intermingled in same deposit.

### THE LATE-PLEISTOCENE FLORA AND ITS PALEOECOLOGY

Studies of late-Pleistocene plant fossils have been undertaken at sites from northern and eastern Maine to western New York State (Anderson et al., 1986; Jackson, 1989; Miller, 1973, 1987; Miller & Thompson, 1979; Tolonen & Tolonen, 1984), and I have recently summarized what is known about the late-Pleistocene floras of New England, emphasizing biogeographic relationships and paleoecology (Miller, 1989). Full details about the composition of the paleofloras are found in these papers.

Rare, threatened, or endangered species in the contemporary floras of New York and New England (Mitchell & Sheviak, 1981; Crow et al., 1981) that are represented in the fossil floras include *Selaginella selaginoides* (L.) Link, *Potamogeton alpinus* Balbis, *Salix argyrocarpa* Anderss., *S. herbacea* L., *S. uva-ursi* Pursh, *Betula glandulosa*, Michx., *Oxyria digyna* (L.) Hill, *Silene acaulis* L. (Figure 1), *Ranunculus cymbalaria* Pursh, *Saxifraga aizoides* L. (Figure 2), *S. paniculata* Mill. (Figure 3), *Sibbaldia procumbens* L., *Rubus chamaemorus* L., *Shepherdia canadensis* (L.) Nutt., *Rhododendron lapponicum* (L.) Wahlenb. (Figure 4), *Harrimanella hypnoides* (L.) Coville, and *Gentianopsis* sp. In addition, the following occur as fossils in New York-New England but are now native north of this region: *Carex bipartita* Bellardi, *Salix vestita* Pursh, *Arenaria* cf. *dawsonensis* Britt., *Saxifraga* cf. *flagellaris* Willd., *Parnassia* cf. *kotzebuei* Cham., *Dryas drummondii* Richards. (Figure 5), *D. integrifolia* Vahl, and *Elaeagnus commutata* Bernh. ex Rydb. (Figure 6). How frequent and abundant these species were in the northeastern United States during the late Pleistocene can not be established with the data now available.

Species with late-Pleistocene fossil records are distributed in a number of different patterns in the contemporary flora. Most are alpine or subalpine plants (e.g., *Salix herbacea*, *Betula glandulosa*, *Oxyria digyna*, *Rhododendron lapponicum*), the present high-elevation stations of



**Fig. 1.** *Silene acaulis* L., Mont Albert tableland, Gaspé Peninsula, Quebec, Canada (15 June 1979); **Fig. 2.** *Saxifraga aizoides* L., Rhone Glacier Vorland, Bernese Oberland, Switzerland (25 August 1979); **Fig. 3.** *S. paniculata* Mill., Mont St. Pierre, Gaspé Peninsula, Quebec (15 June 1979); **Fig. 4.** *Rhododendron lapponicum* (L.) Wahlenb., near summit of Mt. Marcy Essex County, New York, U.S.A. (10 June 1985); **Fig. 5.** *Dryas drummondii* Richards., Petit Cascapédia River, Gaspé Peninsula, Quebec (18 June 1977); **Fig. 6.** *Elaeagnus commutata* Bernh. ex Rydb., Mont St. Pierre, Gaspé Peninsula (15 June 1979). (Photographs by the author, from color transparencies.)

which contrast with the low elevation and lower latitude occurrences of the late-Pleistocene fossils. Other species (*Selaginella selaginoides*, *Ranunculus cymbalaria*, *Shepherdia canadensis*) are northern (boreal, subarctic), lowland plants with specific edaphic requirements. While currently rare in New York-New England region, all the species in this category are at the southern limit of distributional areas that extend northward and across North America and beyond, in some cases to include parts of northern Europe and Asia.

From the large list of plant species known to have been present in the late Pleistocene of New York-New England, one can confidently infer considerable detail about habitats that were present during the several millennia immediately following deglaciation. Standing or flowing water (ponds, lakes, streams) was circumneutral or slightly alkaline and had high concentrations of cations. Terrestrial habitats (mineral soil, seeps, flushes) were also base-rich, and they probably remained so until weathering or changes in hydrology removed the sources of cations. Terrestrial habitats were suited edaphically to early successional species because of frequent site-renewal events, involving, for example, deflation, mass-wastage, and other types of erosion. Another factor was the absence or sparse occurrence of trees, with only spruce and pine present, based on macrofossil evidence. These conditions help to account for the preponderance of light-demanding early successional calcicoles in the late-Pleistocene floras. Calcifuge plant species are poorly represented in the paleofloras, but the few that are present suggest that humus accumulation, leaching, or other pedogenic processes leading to soil acidification were also occurring.

Mixtures of sites for calcicoles and for calcifuges (with species in the former category dominating all known fossil assemblages) is the pattern, regardless of whether the fossils come from an area of bedrock or glacial till that is calcareous or noncalcareous. Moreover, the pattern is evident in independent analyses of bryophyte (mostly moss) assemblages, which indicate the existence of xeric, mesic and hygric, mostly calcareous habitats, possibly occupying micro-scale topographic/moisture gradients.

The fate of this late-Pleistocene, heliophytic, pioneer flora is under active study. At Tom Swamp, Massachusetts (Miller, 1989), and Upper South Branch Pond, northern Maine (Anderson et al., 1986), for which both radiocarbon-dated pollen and plant macrofossil stratigraphies are available, the transition from herb/shrub-dominated to tree-dominated assemblages is abrupt, suggesting that the change was an outcome of increases in the sizes of populations of spruce. The transition took place 2,300 radiocarbon years later in Maine, ca. 500 km northeast of Tom Swamp, which may mean that process is more significant in explaining the change than is the timing. Both depositional basins ceased receiving fragments of plants of open habitats during a period of transition to a more closed vegetation type. For some plant species, such as *Elaeagnus commutata*, that are not now members of the New York-New England flora, extirpation may have taken place as forest communities developed across the landscape. For others that are now rarities in the contemporary flora, it seems possible that they may have persisted at places such as gorge walls and alpine habitats that were immune to competition from trees.

## DISCUSSION

That certain contemporary plant rarities in the northeastern United States have a late-Pleistocene fossil record is significant, because of ecological and historical implications of this discovery. The ecological message is a simple one: habitats of those rare species in the contemporary landscape must be comparable to habitats that were widespread

during the late Pleistocene. The similarities may be explained by the existence of equivalent ecological factors at both times, involving, for example, reduced competition from trees or their absence, mechanisms that maintained raw mineral soil for repeated establishment, and other edaphic factors such as a consistent source of calcareous parent material.

Contemporary stations of the species discussed in this paper are few because the requisite habitats are rare and scattered in the northeastern United States, a region now dominated by forest vegetation. There is evidence that alpine vegetation existed throughout the Holocene in the Adirondack Mountains (Jackson, 1989), and presumably in the high mountains of New England as well, thereby providing a plausible basis for populations of alpine plants, including those of species discussed here, to persist through the Holocene in areas where they now grow. We can be less certain that the few lowland stations for plant rarities such as *Saxifraga aizoides* in New York and Vermont represent a similar case of persistence. The New York stations of this species are of special interest because they are the vertical walls of deep gorges (House, 1924) that were cut during the Holocene by stream or river erosion. Such habitats may have served as continuous Holocene refugia for certain early successional species. However, this possibility requires evaluation by paleobotanical or other criteria.

The paleobotanical, floristic, and ecological data from the northeastern United States are strikingly similar to data presented by Pigott and Walters (1954; see also Godwin, 1975: 452-455) in an analysis of the same phenomenon in Britain. Many of the floristically richest areas in England, Scotland and Wales have calcareous soils or rock (sometimes base-rich drainage water) and lack woodland. Further, late-Pleistocene fossil occurrences are known for many of the plant rarities of such areas and from numerous places where the species do not presently occur. The Pigott and Walters hypothesis proposes that these species were of wide and more or less continuous distribution in the southern lowlands of Britain during the late Pleistocene but were eliminated nearly everywhere as forest, peatland, and soil development proceeded. In Britain, some of the species have spread recently in response to the return of ecologically favorable conditions brought about by agricultural and other anthropogenic activities, but this does not seem to be true in the northeastern United States, except for the occasional establishment of alpine plants at low elevation streamside habitats in some mountain areas (see, for example, plant distributional data presented in Scoggan, 1950).

The Pigott and Walters hypothesis pertains not only to tracheophytes in Britain and northeastern United States but also to the bryophyte floras of the two regions. Dickson (1973) presents a detailed account of the loss, or reduction in area, of rare alpine and calcicole elements during the late Quaternary in Britain, where habitats for base-rich fen species have been severely reduced by two additional factors, peatland drainage and peat cutting. In the contemporary bryoflora of northeastern North America many examples are known of rare alpine species and calcicoles that, from the evidence of late-Pleistocene fossils, were widespread beyond the few stations at which they now occur (Miller, 1984).

These observations may prove useful to managers of areas containing rare plants, especially calcicoles and other species of early successional habitats. Maintaining or encouraging, by either natural or artificial means, processes that lead to the periodic renewal of mineral soil may be necessary for the survival of established populations of some rare plants. Equally important, especially at lowland stations, is the exclusion of trees and shrubs. At some places, such as stream and river bluffs, weathering and erosion should insure the continuation of natu-

rally occurring open habitats for early successional species, but at low-land sites that are more stable, it may be necessary to restrict tree and shrub establishment artificially and perhaps also to encourage recruitment from established populations of the rare plants by whatever means are practical.

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# The Biology and Management of *Potentilla robbinsiana*, an Endemic from New Hampshire's White Mountains

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**Abstract:** Dwarf cinquefoil (*Potentilla robbinsiana* Oakes) is a small perennial plant endemic to the alpine zone of New Hampshire's White Mountains. At the turn of the 20th century, botanical collecting became a serious threat to the survival of *P. robbinsiana*, and some small colonies were eliminated by heavy collecting. Listed as federally endangered in 1980, the species now has only two known natural populations. One population consists of approximately 1800 adult plants, and the other of only three adults. Since 1984, the species has been the subject of an intensive demographic and ecological study. The information gained is being used to manage the natural populations and establish new colonies in suitable habitats. The main population is found in a 1 ha barren area on Monroe Flats (elevation 1500 m), approximately 2 km south of Mt. Washington. Ongoing demographic studies indicate that the population is stable. The plant appears adapted to relatively barren alpine habitats that are swept clear of winter snow. Such habitats are relatively uncommon in the White Mountains. Disturbance is the principal current threat to the species' survival, whether it be from hiker traffic, frost heaving, or wind and water erosion. Research into the breeding biology of *P. robbinsiana* suggests that the plant is apomictic and pseudogamous. Isozyme studies have indicated little or no genetic diversity in the two remaining populations. Management of the species has several components. The principle population is located adjacent to the Appalachian Trail, and the area was once subjected to heavy hiker traffic. A minor trail relocation in 1983 and ongoing hiker education efforts have nearly eliminated this threat. However, because the species grows in a limited area, it is still very vulnerable to catastrophic disturbances. To improve the species' chances of survival, plants are being grown in a greenhouse from seed collected at Monroe Flats and transplanted in suitable habitats. Early results are promising, but due to the slow growth and maturation of *P. robbinsiana*, it will be several years before we can determine if the transplant populations will be self-sustaining.

Pages 163 - 166. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The dwarf or Robbins' cinquefoil (*Potentilla robbinsiana* Oakes) is a rare endemic found only in the alpine zone of New Hampshire's White Mountains. It is a small plant that forms rosettes with up to 50 showy, yellow flowers. This plant's historical interest to collectors, its occurrence in a very harsh environment in close proximity to heavily used recreational facilities, and its unusual breeding biology make for a very interesting story. Highlights are covered in this paper.

## HISTORICAL DISTRIBUTION

*Potentilla robbinsiana* was discovered in 1825 by James Robbins, approximately 2 km south of the summit of Mt. Washington, near the base of Mt. Monroe (Graustein, 1964). The species was described later by William Oakes, one of a group which crossed Monroe Flats in July 1825. Oakes named the plant for James Robbins, who was working for him in the region in 1829.

The Monroe Flats population was the best known location for the cinquefoil, and there was extensive collecting at that site over the years. A search of 29 herbaria turned up 421 specimens of *P. robbinsiana* that had been collected from this population (Cogbill, 1984). The majority of these plants were collected between 1870 and 1910, sometimes in large quantities that were offered for sale.

In 1915 the Appalachian Mountain Club constructed Lakes of the Clouds Hut approximately 150 m north of Monroe Flats. This area was already traversed by Abel Crawford's path to the summit of Mt. Washington, which was constructed in 1819 (Burt, 1960). Shortly after the hut was completed, the Crawford Path was relocated through another part of the *P. robbinsiana* population so that it would pass by the new facility. By the 1960's another hiking trail had been construct-

ed along one edge of the population, further increasing human traffic in the area.

After the relocation of the Crawford Path over the dome of Monroe Flats in 1915, the main population of *P. robbinsiana* was located to the east of the trail. There was also a small subpopulation, of perhaps 12 to 40 plants (Cogbill, 1984), located to the west of the trail. This population has been extirpated, accounting for the reduction of area occupied by *P. robbinsiana* on Monroe Flats (Graber, 1980).

There have been at least three stations for *P. robbinsiana* on Franconia Ridge. In 1897 Endicott, and in 1915 Fernald collected plants at different locations on the ridge (Steele, 1964). In 1963 Fred Steele and Harry McDade relocated what they felt was Fernald's station (Steele, 1964). As it turns out, this was probably a new colony that is still extant (Cogbill, 1984). In 1988 this colony consisted of only 3 flowering adults and 10 to 15 juveniles and seedlings.

There have been a few other reports of *P. robbinsiana* in other locations that have turned out to be erroneous or are impossible to document. According to herbarium labels, Edward Tuckerman collected specimens from various locations in the White Mountains (Cogbill, 1984). There is also a specimen that is annotated "Mansfield?", perhaps referring to Mt. Mansfield in Vermont, that has been widely referenced in the literature. Research into Tuckerman's original field notes indicates that all these specimens were probably collected from Monroe Flats. There was also a report of *P. robbinsiana* on Mt. Equinox in southern Vermont (Countryman, 1980), but the plants turned out to be *P. norvegica* L. (Graber and Brewer, 1985).

## ECOLOGY

The principal habitat of *P. robbinsiana* is in a well-developed system of turf-banked terraces at Monroe Flats, the largest well-developed

**Table 1** Phenology of Monroe Flats *P. robbinsiana* population. Values are date of first observation.

Year	Dormancy Release	First Flowering	First Seeds Released	First Seedlings
1984	ca. 1 Jun	11 Jun	ca. 20 Jul	3 Jul
1985	ca. 28 May	2 Jun	20 Jul	17 Jun
1986	14 May	ca. 21 May	negligible <sup>1</sup>	22 Jun
1987	ca. 22 May	28 May	ca. 9 Jul	ca. 10 Jul

<sup>1</sup>Heavy frost in June caused seeds to abort in most plants.

terrace system in the White Mountains. This system consists of barren, stony terraces joined by vegetated risers. The soils and topoclimate of this area are fairly typical of those found in the vicinity of Mt. Washington, and are apparently not important in controlling the species' occurrence. (Cogbill, 1987).

Disturbance, however, does appear to be a factor in controlling *P. robbinsiana*'s distribution. By virtue of a deep tap root, the species appears to be adapted to a moderate level of frost heaving that limits other species. At the same time, it cannot tolerate frost-induced movement of more than 18 mm yr<sup>-1</sup>, or frost action sufficient to produce stone stripes or other patterned ground (Cogbill, 1987). It also appears that *P. robbinsiana* does not compete successfully with other species in undisturbed sites.

Frequent high winds and lack of winter snow cover are characteristic of all past and present sites known to support *P. robbinsiana*. In addition to high winds, the species is also known to be tolerant of needle ice action (Cogbill, 1987). *P. robbinsiana* appears to be well adapted to such conditions, as are several other alpine plants, including *Diapensia lapponica* L. and *Minuartia groenlandica* (Retz.) Ostenf.

The weather regime experienced by *P. robbinsiana* is highly variable from year to year. In 1986, an unusually warm spring caused the plants to break dormancy and flower in mid-May (Table 1). A freeze in early June then aborted most seed production for that year. In 1987, heavy rains in early April caused extensive sheet erosion that was particularly damaging to juvenile plants that lacked well-developed root systems, but affected larger plants as well. A severe drought persisted through much of the summer of 1988, but its effect on seedling germination and mortality has yet to be determined.

## REPRODUCTIVE BIOLOGY

Adult *P. robbinsiana* plants growing at Monroe Flats produced an average of 5.9 flowers each in 1988, and have been observed to produce as many as 52 flowers on a large plant (Kimball and Paul, 1986). In favorable years, ovaries may mature into single-seed fruits called achenes (Lee, 1984); Graber (1980) determined that field-produced achenes are viable.

Emasculating experiments performed since 1984 indicate that self-pollination is prevalent in the species, and that pollination is required for achene production to occur (Lee, 1986). However, observations of ovule and megagametophyte development in *P. robbinsiana* support the hypothesis that the species is also apomictic, producing seeds via agamospermy (Lee and Greene, 1986). Isozyme analysis indicates that there is very little genetic diversity in the Monroe Flats population (D. O'Malley, unpublished).

Research to date indicates there is little or no genetic diversity or gene flow in the species. Species that exhibit such characteristics often

compensate with considerable phenotypic plasticity, which may indeed be the case with *P. robbinsiana*. If operative, plasticity would be advantageous to the species, since it lives in an environment subject to extreme year-to-year weather variations.

## Demographics:

In 1984, 94 permanent, 1 m<sup>2</sup> quadrats were established in the Monroe Flats area. Each quadrat was carefully mapped, and all *P. robbinsiana* plants were located and placed in one of three life-stage categories. Seedlings were plants judged to have germinated in that season; juveniles were non-flowering plants at least 1 year old, and adults were plants that had flowered at least once. The habitat, vegetated riser or bare terrace associated with each *P. robbinsiana* individual was also noted. The quadrats have been re inventoried twice a year since 1984. In addition to noting the presence or absence of individual plants and their life stage, flowers have been counted on adults during the spring survey, and rosette diameters of all plants have been measured in the fall surveys.

## Phenology:

The phenology of *P. robbinsiana* is typical of many alpine species, with dormancy release and flowering occurring early in the season (Table 1). Seed germination also occurs relatively early, allowing the young plants to become established before the onset of winter. Annual growth typically ends with the onset of heavy frost in late August or early September.

## Survivorship:

Seedling survivorship studies suggest that survival is higher during the first summer in the pebble terrace than in the vegetation mats, but that the reverse is true during the first winter.

Survivorship of juveniles and adults can be estimated by following the original cohort of juveniles and adults recorded during the first survey in June 1984 (Table 2). Annual juvenile survivorship appears to be lower in the pebble terraces, probably because the pebble terraces are subjected to greater frost heaving action than the vegetated habitats. This difference in survival between habitats is less obvious for the adults, which are anchored by larger root systems. Survivorship is somewhat lower during the winter than the summer, particularly for juveniles (Table 2). Desiccation from strong winds in the alpine habitat has been proposed as an important cause of summer mortality for *P. robbinsiana*, but it is highly unlikely as the Monroe Flats population is immersed in clouds over 40% of the time. Measurements of soil moisture also indicate that the soil moisture content in the essential habitat is high throughout the summer (C.V. Cogbill, unpublished). Physical uprooting of the plants by frequent frost heaving during the autumn and spring appears to be the principal cause of mortality. Since 1985,

**Table 2.** Survival for 1986 and 1987 of plants originally measured during 1984 survey. Values are live/total, per cent survival in parentheses.

Season	Stage	Pebble Terrace	Vegetation Mat	Total
Winter	Juvenile	85/108 (79)	92/100 (92)	177/208 (85)
Aug 86	Adult	70/74 (95)	60/63 (95)	130/137 (95)
to				
Jun 87	Total	155/182 (85)	152/163 (93)	307/345 (89)
Summer	Juvenile	83/85 (98)	91/92 (99)	174/177 (98)
Jun 87	Adult	69/70 (99)	59/60 (98)	128/130 (98)
to				
Aug 87	Total	152/155 (98)	150/152 (99)	302/307 (98)
Annual	Juvenile	83/108 (77)	91/100 (91)	174/208 (84)
Aug 86	Adult	69/74 (93)	59/63 (94)	128/137 (93)
to				
Aug 87	Total	152/182 (84)	150/163 (92)	302/345 (88)

the annual survival rate for the combined juvenile/adult population has ranged from 85 to 91%. Of those plants originally surveyed in 1984, 16 adult plants died between August 1986 and June 1988. During this same period, 33 juvenile plants of this same group flowered for the first time, 13 in 1987 and 30 in 1988. This resulted in a net increase of 17 adult plants. Based on these limited data, the adult population appears to be stable.

#### Growth Rate:

An indirect estimate of the time it takes a seed to mature into a flowering adult can be made from the data. Of the 73 seedlings measured in the quadrats during 1984, 18 (24%) were surviving in 1987. Of these, only 11 were of measurable size in 1987 (4 growing seasons after germination) and they were  $15 \pm 6$  mm in diameter. The 1984 seedling attaining the largest size by August 1987 was 25 mm in diameter.

Plants in the original 1984 cohort that flowered for the first time during 1986 or 1987 had rosette sizes ranging from 8 to 51 mm in diameter ( $29 \pm 11$  mm). Theoretically, a seed germinating in 1984 could reach the minimum flowering size of 8 mm by 1987, a 3 year time span. However, no 1984 seedling flowered in 1987.

We calculated an average rosette growth rate of 2.2 mm/yr between 1984 and 1987 for juveniles or adults that first flowered in 1987. At this rate, on average, it would take a seedling 13 years to obtain the average flowering size of 29 mm. This contrasts with seeds grown in the greenhouse, which can grow to 30 to 50 mm in diameter after five months (Graber, 1980, T.D. Lee, unpublished).

#### Seedling Dormancy:

During 1986, an unusually warm spring caused *P. robbinsiana* to flower about one month early. There were two severe frosts with strong winds and icing during June of that year as well. As a result, much of the seed crop for *P. robbinsiana* (possibly 90% or greater) was aborted. We had hypothesized that the majority of *P. robbinsiana* seeds germinate the year following seed set. To test this hypothesis, we compared 1986 with 1987 seedling emergence in seven 1 m<sup>2</sup> seedling study plots. Since few if any seeds were produced in 1986, this hypothesis would predict negligible seedling emergence in 1987. In 1986, a total of 97 seedlings entered the population, following a typical flowering year in 1985. In contrast, a total of 14 seedlings entered the popula-

tion in these same study plots during 1987. Successful germination in 1987 was therefore 14% of that measured in 1986 for these seven study plots.

These data do not support the hypothesis that all seeds overwinter and germinate the following year. An extreme 1987 spring runoff event had the potential to wash out the few viable seeds overwintering from the 1986 seed crop. The presence of 14 seedlings in our seven seedling study plots during 1987 is relatively high, considering the poor seed production of the previous year. In addition, four adult plants died in the seven plots between August 1985 and June 1986, reducing (from 32 to 28) the number of plants capable of contributing to the 1986 seed crop. We interpret these data to suggest that a reservoir of seeds from 1985 or an earlier seed crop contributed to the presence of 1987 seedlings. It is important to note that seeds of this species germinated in the laboratory have exhibited variable germination times, even when stimulated with auxin (T.D. Lee, unpublished).

## MANAGEMENT

#### Monroe Flats Population:

Management factors concerning *P. robbinsiana* on Monroe Flats consist of several components. First, the population is located in the White Mountain National Forest, which is administered by the U.S. Forest Service. In 1983 the Crawford Path, which is also the Appalachian Trail in this area, was relocated out of the plant's habitat. The essential habitat was then closed to the public. The area was marked by a low rock wall constructed to completely surround the essential habitat with posting signs requesting hikers to stay outside the wall. The compliance rate by hikers has been estimated at more than 96% since the protection program began. In addition, the Appalachian Mountain Club initiated an education program at Lakes of the Clouds Hut, consisting of educational posters and interpretive talks for hut guests (Weathers, 1983).

#### TRANSPLANT PROGRAM

Lanier and Hill (1983) called for the establishment of four new *P. robbinsiana* populations. We have collected seeds from the Monroe

Flats population, which were then germinated in a greenhouse and later kept cold over the winter.

Our transplant efforts have now focused on Franconia Ridge and Boott Spur. In 1987 we resurveyed Franconia Ridge for potential transplant sites. Because of the limited area and steep, cliff-like nature of the existing population's habitat there, we did not initiate transplant efforts to try to expand this population. There is another site on the ridge that the species may have historically occupied, where we transplanted 45 plants in June of 1988. This site will probably support a small population of 50 - 150 plants.

In 1986, 70 plants were planted at two locations on Boott Spur. No additional plants were put out in 1987, because of failure of the seed crop following the June 1986 frost. Site 1 had only an 18% (7/40) survival rate after one year, so we have abandoned further efforts at that site. The site has well-formed terraces, but the configuration of adjacent topography subjects the terraces to heavy runoff. Site 2 had a 50% (15/30) survival rate after one year. Considering the seedling-like condition of the plants used, this survival rate is similar to the 68% survival of overwintering seedlings in the Monroe Flats population. In 1988 we continued our effort at this site by transplanting 80 plants in June. The potential habitat at Site 2 is quite extensive, and we plan to continue our efforts at there.

### CONCLUSIONS

Although the Monroe Flats *Potentilla robbinsiana* population is confined to a limited area subject to severe weather and human impact, management efforts to date appear to have stabilized the population. However, to make the species less vulnerable to extinction it is necessary to establish viable populations in other areas. This is a difficult task, due to the specialized habitat requirements of the species. Results of the transplant effort to date are promising, but it will be several years before we can declare that successful colonies have been established. Only after offspring of the transplanted individuals have matured and produced additional generations of offspring will we know that *Potentilla robbinsiana* has been established successfully outside of Monroe Flats.

### ACKNOWLEDGMENTS

Financial support for this project has been provided by the U.S. Fish and Wildlife Service through contract 50181-00446-6 to the Appalachian Mountain Club, and through a grant from the Rowland Foundation, Inc. Our thanks to Rose Paul, MaryBeth Keifer, and Joseph Doucette for their dedication to this project, and their capable assistance with field work and data analysis. We also thank Dick Dyer of the U.S. Fish and Wildlife Service for his assistance and concern for

endangered plants since the beginning of this study. This paper has not been submitted to U.S. Fish and Wildlife Service for review, so its contents do not necessarily reflect the views of that agency.

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# Genetic Diversity and Population Structure of a Rare Plant, Northern Monkshood (*Aconitum noveboracense*)

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**Abstract:** Genetic variation at 38 sites of Northern Monkshood (*Aconitum noveboracense* Gray) was examined by starch-gel electrophoresis. Two of 19 loci were found to be polymorphic. Average heterozygosity was 3.1%, averaged over 19 loci, but there was no consistent relationship between heterozygosity and estimated adult population size. Sites differed considerably in allele frequencies; 53% of the total genetic variation was associated with between-site differences ( $F_{st}$ ). Significant genetic differences existed among river drainages within a state and between New York populations and western populations in Iowa and Wisconsin. There were no statistically significant differences among sites within a drainage or between western limestone and sandstone populations. Northern Monkshood's long adult lifespan, clonal propagation, and dispersal within drainages probably reduced potential inbreeding effects due to small population sizes. Conservation of genetic variation in this species should emphasize protecting at least one site in many drainages rather than many sites in one drainage. Weaker evidence suggested that dispersal among multiple sites in a drainage should be protected to maintain heterozygosity.

Pages 167 - 175. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Northern Monkshood (*Aconitum noveboracense* Gray), is a relatively rare, herbaceous perennial found in cool, damp microsites in Iowa, New York, Ohio and Wisconsin (Read and Hale, 1983). It has a wide geographic range but is restricted to specific microhabitats and often has small population sizes, from 5 to 500 plants in a site (Read and Hale, 1983). In 1978 the species was listed as Threatened under the U.S. Endangered Species Act because of potential inundation of large populations along the Kickapoo River in Wisconsin. Since 1978, a recovery plan has been written (Read and Hale 1983), many new monkshood sites have been discovered, especially in Iowa, experimental transplants have been used to start new populations, and northern monkshood life history has been studied in more detail (Dixon and Cook, 1986; Dixon and Cook, 1989; Kuchenreuther et al., 1986; Kuchenreuther, 1988). However, we have little understanding of the genetic structure of monkshood populations, especially the genetic relatedness of populations across a wide geographic range and the consequences of its small population size.

Northern Monkshood is one of five native species of *Aconitum* in the United States: *A. delphinifolium* in Alaska, *A. columbianum* throughout the Western U.S., *A. uncinatum* and *A. reclinatum* in Southeastern mountains, and *A. noveboracense* in the Northeast and upper Midwest. However, monkshood taxonomy is somewhat confused and the specific validity of *A. noveboracense* is uncertain (Brink, 1982). Northern Monkshood has been variously considered a unique species, a subspecies of *A. columbianum* (Brink, 1982) or a subspecies of *A. uncinatum* (Hardin, 1964), based on similarity of leaf, flower, and root morphology.

Although northern monkshood sites are usually cool, damp and semi-shaded, details of the habitat vary across its range. In Iowa, monkshood is found on algalic talus slopes (Hallberg et al. 1984). These slopes are formed when limestone dissolves, leaving numerous air vents, often leading back to an ice cave. Although some Iowa

monkshood slopes are in the full sun, cold air flowing through the vents maintains cool subsoil temperatures. The largest known Northern Monkshood populations are found in Iowa (with one population estimated at 10,000 individuals). In Wisconsin, one group of sites at Chase Creek is found on limestone, similar to the Iowa populations, but most sites are partially shaded horizontal ledges in sandstone cliffs. Population sizes are smaller than in Iowa, ranging from 65 to a thousand individuals. Two small populations (ca. 100 individuals) are found on sandstone ledges in Ohio, but very little is known about them, and they were not included in this study.

In New York, monkshoods are found in three separate habitats in the Catskill Mountains. The two largest populations in the State are at a small sandstone cliff on Dry Brook Ridge (ca. 1000 individuals) and in Peekamoose gorge, a sandstone gorge kept cool by an underground flow of cold air and water (ca. 500 individuals). The majority of New York sites are small populations (each <10 - 100 individuals) scattered along three streams, the Dry Brook, Beaverkill and Neversink creeks, which all have their headwaters on the slopes of Double Top Mountain. In these sites, plants are usually found in gravel beds and cobble point-bars along stream banks or on moss-covered rocks in the headwater creeks.

In both Iowa and Wisconsin, monkshoods are found only in the driftless area, largely left unglaciated during the Wisconsin glaciation. In New York, the Catskills were near the southern limit of the Wisconsin glaciation, but the detailed glacial history of the southern Catskills is poorly studied (Rich 1934). Dry Brook, Beaverkill, and Neversink valleys were possibly south of the last continental glaciation, but they were occupied by small valley glaciers, while the peaks and ridges around those valleys were possibly snow free "nunataks" (Rich 1934). The current monkshood distribution suggests the possibility that Double Top Mountain was a refugium from which monkshoods dispersed into the three surrounding valleys.

Drawing population boundaries can be a problem for managers of any rare species. It is a serious concern for monkshood in New York, where many small clumps of plants are scattered along 20 miles of

creek bank. It isn't clear whether an entire drainage is a population, whether each individual clump is a population, or whether a drainage can be divided into a small and manageable number of populations. One biological solution to this problem is to look at genetic similarity at individual sites and choose population boundaries that match genetic subdivisions of the population.

Knowledge of genetic variation in populations can contribute to the management of a rare species in other ways. Experimental transplants have been used to establish new populations of Northern Monkshood in Wisconsin, for instance. Choosing source populations of high within-population variability may help improve transplant success and maintain genetic variation.

Genetic structure of natural populations can be studied using variation in morphological traits, protein characteristics (allozymes), or DNA sequences. Leaf and flower characteristics of Northern Monkshood are quite variable, with much intrapopulation variation and little differentiation among populations (Brink, 1982). In this study, we chose to analyze allozyme variation within and among populations. Our specific goals were: 1) to characterize variation within and among populations as an aid in choosing transplant sources, 2) to determine whether population structure is related to geographic distance, substrate differences, or dispersal within drainage basins, and 3) to examine the structure of populations in New York State in order to group them into manageable units.

## METHODS

Recently expanded leaves were collected from 38 sites of Northern Monkshood in Iowa (8 sites: IA - IJ), Wisconsin (11 sites: WA-WK), and New York (19 sites: NA - NS). Small populations in New York and Wisconsin were sampled by collecting leaves from all large plants present; larger populations in Iowa and Wisconsin were sampled by arbitrarily choosing 40 individuals from throughout the population. Leaves were collected between 20 May and 3 June 1987, immediately frozen on dry ice and stored at -70 C until processed.

Leaves were ground in an extraction buffer (0.05 M Tris-HCL at pH 8.0, 0.4% mercaptoethanol, 0.00018 M EDTA), then proteins in the supernatant were separated by horizontal starch-gel electrophoresis. Electrophoretic methods, staining protocols, banding pattern interpretations, and genic nomenclature followed May, Wright, and Stoneking (1979) and Harris and Hopkinson (1976). A preliminary screen of 80 individuals from populations across the range of Northern Monkshood was used to optimize extraction methods and buffer systems. Banding patterns for 19 enzyme systems could be resolved clearly, of which 17 were monomorphic (Table 1). Two enzymes, Acid phosphatase (ACP) and Esterase (EST), exhibited clear polymorphic banding patterns consistent with their protein structures and appeared to be coded by a single locus each. All 1321 individuals were then stained and scored for ACP and EST. Alleles are identified by the mobility of their homomeric protein product relative to that of the commonest allele in New York.

Data were analyzed with the aid of the BIOSYS-1 program (Swofford and Selander, 1981). Genetic variation within populations was summarized by mean number of alleles per polymorphic locus and mean direct-count heterozygosity. Within-population fit to Hardy-Weinberg expectations was verified using Chi-square tests with Levene's (1949) correction for small sample sizes. Rare alleles were pooled when expected genotype frequencies were less than 1.0. Populations were clustered using the UPGMA algorithm, with genetic distances between populations measured by Roger's distance. Wright's hierarchical F statistics (Wright, 1969, 1978) and Nei's genetic diversity measures (Nei, 1973) were used to examine the distribution of genetic variation among population subdivisions.

## RESULTS

### Within-population variation:

Phenotypes for EST and ACP could be resolved for most individuals. In Wisconsin and Iowa, where approximately 40 individuals were collected from each population, from 31 to 41 individuals were suc-

**Table 1.** Results of enzyme screen on 80 individuals of Northern Monkshood. The buffer system used for each enzyme are indicated in parentheses. Buffers are 4: continuous Tris-citrate type 4 of Selander et al. (1971), C: CT buffer of May, Wright, and Stoneking (1979), R: (Ridgway, Sherburne, and Lewis, 1970), M: (Markert and Faulhaber, 1965).

Isozymes clearly resolvable		Limited Activity or poor resolution, loci appear to be:	
Monomorphic	Polymorphic	Monomorphic	Polymorphic
GR (C)	EST-A-1 (4)	CAT (M)	IDH (4)
HA (R)	ACP-1 (4)	PGK (4)	ME-2 (C)
GDK (4)		AK (R)	SKDH (4)
AAT (R,C)		BGLU (4)	GPI-1 (4)
MDH-1 (C)		G6PDH (R)	
MDH-2 (C)		GK (R)	
SOD-1 (R)		ALD (4)	
SOD-2 (R)			
LAP (R)			
PGM (R)			
GAPDH (C)			
ACP-2 (4)			
GPT-1 (R)			
GPT-2 (R)			
PGD (C)			
ME-1 (C)			
TPI-3 (4)			

**Table 2.** EST and ACP allele frequencies in Northern Monkshood. Alleles are labelled by percent mobility of the protein relative to that of the most common NY allele. Iowa site names and numbers are from Whitson, Stanislav and Watson (1985).

Population	N	EST					ACP				
		70	100	118	139	68	100	120	175	220	
<b>New York</b>											
NA: Dog Hollow	30		1.0				.867		.133		
NB: Dog Hollow	21		1.0						.333		
NC: Dog Hollow	29		1.0			.119	.548				
ND: Dog Hollow	38	.023	.977				.810		.190		
NE: Dry Brook	26		1.0				.288	.173		.538	
NF: Dry Brook	12		1.0			.167	.208	.250		.375	
NG: Dry Brook	31		1.0			.097	.290	.306		.306	
NH: Dry Brook	32		1.0			.133	.200	.333		.333	
NI: Beaver Kill	40		1.0			.400	.350	.038		.213	
NJ: Beaver Kill	10		1.0			.250	.500			.250	
NK: Beaver Kill	22		1.0				.455	.227		.318	
NL: Beaver Kill	41		1.0			.110	.329	.085		.476	
NM: Beaver Kill	39	.081	.919			.013	.603	.026		.359	
NN: Peekamoose	36	.014	.972	.014			.500	.097		.403	
NO: Peekamoose	40	.016	.952	.032			.438	.013		.550	
NP: Peekamoose	38	.048	.952				.566			.434	
NR: Dry Brook R.	41		1.0							1.0	
NS: Dry Brook R.	39		1.0							1.0	
NT: Dry Brook R.	57		1.0							1.0	
<b>Wisconsin</b>											
WA: Loddes Mill	37		.542	.458				.959			.041
WB: Hay Valley	35			1.0			.357	.286		.357	
WC: Hay Valley	37			1.0			.459	.176		.365	
WD: Hay Valley	31			1.0			.097	.597		.306	
WE: Hay Valley	40			1.0			.125	.412		.450	.012
WF: White Hollow	34			1.0				.206		.544	.250
WI: Parfrey's Glen	27		1.0					1.0			
WG: Chase Creek	34		.617	.383				.551			.449
WH: Chase Creek	39		.571	.429				.667			.333
WJ: Chase Creek	40	.087	.250	.663				.800			.200
WK: Chase Creek	39		.414	.586				.692		.154	.154
<b>Iowa</b>											
IA: Elkader N (73)	39	.705	.026	.269			.372	.038			.590
IB: St Olaf (218)	35	.212		.500	.288		.257	.014			.729
IC: Howard Ck (245)	39	.423	.038	.538			.885	.024	.026		.064
ID: Elkader NW (261)	39	.149		.851			.026		.846		.128
IE: Elkader S2 (166)	41			1.0			.183	.037	.122		.659
IF: Kline Hunt (162)	38		.278	.722				.947			.053
IG: Kline Hunt (161)	35		.617	.383				.743			.257
II: Dubuque (299)	32	.518	.411	.071				.203	.047		.750

successfully scored (Table 2). Sample sizes for the Parfrey's Glen (WI) and most New York populations are smaller because fewer than 40 plants were collected from those sites. Generally 75% - 100% of individuals could be scored at each locus. The banding patterns of unscorable individuals were usually too faint at both loci, suggesting that protein activity in that leaf was low. Representative zymograms for the ACP and EST loci are shown in Figure 1.

Most *A. noveboracense* populations were in Hardy-Weinberg equilibrium. Three populations deviated significantly from HWE at the ACP locus and one deviated at the EST locus (Table 3). Excess heterozygotes were found in population B (ACP), while excess homozy-

gotes were observed in populations IC (ACP), ID (ACP), and WA (EST). Although the number of loci not in HWE exceeds that expected by chance alone, no consistent patterns could be discerned and no population was out of HWE at both loci.

Most populations were polymorphic, with up to 3.5 alleles per polymorphic locus and an average direct-count heterozygosity of 3.1% (Table 3). Generally, New York populations were fixed for EST allele 100 and polymorphic at the ACP locus, with up to 4 alleles present in Dry Brook populations, while both loci were polymorphic in most Wisconsin and Iowa populations. There was no observed genetic variation in three areas of the large population on Dry Brook Ridge in New

York and in the small population at Parfrey's Glen in Wisconsin (Tables 2, 3). Iowa populations had significantly more alleles per locus (mean 2.70) than Wisconsin (mean 2.04) and New York (mean 2.08,  $F=3.77$  with 2,35 d.f.,  $p < 0.05$ ); similarly, Iowa populations tended to be more heterozygous (mean 3.7%, compared to 3.5% and 2.7%), but the differences were not statistically significant. Across all samples, there was no correlation between log transformed adult population size and per cent heterozygosity (Figure 2), although there was a significant positive correlation in Wisconsin samples, a significant negative correlation in New York samples, and no significant correlation in Iowa

samples (Figure 2). The lack of a consistent relationship across the three states suggests that heterozygosity is more influenced by other differences between the sites than by adult population size.

#### Population structure:

Allele frequencies differed considerably among populations of Northern Monkshood, although nearby sites were often quite similar (Table 2). These differences in allele frequencies were statistically significant for each allele and group of populations (Table 4). Pooling both alleles in all populations, 53% of the genetic variation was

**Table 3.** Within-population genetic variation and hierarchical arrangement of monkshood sites. Estimated population sizes are from Dixon and Cook (1989) for New York, Whitson et al. (1985) for Iowa, visual estimation in May 1986 for Wisconsin.

Population	Drainage	Substrate	Not in HWE	Mean number alleles/polym. locus	Mean heterozyg.	Estimated pop. size	
New York							
NA:	Dog Hollow	shale/sandstone	ACP	1.5	0.014	90	
NB:	"	"		2.0	0.043	50	
NC:	"	"		1.5	0.016	70	
ND:	"	"		2.0	0.019	53	
NE:	Dry Brook	"		2.0	0.036	70	
NF:	"	"		2.5	0.040	50	
NG:	"	"		2.5	0.032	57	
NH:	"	"		2.5	0.042	100	
NI:	"	"		2.5	0.036	210	
NJ:	Beaver Kill	"		2.0	0.039	15	
NK:	"	"		2.0	0.038	36	
NL:	"	"		2.5	0.038	108	
NM:	"	"		3.0	0.030	110	
NN:	Peekamoose	"		3.0	0.032	200	
NO:	"	"		3.0	0.031	500	
NP:	"	"		2.0	0.031	500	
NR:	Dry Brook Ridge	"		1.0	0.000	1000	
NS:	"	"		1.0	0.000	200	
NT:	"	"	1.0	0.000	1000		
				mean: 2.08(0.65)	0.027(0.014)		
Wisconsin							
WA:	Loddes Mill	sandstone	EST	2.0	0.018	100	
WB:	Hay Valley	"		2.0	0.032	200	
WC:	"	"		2.0	0.028	500	
WD:	"	"		2.0	0.037	300	
WE:	"	"		2.5	0.038	500	
WF:	White Hollow	"		2.0	0.028	200	
WI:	Parfrey's Glen"	"		1.0	0.000	40	
WG:	Chase Creek limestone	"		2.0	0.052	1200	
WH:	"	"		2.0	0.054	1000	
WJ:	"	"		2.5	0.047	300	
WK:	"	"		2.5	0.047	100	
				mean: 2.04(0.41)	0.035(0.016)		
Iowa							
IA:	Roberts Creek	limestone	ACP	3.0	0.049	750	
IB:	"	"		3.0	0.054	10000	
IC:	"	"		3.5	0.030	3000	
ID:	Turkey Creek	"		2.5	0.018	200	
IE:	"	"		2.5	0.024	1000	
IF:	Kline Hunt H.	"		2.0	0.023	7500	
IG:	"	"		2.0	0.047	400	
II:	Lytle Creek	"		3.0	0.049	1000	
				mean: 2.69(0.53)	0.037(0.014)		

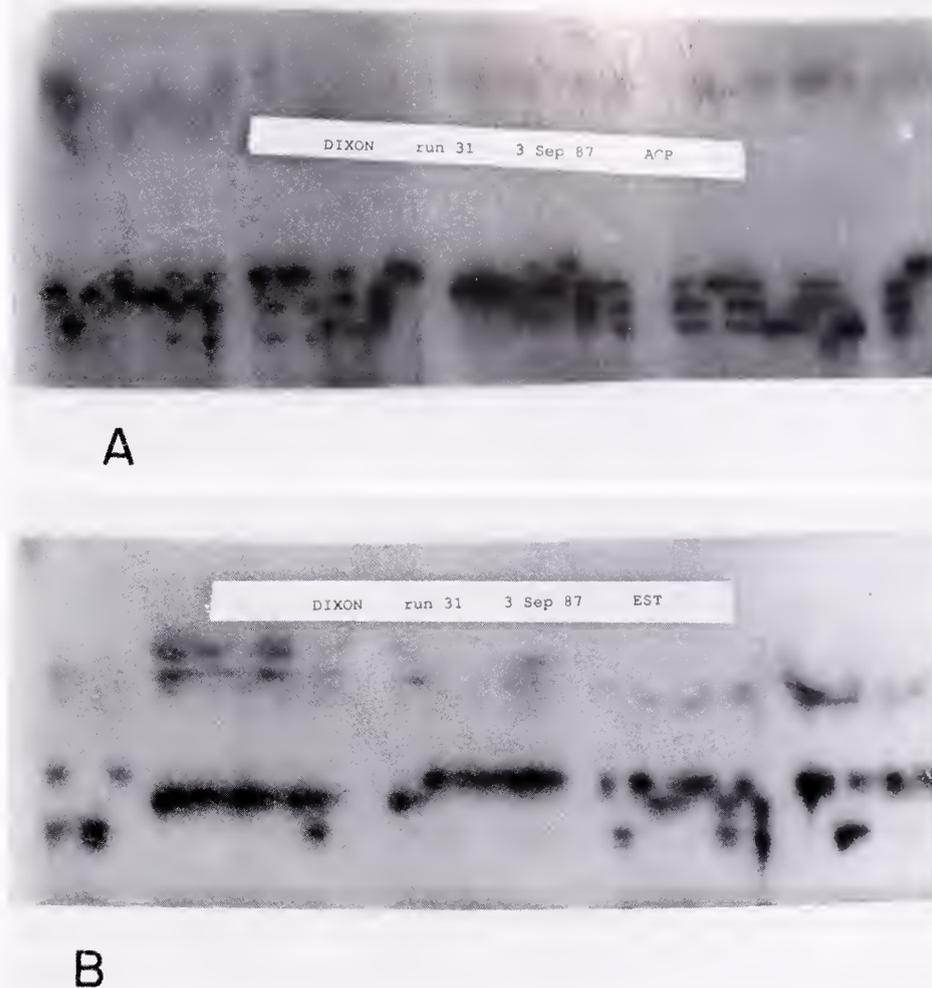


Figure 1. Representative zymograms for a) ACP and b) EST.

attributed to differences among populations (Table 4).  $F_{is}$  statistics were small or negative, except in the Iowa populations (Table 4).

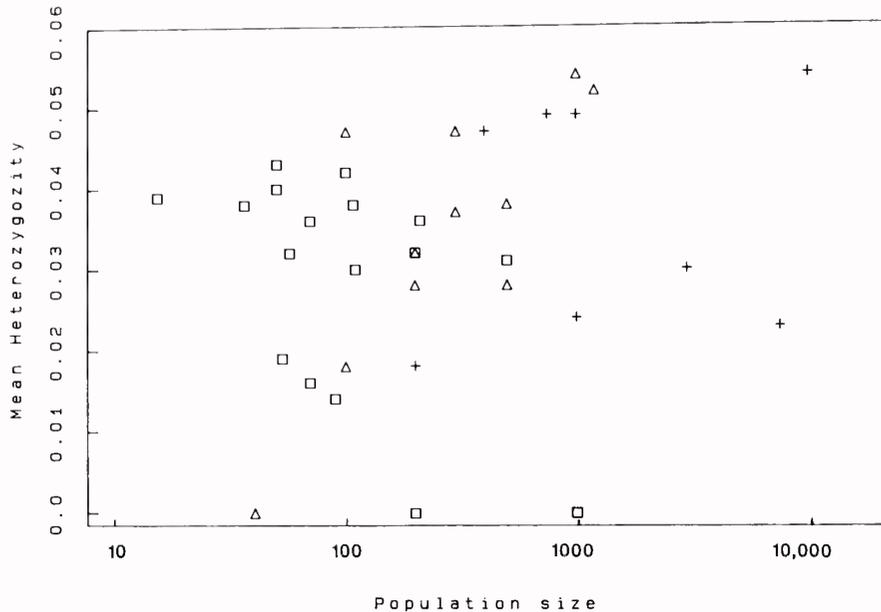
**Table 4.** Wright's F statistics for population subdivisions of Northern Monkshood.  $F_{is}$  measures the reduction in heterozygosity due to inbreeding;  $F_{st}$  measures the apparent inbreeding due to genetic differentiation.

Significance levels for  $F_{st}$ : \*\*\*\*:  $P < 0.0001$

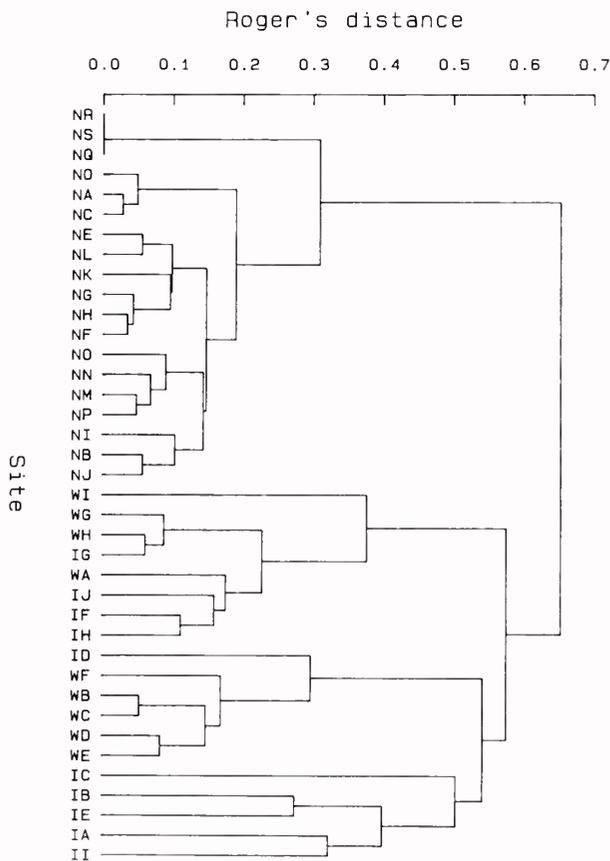
State	EST		ACP		Combined	
	$F_{is}$	$F_{st}$	$F_{is}$	$F_{st}$	$F_{is}$	$F_{st}$
New York (19 pops)	-0.054	0.042 ****	-0.057	0.252 ****	-0.057	0.244 ****
Iowa (8 pops)	0.118	0.334 ****	0.027	0.510 ****	0.076	0.428 ****
Wisconsin (11 pops)	0.076	0.494 ****	-0.027	0.268 ****	0.008	0.363 ****
All Northern Monkshood (38 pops)	0.089	0.686 ****	-0.034	0.416 ****	-0.001	0.527 ****

A UPGMA cluster analysis of the Roger's distance matrix portrayed the differences among populations (Figure 3). The major division in the cluster tree was that between New York populations and Iowa/Wisconsin populations. Within New York, genetic distances between sites were small; populations from a drainage clustered together, except for populations in the Beaverkill, which fell into 3 other clusters. Within the Iowa/Wisconsin cluster, only the Hay Valley populations and the nearby White Hollow population formed a unique cluster that was related to geography. Although the Chase Creek populations fell into one cluster, that cluster included the Kline Hunt Hollow populations that were nearby but across the Mississippi River and the geographically distant Loddes Mill Bluff population. In general, Iowa and Wisconsin populations were less similar to each other than were New York populations, and clusters of populations in Iowa/Wisconsin were more distinct from each other. The robustness of the clustering was tested by jackknifing. Thirty-eight jackknife clusterings were performed, omitting each locality one at a time. Each jackknife clustering pattern was essentially the same as the original clustering.

Up to three levels of population structure may be distinguished in these data: among locations within drainages, among drainages within substrates, and among substrates (Table 3). Across the range of Northern Monkshood, two levels accounted for most of variation among populations (Table 5): variation among drainages ( $G = 0.221$ ) and variation among substrates ( $G = 0.229$ ). Genetic variations noted among substrates were exclusively due to the differences between New York and the other two states; there was no variation between lime-



**Figure 2.** Relationship between direct-count heterozygosity and population size (log scale) for Northern Monkshood populations. Combining all states:  $r = 0.18$ , n.s. New York alone ( $\square$ ):  $r = -0.57$ ,  $p = 0.011$ ; Wisconsin: ( $\Delta$ ):  $r = 0.75$ ,  $p = 0.0079$ ; Iowa (+):  $r = 0.158$ , n.s.



**Figure 3.** UPGMA clustering of populations, based on Roger's distance calculated from EST and ACP loci. Cophenetic correlation for this phenogram is 0.906. See Table 2 for full names of sites.

stone and sandstone substrates in Wisconsin and Iowa. If each group of states is considered separately, most of the population differentiation is between drainages ( $G = 0.196$  in New York and  $0.331$  in Iowa/Wisconsin). New York populations had more genetic variation within populations ( $G$  individuals =  $0.772$ ) than did Iowa/Wisconsin populations ( $G$  individuals =  $0.537$ ), less variation among localities, and less variation among drainages.

### DISCUSSION

Small populations of northern monkshood have low heterozygosity within populations, but they have maintained large amounts of genetic variation among populations. The overall  $F_{ST}$  value of 53% is considerably larger than values for among-population differentiation reported for three endemic trees: California Fan Palms (McClenaghan and Beauchamp, 1986), Monterey Cypress (Conkle, 1988), and Caledonian Scots Pine (Kinloch, Westfall and Forrest, 1986). The among-population variation is also high, compared to averages reported for species with similar ecological traits: bee pollinated (mean 22%), long-lived perennial (mean 7.7%), polycarpic (mean 14%), late successional habitat (10%), and regional distribution (23%) (Loveless and Hamrick, 1984). However, the among-population variation in Northern Monkshood is quite similar to that reported for another deciduous forest herbaceous perennial, *Desmodium nudiflorum*, which has 47% of its genetic variation associated with among-population variation (Schaal and Smith, 1980).

Low levels of heterozygosity are expected for populations with small effective population sizes (Wright, 1969). Northern Monkshood populations are less heterozygous and have fewer polymorphic loci than is typical for plants, but there are more alleles per polymorphic locus (Hamrick et al., 1979). The number of polymorphic loci in Northern Monkshood is probably underestimated in these analyses. If a rare allele was missed in the initial scan of 80 individuals, the locus

would be erroneously considered monomorphic. A sample of 80 individuals has a statistical power of 95% of detecting a polymorphism if the frequency of the rare allele is 1.9% or more; however, if rare alleles are present at only a few sites, the power of detecting them is much lower.

**Table 5.** Proportion of total genetic variation associated with different levels of population structure. Statistics for subdivisions are Nei's G statistics; statistics for total genetic variation are Nei's mean  $H_t$  for polymorphic loci.

Subdivision	New York Only		
	EST	ACP	Combined
Individuals within sites	0.975	0.764	0.772
Sites within drainages	0.022	0.033	0.032
Drainages within total	0.003	0.203	0.196
Total variation	0.024	0.626	0.325

Subdivision	Wisconsin and Iowa		
	EST	ACP	Combined
Individuals within sites	0.510	0.557	0.537
Sites within drainages	0.084	0.169	0.132
Drainages within substrates	0.406	0.274	0.331
Substrates within total	0.000	0.000	0.000
Total variation	0.535	0.696	0.616

Subdivision	All populations		
	EST	ACP	Combined
Individuals within sites	0.319	0.594	0.482
Sites within Drainages	0.036	0.090	0.068
Drainages within Substrates	0.214	0.226	0.221
Substrates within Total	0.431	0.089	0.229
Total variation	0.515	0.746	0.630

The amounts of heterozygosity and polymorphism in northern monkshood are typical of selfing plants (mean heterozygosity: 0.058, polymorphic loci: 19%, Hamrick et al., 1979), but the lack of divergence of genotype frequencies from Hardy-Weinberg Equilibrium (Table 3), lack of consistent correlations between heterozygosity and population size (Figure 2), and the low  $F_{is}$  values (Table 4) suggest that inbreeding is not important in this species' population structure. Detailed studies of seed dispersal, pollen flow, and reproductive biology are needed, however, to confirm this. In particular, the lack of consistent correlation between population size and heterozygosity suggests that small population sizes are less important than other factors determining heterozygosity in this species.

Plants of Dry Brook Ridge in New York are unusual in having large populations with no electrophoretic variation. During an especially dry summer, 80% mortality was observed in one part of this site, higher than at any other New York site (Dixon and Cook, 1990), so this site may experience severe population bottlenecks. Founder effects may also explain the low heterozygosity. Dry Brook Ridge is unique among New York sites in not being along a creek; one chance colonization event may have initiated the population.

Levels of heterozygosity in Northern Monkshood may be explained by aspects of its ecology. Monkshoods are long-lived slow-growing perennials with the capacity to reproduce clonally. Ramets can not be directly dated, but indirect evidence from growth rates suggests that

large plants in New York are 40-50 years old or older (Dixon and Cook, 1989), and the effective generation time may be even longer because of clonal propagation on rates. Annual loss of genetic variation due to drift or inbreeding should be slower in populations with long generation times (Wright, 1969). Seedling survival is low in many populations, especially in New York, so even if individuals in a small population are breeding with their relatives, their inbred progeny are unlikely to join the breeding pool. Even in the small populations in New York, the loss of genetic variability due to inbreeding may be extremely slow.

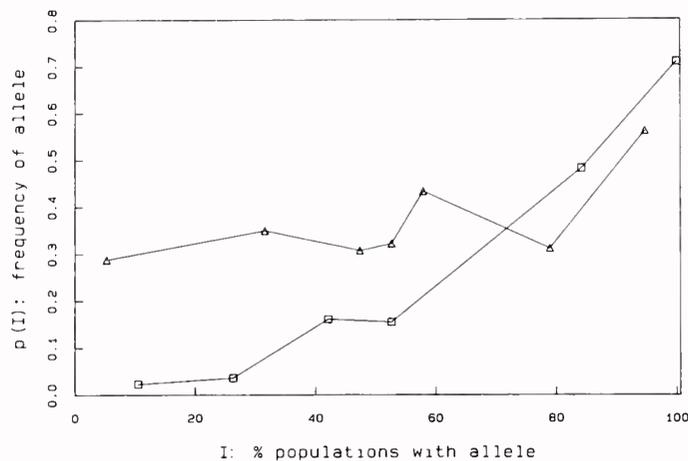
Gene flow among sites probably occurs primarily within a drainage. We observe genetic similarity within a drainage, but high differentiation among drainages (Table 5). Sites in New York drainages are more similar to each other than are those within a Wisconsin or Iowa drainages (Table 5) suggesting that gene flow is higher in New York. Analysis of conditional average allele frequency (Slatkin, 1981) also suggests that gene flow between populations is higher in New York. In this analysis, alleles are grouped by the number of populations in which each is found. The average allele frequency is calculated for those alleles found in one population, two populations, and so on. In a system with high gene flow between populations, an abundant allele is likely to spread to other populations, so the average frequency of alleles found in few populations is lower than in a system with low gene flow. Alleles found in New York are shared among more populations than are Wisconsin and Iowa alleles (Figure 4), a pattern that is consistent with higher gene flow within New York sites (Slatkin, 1981). The higher rate of gene flow may be due to differences in dispersal rates among states or the shorter distances between most (but not all) sampling sites in New York.

All our analyses and conclusions are necessarily limited because they are based on just two polymorphic loci. All electrophoretic studies sample only a portion of the genome (Richardson, Baverstock and Adams, 1986), but when the number of polymorphic loci is small, it is possible that the observed patterns are due to chance events or locus-specific effects and are not general patterns in genetic structure.

Assuming that the patterns of the two loci studied here are representative, knowledge of the genetic structure and within-population variation can help us to understand the evolutionary history of the taxon and also help to design conservation strategies. The similarity among populations within a drainage suggests that all colonies within a given drainage are part of one genetic population, and it is not necessary to give separate consideration to each small colony in New York. This strategy is being followed, except in the longest drainage in New York, the Beaverkill, where populations do not fall into one cluster (Figure 3). Population boundaries in the Beaverkill are being determined by land-ownership patterns and ecological factors (U.S. Fish and Wildlife Service, 1987).

Similarity among sites within a drainage also suggests that transplants to reestablish extirpated populations or establish new populations should be taken from source populations within the same drainage, wherever possible. If genetic variation is a concern, the Dry Brook Ridge area is a poor source of transplants, even though it contains the largest New York population. Other large populations in Iowa and Wisconsin are genetically variable and would be more suitable sources for transplants.

One monkshood site (or even all the sites in a single drainage) represent only a small fraction of the genetic variation within northern monkshood. Conserving a large fraction of genetic variation in northern monkshood will require conserving many populations distributed



**Figure 4.** Conditional average allele frequencies for New York (□) and Wisconsin/Iowa (Δ) populations. I is the proportion of populations in which an allele occurs; p(I) is the average frequency of those alleles. Systems with higher gene flow have lower p(I) values for rare alleles (I < 0.2). See discussion for further details.

across the range of the species. If dispersal among sites is necessary to maintain the levels of variation within each site, then it will also be necessary to maintain the continuity among sites in a drainage. A good strategy would be to conserve at least one population from representative drainages in each state. Deciding the number of conserved populations in a drainage requires better information on the processes maintaining heterozygosity within a site.

#### ACKNOWLEDGMENTS

Janet Dixon, June Dobberpuhl, Val Kirsis, Margaret Kuchenreuther, Holliss Marriot, Ethan Perkins, and Susan Winters helped find populations, collect plants, and run electrophoretic gels. Various state officials, federal officials, and landowners granted collection permission. Electrophoretic analyses were done at the Cornell Laboratory for Ecological and Evolutionary Genetics. Peg Mulvey and Margaret Kuchenreuther provided comments on the manuscript. Field work and electrophoretic analyses were funded by a McIntire-Stennis grant to Robert Cook and Philip Dixon. Data analysis and manuscript preparation was funded by contract DE-AC09-76SROO-819 between the U.S. Department of Energy and the University of Georgia's Savannah River Ecology Lab.

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# Species Biology of *Lindera melissifolia* (Walt.) Blume. in Northeast Arkansas

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**Abstract:** The rare pondberry is characterized in the Arkansas portion of its range, where it occurs in isolated populations in dune-field depressions of the northeast portion of the state. The species is well adapted to maintain itself vegetatively where it now grows, but incapable of effective seed reproduction. Management to maintain present habitat characteristics is recommended, along with steps to increase the success of reproduction by seed.

Pages 176 - 179. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Pondberry is an appropriate common name for this understory shrub of seasonally flooded bottomland hardwood forests in the southeastern states. Tucker (1983) reported occurrences of *Lindera melissifolia* in North Carolina, South Carolina, Georgia, Mississippi, Missouri and Arkansas. The common name appears to have been first reported by Steyermark (1949), and may have originated in Missouri or Arkansas, where all the stands are found in topographically isolated depressions in dunefields.

Saucier (1978) described the probable origin of the 1025 km<sup>2</sup> of sand dunes where the plants occur as derived from glacial outwash, delivered by the Mississippi River system principally between 22,000 and 18,000 years B.P. The dunefields extend across northeastern Arkansas into southeastern Missouri.

Individual stands of pondberry observed in this study cover areas of 1000 m<sup>2</sup> or less, and have from 175 to over 10,000 stems each. They tend to occur close to pond edges under complete or partial tree canopies. Dune slopes surrounding the ponds have all been cleared for farming. There is shallow winter flooding over most of each stand. During the winters of 1987 and 1988 the maximum water depth over any pondberry stems was approximately 30 cm, although some ponds reached greater depths outside the vicinity of the plants.

Flowering occurs in mid-March while the ponds are still flooded, and shoot growth begins in late April concurrent with leaf expansion in the tree canopy. During 1987 and 1988 water receded from the lowest stems of pondberry in April, May or early June. In 1987 the ponds were dry for the remainder of the growing season. Fruiting occurs in October, with most fruits shed within one to two months.

Pondberry has always been a rare plant (Steyermark, 1949). It received Federal listing as threatened in 1986. This study investigated its prospects for survival, and management strategies that may be required.

## VEGETATIVE ADAPTATIONS

The following observations covered the period January 1987 to May 1988, during which data were accumulated over 20 field days. They indicate that the species possesses both hydric and xeric adaptations.

## HYDRIC ADAPTATIONS

### Root environment:

Excavation revealed a shallow root system, with principal rhizomes and roots in the top 20 cm of soil. Rhizomes are thicker than roots and

grow horizontally within a few cm of the surface. Root growth is not vigorous, and extends from rhizome nodes only for a few additional cm. Root growth and metabolism are presumably favored by soil aeration, although some metabolic activity may be maintained under hypoxic conditions. Living roots were found in saturated soils where dissolved oxygen was below 1.5 ppm in the soil solution. Hypoxic conditions can be tolerated by roots and rhizomes of a number of wetland species (Schat, 1985; Armstrong, 1978). Microscopic cross-sections of rhizomes and roots of pondberry revealed tissues with regular intercellular spaces (lacunae) that apparently enhance diffusion of oxygen to underwater parts (see Armstrong, 1978).

Soil characteristics in and around the ponds contribute to acquisition and retention of pond water. Although no surface drainage network could be found, abrupt transition to lenses of finer particle sizes, along with the presence of considerable clay and silt, would be expected to contribute to ponding (H. Don Scott, pers. comm.). Table 1 confirms such soil properties for two sites (sites are designated by Element Occurrence numbers assigned by Arkansas Department of Natural Heritage biologists). It is possible that ground water recharge supplies water to the ponds through these soils, as documented by Winter (1986) in the sandhills of Nebraska and by Mills and Zwarich (1986) for sloughs in a Manitoba till plain.

**Table 1.** Soil particle sizes from hydrometer analysis of two pondberry (*Lindera melissifolia*) habitats.

Site 10	%sand	%silt	%clay	Site 11	%sand	%silt	%clay
10 cm	44	36	20	46	26	28	
45 cm	64	15	21	50	10	40	
90 cm	62	14	24	54	14	40	

Soil Conservation Service classifications for several sites are given below:

- Site 6 - Wardell fine sandy loam
- Site 8 - Patterson fine sandy loam
- Site 10 - Tuckerman fine sandy loam
- Site 11 - Patterson fine sandy loam
- Site 12 - Patterson fine sandy loam
- Site 13 - Dundee silt loam

With the exception of Dundee silt loam, these soils are classified as hydric, meaning that they remain saturated for too long a period into the growing season to permit cropping. Hydric classification also discourages drainage and clearing of the sites under the provisions of the "Swamp Buster" portion of the Federal farm act for 1985.

### Shoot environment:

Canopy closure over most sites promotes the development of typical shade leaves that are extremely thin, with a single cell layer of palisade cells and a thin, loose, spongy mesophyll. Their response of net photosynthesis to light intensity was also characteristic of shade leaves. Response was measured with a LI-COR 6200 portable photosynthesis system in May 1988, when soil water potential was near -0.1 MPa and leaf water potential was above -1.0 MPa as determined by thermocouple psychrometry. Figure 1 demonstrates that net photosynthesis occurs with PAR values as low as  $15 \text{ } \mu\text{mol m}^{-2} \text{ s}^{-1}$ , and that light saturation may be approached at around  $200 \text{ } \mu\text{mol m}^{-2} \text{ s}^{-1}$ . The latter value is within the range for shade leaves cited by Bohning and Burnside (see Boardman, 1977).

### XERIC ADAPTATIONS

When severe late-summer drought develops at a site, response of the pondberries indicates they are adapted to survive conditions that tend to limit competing plants. Although the leaves wilt, they do resist wilting better than one of the competing vines, *Brunnichia ovata*. When rains come, the pondberry also recovers turgidity more quickly and completely. Under conditions of extended drought, pondberry is facultatively deciduous, and may experience partial leaf senescence and twig dieback while retaining some healthy foliage. *Brunnichia* stems die to the ground under such conditions (observations were made in August and September, 1987).

Where the tree canopy is approximately 50% open, the pondberry develops sun leaves that are thicker and more rugose than its shade leaves. One such stand (at Site 6) fruited heavily in 1987, with an estimated 100,000 fruits produced in an area of 5 x 30 m. Other fruiting populations observed in 1986 and 1987 produced an estimated crop of 100 to 2000 fruits per population.

The fringes of a number of stands extend up the pond banks to well above the high water mark, where surface soils tend to be sandy and summer drought more severe. Their advance seems to be checked by

competition rather than lack of moisture, as they become overgrown by a number of woody understory species.

### HERBIVORY

Browsing by vertebrates was detected at only one stand, where a few stems appeared to have been shortened by rabbits during the winter. Insect damage to foliage was seen as minimal, amounting to less than 1% of leaf area. Pondberry is an aromatic member of the Laurel Family (Lauraceae), and it may possess chemical defenses against herbivores. Also, very few of its fruits appear to be eaten.

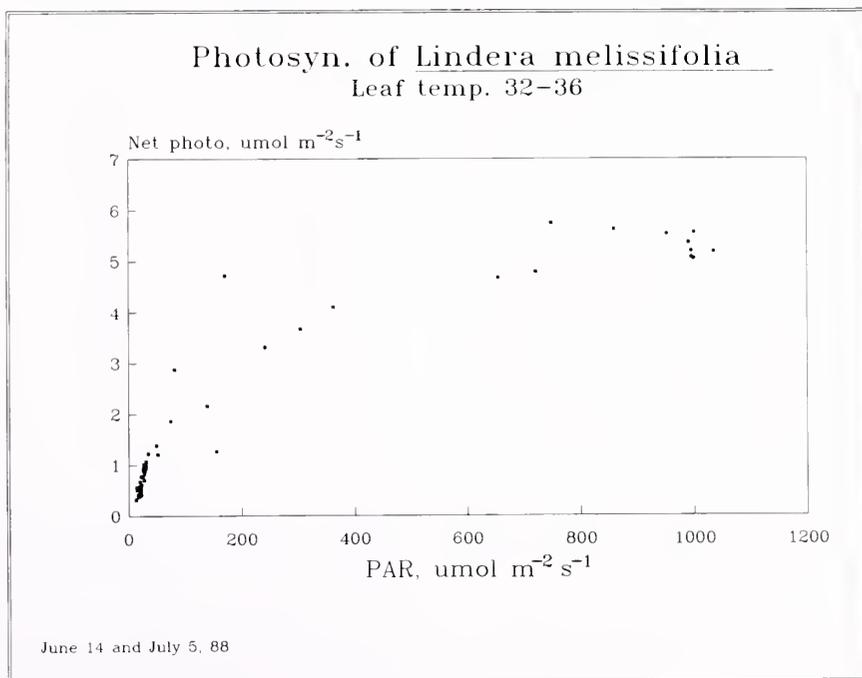
### REPRODUCTIVE POTENTIAL

#### Vegetative reproduction:

This species appears to rely on vegetative reproduction to maintain substantial stands. At all sites sprouts were readily produced from both rhizomes and stems, maintaining a stem density of 4 to  $10 \text{ m}^{-2}$ . Stems on the periphery of a population were sometimes isolated by one to two meters, indicating spread of the stand by vegetative means. At the upper margins of populations there was typically considerable competition from other woody species, but at their lower margins there was little else in the shrub layer. Species of *Brunnichia*, *Smilax* and *Rubus* were the main competitors, but they were present in significant numbers in only a minority of stands.

#### Seed Production:

Seed production in 1986 and 1987 was sporadic, and, with the one exception cited above, light. During the flowering period in March, 1988 the distribution of sexes was determined for several stands (pondberry is dioecious), and results are presented in Table 2. Seed set was foreclosed in stands of one sex, since several such stands observed were of all males plants. In stands where both sexes were present, the size of the single-sex patches indicates that clonal reproduction predominates over reproduction from seed. Doust and Doust (1988) sug-



gested that with dioecious clonal shrubs the energy burden of reproduction may depress female survival; this is borne out for pondberry by the data in Table 2.

**Table 2.** Distribution of sexes in pondberry (*Lindera melissifolia*)

Site	Sexes	Clone diam.	Fruiting, 1987
6	M F	M 15 m F 20 m	heavy
8	M		none
10	M		none
11	M F	M 20 m F 7 m M 16 m M 16 m F 5 m	sparse
12	M		none

Although fruiting is sporadic and appears to fluctuate between light and heavy production in alternate years (the 1988 crop set at Site 6 was light), pollinator activity appears to be adequate. Hymenopterans, Dipterans, Lepidopterans and Coleopterans have been seen on the flowers of both sexes.

Seedlings are very uncommon in nature. The stand at Site 6, estimated to have produced 100,000 seeds in 1987, was searched for seedlings on June 2, 1988 and only 13 were found; 12 germinated that current year and one was two years old. All were growing within the stand under heavily shaded conditions. There was no apparent relationship between seedling location and canopy openings or disturbance. Most seedlings were grouped in two small areas 1-2 m in diameter, and had emerged from seeds 1-3 cm beneath the surface of the litter. There were at least 500 new sprouts in this fruiting stand, mostly several times as tall as the seedlings. These seedlings and seven 2-year seedlings at Site 11 were uniformly less than 10 cm tall, typically with only three or four small leaves, and they had apparently completed their growth for the season. Seedlings in the greenhouse, under a natural light regime between 250 and 400  $\mu\text{mol m}^{-2} \text{s}^{-1}$ , by contrast, have grown continuously for five months in native soil, reaching heights exceeding 60 cm with numerous full-sized leaves. It appears that in nature, competition for light and perhaps water may prevent most of the seedlings from entering the shrub canopy. Allelopathy can also not be ruled out.

Table 3 demonstrates that seeds are highly viable but require rupture of the seed coat and elevated temperatures to germinate promptly, conditions not readily attained under the protection and isolation found in the forest. Germination responses were also obtained with  $10^{-4}\text{M}$  gibberellic acid under some conditions. All tests were done using 20 seeds under sterile conditions on petri plates, and all seeds had the fleshy fruit wall removed. Additional treatments included notching the seed coat, peeling the entire seed coat away to expose the cotyledons, or stratification for two months.

In another test, peeled seeds were submerged under 2 cm of water where they failed to germinate over time periods predicted by the above results. When the water was then poured off, they germinated promptly.

Seeds with the fruit wall in place were exceedingly slow to germinate in native soil in the 29/18° greenhouse. The first emergence was at 60 days, and germination reached 18% at 110 days; after 180 days germination was 28%.

## DISTURBANCE

Around 1985, heavy equipment was driven through one site. By 1987 regrowth had largely replaced the many crushed stems, but, where the ground had been disturbed to any depth, regrowth and recovery were poor. In November 1987 a ground-fire killed all the stems in two adjacent sites. When the pond level receded in May 1988 vigorous sprouting began, approximately restoring the original stem numbers. These incidents demonstrated the vigor with which the rhizome system of pondberry regenerates stems after moderate disturbance; however, Tucker (1983) observed that crown fire exposed the plants to heavy competition and potentially damaging light intensity levels.

Hydrologic changes might produce severe effects. It seems clear that winter flooding is involved in reducing competition in the shrub layer, so that drainage of sites would be deleterious to pondberry. Although deep flooding, exceeding one meter, has been reported for a site in Delta National Forest in Mississippi (Becky Banker, pers. comm.), the water had been artificially withdrawn before the start of the growing season. Thus, critical maximum pond depths of 30 cm observed in this study may relate to the time of pond drying rather than depth of flooding. Both flowering and leaf expansion were observed while plants were still shallowly flooded.

## SUMMARY AND MANAGEMENT CONSIDERATIONS

In the dunefield ponds where it grows, pondberry seems well adapted to survive the natural conditions of its environment in Arkansas. It tolerates dormant-season flooding with no apparent ill effects, and it

**Table 3.** Germination of Seeds of Pondberry (*Lindera melissifolia*)

Pretreatment	Temp	Days	%Germ
strat/notch/water	21	10	30
strat/notch/GA	21	10	45
notch/water	21	10	30
notch/GA	21	10	25
strat/peel/water	21	8	80
peel/water	21	8	90
strat/notch/water	29/18	10	45
strat/notch/GA	29/18	10	70
notch/water	29/18	10	35
notch/GA	29/18	10	35
peel/water	13	13	15
peel/GA	13	13	90
peel/water	21	9	90
peel/GA	21	9	95
peel/water	29	6	100
peel/GA	29	6	90
notch/water	21	15	5
notch/GA	21	13	75

carries on enough photosynthesis under the tree canopy to permit vigorous vegetative growth. When the ponds dry, it survives as well as or better than competing woody plants; since 1980 the stands have survived several severe summer droughts. It suffers little damage from herbivores. One feature of a vigorous species is clearly missing, however: vigorous seed reproduction. Some stands are all male, and even where both sexes are present, fruiting is sporadic. In the one case where heavy fruiting was observed, seedling production was

minimal the following spring. Seedlings also grow very slowly in nature, in the presence of mature plants. During the time the bright red fruits remained on the plants there was little evidence of frugivory. It may be that, as with the congener *Lindera benzoin* (L.) Blume, spicebush, abscised fruit is eaten by ground-foraging birds such as grouse, quail and pheasants (Grimm, 1957). However abscised fruits of pondberry quickly darken and become well camouflaged among newly-fallen leaves from the tree canopy. In any case, the clonal nature of populations indicates the rarity with which seedlings reach the shrub canopy.

To sustain the vigor of existing populations of pondberry, the tree canopy should be kept at least partially intact. Although openings in the canopy may favor growth of competing plants, winter flooding seems critical in controlling potential competitors. Therefore, a stable hydrological regime should be maintained. Until experimentally verified, the surrounding dune slopes should be considered part of the hydrological system of a given pond. Since these have been in row crops for some years without any current evidence that the clearing adversely affected the pondberries, agricultural use of the dune slopes may perhaps be safely continued. Any chemicals applied nearby, including herbicides, should be presumed capable of reaching the ponds through underground lateral movement of water.

Enhancing seed reproduction might be of major value to the species, by way of enriching the gene pools of existing populations and fostering the establishment of new populations. This, however, will be difficult to manage. Greenhouse-grown seedlings might be transplanted into the field, but their survival capabilities in the wild have yet to be determined. Seeds could be pretreated and then sown in the field, but, like potential transplants, this could be costly on a large scale, and no success is guaranteed. Existing seed banks might be stimulated to germinate by low levels of disturbance, which could then be simulated in patches where seed was sown. Future studies should explore these and other possibilities for promoting reproduction from seed.

## ACKNOWLEDGMENTS

This study was supported by grants from the Arkansas Nongame Preservation Committee and the University of Central Arkansas Research Council. Assistance in field work was given by M. Brinkley, W. Chambers, C. Christy, D. Culwell, C. Hunter, K. Kegley, J. Marr, D. Priest, J. Thomas, D. Short and L. Walls.

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# The Significance and Management of Relict Populations of *Chamaelirium luteum* (L.) Gray

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**Abstract:** Relict populations may represent strongholds of important genetic variability and should be protected and actively managed, even if the concerned species is currently common elsewhere in its range. *Chamaelirium luteum*, a dioecious, perennial lily found in rich deciduous forests of southeastern United States, persists in fewer than a dozen known sites in New York and New England, although its historic range was much more extensive. Four of these northern populations occur in Dutchess County, New York and have been studied for three to eight years. These four populations range in size from 1200 to 3600 individuals and occupy forested and old field habitats. Annual flowering rates vary from 0 to 5% of the populations. Our three-tiered monitoring system includes; (1) periodic mapping of all individuals in each population; (2) detailed measurements of plants and environmental characteristics in 1 m plots; and (3) longitudinal studies of all individuals that have flowered since the project was initiated. Canopy cover is estimated annually using fisheye lens photography. The four populations show some significant qualitative differentiation from southern populations, i.e., different optimal habitat, occurrences of vegetative reproduction, and unusual, successful seed production by hermaphroditic individuals. Such differences document the significance of marginal populations to intraspecific diversity. Determination of management strategies have centered on enhancement of sexual reproduction and individual plant survival. Closed-canopy woodland populations have lower and more variable flowering rates and individual flowering schedules than do field populations, suggesting that light is the critical variable controlling sexual reproduction. Estimates of light availability within populations support this contention. The primary factor controlling survival appears to be light availability. Management goals for *Chamaelirium luteum* should foster the maintenance and enhancement of genetic diversity and maintain positive long-term population growth rate. Our data suggest that controlling the canopy to maintain 20-40% cover is one way to achieve both of these goals, assuming that increased rates of sexual reproduction enhance heterozygosity and genetic diversity. We propose that studies and tests of management strategies on relict populations such as these can serve as model systems for managing certain endangered species. Such populations also deserve management priority in their own right.

Pages 180 - 184. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The goal of conservation biology as stated by Soule (1985) is "to provide principles and tools for preserving biological diversity;" this mandates "maintaining the fitness of individuals and populations." A corollary might be to preserve the capacity of species to respond to environmental challenges in an evolutionary fashion. Long-term fitness will depend upon maintaining diversity in the gene pool of the species. Such genetic diversity includes polymorphic loci (degree of heterozygosity), presence of unique or rare alleles, chromosomal polymorphism, variations in linkage groups or in karyotypes, degrees of heterosis, dominance relationships among alleles, and epistasis. Some genetic diversity is inherent in locally adapted populations occupying atypical habitats (e.g. Antonovics, 1984; Bradshaw, 1984; Shumaker and Babbie, 1980) or isolated populations found along the periphery of a species' range (Mayr, 1963). It is therefore important to protect and manage such populations even though the species represented by these populations may be more abundant elsewhere in its range.

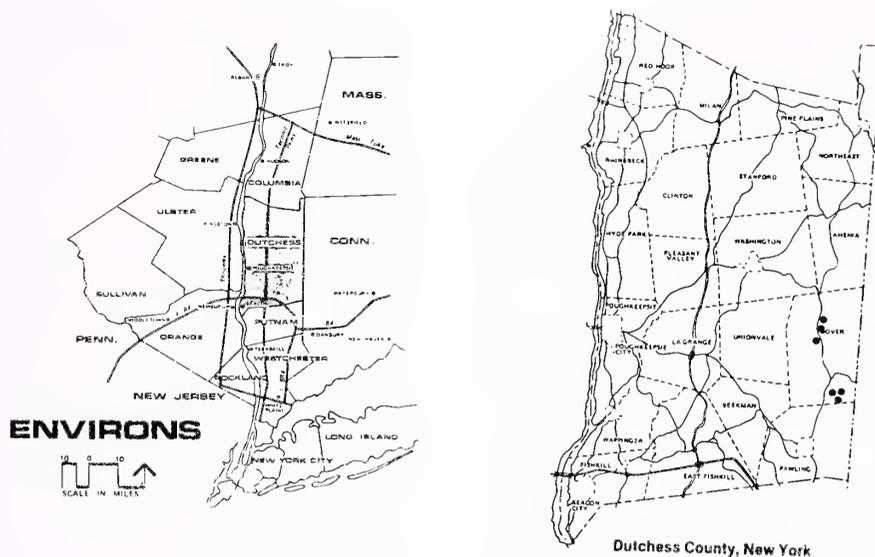
Rare species lists prepared on a statewide or local basis often provide recognition of such populations. One recurring problem, however, is that distributions of many species have been reduced during the past 50 to 100 years without the decline even being noticed. This is because numerous historical records exist for the species, and searches for it are deemed unnecessary. Therefore, careful validation of current distributions is important if endangered gene pools are to be identified and preserved. Natural Heritage Programs are making a significant contribution in identifying and verifying such rare elements in many states.

The crucial first step towards the protection of these isolated, potentially important populations is recognition of their occurrences. The second consideration is one of land ownership. Unlike wildlife, the landowner owns the plants growing on his/her land. Thus protection

and management depend on ownership, which renders the Natural Areas network a crucial link in plant conservation. A third problem to be addressed is that simply protecting an identified population may not be sufficient to sequester its genetic diversity. Thus, management of a population to maintain and possibly enhance genetic diversity is an important consideration. A serious detriment, however, is that specific knowledge is often lacking on how to manage populations effectively towards these goals. Information is needed on life history characteristics, population dynamics, population responses to environmental perturbations, gene pool responses to population changes and the effects of gene pool changes on the population dynamics.

The perennial dioecious lily, *Chamaelirium luteum* (devil's bit or blazing star) is a species that lends itself to the study of such considerations. It is fairly widespread in the southeastern United States, where it occurs in rich forest habitats. It is rare in northeastern North America, where it is at the periphery of its range, listed as an endangered species in Massachusetts and known from only a few sites in Connecticut; it is rare in northern Pennsylvania and New Jersey (Carrolan, 1982). Only eleven extant populations have been documented in New York State, and six of these are within a seven mile long area in southeastern Dutchess County, about 90 miles north of New York City (Figure 1). These populations are relicts, persisting from a more widespread distribution, as evidenced by over 60 historical site records in New York.

Our ability to detect unusual features of these populations and to use *Chamaelirium* as a model species is a result of extensive research on its life history, population demography, and genetics in North Carolina by Thomas Meagher (Meagher, 1978, 1982, 1986; Meagher and Antonovics, 1982a, 1982b). It is known that in *Chamaelirium* individuals are long-lived (genets persisting perhaps as long as the forest trees), requiring at least five years to reach sexual maturity; they repro-



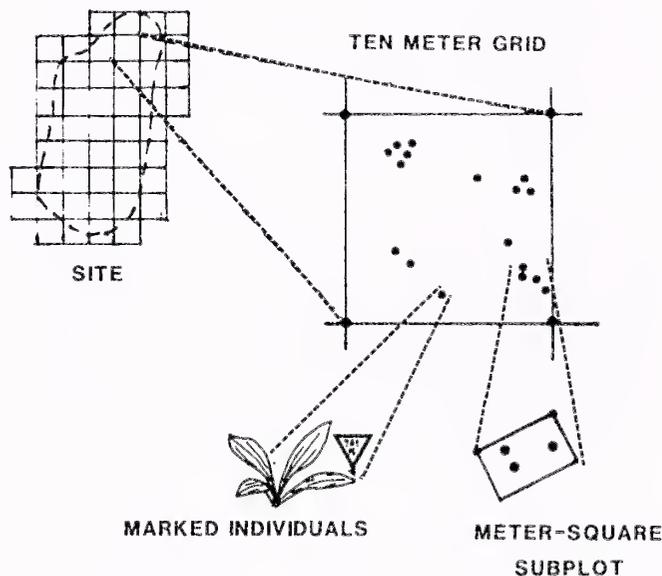
**Figure 1.** Location of the six populations of *Chamaelirium luteum* in Dutchess County, New York. (Maps reprinted with permission from Dutchess County Natural Resources Inventory, 1985)

duce at intervals of two to several years, and have very limited seed dispersal distance ( $\bar{x}=10$  meters) [Meagher, 1984; Meagher and Antonovics, 1982a, 1982b-; Meagher and Thompson, 1987], and, even within suitable habitat, they occur in island-like patches. These characteristics are typical of many endangered species such as northern monkshood, *Aconitum noveboracense* (Kuchenreuther, Cervelli, and Stearns, 1986) and heart-leaved plantain, *Plantago cordata* (Meagher, Antonovics and Primack, 1978).

Three of the six known extant Dutchess County populations are ownership-protected by The Nature Conservancy and The National Park Service (Appalachian Trail corridor), but the other three are on private land and vulnerable to destruction. One is on the site of a proposed housing development currently before the local planning board.

Four of these Dutchess County populations have been the subjects of an ongoing monitoring and research program (Carrolan, 1982; Blau and Venezia, 1983; Utter and Venezia, 1984; Utter, Napolitano, and Hurst, 1986; Hurst, 1987). Early studies were initiated on two populations by Thomas Carrolan and James Utter in 1980 for The Nature Conservancy's Pawling Nature Reserve. During the ensuing eight years the area covered at each site has increased, and two other populations have been added to the studies. The populations range in size from 1200 to 3600 individuals; two of the populations are in forest habitats, and the other two are in old fields.

The monitoring system that has evolved is three-tiered (Figure 2) based on a 10 meter, permanently staked grid that includes each population in its entirety. The first monitoring level is the mapping of all individuals in the grid. Accuracy and precision are enhanced during this tedious effort by dividing each 10 m<sup>2</sup> cell into one or two meter wide strips using tape measures, searching each strip systematically, and locating the plants on the map, using coordinates approximated from the tape measures. Repeated mapping of each site for two or three consecutive years has proven necessary because individuals can be covered with litter one year and remain undetected, yet be visible the next year. The second level involves measurement of each flowering rosette and marking it with a numbered plastic plant tag. Each indi-



1. ALL INDIVIDUALS MAPPED ON A TEN METER GRID.
2. EACH FLOWERING INDIVIDUAL MARKED WITH PLANT STAKES AND MONITORED ANNUALLY.
3. ALL INDIVIDUALS IN PERMANENT SUBPLOTS MAPPED AND MEASURED ANNUALLY.

**Figure 2.** Schematic representation of the three-tiered monitoring system developed for the *Chamaelirium luteum* study in Dutchess County N.Y.

vidual is relocated in subsequent years and its size and flowering status recorded, to provide a longitudinal history of these mature individuals. The third level involves subsampling from the population using permanently marked 1 m<sup>2</sup> area (120 cm × 80 cm) sites to assess environmental effects on flowering, seedling production, annual growth, survivorship and other population parameters. These sites were established using a stratified random design based on the density of tagged plants (those rosettes that had flowered during the previous four years). Control plots were established in areas where there had been no flowers during four previous years, and were stratified on the basis of plant density.

Our studies have shown that there are at least three differences between the Dutchess County, New York, populations and those in North Carolina studied by Meagher: (1) the optimal habitat is old-field; (2) hermaphroditic individuals often produce seeds; and (3) clonal vegetative reproduction occurs.

Flowering is our indicator of habitat quality, and flowering occurs more often in open habitats than in wooded ones. Each year a higher proportion of individuals flower in the field populations (5.1-5.9%) than in the forest populations (0.-4.1%) (Table 1). In addition, the proportion of plants flowering in the field population is nearly constant from year to year, but the proportion flowering in the forest populations are highly variable. Large fluctuation in flowering is also the general pattern for woodland, North Carolina populations, but the proportion of plants flowering (1.6-38.9%) (Meagher, 1984) is much higher than in New York woodland populations. The apparent decline of flowering in the wooded National Park Service (NPS) site after 1983 can be interpreted, using the longer run of data collected for the portion of the population initially studied in 1980. There is evidence that the decline is actually a return to steady state-conditions following a flowering peak in 1982 and 1983. The peak appears to have been a result of gypsy moth (*Porthetria dispar*) defoliation of the canopy in 1980 and 1981, which resulted in a semi-open environment. Optimal habitat is also inferred from the shorter flowering intervals of males and females in the open areas, as compared to those intervals in forested habitats. In the old-field (WM) population, 64 males have flowered three or more times, but in the forested NPS site, only one male flowered three times. At WM, 16% of 100 females have flowered twice, but none of the 43 marked females at NPS have flowered again. We

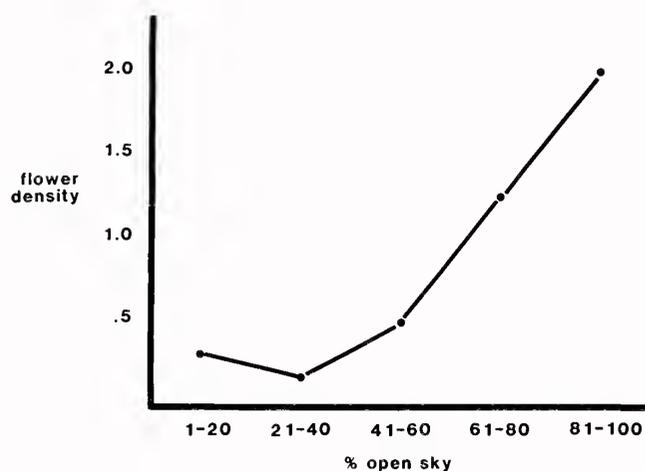


Figure 3. Average number of *Chamaelirium luteum* flowers in each m<sup>2</sup> plot in 1987 as a function of canopy cover in 1986.

therefore concluded that the more open, old-field habitat is the optimal habitat in the north, while in the south, the forest habitat is better, and *Chamaelirium* does not normally occur in old fields (Meagher, pers. comm.).

In the northeast, some predominantly male plants are hermaphroditic and produce seeds. Approximately 10% of the flowering male plants in the WM, and 2% in the NPS populations, have hermaphroditic florets on the lower portion of the raceme, and many of these develop mature ovaries with seeds. Seed set was not seen in the North Carolina populations, although some males did produce hermaphroditic florets (Meagher, pers. comm.).

Approximately 20% of the ramets in samples from our subplots were connected to another ramet by a common rhizome (Nudelman, 1987). Rhizomes had as many as eight rosettes, and one excavated rhizome was connected to another, several centimeters below the surface. Meagher (1978) excavated more than a thousand plants and found no evidence of such clonal growth in the North Carolina populations.

We suspect that these biological differences between northern and southern populations are partly due to environmental plasticity, but they are also likely to involve significant genetic differences. Environments, and physiological responses to them, represent different selective backgrounds within which genes act. Selection moves genetic systems towards the maximization of fitness within each specific environment; thus, different environments foster genetic differences (Bradshaw, 1984). Isolation of populations reduces or eliminates gene flow, which enhances such genetic differentiation through selection. Concurrently, the random changes resulting from the founder effect and genetic drift in small isolated populations can lead to additional differences between peripheral and central populations.

Reduced genetic variation may result from the reduction of both population size and gene flow, increasing the risks of extinction (Gilpin, 1987; Gilpin and Soule, 1986). To avoid this, active management has an important role. Preservation and perhaps even "enhancement" of the genetic diversity in small populations may be critical to their survival. One management option is to increase gene flow by transplanting individuals between populations, though there is a risk that locally adapted genotypes are incompatible. Alternatively, increasing: (1) the rate of reproduction, and (2) the proportion of the population reproducing, should be an effective means of countering the trend towards reduced genetic diversity, while simultaneously increasing the population size. One classic approach to enhancing population growth requires determination of the factors that limit reproduction and subsequent manipulation of them.

Based on the following observations, we hypothesized that light is the primary factor controlling reproduction in northern populations of *Chamaelirium*: (1) populations in open sites flowered more abundantly than populations in wooded sites; (2) at NPS, flowering increased after gypsy moths defoliated the trees but decreased as the canopy recovered; (3) more seedlings were observed in open sites than in wooded sites.

Canopy cover is an indirect measure of light availability, and we have analyzed it using fisheye lens (7mm) photographs of the canopy directly over our meter-square sampling plots. Photographs were analyzed by projecting them onto a standard grid of six concentric bands of equal width. Fifteen randomly located sampling points were located in each of the five innermost bands. We counted each point that was not covered by foliage to estimate the percentage of open sky over the plot (a method developed by Charles Canham, pers. comm.). The number of flowers in each subplot in 1987 was then compared to the proportion of open sky above the site in 1986. The effect of light on flow-

ering had a lag of one or two years, apparently due to the need to store sufficient energy reserves to trigger flower bud formation; the bud is formed in the summer or fall prior to flowering. Flowering increased as the canopy became more open above a threshold of approximately 40% open sky. The number of seedlings in the plot depended on proximity to recently flowering females, germination of seeds and survival of the seedlings. Seedling density was greatest between 40% and 80% open sky, suggesting an 'optimum' pattern of response. These data supported the conclusion that available light is a major causal factor in the flowering of *Chamaelirium*, and in seedling success, and we suggest that management of canopy to maintain 20 to 60% cover can help achieve the goal of increased reproductive activity.

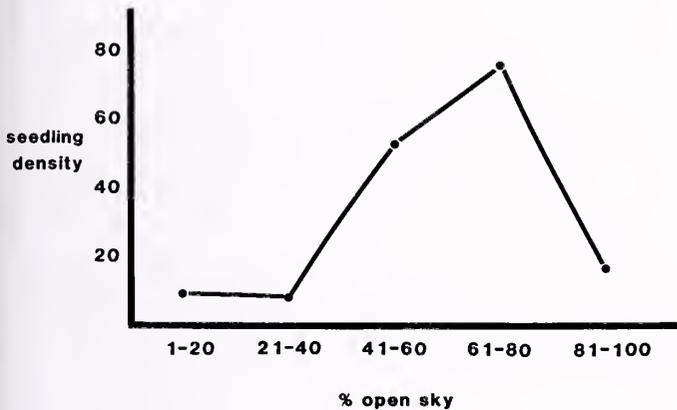


Figure 4. Average number of *Chamaelirium luteum* seedlings in each  $m^2$  plot in 1987 as a function of canopy cover in 1986.

### SUMMARY AND DISCUSSION

We argue that identification, protection, and management of isolated populations that are ecologically or geographically peripheral to the central portion of the species range, have value to long-term survival of that species. This value in preserving such populations lies primarily in the genetic diversity they contribute to the species' gene pool. Identification of peripheral and disjunct populations is proceeding via Natural Heritage Programs and other agencies involved in plant exploration, but vigilance is necessary to ensure that recently declining species be detected against a backdrop of numerous historical records. Protection of such plants and management opportunities depend upon ownership of the land on which they occur.

Rare plants are often not adequately protected by state law. In New York, for example, the current law only protects the landowner from damage or theft of listed vulnerable plants on his/her property. An amendment to this law (1989) lists rare species, but offers no further protection than was given for "exploitably vulnerable" species. It does, however, present a list of rare species to be sought out whenever an area is to be impacted by a development proposal (Richard Mitchell, New York State Botanist, pers. comm.). It is unclear whether invoking this list will only delay impacts or will truly protect the plants. Working with developers and planning boards can, of course, be productive in species protection. Clearly, acquisition of land parcels and conservation easements is the strategy most likely to succeed.

One management goal for rare plants that we suggest is to increase

the frequency of flowering and the proportion of the population actively reproducing; this should help maintain and/or increase genetic diversity within populations, as well as potentially increasing population size.

*Chamaelirium luteum* is a model example of a species in which this kind of approach can be used. It is a rare species in the northeast, whose populations studied in New York, on the periphery of its range, show characteristics that differ from those seen in populations more central to its distribution. These characteristics are likely to reflect stable, genetic differences in addition to environmentally induced modifications. Light availability, as controlled by canopy cover, appears to determine flowering frequency and survival of young plants. A management approach that maintains canopy cover between 20 and 60% is likely to provide optimum conditions for flowering and survival. Such a management plan should enhance both genetic diversity and population growth, and in this way contribute to greater long-term species security.

Populations such as those of *Chamaelirium* that we are studying could serve as good model systems for exploring management options for certain endangered species. The life history characteristics of this species are similar to those of a number of endangered species, and there are many significant management questions that might be studied effectively utilizing such populations. Questions to be asked include: what is the minimum viable population size for such a species? What are the effects of flowering-rate and population sizes on genetic diversity? A variety of such questions, and their reciprocals, could be posed and approached experimentally with minimum risk to biodiversity and with the likelihood of important contributions to scientific management of rare species.

### ACKNOWLEDGMENTS

We thank all of those who have helped with the *Chamaelirium* study, especially Thomas Carrolan, Alfred Feldman, Kathryn Venezia, Terri Blau, Kathleen Napolitano, and Eve Nudelman. Robert Zaremba of The New York Natural Heritage Program and The Nature Conservancy searched for historical populations and located two in Dutchess County. Financial support during the period of the study (1980-1987) has been provided by The Nature Conservancy, Saw Mill River Audubon Society, State University of New York at Purchase, and the Sussman Foundation. Lee Ehrman made very helpful suggestions concerning the manuscript. Some of the work presented was developed while the senior author was on leave at the Institute of Ecosystem Studies of The New York Botanical Garden.

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**Table 1.** Annual flowering of two New York *Chamaelirium luteum* populations.

Population		1982	1983	1984	1985	1986	1987	$\bar{x}$	S
WM Field (3016)*	N	174	155	174	171	179	164	164	8.6
	%	5.8	5.1	5.8	5.7	5.9	5.4	5.6	.31
WM Woods (649)*	N			3	4	5	1	3.3	1.71
	%			0.5	0.6	0.8	0.2	0.5	.25
NPS Woods N (3118)*	N		129	6	9	30	0	34.8	53.8
	%		4.1	0.2	0.3	1.0	0	1.12	1.71

\* 1984 population size

# Recovery of the Endangered Virginia Round-leaf Birch (*Betula uber*): A Decade of Effort

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**Abstract:** The Virginia round-leaf birch (*Betula uber*), rediscovered in 1975, is known from a single population in southwestern Virginia. It was officially given endangered status under the federal Endangered Species Act in 1978. A federally approved recovery plan was implemented in 1982, with the prime objective of increasing the number of individuals of round-leaf birch in the wild to a level where the species could be removed from endangered status. Top priority was given to maintenance and expansion of the natural population and establishment of additional populations in the wild. Somewhat lower priority was given to retention of existing germ plasm through cultivation, determination of systematic relationships, implementation of educational programs and searching for additional wild populations. After a decade of coordinated effort by federal, state and private agencies and institutions, as well as private landowners, the original population remains in imminent danger of extinction, due largely to acts of vandalism. However, the establishment of 20 additional populations in the same watershed where the natural population occurs, together with mass propagation and distribution of materials to arboreta and private nurseries, mitigates potential reduction of the original population. Continued recovery of round-leaf birch could be enhanced substantially by legislation prohibiting the taking of endangered plants on private lands by any individual.

Pages 185 - 188. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The Virginia round-leaf birch, *Betula uber* (Ashe) Fernald, was originally described as a variety of the common sweet birch (*B. lenta* L.) in 1918 by W. W. Ashe from trees he found growing along the banks of Dickey Creek in Smyth County, Virginia. The taxon was subsequently elevated to the species level by M. L. Fernald (1945) and transferred to the section *Humiles* because of presumed affinities to the shrub birches. When it was not collected or observed for several decades, round-leaf birch was assumed to be extinct, until it was rediscovered in 1975 along the banks of Cressy Creek, approximately 2 km from the type locality (Ogle and Mazzeo, 1976). The habitat consists of a 100 meter-wide band of highly disturbed second-growth forest along a 1 km stretch of the Cressy Creek floodplain, a site nearly surrounded by agricultural land (Sharik and Ford, 1984; Ford et al., 1983; Ogle and Mazzeo, 1976). The general consensus among botanists working with round-leaf birch is that Ashe probably erred in his original reference to Dickey Creek (Sharik and Ford, 1984).

There were an estimated 41 individuals in the natural population of round-leaf birch in 1975, including 18 reproductively mature adults and 23 subadults (seedlings and saplings). Two adults were growing on public land, while the remainder of the population was on two adjacent private land holdings. The population declined to 26 individuals by 1977 (Fig. 1). Most of the losses (12 individuals) in the subadult segment of the population were due to vandalism on private property, but they also included the transplanting of several plants by one of the private landowners (Sharik, 1980). By contrast, the loss of three individuals from the adult portion of the population on private land was due mostly to natural causes. It is worth noting that the vandalism noted

above occurred despite the fact that the federal government and private landowners had erected protective fences around their respective segments of the population in 1976 (Sharik, 1980).

Round-leaf birch recovery efforts began in earnest with the formation of the *Betula uber* Protection, Management and Research Coordinating Committee in May 1977, spearheaded by the U. S. Forest Service, Jefferson National Forest (Sharik, 1980). The committee, comprised of members from federal and state government, conservation organizations, universities, and the private sector, has met annually or semi-annually since 1977 and continues to function as an informal recovery team. The committee was aided in its efforts in 1978 and 1979 when round-leaf birch was listed as endangered under the federal Endangered Species Act of 1973 and the Virginia Endangered Plant and Insect Act of 1979, respectively. However, it is noteworthy that the "to take" clause in the federal act did not apply to endangered plants until 1982, and then only on federal lands. In contrast, the state law prohibited taking of endangered plants on both public and private lands, except by the private landowner. In 1982, the U. S. Fish and Wildlife Service approved the Virginia Round-leaf Birch Recovery Plan (Sharik, 1982), which was revised in 1986. The prime objective of this plan was to increase the number of round-leaf birches in the wild to a level where the species could be removed from endangered status—estimated at 1000 individuals in each of 10 populations. Major sub-objectives and their priority ratings, where a rating of 1 was highest, included: maintenance and expansion of the natural population (priority 1), establishment of additional populations in the wild (priority 2), continued searches for other natural populations (priority 2), determination of systematic relationships (priority 3), retention of existing germplasm through cultivation (priority 3) and implementa-

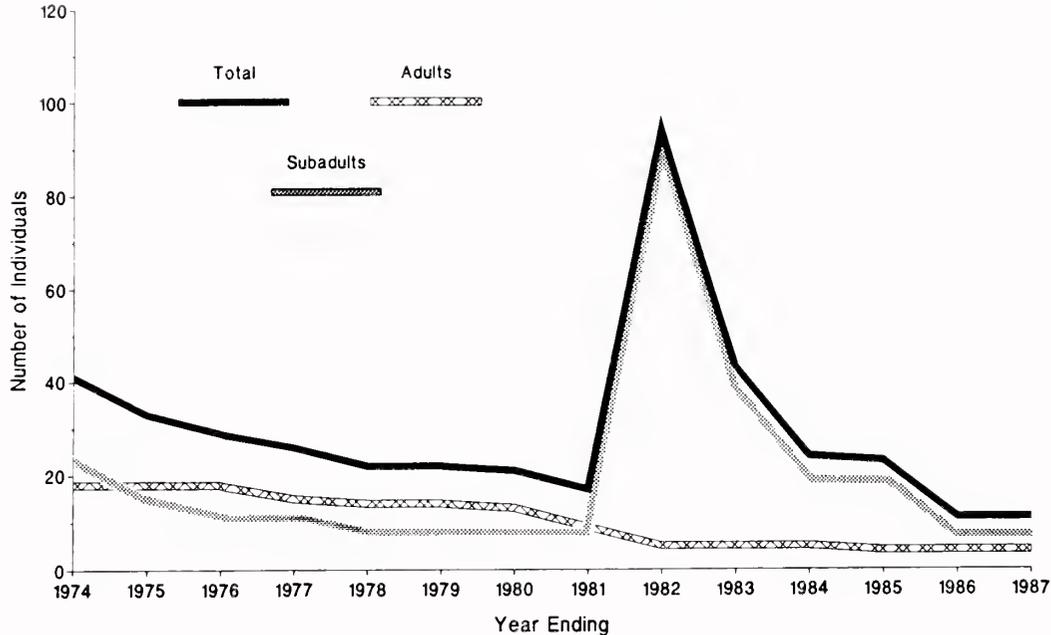


Fig. 1. Annual net changes in the Cressy Creek *Betula uber* population.

tion of educational programs (priority 4). The objective of this paper is to describe round-leaf birch recovery efforts over the past decade, and to outline future endeavors.

#### MAINTENANCE AND EXPANSION OF THE NATURAL POPULATION

Of the approximately 650 dark-barked birches (41 round-leaf birches, 610 sweet birches, and eleven yellow birches (*B. alleghaniensis* Britt.) recorded in the Cressy Creek population in 1975, about 200 have been monitored closely since 1978. This monitoring effort has been aided more recently by the development of a computerized data management system that provides annual summaries of flowering, fruiting and growth rates of individuals by species. In addition, there has been increased surveillance of the population by federal and state personnel. By 1981 the round-leaf birch population had declined to 17 individuals (9 adults and 8 subadults), the more recent decline due exclusively to natural causes (Fig. 1). Given this heavy mortality, the decision was made to implement a planned disturbance adjacent to seed sources with the intent of encouraging natural regeneration. Birches are pioneer species that often invade rapidly, following disturbance, and there was no reason to infer that round-leaf birch was an exception, given its co-occurrence with sweet and yellow birch.

The disturbance treatment was preceded by a dispersal study in 1978-1979, that provided information on the direction, distance and timing of dispersal relative to seed sources (Ford et al., 1983). In November 1981, two areas were cleared within 60 m of potential seed sources, one about 1000 m<sup>2</sup> on public land and the second about 400 m<sup>2</sup> on private land. At the end of the 1982 growing season, approximately two-thirds of the seedlings generated in these plots were birches (Sharik et al., 1989). Of the birch regeneration, about 8% of seedlings were round-leaf birch, about 92% sweet birch, and less than 1% were yellow birch. This contrasts with round-leaf birch percentages of about 6.8% of the dark-barked birches in 1975 and 4.6% in 1981. In all, 81 round-leaf birch seedlings were recorded, all on private property.

The fact that no round-leaf birch seedlings occurred in the disturbed area on public property was attributed to the absence of a pollen source for the relatively isolated round-leaf birch mother trees growing there

(Sharik et al., 1989). The 37% first-year survival of round-leaf birch seedlings in cleared areas was twice that of sweet birch seedlings. There was no difference in height of seedlings among the two taxa after two growing seasons, suggesting that fitness in round-leaf birch may be as high as that in sweet birch (Sharik et al., 1989). All of the 30 round-leaf birch seedlings remaining after the end of the second growing season (1983) were gone by the spring of 1986, the apparent result of vandalism, as whole plants (roots and shoots) were found missing.

In July 1984, The Nature Conservancy acquired through public auction 14 ha of land adjacent to the natural population and to U. S. Forest Service property. The land was in turn purchased by the U. S. Forest Service in October 1986 and managed as part of the round-leaf birch recovery program. Such land acquisitions are highly valued, given the limitations of federal and state legislation regarding protection of endangered plants on private land.

#### ESTABLISHMENT OF ADDITIONAL POPULATIONS

Given the results obtained from experiments with the natural population, the committee concluded that additional populations could be established and that they could be self-sustaining given periodic disturbance. The approach was to select 20 wooded locations in the Cressy Creek watershed where sweet birch was abundant, clear these areas and plant seedlings. Establishment of additional populations was limited to the approximately 20 km<sup>2</sup> Cressy Creek watershed because round-leaf birch was not known outside this area and because confinement to a single watershed would facilitate population maintenance. Seeds were collected from six round-leaf birch mother trees and four sweet birch mother trees within the area of the natural population, germinated in the greenhouse in 1982, and held in cultivation for two to three growing seasons before transplanting in the field in 1984 and 1985. Additional seeds were germinated in 1985 for transplanting in 1986 and 1987.

Twenty populations (five per year) were established over the four-year period, with the hope that a minimum of 10 would be self-sustaining as specified in the recovery plan. Sweet birch progeny were included in the planting to allow comparisons in growth rates and fecundity with round-leaf birch progeny. Each additional population consisted of 96 individuals established on a 3.2 x 3.2 m spacing grid. Individual

trees were fenced off from browsers, and competing vegetation was removed from around individual transplants annually. Nearly 76% of the 1,920 transplants had round leaves; 74% of the transplants were from round-leaf birch mother trees.

At the end of the 1987 growing season, survival averaged 81% for all populations, and ranged from 21-99%. There was no difference in survival associated with the species of the mother tree, leaf shape of transplants, or their location within plots. Likewise, growth in height was not affected by the species of the mother tree or by the leaf shape of transplants. There were significant differences in height growth among populations and among locations within populations, however, with border trees growing slower than interior trees. The slower growth of birches in the outside rows in each plot, compared to trees in the interior positions, led to a decision in late 1987 to remove competing vegetation from the forests bordering the 10 populations established in 1984 and 1985. The width of removal averaged 5-10 m, but varied as a function of plot orientation and height of the surrounding vegetation. All stumps were injected with a systemic herbicide to prevent resprouting. The 10 remaining populations were similarly treated in 1988 and 1989 at the rate of five per year.

### CONTINUED SEARCHES FOR OTHER NATURAL POPULATIONS

A coordinated search for other natural populations of round-leaf birch was conducted in 1977, utilizing personnel from the U. S. Forest Service trained in identification of the species. Searches in Cressy Creek and other watersheds over a three-county area failed to reveal other locations where round-leaf birch grows. No further searches have been scheduled.

### DETERMINATION OF SYSTEMATIC RELATIONSHIPS

About 10,000 progeny of round-leaf birch and sweet birch, originating in 1982 from the ten open-pollinated mother trees noted above, were analyzed for morphological differences. Over 99% of the progeny exhibited either round leaves, typical of round-leaf birch or ovate leaves, characteristic of sweet birch. Less than 1% of the seedlings had leaves intermediate between the two taxa. Round-leaf mother trees averaged 3.0% round-leaved progeny and ranged from 0-11.7%. In contrast, sweet birch mother trees produced an average of 0.3% round-leaved progeny, ranging from 0-0.9%. Another sweet birch, assessed for the first time in 1985, produced 4.4% round-leaved progeny.

The frequency of round-leaved progeny among round-leaf birch mother trees was inversely related to distance to the closest pollen source. The observations reported above lead to the hypothesis that the round leaf shape is a case of simple Mendelian inheritance, resulting from the mutation of a single gene in the sweet birch population. The hypothesis specifies that the ovate leaf shape characteristic of sweet birch exhibits complete dominance over the round leaf shape, with some modifier effects to account for the low frequency of progeny with intermediate leaf shapes. This hypothesis has been tested recently with controlled crosses and we have so far uncovered no reason to reject it (Feret and Sharik, unpublished data).

A breeding orchard, established in 1985, contains trees with genotypes from several species of birches. When the trees become fertile, crosses will be made between these genotypes to provide a better understanding of the genetic and evolutionary nature of round-leaf birch. The situation with round-leaf birch appears very similar to that with simple-leaved white ashes (*Fraxinus americana* L.) reported from

northern Lower Michigan (Wagner et al., 1988). Most plant taxonomists would treat such local variants of widespread species as forms of these species, and we would agree with such a treatment in the case of the Virginia round-leaf birch. However, we also see no reason to discontinue conservation efforts for this taxon. From a legislative standpoint, the federal Endangered Species Act of 1973 (Sec. 3-11) was designed to accommodate infraspecific taxa as threatened or endangered, at least to the level of variety. Furthermore, it is scientifically interesting to examine the role that single-gene mutations play in structuring plant populations and communities, and in the evolution of higher plants (Wagner et al., 1988). Hilu (1983), based on a review of the horticultural literature, argued that we have substantially underestimated the magnitude of changes in higher plants as a result of such mutations.

### RETENTION OF GERMPLASM THROUGH CULTIVATION

Efforts aimed at retention of round-leaf birch germplasm began in 1975 when the U. S. National Arboretum transplanted three seedlings from the wild to their grounds in Washington, D.C. Approximately 50 plants were produced from rooted cuttings and grafted scions of plants of those three genotypes. The resulting plants were distributed to over a dozen arboreta, botanical gardens and nurseries in the United States and Europe (Sharik, 1980). A mass propagation effort, initiated in 1988 at Virginia Polytechnic Institute and State University and funded by the U. S. Fish and Wildlife Service and the Commonwealth of Virginia, was designed initially to provide materials to arboreta and botanical gardens for teaching and research. The plan was to produce up to 2,000 seedlings from crosses of selected genotypes.

The committee is now developing a policy for distribution of round-leaf birches under the guidelines of the Virginia Agricultural Experiment Station with regard to the release of plant materials. The current plan is to initially provide three plants to interested organizations in the spring of 1989. Additional individuals will be available upon special request, assuming an adequate supply of plants. Availability of plants will be announced in the journal of the American Association of Arboreta and Botanical Gardens. Recipients will be asked for a small fee to cover costs of packing and mailing, and will be required to sign a waiver that they will not sell the plants received or their offspring. Once requests for round-leaf birches from botanical gardens and arboreta have been filled, plants will then be made available to the general public. It is likely that this aspect of the propagation and distribution program will be handled by private nurseries, though details have not yet been worked out.

### IMPLEMENTATION OF EDUCATIONAL PROGRAMS

Early in the round-leaf birch recovery effort, the decision was made to allow the public access to the trees occurring naturally on public land. This was seen as a way of increasing awareness of endangered species in general and of minimizing human impact on the majority of the natural population of round-leaf birch located on private property. Thus, a sign was erected by the U. S. Forest Service, giving the location of the largest round-leaf birch in the population. A ramp provided a close-up view of the tree, which was enclosed by a chain-link fence. A slide-tape program was developed for viewing at the Forest Service interpretive center, telling the round-leaf birch story from its discovery through current recovery efforts. It is not clear what impact this educational thrust has had on the public's impressions of endangered species in general or of round-leaf birch in particular. The program did not

eliminate vandalism, although it is conceivable that it may have reduced it.

### SUMMARY AND CONCLUSIONS

Despite a decade of coordinated effort by federal, state and private agencies and institutions, as well as private landowners, the single wild population of the Virginia round-leaf birch remains in imminent danger of extinction, due largely to acts of vandalism. However, the establishment of 20 additional populations, together with mass propagation and distribution of plants to arboreta, botanical gardens, private nurseries and the general public mitigates loss of this rare birch. The continued recovery of round-leaf birch would be enhanced by state and federal legislation which prohibits taking of endangered plants on private land by any individual, including the landowner.

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# The Establishment of a New Population of *Pediocactus knowltonii*: Third Year Assessment

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**Abstract:** An endangered plant, *Pediocactus knowltonii* (Knowlton cactus), is known from only one site in the Four Corners area of New Mexico. Since 1985, the U.S. Fish and Wildlife Service has collaborated with the state of New Mexico to establish a transplanted population of this rare species. Cuttings were obtained from the type locality, rooted in a greenhouse, and planted into a new site. In addition to the cuttings, a seed plot was established in 1987. Data on survival, reproductive activity, growth, and seed germination are being collected semi-annually. For comparative purposes, monitoring plots have been set up at the type locality to document population trends. Information obtained from these studies will be used to assess recovery techniques and as baseline biological data on the species.

Pages 189 - 193. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

*Pediocactus knowltonii* L. Benson was listed as Endangered on November 28, 1979 (U.S. Fish and Wildlife Service, 1979). This rare cactus occurs at only one locality on private land in northwestern New Mexico at the eastern edge of the Colorado Plateau. Collection and habitat destruction from oil and gas development are the primary threats to the species.

To ensure survival and conservation of *Pediocactus knowltonii*, many agencies and individuals are collaborating on actions directed toward recovery of the species. The majority of the work is being conducted by the New Mexico Department of Energy, Minerals, and Natural Resources under a Section 6 (Endangered Species Act) agreement with the U.S. Fish and Wildlife Service (FWS); however, others involved in the recovery efforts include: The Nature Conservancy (TNC), Public Service Company of New Mexico (PNM), The Arboretum at Flagstaff, Bureau of Land Management (BLM), Bureau of Reclamation (BR), University of New Mexico (UNM) personnel, and private individuals.

In an effort to protect the only known locality of this plant, PNM donated 25 acres of land surrounding the population to TNC in 1983. The Nature Conservancy raised \$5,000 from private sources for fencing to protect the population from livestock and to deter collectors. The fence was constructed in May 1984.

The population covers an area of less than 15 acres, with most of the estimated 9,000 plants occurring within less than 2.5 acres. The plants grow on the top and slopes of a single hill composed of Tertiary alluvial deposits, at an elevation of 1,890 meters. The dominant woody species include *Pinus edulis* Englm. (pinyon pine), *Juniperus osteosperma* (Torr.) Little (Utah juniper), and *Artemisia tridentata* Nutt. (big sagebrush).

In March 1985, a recovery plan was developed for *Pediocactus knowltonii* (U.S. Fish and Wildlife Service, 1985). With only one viable population, and that population well known to cactus collectors, the survival of this species is tenuous. Reintroduction and monitoring programs were designed as part of the recovery process to ensure the continuation of the species and to alleviate potential threats to the population.

## METHODS AND MATERIALS

Several areas on federal and New Mexico state lands were surveyed to identify an adequate transplant site. Criteria for site selection were established prior to the evaluation of potential sites. A reintroduction site should be located within the probable historic range relative to the type locality; it should be at approximately the same elevation, and have topographic features and soil structure similar to those of the type locality. To curtail visits by cactus collectors, the site should be inaccessible to vehicles, and the site must also be on federally administered land so that the Endangered Species Act is enforceable. Because oil and gas development occurs throughout the historic range of *Pediocactus knowltonii*, the site needs to be fully protected from these activities as well. Both BLM and BR advised FWS on appropriate areas and reviewed site selection.

The area selected duplicates most of the environmental parameters that exist at the type locality. The site is on federal land within five miles of the type locality and access is restricted except by hiking. The elevation at the site ranges from 1,890-1,900 meters, and the site exhibits similar soil structure and cobble density to the type locality. Although it is somewhat drier than the type locality, the same dominant perennial species occur at the reintroduction site.

The transplant program began in May 1985, when 250 cuttings were taken from wild plants at the type locality. For a more detailed description of the techniques, see Olwell *et al.* (1987). Of the 250 cuttings taken, 240 (96%) developed roots in the greenhouse. Information on survival and response of donor plants was not recorded, although subsequent site visits indicate that no damage to the population occurred.

Plants designated "Group A" (103 individuals) were transplanted onto the reintroduction site in fall 1985. Each plant was tagged, watered, and the stem diameter recorded. In spring of 1986, 47 additional plants (Group B) were transplanted and each plant received the same treatment as those planted in fall of 1985. The plot was visited each fall and spring during 1986, 1987 and 1988.

In September 1987, a seed reintroduction plot was established about 200 m downhill from the transplant plot. The seed reintroduction site was selected because it has the same favorable exposure, elevation, and habitat as the transplant site, although it is separated by a small drainage corridor. The seed plot is arranged in a 10x10 m grid, with its principal axis aligned at magnetic 270 degrees. Seeds were planted every meter north to south and east to west.

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The seeds used in this effort were acquired from the original transplant cuttings that were collected at the Knowlton cactus type locality in 1985. Not all of the plants were used for transplanting, and the remainder are presently being maintained at the Mesa Gardens Greenhouse in Belen, New Mexico. The original collection of these parent plants represented individuals from all microhabitats of the type locality. The introduced seeds are from a cross-section of parent plants, and they should represent most of the genetic diversity of the population at the type locality.

At each of the grid points of the seed plot, a nail and an aluminum tag were installed to identify the grid point precisely. Utilizing each grid point as a subdatum, a wire mesh template aligned to a north axis was utilized to determine the precise spots for seed planting. The template was of galvanized mesh with 1 cm square apertures cut into it, so that, when properly situated at a grid point, it would allow planting of seeds at three predetermined locations. These three locations were situated 10 cm from the grid point along the north, south, and west axes respectively. Steve Brack (pers. comm., 1988) found that the germination rate for Knowlton cactus exceeds 75%. Taking this germination rate into account, two seeds were planted at each location to ensure that at least one would sprout. The three planting sites at each grid point were chosen to test germination success when seeds are planted at different depths. At the south axis location, seeds were left on the surface and lightly covered with a coating of fine soil. Along the west axis, the seeds were planted at 0.5 cm depth, and at the north axis, the seeds were planted 1 cm below the surface. To ensure accuracy at these various depths, a specialized seed planting tool was developed. This tool consisted of a stainless steel bar with 0.5 cm calibrations on its long axis. The bar was then inserted into a large metal disk that was designed to slide up and down its length with the capability to be tightened and locked into specific positions for desired depths. The bar acted like a soil drill, and the depth that it penetrated was determined by the calibrated position into which the metal disk was locked.

The initial fall, 1987, seed planting consisted of placing seeds at 48 grid point locations. In all, 288 seeds were planted at that time. Seeds

were planted again at 32 grid points in May, 1988. The procedure was identical at all the grid points, and a total of 480 seeds have been planted thus far. The seed grid will be read every spring and fall. Data from spring and fall plantings will be compared, and the success of planting at different depths will be assessed. No germination of those seeds planted in September 1987 was found in May 1988. Eventually, the success of establishment of a new population by seed will be compared to that of the transplanted population.

For comparative purposes, 24 monitoring plots 10 m square were established at the type locality. Plots were set up depending upon aspect, position on hill, soil type, and associated vegetation type. Each *Pediocactus knowltonii* plant within the plots was identified, and stem diameter, number of buds, flowers or fruits were recorded. These plots have been studied every spring since 1986.

## RESULTS AND DISCUSSION

The growth patterns of introduced cuttings were distinctly different, depending on time of transplant: fall-planted (Group A) versus spring-planted (Group B), (Table 1). The stems of Group A shrunk from spring to fall 1986. In contrast, Group B stems increased in diameter from spring to fall 1986; however, these differences appear to be smoothing out over time. From fall 1986 to the present, the pattern of growth and the total growth since planting are similar for both groups. The initial growth spurts for both groups are most likely due to the application of fertilizer in the greenhouse before transplanting. Overall, plants of Group B grew more from the initial planting to the time of last measurement. These growth patterns are readily apparent in Figure 1. The straight line in Figure 1 indicates overall average growth for all plants. Since the time these individuals were transplanted, there has been an average growth of 5.0 mm. Although this may not appear significant, this is appreciable growth for plants that average only 13.1 mm in size.

The graph in Figure 2 illustrates differences in average growth along the slope of the hill. The top of the hill is at row 1 and the base at row 10. The majority of growth (78%) occurred at mid slope in rows 3-6

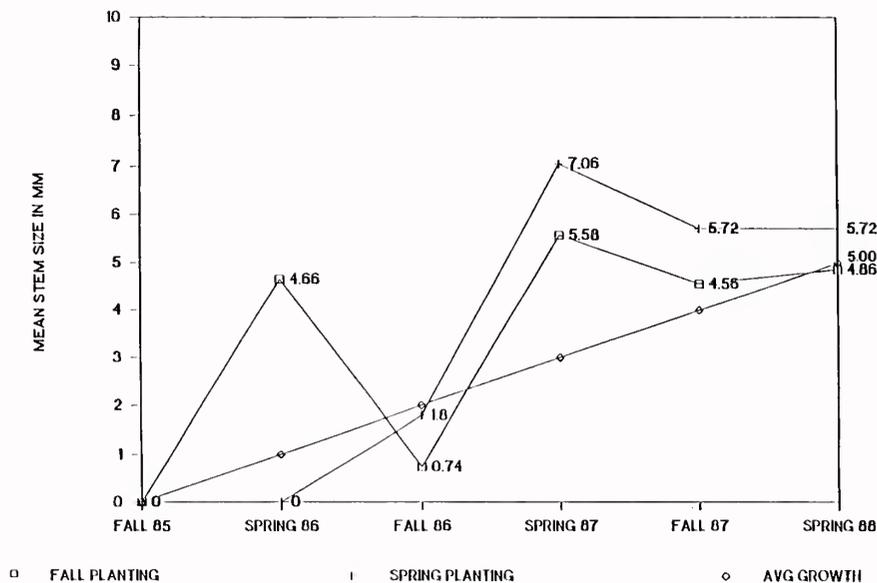


Figure 1. Average change in stem diameter for *Pediocactus knowltonii*

and the least amount of growth took place at the base of the hill in rows 8-10.

Until spring 1988, mortality was surprisingly low, with an overall loss of only 8% of the plants. Group A had a mortality of 4%, compared to Group B, which had a mortality of 17%. Considering this difference in mortality, it was believed that fall was the best time to transplant Knowlton cactus. However, in spring of 1988 we had the highest loss of plants ever and most of the loss was in Group A which had been planted in the fall (Table 2). With that eventuality taken into account, it appears that both groups have had similar mortality.

**Table 1.** Change in average stem diameter (mm) of transplants at reintroduction site.

	FALL 1985	SPRING 1986	FALL 1986	SPRING 1987	FALL 1987	SPRING 1988	TOTAL GROWTH
Group A	0	+4.66	-3.92	+4.84	-1.02	+0.3	+4.86
Group B		0	+1.84	+5.22	-1.34	0	+5.72

The graph depicting mortality along the slope of the hill (Figure 3) shows that 52% of the dead plants occurred in rows 7-10 at the base of the hill. This is not surprising, considering that these rows contain most of the trees and shrubs in the plot. Knowlton cactus plants have been found buried in as much as 5 cm of litter from surrounding plants. In addition, sediments from up slope accumulate around the trees and shrubs and tend to bury *Pediocactus*. Other factors of mortality appear to be rodent and insect damage and fungal diseases. As of spring, 1988, 25 plants of 151 (17%) had died.

The percentage of plants at the reintroduction site with flowers or fruits was down considerably from 52 per cent in 1987 to 23 per cent in 1988 (Table 3). The high mortality may play a part in this; however, there were many aborted flower buds observed in 1988. This may be

**Table 2.** Numbers of dead plants and percent mortality at reintroduction site.

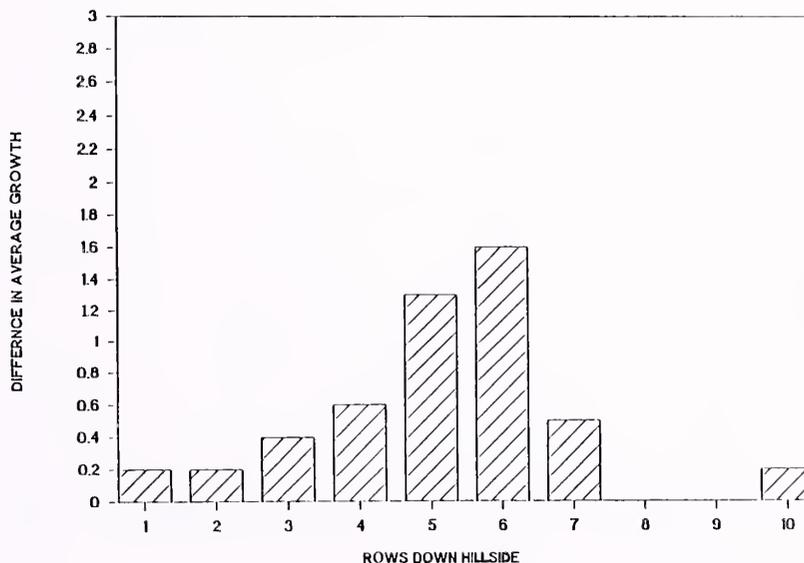
	FALL 1985	SPRING 1986	FALL 1986	SPRING 1987	FALL 1987	SPRING 1988	PER CENT (%) MORTALITY
Group A	0	0	2	2	0	12	16
Group B		0	4	3	1	1	19
All Plants	0	0	6	5	1	13	17

**Table 3.** Plants with flowers and fruits.

	NEW POPULATION	TYPE LOCALITY
1986	59%	34%
1987	52%	31%
1988	23%	21%

due to a period of warm weather at the end of March that may have induced the buds to set. The warm weather was followed by a cold spell in April that may have damaged the buds. This same observation was made at the type locality. As suggested in Table 3, this is the first year that the number of plants with flowers or fruits at the reintroduction site is very close to that of the type locality. The plants at the reintroduction site were fertilized prior to transplanting, and perhaps the residual effects from fertilizer were the reason flowering and fruiting were high during the first two years.

Figure 4 illustrates reproductive activity in response to location of the transplants on the hill. Sixty-nine per cent of those plants flowering or fruiting occur in rows 3-6, which is also where the least amount of



**Figure 2.** Differences in average growth versus slope of hill.

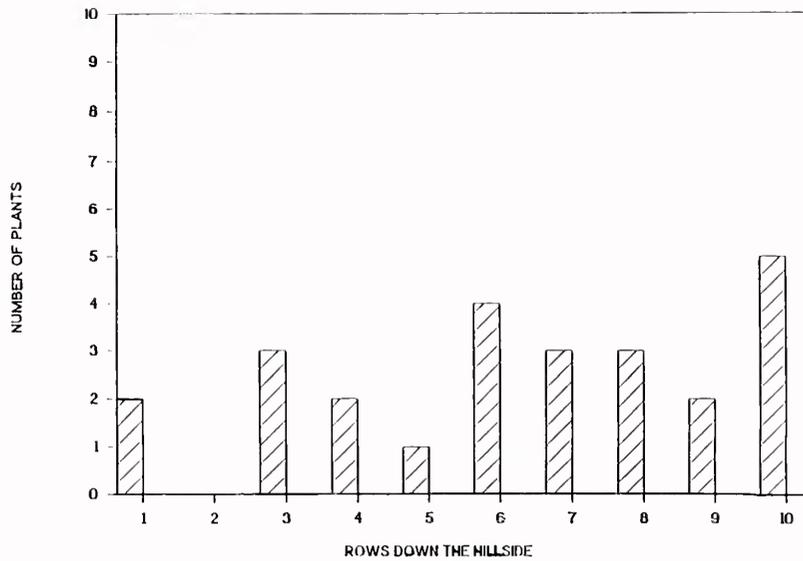


Figure 3. Mortality versus slope of hillside.

mortality and the largest amount of growth occurred. From these data, it appears that the mid-slope location is the best place to transplant *Pediocactus knowltonii*.

The monitoring plots at the type locality are especially valuable in supplying data for comparison with those from the new population. While the reproductive activity has been higher in the new population, trends have been the same at both locations. An unknown fungus was observed at both plots during 1988. These observations suggest that the new population is located in a place that is similar in critical ways to that of the type locality. With further years of data gathering, the comparisons will be increasingly useful. The amount of reproductive

activity, rate of growth, rate of recruitment and other population characteristics will be compared, and the relative success of the new population more accurately assessed.

The transplanted population has been in place now since 1985. The survival rate of 83% is encouraging, as is the average growth of 5 mm for each plant. Flowering and fruiting are occurring, suggesting a favorable environment and the presence of pollinators. We have not seen any juveniles, and we cannot say the project is biologically successful with any certainty until we see a reproducing population. Preliminary data, however, give us reason to have hope that juveniles will appear.

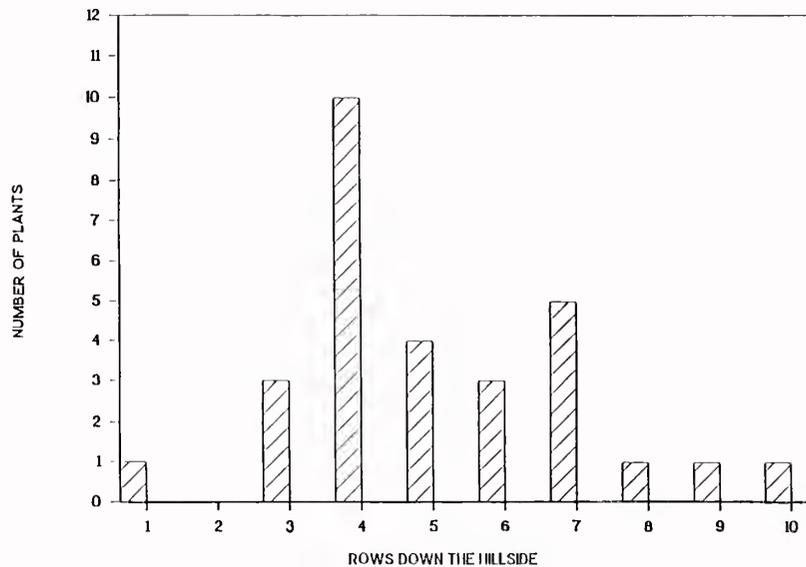


Figure 4. Reproduction versus slope of hillside.

### ACKNOWLEDGMENTS

We would like to thank The Nature Conservancy for allowing us to remove cuttings from the type locality, and the New Mexico State Parks Division for their logistical assistance.

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# Biological Considerations in the Management of Temperate Terrestrial Orchid Habitats

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**Abstract:** In the choice of natural areas for preservation, the rarity of orchid species frequently becomes a consideration, and management should reflect their needs. Unfortunately, management practices are often undertaken without knowledge of certain biological features of these plants, and populations may be easily destroyed by misguided efforts. Important aspects to be considered include a growth pattern which precludes, until the following year, the regeneration of shoots lost through accident or design, and the intrinsic colonizing nature of many orchid species.

Pages 194 - 196. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The immense popular appeal of orchids began over a century ago with the first blooming in a British hothouse of a tropical American *Cattleya*. Immediately orchids gained a reputation for exotic beauty and remote, mysterious origins. Increased horticultural and scientific experience with them only increased this perception, since the plants were characteristically very difficult to cultivate, and seed germination was nearly unknown. Orchids, especially terrestrials of temperate regions, came to be known as plants with very precise, but largely unknown, habitat requirements, these being met only in the most pristine (read "remote and mysterious") of sites. Hence, the general rarity of orchids was explained, if not understood, by rarity of suitable habitats. In some cases this reputation was deserved: the habitats of certain species are restricted, unusual, and often fragile. Perhaps the earliest tangible indication that the popular notion was in some cases incorrect, is found in a brief note by Gleason (1909), who reported the discovery of *Liparis lilifolia* in a plantation on the University of Illinois campus. At that time the species was thought to be one of the rarest in eastern North America, but the significance of Gleason's discovery went unappreciated. Indeed, it was really not until the publication of Case's *Orchids of the Western Great Lakes Region* (Case, 1964) that the occurrence of a number of species of orchids in disturbed sites and successional communities came to be recognized as characteristic.

In my own subsequent work in Illinois (Sheviak, 1974), I found that a substantial number of orchid species occurring in the state were truly colonizers, and that a number of species could be found with considerable frequency in a variety of disturbed habitats. These observations are consistent with those of others (Case, 1964, 1987; Stuckey, 1967). In the case of *Liparis lilifolia*, I found the natural habitats of the species to be rather open areas within otherwise continuous forest cover. I could identify only two such communities: the open, often scrubby forest at the rim of ravines and windthrow openings in dense forest. Populations in the former type of site can remain quite stable because the community is physiographically determined. In windthrows, however, *Liparis* seeds into the area within a few years of initial disturbance, increases in numbers for a time, then dwindles and finally dies out as the canopy closes again. The principal habitat of the species in Illinois and elsewhere in the region, however, is in old fields and abandoned pastures which have grown up to thickets of *Crataegus*, sometimes with a mixture of saplings of forest trees. Here

the species occurs in great abundance and with a high frequency. Other orchids that typically share this habitat include *Galearis spectabilis*, a forest species of spotty occurrence across much of its range, but rather frequent in successional thickets in Illinois and Iowa; *Corallorhiza odontorhiza*, a leafless, saprophytic plant that would at first seem an unlikely colonizer of disturbed mineral soils; *Spiranthes ovalis*, another plant originally exceedingly rare throughout its range but now locally abundant in these successional communities; and *S. cernua*. A number of other species could be mentioned, but the ones cited here are actually predictable components of this disturbed community type.

This association of orchids in old field and pasture thickets is not an isolated example. Throughout the East, portions of the Midwest and South, and the montane West, *Spiranthes* species are characteristic colonizers of raw soils in excavations such as borrow pits and roadcuts. On one roadcut bank in the southern Adirondacks, I have repeatedly viewed an incredible density of orchids, such that much of the green color evident on the bank, as viewed from the road, is contributed by the leaves of *Spiranthes*. Five species of this genus occupy the bank, together with three of *Platanthera*. A colleague estimated that, at one time, 90,000 *Spiranthes* were in bloom in an area about 80 meters long and at most 10 meters in depth. This was one day out of a five week period of bloom, in which species and individuals bloomed sequentially. At this site, a coarse soil and slow but steady rate of erosion may favor the persistence the orchids. Over time, the ditch at the base of the slope fills with sediment, and the erosion of the slope lessens. However, the ditch is then dredged and significant erosion of the slope resumed. *Spiranthes* plants also are favored by mowing, and they frequently become abundant on roadsides. In the prairies, mowing of prairie hay meadows similarly promotes the development of large populations. Other factors favoring their presence are fire and grazing.

Truly extreme habitats can be found, such as the edge of railroad ballast, where herbicide treatment retards competition. At one such site in Illinois, I found *Platanthera hyperborea* occupying a narrow zone between the area subjected to lethal spray dosage (some plants had been killed) and the untreated, dense vegetation of the right of way. Like *Spiranthes*, *Platanthera* species are frequent on roadsides and excavations in many areas. The inconspicuous nature of the flowers of many species is all that prevents their detection from passing cars.

At this point, I suspect that the reader might be thinking, "Yes, but these are just little green and white things, not *real* orchids." On the contrary, the intrinsic colonizing behavior cuts across the breadth of our orchid flora. In recent years *Platanthera leucophaea*, a rare plant of the eastern prairies and Great Lakes region fens and one of the most striking of our species, has appeared in considerable abundance in a

number of disturbed sites, such as old fields. In the Great Lakes region and the Northeast, *P. psycodes* is commonly seen in wet roadside ditches, as is *P. ciliaris* in the Southeast. In the West great masses of *Epipactis gigantea* develop on roadcut seepage banks, sometimes to the exclusion of all other vegetation. *Cypripedium* similarly displays colonizing potential. Again, in the Midwest, successional, open woodland, developing on old fields and abandoned pastures, is sometimes colonized by *C. calceolus* var. *pubescens*. In the Great Lakes region and on the northern prairies, roadside ditches frequently support *C. calceolus* var. *pubescens* or *C. candidum*. Throughout the northeast *C. acaule* is a characteristic plant of white pine stands established naturally in old fields, pastures, excavations, and other disturbed sites or planted in plantations. Because of the present abundance of such communities in the Northeast today, *C. acaule* probably is more abundant now than it was at the time of European settlement. Indeed, it is likely that, within favorable portions of their ranges and with appropriate soil and moisture conditions, most species of orchids might act as colonizers of disturbed sites.

The message here is simple, but its application is not necessarily straight-forward. Many orchids are colonizers. They are not successful competitors, but they often become established in disturbed sites that are relatively open. Large populations may develop if conditions are favorable, but if successional processes lead to development of dense vegetation, the orchids will diminish in numbers and ultimately disappear. If, on the other hand, succession is arrested, for instance by erosion or mowing of a roadcut bank, or if a natural community is maintained in an unnaturally open condition, such as by thinning of a pine stand or mowing of a prairie, or if litter accumulation is reduced by burning a prairie or upland woods, then the populations of orchids in those sites may be maintained indefinitely. This topic is discussed very cogently and with numerous examples in Case's new revised edition (Case, 1987).

Orchid populations, then, are often ephemeral in nature; a species appears, increases, dwindles and disappears, but in the interim it has often colonized another area. This colonizing habit is made possible by the nature of orchid seeds, which are minute, dust-like, and very buoyant. Produced in prodigious quantities (e.g., commonly over 100,000 per plant per year in many *Spiranthes*), they are widely dispersed by the slightest breeze. This mobility has a price, however: the seeds are virtually devoid of food reserves, and are wholly dependant on mycorrhizal fungi for germination and establishment. Even as mature plants, orchids have greatly reduced root systems and evidently require mycorrhizae for proper water uptake and at least some nutrition. An understanding of orchid mycorrhizal relationships is only very slowly developing, but it appears that they are, at best, tenuously balanced interactions that are easily upset (Sheviak, 1983). The relationship between fungus and orchid is dynamic. Any change in the physical or biological environment, such as those brought about by succession, soil stabilization, or litter accumulation or removal, is likely to affect the orchid or the fungus, with the result that the orchid population responds in one way or another. Mere preservation of habitat, therefore, is often not enough. Indeed, if the nature of the habitat is not understood, so-called protection may very well be the wrong thing to do. Often the presence of orchids is cited among the reasons why an area should be preserved. I won't argue with that, but, too often, if the site is not in a stable climax condition, the perception is often that the orchids are "just hanging on." The area is then protected, the cattle removed, the mowing stopped or timber management halted, and the orchids disappear. "Too bad, we didn't get them in time." The real cause of the extirpation is not realized. The culprit is the removal of

unintended management, or, rather, the introduction of a management regime that does not include appropriate factors for disturbance.

Unfortunately, the management technique to employ in a particular situation may not be obvious. It will vary with the site, past management in the area, and the species under consideration (see Case, 1987). If the population is strong under a grazing regime or with mowing or selective timber harvest, then consideration should be given to continuing these practices. If the plants are limited to erosion scars or artificial sites, repeated excavation may be necessary to maintain populations. In the absence of previous formal management of a site, an appropriate course of action is often unclear.

If monitoring of a population discloses a steady decline, consideration should be given to introducing specific types of disturbance aimed at opening up the cover. Methods could be chosen from those suggested in the orchid literature as favorable to the species in question. If at all possible, management should be introduced in a formal experimental format and the results published. Unfortunately, much of what is known about orchid management has been learned on an *ad hoc* basis. At present, I know of only a single formal experimental study for which the results were published, and this over forty years ago! In Wisconsin, Curtis (1946) investigated mowing of sedge meadows as a means of retarding woody invasion and promoting the perpetuation and increase of *Cypripedium candidum*. To my knowledge, the rest of the available information on management of orchid populations is hypothetical, anecdotal, or informally obtained in the course of other routine management programs.

Given the lack of strict guidelines for the management of orchid populations, I think it is important to point out a particular feature of the growth of these plants which must be considered when any management activity is planned. This has been known in horticultural circles since the last century, but for the most part it has not been appreciated by botanists or managers. As with most perennials, spring growth of orchids arises from overwintered buds produced the preceding growing season. Unlike most other plants, however, if growth is destroyed by late frost, foraging animals, disease, accident, or ill-advised management practices, an orchid cannot replace the lost tissues until the following year. Although dormant buds may be present, they will not initiate growth. The root system will remain, and a new bud may form, or a dormant bud enlarge, but at best the plant will suffer a major setback, and it may die. *Cypripedium* plants that lose their growth before midsummer will commonly appear the next year but will not bloom (Whitlow, 1983). Depending on how severely depleted were their energy reserves, they may require two or more subsequent vegetative seasons before blooming (Case, 1987; Whitlow, 1983). Plants in other genera may respond similarly, but many are less resilient and the result of such damage is often death. My own monitoring of individuals of *Platanthera hyperborea*, *P. ciliaris*, *P. psycodes*, and *P. peramoena*, following destruction of the shoot by animals, accident, or collection for the herbarium indicates that the usual result of shoot loss before and during bloom is death of the plant. *Platanthera leucophaea* and *P. praeclara*, however, commonly survive such treatment (Bowles, 1983). The different sensitivities of these related species may be due to the timing of the formation of the subsequent season's buds. Bowles (pers. comm.) observed that, at the time of bloom, the development of buds of *P. leucophaea* and *P. praeclara* appears to be markedly more advanced than in related species. Predictions regarding the responses of particular species are consequently difficult. Recovery may furthermore be aided by mycorrhizae, but the tenuous nature of the balance between fungus and orchid makes this very uncertain. The shock of the loss of a shoot may upset

the balance between plant and fungus and lead to pathological destruction of the plant by the fungus.

It is vital to understand the pattern of growth of any orchids which are to be managed, either as direct target organisms or merely incidentally, as part of a community. A population can easily be destroyed following only a few seasons of misdirected management practices. Furthermore, not all orchid species at a site will necessarily respond to a particular management practice in the same manner, and some difficult choices may need to be made.

In summary, when areas are preserved or management is to be undertaken or altered, it is essential that orchid populations be given special attention to insure that needed management is received and that whatever management is undertaken on a site be conducted in a manner consistent with the orchids' cycle of growth.

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## **Chapter 6.**

### **ENDANGERED BRYOPHYTES**



# Why Rare and Endangered Bryophytes?

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**Abstract:** Rare and endangered bryophytes should be recognized and preserved for the same reasons as other organisms: they are unique and irreplaceable. They also differ from vascular plants enough in life history, distribution and ecology to justify separate listing. Bryophytes are probably no more ancient as a group than vascular plants, but they differ in that their haploid generation is dominant. Their evolution is much slower and distributional patterns are quite different as a result. Bryophytes typically occur in spatial or temporal niches in which vascular plants do not occur, although, in some situations, like weakly mineralized wetlands, they can dominate ecologically. Bryophytes are sources of information and aesthetic enjoyment in ways quite different from vascular plants, and, because of their habitat preferences, they sometimes indicate additional rare habitats in need of preservation.

Pages 199 - 201. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Vascular plants have been recognized on national and state lists of rare and endangered plants for some years now, but bryophytes have been almost completely ignored. Although a few state conservation groups have produced unofficial lists, I know of only two states, North Carolina and Minnesota, for which rare bryophytes are given official status. Even in these states, a total of only six species is listed, and no federal bryophyte list currently exists.

Failure to list bryophytes is an oversight that should be corrected, since in several respects they are organisms quite different from vascular plants. The basic justifications for preserving rare and endangered organisms of all types are the same, but protecting unusual bryophytes preserves different kinds of information and aesthetics than does the preservation of rare vascular plants.

Bryophytes are unique among truly terrestrial plants in having the gametophytic haploid generation as the ecological dominant. The ramifications of this are manifold. Sexual recombination can occur only in the sporophytic generation, which is parasitic on the gametophyte and essentially irrelevant ecologically except for its reproductive function. Many bryophytes largely forego sporophytic reproduction and utilize vegetative reproduction instead (Anderson, 1963; Schuster, 1966).

In a haploid organism, all genes are functionally dominant due to the hemizygous condition. Since recessive alleles cannot be carried from one haploid generation to another, the possibilities for natural selection are greatly limited. In addition, many mosses are monoicous and probably mostly self fertile, thus further limiting natural selection (Gemmell, 1950). However, Longton (1976) has noted that many bryophytes are polyploid and thus functionally diploid, partly overcoming the disadvantage of a dominant gametophytic generation by allowing recessive alleles to be carried. Dioicous mosses produce many fewer sporophytes than monoicous mosses (Rohrer, 1982), but the ecological and evolutionary significance of the differences between them remains unclear.

The fossil record indicates that bryophytes appeared about 480 million years ago, about the same time as vascular plants, but their more conservative rates of evolution make them useful in providing different and sometimes better clues to phytogeographic, paleoecological and geologic history (*cf.* Miller, 1980; Birks, 1982; Fife, 1985). Rarer and especially relict species often provide particularly valuable historical clues.

Distribution patterns of bryophytes are often significantly different from those of vascular plants. In the first place, there are many fewer species of bryophytes than vascular plants, perhaps 23,000 bryophytes

compared with 350,000 vascular plants in the world. Bryophyte species, however, are more widely distributed and thus fewer are endemic to restricted geographic areas. While the number of bryophyte species is only 6% the number of vascular plants species for the world, the corresponding figure for North America is about 20% and for Canada about 40% (Ireland *et al.*, 1980; Scoggan, 1978). Though there are many fewer species, most bryophytes are much more widely distributed than vascular plants. Schofield (1965) has noted that 47% of the bryophyte species of British Columbia also occur in Japan.

In the bryophytes, species are in some ways comparable to genera of vascular plants. Many genera but few species of vascular plants have similar amphipacific distribution ranges. It should also be noted that bryophytes are sometimes perceived as plants that, because of reproduction by spores, can spread readily over large distances and thus have somewhat capricious distribution patterns that do not particularly reflect historical migrations but chance occurrences. Although some unusual disjunct patterns do exist, bryophyte distributions quite often provide information about the past similar to that afforded by vascular plants. In tundra habitats, where the low vegetation and high winds promote long-distance dispersal, many bryophytes have circumboreal distributions, but it should also be noted that many vascular plant species are similarly distributed.

Bryophytes differ substantially from vascular plants in patterns of both diversity and abundance. Bryophytes are more predominant ecologically where climates are cooler and moister. Steere (1976) noted the great abundance of bryophytes over the moister tundra while noting their scarcity in the drier polar desert. Bryophytes, especially *Sphagnum* species, are very important in muskeg vegetation, an enormously widespread association within the boreal coniferous forest biome (Terasmae, 1973). The massive epiphytic bryophytes stands in the coniferous rain forest of the western North American coast and the so-called liverwort forest of Puerto Rico are other examples. But, in environments where moisture stress becomes greater, bryophytes virtually disappear as an ecologically significant element. In prairie and hot desert regions (Scott, 1982), for example, bryophytes generally occur only as scattered, small stands. Bryophytes also compete poorly with grasses under the moderate moisture stress of prairies, and, in the extreme heat and aridity of hot deserts, bryophyte biomass may decrease to less than 1 gm/m<sup>2</sup> (Nash *et al.*, 1977).

Patterns of diversity in bryophytes are somewhat unusual, in that bryophyte diversity does not necessarily parallel ecological prominence. If we look at mosses alone, for which somewhat better data exist than for liverworts, we find the number of species for such diverse North American areas as New York [461 spp.] (Ketchledge,

1980), North Carolina [458 spp.] (Anderson in Steere, 1978), Alberta [466 spp.] (Vitt & Horton, 1979), Illinois [391 spp.] (McKnight, 1987), Oregon [411 spp.] (Christy et al., 1983), and arctic Alaska [415 spp.] (Steere, 1978) are quite similar, though the overall abundance of mosses certainly differs greatly between the areas. All of these states, however, have at least some forest habitat.

For North American states and provinces that are relatively dry and have little or no forest, diversity parallels abundance in being low. Examples here would include Nevada [153 spp.] (Lawton, 1958; Lavin 1981), Nebraska [174 spp.] (Jacobsen, 1979; Churchill, 1982), Iowa [272 spp.] (Conard, 1956), Kansas [151 spp.] (Churchill, 1985), North Dakota [96 spp.] (Churchill, 1976; 1982). Although other factors such as habitat and substrate diversity as well as glacial history affect diversity, bryophytes are clearly less abundant and diverse in dry climates than moist ones. In this respect they differ significantly from vascular plants, which are very prominent ecologically in prairie and warm desert ecosystems.

The reasons for the geographic differences in ecological importance between bryophytes and vascular plants lie in fundamental differences in their biology. Vascular plants, by virtue of their root systems and complex conduction systems, have evolved groups well-adapted to moisture stress. Such adaptations are not particularly advantageous in dealing with cold, however, as in the tundra, where the rooting zone is greatly restricted by permafrost. The ability of bryophytes to deal with moisture stress is considerable (cf. Proctor, 1982), but they are relatively better adapted to environments in which both cold temperatures and moisture stress are important factors.

Bryophytes, especially *Sphagnum* species, also enjoy a competitive edge over vascular plants in nutrient-poor wetland habitats over a large geographic area, and in such habitats they may create even more acidic conditions, favoring their own growth (Andrus, 1986). Although bryophytes of shady habitats can have very low compensation points, these are no lower than those of vascular plants from similar habitats (Larcher, 1980). It is a combination of low compensation points and adaptation to moisture stress that allows bryophytes to survive in habitats such as forest tree trunks and shaded rock faces for which vascular plants are not well adapted.

Rare and unusual habitats often contain rare and unusual plants and animals, and preservation of the habitats will protect the organisms almost automatically. Although this is not a bad principle to follow, it must be recognized that our basic indicator of habitat-rarity is often the presence of rare and unusual species, especially vascular plants. Because of genetic and morphologic differences, rare bryophytes sometimes occur in habitats where no rare vascular plants are found. Rock outcrops, seep areas and fens can fall into this category. Thus, broadening the rare plant information base to include bryophytes will increase our ability to recognize and preserve additional, special natural areas.

The listing of rare vascular plants has been occasionally, and justifiably, criticized for drawing the attention of unscrupulous collectors. The numbers of bryologists, professional and especially amateur, are usually far too small for this to be a serious problem for mosses and liverworts. The effect of a listing would instead be a distinctly positive one, encouraging individuals to try and make new discoveries of rare bryophytes, thereby broadening the information base on these organisms and their habitats.

A final and somewhat curious point should be made. A resource cannot be protected unless its existence is known to those who could take action that might threaten it. Unusual habitats and rare organisms may be inadvertently destroyed because their existence is unknown or inadequately recorded. A personal experience is worth recounting. The

Massasauga rattlesnake is an endangered species in New York State. Recent studies of its biology have suggested that one habitat improvement technique that might increase its population would be to control burn at least part of the bog habitat in which it occurs. The site suggested for this, however, is the most southerly location known for a species of moss and also its type locality. A burn would have serious consequences for the moss population. The story has a happy ending, because a state biologist involved in protecting the snake is also a personal friend of mine. A field trip to the area located the moss so that any burning can now be arranged so as not to affect it. This case, however, was a lucky accident. Formal listing of endangered and threatened species would assure that such cases of potential, accidental destruction of rare bryophytes would be far less likely to occur.

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# Rare Species of Pennsylvania Mosses: An Assessment

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**Abstract:** In 1982, the State of Pennsylvania enacted the "Wild Resources Conservation Act," providing for the conservation of all native plants recognized as endangered, threatened or vulnerable. Although mosses are not currently included on the protected list, rare mosses might in due course be protected. A comprehensive moss checklist is not available for Pennsylvania; thus, an immediate assessment of the status of less frequently reported mosses in the state is not possible. A flexible data base, currently being prepared from literature reports and herbarium records, is providing information on infrequently reported mosses that might prove to be rare. Additional moss collecting throughout Pennsylvania is now being focused on specialized, restricted and sensitive habitats and on areas that current data show to be undercollected.

Pages 202 - 204. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Although bryological collecting in Pennsylvania spans 250 years (Moul, 1952; Buck and McLean, 1985; Manville, 1987), a statewide inventory of the mosses is not available. The works of Porter (1904), Jennings (1951), and Moul (1952) serve as the foundation for our current understanding of the taxonomy and distribution of the moss flora of Pennsylvania, although these works are limited to particular regions within the state. Porter (1904) cited many early collections that clearly emphasized eastern Pennsylvania, including those of Muhlenberg, James, Rau, Burnett and McMinn. Jennings (1951), in the second edition of his manual of western Pennsylvania mosses, provided the best source of information on that part of the state. Although Jennings manual supplied analytical keys and illustrations, it is out of print and now nomenclaturally outdated. Moul (1952) focused on the eastern and central regions of the state and incorporated the collections and publications of earlier workers, then augmented those records with his own collections.

Subsequent to the publications of Moul and Jennings, there have been shorter reports that add to our knowledge of the Pennsylvania moss flora. Pursell (1956, 1973, 1975) collected extensively in the central portion of the state, as well as elsewhere in Pennsylvania. The herbarium at the Pennsylvania State University (PAC) houses his collections. The combined efforts of Allen (1979), Manuel (1975) and Boardman (1977) provide important new records, as do unpublished theses and dissertations (Tees, 1933; Williams, 1971). All of these efforts, however, have not to date resulted in a new general account of Pennsylvania mosses.

## COMPUTERIZED INVENTORY OF PENNSYLVANIA MOSSES

The Pennsylvania Legislature enacted the Wild Resources Conservation Act in 1982. This act established legal mechanisms for the conservation and management of sensitive species of the flora and fauna, and also directed the Pennsylvania Department of Environmental Resources to conduct studies to determine measures to manage wild plants in order to conserve or perpetuate such species within the state, and to establish categories for the classification of the taxa as rare, threatened or endangered. Bryophytes are interpreted as native plants within the provisions of the act; thus they can be protect-

ed and managed under the provisions of the Wild Resources Conservation Act.

The lack of a current data base on the mosses of Pennsylvania has so far prevented an assessment of the moss flora with respect to conservation. The authors initiated and developed a computerized inventory of mosses in Pennsylvania, emphasizing those that were infrequently collected or cited in the available literature.<sup>1</sup> Ideally, a data base should be accessible to other bryologists, to those with an interest in the state flora and to state agencies. When complete, our data base will be incorporated into the Pennsylvania Natural Diversity inventory (PNDI) and possibly the Flora of Pennsylvania project, both of which are supported by a computerized inventory of vascular plants. Because of the flexibility and expandability of computerized data bases, we are presently able to incorporate features of the PNDI and Flora of Pennsylvania data bases into our own system. Our data base includes species lists, ecological and distributional records and references to source collections. Thus, the recommendation of Loucks (1986) that data bases be beneficial to both primary research interests and to secondary users is satisfied. The bryophyte data base will also augment our understanding of sensitive, restricted or unique habitats that may contain rare and infrequent mosses.

Examples include:

(1) Wattsburg Fen (also known as Weber Bog) in Erie County, Pennsylvania, is a National Natural Landmark. Wattsburg Fen is the site of the only known collection of *Tomenthypnum nitens* (Hedw.) Loeske in Pennsylvania (Boardman, 1977). The occurrence of this boreal moss helps confirm the northern affinities of the rest of the flora found at the site.

(2) Bear Meadows National Natural Landmark is a relic bog community near State College, Centre County. *Sphagnum torreyanum* Sullivant was reported from Bear Meadows by Boardman (1977) and represented the sole published report of this species for the state, although Richard E. Andrus (SUNY-Binghamton) has identified *S. torreyanum* from other counties in Pennsylvania (Andrus, personal communica-

<sup>1</sup>The Wild Resources Conservation Fund was established as part of the Wild Resources Conservation Act and is funded principally through a "tax check-off" system on individual state income tax return forms. This fund has supported our study.

tion). *Leucodon brachypus* var. *andrewsianus* Crum & Anders. was collected at Bear Meadows by Pursell (1973), representing the second report of this taxon for the state.

Other unique habitats will undoubtedly reveal rare and infrequent taxa when they are more fully explored. Examples of areas for further exploration include: Titus Bog in Erie County, Slippery Rock Gorge in McConnell's Mill State Park, Lawrence County and Biglers Rocks in Clearfield County. Collaborative work with researchers studying sensitive species of Pennsylvania's flora and fauna will undoubtedly facilitate the location of unique and bryologically unexplored habitats in the future.

## DATA BASE STRUCTURE

Loucks (1986) reviewed the categories of documentation for data bases. Our moss data base is consistent with his recommendations, although it does not follow his exact format. As mentioned earlier, the moss data base is structured to be compatible with the Pennsylvania Natural Diversity Inventory, which is under the control of the Department of Environmental Resources. Some of its specific characteristics are worthy of note. Nomenclature is based on Crum and Anderson (1981), but alternative synonyms and originally reported names are being included, particularly where monographic revisions warrant consideration. Where possible, each county record is based on a designated, verified herbarium specimen. Reliance on voucher specimens remains a long-term goal, as available time does not permit this policy to be strictly followed initially. The data base includes a field to indicate validation in addition to fields for information on site descriptions, site location, longitudinal and latitudinal data for mapping and comments. At present, the preliminary version of the data base is being edited and converted to Ashton-Tate dBase III/dBase III Plus files. New data from recent field collections and ongoing herbarium studies will be incorporated as more data become available.

## DATA SOURCES

### Literature:

Our development of a computerized data base began with the entry of published reports in the references cited earlier. Newly published records of Pennsylvania mosses will be continuously incorporated into the data base. The initial literature survey was used to create a list of "infrequently reported" taxa. For a taxon at the species or varietal level to be included on that list, its occurrence within Pennsylvania had to be limited to reports from six or fewer counties. At present, 150 species, nine varieties, and one form satisfy that criterion. Table 1 summarizes the frequency of reports for taxa on the "infrequently reported" list. All specimens of these taxa will be carefully examined to verify their identities. Further studies have been initiated to locate more records and to develop an understanding of phytogeographic and ecologic relationships of these taxa. New county reports for all mosses known to occur in the state are also being sought. A number of new state records are expected to be found, particularly in underinvestigated counties. Recently, Andrus (1988) described *Sphagnum rubroflexuosum* as a new species, with the type location near Cleremont, McKean County, PA.

### Herbaria:

Major herbaria that contain significant holdings of Pennsylvania mosses are being surveyed for new county reports of all taxa and for specimens of taxa on the "infrequently reported" list. The herbaria are: The Pennsylvania State University (PAC), the Academy of Natural

Sciences of Philadelphia (PH), the Missouri Botanical Garden (MO), the New York Botanical Garden (NY), the Smithsonian Institution (US) and the Farlow Herbarium at Harvard University (FH). In addition, smaller herbaria will also be surveyed. To facilitate this end, a questionnaire was prepared and sent to all the colleges and universities in Pennsylvania to determine their holdings of Pennsylvania mosses. Private collections are also being sought that have significant collections of Pennsylvania mosses. When completed, literature and herbarium surveys will provide the most comprehensive review of Pennsylvania mosses yet undertaken.

### Other Data Sources:

Bryological field studies are in progress and will continue to be carried out. Our ongoing collecting efforts focus on locating new county records for all taxa and confirming records of rarely reported taxa. In particular, undercollected regions of the state and unique or restricted habitats are being emphasized. A review of reported collections of some common taxa: *Dicranum scoparium* Hedw., *Ceratodon purpureus* (Hedw.) Brid., *Polytrichum commune* Hedw., *Leucobryum glaucum* (Hedw.) Angstr. ex Fries, and *Thuidium delicatulum* (Hedw.) BSG, shows a paucity of records in certain areas, suggesting that the north-central counties along the New York border and the central counties around Harrisburg are undercollected, or that collections from those areas have not been reported in the literature.

Where possible, collections are being made in habitats that enjoy some degree of protection from development and disturbance. These locations include properties owned by private conservation organizations (e.g. the Western Pennsylvania Conservancy, Northern Allegheny Conservation Association, and Presque Isle Audubon Club), state parks, state forest and game lands, the Allegheny National Forest, and Delaware Water Gap National Recreation Area. Field notes on each site, the habitat and associates of each specimen are entered into the data base to enhance ecological and distributional knowledge of the moss flora. These data will be valuable in developing management strategies for mosses that might be designated rare or endangered in the future. For example, some infrequently reported taxa are successional species that grow on substrates available only for relatively

**Table 1.** Frequency Distributions of the Number of County Reports for "Infrequently Reported" Taxa of Pennsylvania mosses.

Number of Counties	Number of Taxa
1	49
2	37
3	27
4	17
5	27
6	3
Total	160

short time periods (e.g., rotting logs, exposed soil). Others are glacial relicts that may disappear with climatic warming; still others are species found to the south with the potential of expanding their known ranges with (or without) climatic warming. Such taxa can be maintained in areas only if suitable habitats are constantly available. Data on the minimal habitat size for sensitive bryophytes are lacking, but should be determined if the conservation and management of sensitive populations are to occur.

The A. LeRoy Andrews Foray of the American Bryological and Lichenological Society is periodically held in Pennsylvania. These forays attract amateur and professional bryologists to specific collecting sites. The results can significantly add to the understanding of rare moss occurrences at these locations as well as stimulate an exchange of information. Many of the participants have collected elsewhere in Pennsylvania and almost certainly represent untapped sources of collection information and technical assistance on selected taxa.

Recently, a Pennsylvania Biological Survey was organized to stimulate the study of the biological resources within the state. The Bryophyte and Lichen Technical Committee was established with its chairman also serving on the Steering Committee. The work of the Bryophyte and Lichen Technical Committee and the goals of the Pennsylvania Biological Survey are consistent with the current course of study of the mosses of Pennsylvania, and the present authors have been active with the Biological Survey since its inception in 1987-1988.

### CONCLUSIONS

Pennsylvania, through its "Wild Resources Conservation Act" provides for the conservation and management of rare, threatened and endangered plants, including mosses. To assess which mosses have restricted ranges in Pennsylvania, a multiple-use, computerized data base has been developed using literature and herbarium records, augmented by field work in diverse and unusual habitats. When complete, this moss inventory will facilitate decisions on taxa eligible for protection under existing Pennsylvania legislation. Furthermore, work on mosses will contribute additional information on sensitive and endangered habitats that support plant and animal species of special concern, making decisions on site or habitat preservation more acceptable to the general public as well as the conservation community.

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# An Endangered Species List for Bryophytes: Endangered and Threatened Species of Ohio

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**Abstract:** Bryophytes are generally ignored as important tools that can be utilized in conservation and preservation efforts. The State of Ohio is one of the first to attempt to rectify this by including a section on bryophytes in an endangered-species list. Sixty-four of approximately 500 bryophyte taxa reported from Ohio currently appear on the list. The preparation of an imperiled bryophyte list should include the following: I. Compilation of a preliminary computerized data base of all taxa reported in the literature from the area under study. II. Initiation of herbarium studies to verify and enhance the data base. III. Establishment of status designations for imperiled taxa. IV. Development of criteria for status designations. V. Initiation of field studies to assess the status of taxa currently included on the list, to search for new taxa, and contribute to the overall data base.

Pages 205 - 207. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

It is well known that the distribution of bryophyte floras closely parallels that of vascular plants. Like vascular plants, bryophytes have migrated along natural routes under the influence of changing geological and climatic conditions (Sharp, 1939; Crum, 1952, 1972). In addition, they have the ability to persist in micro-habitats in areas where many vascular plants may have been unable to survive (Anderson, 1971). Thus, whether they are evaluated alone or in conjunction with the latter, bryophytes can provide evidence of floristic origins and serve as indicators of threatened or endangered habitats. Still, these organisms continue to be ignored as important tools for conservation and preservation efforts.

There are a number of factors contributing to this oversight. For the most part, bryophytes are small in size and inconspicuous in nature. Even in habitats where they form a dominant part of the vegetation, their appearance is frequently one of "sameness," that is, patterns of subtle hues of greens, reds and browns, providing aesthetic pleasure only to the most astute observer.

Bryophytes are also difficult to identify. They require an extraordinary amount of field and laboratory study and require observations at both macroscopic and microscopic levels, the latter usually requiring the aid of both a dissecting and compound microscope. Thus, unlike groups of larger organisms, only the largest and most conspicuous of bryophytes have received common names, and the use of Latin binomials seems to discourage all but the most stronghearted amateur naturalist. In addition, the relatively small numbers of amateur and professional bryologists in North America are widely scattered. Thus, state and provincial conservation organizations or naturalist groups that might express an interest in including bryophytes in floristic inventories may have difficulty in finding individuals with the necessary expertise to verify identifications. Ohio is one of just a few states to include a section on bryophytes in their endangered species list (Snider, 1982). Missouri includes bryophytes (Wilson, 1984), and a rare bryophyte list was recently completed for North Carolina. Pennsylvania is currently in the process of compiling a list of endangered bryophyte species, and Minnesota, in a recently published endangered flora and fauna list (Coffin and Pfannmuller, 1988) also included a small section on bryophytes.

## DISCUSSION

Two lists of endangered plant species have been compiled for Ohio.

One, published in 1984, was prepared through the Ohio Natural Heritage Program of the Ohio Department of Natural Resources, pursuant to the Ohio Endangered Plant Species Law of 1978 (Amended Substitute House Bill No. 908: Sections 1518.01 to 1518.99 of the Ohio Revised Code; McCance & Burns, 1984). It presently serves as the legal document with respect to all matters concerning endangered and threatened native plant species in Ohio. Although this law does not specifically exclude nonvascular plants, the official list currently includes only vascular plants.

An earlier, comprehensive list, begun in 1973 by the Ohio Biological Survey, was initiated to inventory all endangered and threatened plants in Ohio. This list (Cooperrider, 1982) included lichens, bryophytes and vascular plants. The lists of plants in this publication have no legal standing. Rather, they were designed to serve as comprehensive annotated lists of all native Ohio land plants at risk of being lost from the state flora by destruction of rare habitats or unique natural areas. As unofficial lists, they provide greater flexibility for additions, deletions and the shifting of taxa, and also allow the inclusion of informed but subjective assessments. As such, these lists are not bound by legal strictures that may require them to be immediately and demonstrably defensible (Cooperrider, 1982).

Sixty-four of approximately 500 taxa of bryophytes reported from Ohio are cited in the endangered species list. This compares with 72 of ca. 400 bryophyte taxa placed under similar categories in the Missouri checklist. The Missouri bryophyte list, like the Ohio list, is not a legally binding one.

There are numerous ways to approach the preparation of an endangered bryophyte species list and most methods are likely to provide useful data. However, there are certain features that are highly desirable in preparing such a list. These include the following:

### 1) Compile a preliminary working data base of all taxa reported in the literature for the area under study.

The ready availability of personal computers and numerous data base software packages makes the once formidable task of compiling and sorting information more easily manageable. Lotus 1-2-3 software was utilized to produce the data base for Ohio mosses. Even more sophisticated software has since become available for the personal computer and their use will undoubtedly increase efficiency. Fortunately, the more popular data base software is usually designed so that data files can be transferred between systems with little difficulty. Data placed in the Ohio data base include (when available):

TAXON (current species name according to recent nomenclature)

## FAMILY

GEOGRAPHIC REGION (country, province, state, county, vice county, etc.)

HERBARIUM collection in which specimen is deposited)

REFERENCE SOURCE (collector and collection number, date of collection, literature citation)

SYNONYMY (name on specimen or name cited in reference)

COLLECTION DATA (locality, habitat, substrate)

VERIFICATION (a mark or note to indicate those specimens that have been examined by the investigator)

Such data bases are not only valuable for the preparation of an endangered species list, they are also extremely useful as herbarium management systems, for ecological and floristic studies, preparation of state checklists and floras and for taxonomic monographs and revisions.

The importance of state bryophyte data bases becomes even more significant as the opportunity arises for several state data resources to become merged into a larger, regional data base.

### **2) Conduct herbarium studies to verify taxa cited in the data base, add unreported taxa and enhance data base information.**

To maintain credibility, any floristic study, species list, or taxonomic revision should be carefully documented by accurately labeled voucher specimens placed in established, university, museum or botanical garden herbaria. Such collections frequently represent the only evidence of the existence of rare or endangered taxa in a particular region. They also may be utilized for comparisons made later in follow-up studies, and serve to document changes in floristic composition of a region over time. For bryological studies in particular, the assessment of voucher specimens is frequently the most frustrating. Even in major herbaria, bryological specimens are often poorly curated (if curated at all), inadequately housed, and occasionally in such a poor state of repair that additional handling causes further damage, thus making them almost useless for study. A typical occurrence, once the literature search portion of the data base is established, is that the investigator will be better informed regarding the (potential) bryological holdings of particular herbaria than will the curators of those herbaria. Providing each curator with a printout of their holdings as reported from the literature can generate good will and result in free access to bryophyte holdings which may be in storage or otherwise not readily available. Such a gesture may also serve as a stimulus for renewed curation and maintenance of bryophyte collections.

### **3) Establish status designations for rare and imperiled taxa.**

Status designations will typically encompass the following categories: extirpated, possibly extirpated, endangered, rare, threatened, potentially threatened and status undetermined. Generally two to four categories should be utilized depending upon the size of the flora and how well the flora is known. If a vascular plant or other endangered-species list is in preparation or has already been completed for the same locale (state, region, etc.), the categories selected should agree with those already adopted. This will permit continuity and help establish credibility.

### **4) Determine criteria for assigning taxa to status designations.**

The categories utilized in the Ohio list, with their criteria, are:

*Extirpated*—plants (species, subspecies, variety or population) that continue to survive in other parts of their range but have been com-

pletely eliminated from Ohio within historic times.

*Endangered*—plants (species, subspecies, variety or population) whose prospects for continued survival and reproduction in Ohio are in immediate jeopardy. If not protected, extirpation from the state will probably follow.

*Threatened*—plants (species, subspecies, variety or population) whose continued survival in Ohio are not in immediate jeopardy, but which are represented by so few individuals, or occupy such a small area, that they may become endangered. Regular monitoring of their status and some degree of protection is necessary to assure their retention in the Ohio flora.

*Potentially threatened*—plants (species, subspecies, variety or population) that may become threatened in Ohio in a short period of time. Regular monitoring of their status is necessary to assure that they do not become threatened or endangered.

*Status undetermined*—plants (species, subspecies, variety or population) for which some evidence exists for endangered, threatened, or potentially threatened status, but for which the currently available data are insufficient to permit an adequate evaluation. Further research is needed to determine their present status in the Ohio flora.

The examples provided above are meant to be very general and subjective and would not suffice for a legally binding list (see below). They are, however, based on information provided by knowledgeable experts involved in rare plant studies (Cooperrider, 1982). These definitions permit flexibility for refining and shuffling taxa included in the list, as well as deleting current taxa or adding additional taxa as more information becomes available.

Thus, the criteria utilized in the comprehensive list are especially applicable regarding the bryophyte flora of Ohio. Although the state's bryoflora may be well documented in terms of overall numbers of taxa, it is often difficult to obtain adequate distributional data to assign an accurate status designation for a given taxon. Particularly when preparing a first or preliminary list, the freedom to include taxa which appear to be imperiled on the basis of limited distributional information is critical.

### **5) Initiate field studies to assess the status of taxa currently included in the list, search for new taxa, and contribute to the overall data base.**

Field studies are crucial to further expand, enhance and refine the original data base. Relatively few state, provincial or regional checklists and floras exist for bryophytes in North America, and the existing floras, although good for identification purposes, generally provide inadequate information on the frequency and abundance of most taxa. Many bryophytes are extremely small and may occupy microhabitats only a few centimeters in area. Such species are frequently overlooked, even by experienced bryologists. A case in point is the recent discovery of four rare mosses new to Ohio: *Buxbaumia minakatae* Okam., *Plagiothecium latibricola* BSG, *Diphyscium cumberlandianum* Harvill, and *Tetradontium brownianum* (Dicks.) Schwaegr. All four taxa represent significant geographic range extensions, and they were discovered in areas that have traditionally been heavily visited by bryologists for many years. Recently discovered *Tetradontium brownianum* has turned out to be so common in south-central Ohio that it is doubtful that it will qualify for Ohio's endangered species list, yet its nearest known location is on the south shore of Lake Superior in northern Michigan. In North America it is considered to be "very rare." It is also represented by scattered collections from Newfoundland, the Canadian Maritime Provinces, Maine, New Hampshire, New York, Quebec, Ontario and northern Michigan (Crum & Anderson, 1981).

With bryophytes in particular, collections frequently reflect the

interests of the collector and emphasize local or unusual habitats or regional bias. Of the 88 counties in Ohio, only eight have a hundred or more species of mosses recorded, and 22 counties have ten or fewer species reported. Hepatics are even more poorly known. Only nine counties in Ohio have 20 or more taxa listed, and there are 29 counties for which no reports exist to date. Therefore, caution must be exercised in attempting to distinguish the truly rare and endangered species from those taxa for which rarity is only implied because of restricted collecting or misidentified specimens.

Lists of endangered species frequently evolve from (often inadequate) state floristic inventories. Thus, the more intensive the field work, the more complete the inventory. The more careful the inventory, the more realistic one can be in assigning accurate status designations to the taxa under consideration.

Field inventories for the preparation of lists of endangered species, while not as expensive to conduct as many other types of research, do require funds. It is in this area that the citizens of Ohio are enlightened. Utilizing funds generated from the State Income Tax Refund Checkoff Program, a Natural Areas Mini-Research Grants Program has been established through the Ohio Division of Natural Resources, Division of Natural Areas and Preserves. Funding is available, on a competitive basis, for monitoring and inventory of natural areas, scenic rivers and endangered species in Ohio.

#### SUMMARY COMMENTS

It is not an easy task to compile a list of endangered bryophyte taxa. As with many groups of organisms, one must be realistic in attempting to distinguish the truly rare and endangered species of bryophytes from those taxa for which rarity is only implied because of restricted collections or misidentified specimens. Many bryophytes are overlooked because of their extremely small size or because they may occupy microhabitats of only a few centimeters in area. Furthermore, the microhabitats in which the rarest of species once occurred may have since become altered, or the habitat data from herbarium specimens may be insufficient to aid in relocating the species. Many diverse microhabitats may already be protected, to a limited extent, as unusual geological features, state parks, state forests, nature preserves and national landmarks. Holding deed to these properties, however, provides few guarantees that imperiled bryophytes actually will receive protection. For example, protection of bryophyte populations is rarely considered when initiating management and maintenance policies.

Even the most mundane tasks such as moving information signs, trail signs, rerouting trails and other similar practices can cause irreversible damage to entire populations of these small organisms. Nevertheless, the recognition and subsequent preservation of microhabitats in which imperiled species of bryophytes occur must be encouraged, and the creation of an endangered bryophytes list should only be the first step in the awareness process.

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# Endangered Bryophytes in New Jersey: Determination, Protection and Management

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**Abstract:** Bryophytes have received little attention in most programs dealing with endangered and threatened species. Central among the reasons for this is that they are difficult to identify and easily overlooked, and their is lack of an adequate data base upon which to develop valid conclusions concerning the status of the bryophyte flora. In contrast to vascular plants, bryophytes have also been under-collected in most regions of the United States. In spite of such limitations, the identification and listing of endangered and threatened bryophytes (on local, national and international levels) is essential in order to ensure their inclusion in endangered plant legislation. Even if these lists are not legislatively adopted as being "official," they still provide strong guidelines for those involved in ecosystem management and preservation. Once endangered and threatened bryophyte species are located, information about their ecological requirements should be obtained and management plans established.

Pages 208 - 210. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

This paper is based primarily upon insights gained during a three-year survey of *Sphagnum* in the state of New Jersey (Karlin and Andrus, 1988). Towards the end of that survey, it became obvious that several *Sphagnum* species in the state were rare. Having had very little prior experience with programs on endangered and threatened species, I did not know what specific guidelines were used to determine endangered or threatened status for a species. Seeking the help of specialists in the field, I organized a one-day conference entitled "New Jersey's Rare and Endangered Plants and Animals" which was held at Ramapo College in October 1987. The meeting was sponsored by the Institute for Environmental Studies at Ramapo College of New Jersey; a proceedings based on the conference is available (Karlin, 1989).

At this conference, Dick Andrus and I presented one of the first papers ever to address the subject of rare and endangered bryophytes. The following is an outgrowth of certain aspects of that paper.

## DETERMINATION OF ENDANGERED AND THREATENED BRYOPHYTES

The ranking system that had been developed by the Nature Conservancy for vascular plants also proved suitable for New Jersey bryophytes. Although there are limitations to the system, the benefits gained by utilizing a uniform approach in the determination of the endangered/threatened (E/T) status of any given species far outweigh any inconsistencies. The development of a set of guidelines for the determination of E/T bryophytes differing from guidelines already used for vascular plants might actually be counter-productive since it could serve to confuse and complicate efforts to pass E/T plant legislation.

Once a ranking system has been developed, one then must implement it. For this purpose, an adequate data base listing the occurrence and abundance of the species in question is required. Because such data bases do not often exist for bryophytes, an accurate determination of the rarity of bryophyte species is not possible without extensive field surveys. Such data are lacking primarily because:

1) The bryophyte flora in many areas either has not been studied or has undergone only cursory sampling. For instance, only a handful of *Sphagnum* collections had been made from northern New Jersey (Passaic and Sussex counties) and adjacent New York (Orange and

Sullivan counties) prior to the study by Karlin and Andrus (1986).

2) Some sites in a region may have been thoroughly sampled, but adjacent areas may have been neglected (patchy collecting). Although extensive collections of *Sphagnum* had been made in Burlington, Ocean and northern Atlantic counties of the New Jersey Pine Barrens region, very few collections had been made from the adjacent counties (Cape May, Cumberland and Monmouth) prior to the collecting activities of Karlin and Andrus (1988).

3) Knowledge of the bryophyte flora of a site or region may have been limited to studies by persons relatively inexperienced with bryophytes. As many bryophyte species are difficult to distinguish in the field, it is likely that many are overlooked in such cases. The following two examples will illustrate this problem:

Liverworts are commonly encountered in the peatlands of northern New Jersey, where I often visit to study *Sphagnum*, and most of them have not been noted by hepaticologists. Even though I collect a few of the more noticeable liverwort species, I lack the knowledge (and the time) to perform a thorough sampling of the liverwort flora of each site that I visit. The result is that, while one component of the bryophyte flora is being well documented, another component is only partially addressed.

The second example concerns a rich fen in New Jersey that was studied by researchers from the Natural Heritage Program of New Jersey. The presence and ground cover of both the vascular plants and bryophytes were determined in a vegetation plot that was located in the rich fen community type. Two species of *Sphagnum* were noted in this study. When I first visited the site at a subsequent date, I surveyed not only the specific area that had been sampled, but also the surrounding wetland complex. Although most vascular plant species occurred commonly throughout the rich fen community type, the *Sphagnum* species present had a more sporadic distribution. There were areas where only 0-2 *Sphagnum* species were present, but five species of *Sphagnum* were found to have significant amounts of ground cover in the community. A total of 17 species of *Sphagnum* were collected from the entire wetland complex, three of which were quite rare in New Jersey. This is a surprisingly rich *Sphagnum* flora, and yet there was no evidence that such diversity existed based on results of the prior study. In addition, *Tomenthypnum nitens* (Hedw.) Loeske, another rare bryophyte in New Jersey, also occurred at this site and was not reported in the earlier study.

4) Because of the difficulty in the identification of some bryophyte

species and changes in taxonomic perceptions over time, many historical identifications can not be trusted or require updating. All herbarium records should be verified by competent bryologists. For instance, *Sphagnum quinquefarium* (Braith.) Warnst., *S. russowii* Warnst., *S. squarrosum* Crome, *S. subnitens* Russ. & Warnst. and *S. teres* (Schimp) Angstr. have all been reported from the New Jersey Pine Barrens region but reexamination of herbarium and voucher specimens showed that those species do not actually occur there (Karlin and Andrus 1988).

It should also be noted that an adequate bryophyte survey is not a trivial undertaking and that additional time should accordingly be budgeted into any sampling schedule that includes bryophytes. It might be best to have one person focus on the vascular species and one on the bryophytes whenever possible. If this is not possible, then a competent bryologist should be invited to study the most representative of the ecosystems being studied.

In the case of southern New Jersey, sufficient collections of *Sphagnum* have been made to allow adequate determinations of E/T status. The specimens and data, however, are scattered in several different herbaria, and have needed to be compiled into one data base. In addition, several sites in southern New Jersey counties which had not been adequately surveyed in the past needed to be studied. Northern New Jersey has required different tactics. Aside from C. F. Austin's collections in the Palisades area of Bergen County (made in the mid 1800's), very little data existed for the glaciated portion of New Jersey prior to the study of Karlin and Andrus (1986). It was, in essence, "virgin territory" with regard to *Sphagnum*, and extensive field work was required. All major sites where *Sphagnum* was likely to occur were visited, a process that required much time and effort. About 1500 New Jersey collections were made by Karlin and Andrus (1988), nearly doubling the number of all previous collections.

At that point, sufficient data had been accumulated to make possible accurate E/T rankings of New Jersey's *Sphagnum* species. Because the data base is based in large part on recent collections and because all available previous collections were verified, the list is current with regard to both taxonomic concepts and recently verified sites of occurrence. Nine new state records were established (Karlin and Andrus, 1986, Karlin and Andrus, 1988), all from the previously poorly studied northern portion of the state. Five of these nine species are quite rare in New Jersey, and are currently known from fewer than five sites there (Andrus and Karlin, 1989).

## PROTECTION

The protection of E/T bryophytes is in some respects no more difficult than that of vascular species. Many E/T bryophytes are found in habitats in which other plant and animal E/T species also occur. Therefore, the preservation of habitat for other species will also protect the bryophytes. Two E/T species of *Sphagnum* occur in a peatland in New Jersey which is also home to a threatened dragonfly species (Andrus and Karlin, 1989; Carle, 1989). One threatened *Sphagnum* species occurs at Bennet bog in Cape May County of New Jersey, where several E/T vascular plants also occur (Andrus and Karlin, 1988; Snyder, 1988; Montgomery and Fairbrothers, 1963).

Many E/T bryophytes are also found in relatively protected areas such as state and national parks. It should be noted, however, that many such species apparently had a more extensive distribution in New Jersey in the past, and that extensive development activities have destroyed most of the sites that were not protected. Although C.F. Austin made many significant collections of *Sphagnum* in the

Palisades area of eastern Bergen County in the mid-1800's (including the only New Jersey collection of *S. platyphyllum* (Braith.) Warnst.), apparently none of these sites has escaped development. So species found in state and national parks, wildlife refuges and biological reserves are protected to some extent, but they often represent only a remnant of what were once much more extensively distributed species.

Although bryophytes may not be specifically included when a particular legislative act protecting plants is drafted, the definition of "plant" utilized in the legislation is often broad enough to include bryophytes. The result is that bryophytes may already be protected by existing legislation dealing with E/T plants in some states. However, where no legislation exists to protect E/T plants, the question of providing E/T bryophyte protection may become problematic. There is frequently little interest in protecting E/T plant species in general, and protecting bryophytes meets with even more skepticism. Education of the public and of our colleagues is essential. In the event that there appears to be serious resistance to the concept, attempting to force the issue of E/T bryophyte protection could possibly endanger the passage of E/T species legislation for other plant groups. In such cases, rather than referring to bryophytes directly, efforts might be made to ensure that the E/T species legislation that is being formulated have a broad enough definition to include bryophytes.

The development and enactment of legislation for the preservation of endangered ecosystems represents a powerful and useful approach to protecting bryophytes as well. This may receive greater public support than would the protection of individual species, and should serve to provide protection for a wide range of organisms (fungi, bryophytes, insects) that might not otherwise be granted protection. Habitat protection also circumvents the necessity of determining the E/T status of all species present in a region (an almost impossible task). By preserving examples of the unique and rare ecosystems in a given region, it is likely that a large percentage of the E/T bryophytes (and other taxonomic groups) occurring in that region would be present in the protected sites.

## MANAGEMENT

Once E/T bryophyte species have received protection, the question of management comes up. How does one preserve the habitat required for the preservation of any given species in question? In many cases, it is only necessary to maintain the *status quo*. Simply keeping the ecosystem in its current state and relatively free of human disturbance could be sufficient management. This may not, however, be a simple task. For instance, beaver activity at one site in New Jersey destroyed one of the few rich fens that occurred in the state. One of the five northern dwarf shrub bogs that occur in New Jersey has also recently experienced extensive beaver damage.

Although the "harvesting" activity by humans is not as critical a problem for bryophytes as it might be with vascular plants, the trampling of moss mats (especially in peatlands) may have a serious impact. Finally, bryophytes are often more sensitive to environmental changes than are vascular plants, particularly to changes in water quality (Sjors, 1963; Vitt *et al.*, 1975).

Several bryophyte species are ephemeral or are dependent on perturbation. Such species require active management techniques, and in some cases adequate management may not even be possible. The New Jersey Pine Barrens are a mosaic of communities in which fire has been the dominant influence (Little, 1979). Pine Barrens plants must be adapted to fire to some degree, and the sphagna are no exception. Fire and related disturbance may thus play a role in the maintenance in

the diversity of *Sphagnum* species present in such ecosystems. An atlantic white cedar swamp [*Chamaecyparis thyoides* (L.) BSP.] at Maurice River in Cumberland County, New Jersey provides a good example of this. Half the swamp was mature, with an extensive tree layer canopy. Only two *Sphagnum* species were found in this undisturbed area of the peatland. The other half of the peatland had been logged and was a young cedar swamp with extensive, open clearings. Ten species of *Sphagnum* were found there. This general pattern was observed throughout the Pine Barren region. Clearly, any attempt to manage E/T bryophyte populations in a region like the Pine Barrens must take periodic disturbance and its possible benefits into account. For effective achievement of such management goals, however, more knowledge of the ecology of bryophyte species is needed.

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# Flora Protection: The Question of Rare Mosses in New York State

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**Abstract:** Conservation of endangered bryophytes requires, among other things, a definition of endangerment suited to bryophytes, knowledge of their life histories and taxonomy, and an existing data base on the bryophytes of a region that includes their relative abundance or rarity. We have used the number of past collection locations in New York as the primary criterion to define the degree of rarity of the state's mosses. Other criteria, such as abundance and range, will be used in the future to refine the list presented here. Our list of New York rare mosses presents 133 species with an additional watch list of 25 species. Because the bryoflora of New York is still poorly known, and because bryology is a specialty, the apparent rarity of some species is not necessarily a measure of the rarity of the species, but a measure of the rarity of bryologists working in the state.

Pages 211 - 216. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Conservation of endangered bryophytes requires definitions of rarity and endangerment especially suited to them, an understanding of their life forms and taxonomy, and availability of data on the relative abundance of species within the region of concern. For purposes of discussion, we will focus on the moss flora of New York State and, from this case study, suggest several key steps toward generating a conservation program for bryophytes in general.

There are many possible criteria that might be used to define rarity among mosses of New York, including range, abundance and number of locations for the species in the State. Because most of these criteria require more knowledge than we currently have, we decided to use the number of known locations in the State as the major criterion for inclusion on the list proposed here. This is the only standard that can easily be applied to all species in New York with any degree of equity at the present time.

Our list of rare mosses (Table 1) includes species known historically in New York from 20 locations or fewer. A few other species have been included, even though they have more than 20 historical locations in New York, since some sites of their former occurrence are known to have been destroyed. A second (watch) list (Table 2) includes species reported without vouchered verification from New York and those species apparently too common for active inventory but too uncommon to ignore. Our list contains 133 species with 25 additional species included on the watch list. Although some may consider this too many species for a rare moss list, it must be kept in mind that New York State is one of the most ecologically diverse states in the northeast, with sub-arctic elements occurring in the Adirondack high peaks, calcicoles in abundance along a broad limestone contact zone, and southern Coastal Plain species ranging north to Long Island. The apparent rarity of some species over these geographical and ecological gradients may, however, be merely a measure of the scarcity of bryologists working in local areas over the years.

The occurrences of many species of mosses are closely keyed to conditions at the interfaces of substrate and atmosphere. Regionally rare species are often those at the climatic limits of their ranges where

their specialized habitat requirements are infrequently met. The occurrence of rare species is, therefore, dependent primarily on the presence of specialized microhabitats within the State. Rare habitats, in effect, often indicate rare mosses, and vice-versa, and the presence of rare mosses can be an important indicator of a unusual microhabitats or microclimates. Thus mosses can help in locating rare communities, even when rare vascular plants are absent.

Because of the minute size and unusual morphology of most mosses, their study remains a specialty among botanists, whether amateur or professional. Facility in taxonomic bryology requires at least the equivalent of a year of graduate study. Even so, many mosses can only be identified accurately by specialists and monographers, and only in recent years have good manuals appeared for the generalist. Bryology is not a casual undertaking, but a major commitment. Consequently there are few people capable of, or willing to work with mosses, especially the rare ones.

A history of bryological exploration in New York State has been summarized by Ketchledge (1962). We will report in this paper only on the current catalog of mosses, based on the work of Ketchledge (1957, 1980), assisted throughout by the late Stanley Jay Smith, Senior Curator of Botany at the New York State Museum, Albany. Based on literature reports, the examination of specimens at the major herbaria in the State, and on their systematic explorations in bryologically unknown districts, Smith and Ketchledge produced a checklist of the New York State mosses (Ketchledge, 1980) in which the presence or absence of 503 species and varieties were reported for each "square" degree of latitude and longitude (26 in all) that covers part of the State. Occurrences of *Sphagnum* were based largely on the research of R. E. Andrus. The original purpose of the checklist was to document the state of our knowledge of the moss flora in each section of New York, and thereby identify those areas in greatest need of further bryological exploration. The checklist now serves also as a data base, identifying those species most worthy of protection and conservation.

Stanley Smith and Arthur Holweg used an earlier version of this data base when they developed the first list of rare New York bryophytes (Smith & Holweg, 1974). This unpublished paper lists 249 species and varieties of mosses as rare, endangered or potentially extirpated.

Our current list was developed using Smith and Holweg's list as a starting point and expanding it to include new species, e.g. those

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reported by Eckel (1987), Miller (1987) and Slack *et. al.* (1988), and removing species no longer considered rare. The primary reason for creating such a list was to aid conservation efforts. Our list is now incorporated in the New York Natural Heritage Program inventory. Accordingly the list has been ranked following the Natural Heritage ranking system (The Nature Conservancy, 1982).

Global ranks (i.e. rarity of species throughout their ranges) were assigned by inspection of the literature, particularly Crum and Anderson (1981), and consultation with experts. State ranks were based predominantly on the number of *extant* localities known in New York State. One difficulty encountered, and not yet resolved, was defining "extant" for the purposes of ranking. For vascular plants, species verified as extant by the Natural Heritage Program, or reliably reported from New York within the past 15 years are presumed extant, all other records are considered historical. With some rarely noted mosses (such as *Ephemerum* spp.), this standard may not apply, and could lead to imprecise ranking. Although we are fully cognizant of this problem, we are retaining a 15 year limit for the present.

Other criteria used for ranking elements by Natural Heritage Programs include abundance, range and number of protected occurrences. The first of these (abundance) is not easily applied to mosses and might lead to absurd results if it were; the other two criteria have not been used because of inadequate data on the distribution of most rare mosses in New York. As more information accumulates, these criteria will possibly play a more important part in the ranking of species.

We suggest the following: (1) An initial list of rare bryophytes is best generated from a data base developed by professional bryologists working closely with amateurs from various parts of the region; this should include intense field studies in the region. (2) Initial lists of rare or potentially rare species should be developed using specific criteria, such as 20 or fewer historical collections, as standards for inclusion on the working lists with other criteria used to refine the lists as warranted. Although initial lists of rare bryophytes will often reflect the lack of bryological collection in a region, increased field work and constant updating of the lists will build an understanding of what truly are the rare bryophytes of a region. (3) Bryophytes can be important indicators of rare communities and community quality and should not be overlooked by conservation organizations. (4) Locating rare mosses requires field experts, both amateur and professional, who can explore unusual and remote communities, and who do it with the proverbial fine-toothed comb. We hope that listing rare New York State mosses

will encourage bryologists in their field work, especially in poorly known areas, and assist in the identification and protection of natural areas throughout New York State.

#### ACKNOWLEDGMENTS

Many people have helped throughout the development of the 1988 lists of rare mosses including the following bryologists who have seen the lists or partial drafts: Richard E. Andrus, William Buck, Patricia Eckel, Norton Miller, Carol Reschke, Jon Shaw, William Steere, Nancy Slack, Robert Wesley, and Richard Zander.

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**Table 1.** Rare Mosses of New York State, 1988 List.

Scientific Name <sup>1</sup>	Global Rank <sup>2</sup>	State Rank <sup>3</sup>
<i>Acaulon muticum</i> (Hedw.) C. Müll. <sup>4</sup>	G4G5	SH
<i>Amblyodon dealbatus</i> (Hedw.) B.S.G.	G4G5	S1
<i>Amblystegium noterophilum</i> (Sull. & Lesq. ex Sull.) Holz. <sup>5</sup>	G3G4	S2
<i>Anoetangium aestivum</i> (Hedw.) Mitt.	G5	SH
<i>Archidium alternifolium</i> (Hedw.) Schimp.	G5	SU
<i>Archidium ohioensis</i> Schimp. ex C. Müll.	G5	SU
<i>Arctoa fulvella</i> (Dicks.) B.S.G.	G3	S1
<i>Astomum muhlenbergianum</i> (Sw.) Grout	G5	SH
<i>Aulacomnium androgynum</i> (Hedw.) Schwaegr.	G5	S2S3?
<i>Aulacomnium turgidum</i> (Wahl.) Schwaegr.	G5	SH
<i>Brachythecium turgidum</i> (C. J. Hartm.) Kindb.	G4G5	S1
<i>Bryum algovicum</i> Sendtn. ex C. Müll.	G5	SH
<i>Bryum cyclophyllum</i> (Schwaegr.) B.S.G. <sup>6</sup>	G5	SH
<i>Bryum gemmiparum</i> De Not.	G4G5	SU
<i>Bryum muehlenbeckii</i> B.S.G.	G3?	S1
<i>Bryum pallescens</i> Schleich. ex Schwaegr.	G5	SU
<i>Bryum tenuisetum</i> Limpr.	G5Q	SU
<i>Bryum turbinatum</i> (Hedw.) Turn.	G5	SH?
<i>Bryum weigelii</i> Spreng.	G5	SH
<i>Buxbaumia minkatae</i> Okam.	G3?	S1
<i>Calliergon sarmentosum</i> (Wahl.) Kindb.	G3G5	S1
<i>Calliergon trifarium</i> (Web. & Mohr) Kindb.	G3G4	SH
<i>Campylostelium saxicola</i> (Web. & Mohr) B.S.G.	G3?	SH
<i>Catoscopium nigrum</i> (Hedw.) Brid.	G5	S1
<i>Cinclidium stygium</i> Sw.	G5	S1
<i>Conardia compacta</i> (C. Müll.) Robins. <sup>7</sup>	G5	S1
<i>Conostomum tetragonum</i> (Hedw.) Lindb.	G4G5	S2
<i>Cratoneuron commutatum</i> (Hedw.) Roth	G5	SU
<i>Cynodontium alpestre</i> (Wahl.) Milde	G5	S1
<i>Cynodontium strumiferum</i> (Hedw.) Lindb.	G5	S1
<i>Desmatodon obtusifolius</i> (Schwaegr.) Schimp.	G5	S2
<i>Desmatodon porteri</i> James ex Aust.	G4G5	S1
<i>Dichylema falcatum</i> (Hedw.) Myr.	G4G5	S1
<i>Dicranella cerviculata</i> (Hedw.) Schimp.	G5	SH
<i>Dicranella subulata</i> (Hedw.) Schimp.	G5	SH
<i>Dicranoweisia crispula</i> (Hedw.) Lindb. ex Milde	G5	SX
<i>Dicranum elongatum</i> Schleich. ex Schwaegr.	G5	S1
<i>Discelium nudum</i> (Dicks.) Brid.	G3	S2?
<i>Distichium inclinatum</i> (Hedw.) B.S.G.	G5	S1
<i>Ditrichum flexicaule</i> (Schwaegr.) Hampe	GU	S1
<i>Encalypta rhaptocarpa</i> Schwaegr.	G5	S1
<i>Entodon brevisetus</i> (Hook. & Wils. ex Wils.) Lindb.	G3G4	SH
<i>Ephemerum cohaerens</i> (Hedw.) Hampe	G5	SH
<i>Ephemerum crassinervium</i> (Schwaegr.) Hampe	G5	SH
<i>Ephemerum serratum</i> (Hedw.) Hampe	G5	SH
<i>Ephemerum spinulosum</i> Bruch & Schimp. ex Schimp.	G5	SH
<i>Funaria flavicans</i> Michx.	G5	SH
<i>Grimmia affinis</i> Hoppe & Hornsch. ex Hornsch.	G5	SH
<i>Grimmia agassizii</i> (Sull. & Lesq. ex Sull.) Jaeg. & Sauerb. <sup>8</sup>	G5	SH
<i>Grimmia cribrosa</i> Hedw. <sup>9</sup>	G3?	S1
<i>Grimmia donniana</i> Sm.	G3G5	S1
<i>Grimmia laevigata</i> (Brid.) Brid.	G5	SX?
<i>Grimmia tenerrima</i> Ren. & Card.	G5	SU
<i>Haplohymenium triste</i> (Ces. ex De Not.) Kindb.	G5	SH
<i>Heterocladium dimorphum</i> (Brid.) B.S.G.	G5	SH
<i>Homalotheciella subcapillata</i> (Hedw.) Broth.	G3G5	SH
<i>Hygrohypnum closteri</i> (Aust.) Grout	G3	SH
<i>Hygrohypnum micans</i> (Mitt.) Broth. <sup>10</sup>	G3G5	SU

Scientific Name <sup>1</sup>	Global Rank <sup>2</sup>	State Rank <sup>3</sup>
<i>Hylocomium pyrenaicum</i> (Spruce) Lindb.	G5	SH
<i>Hyophila involuta</i> (Hook.) Jaeg. & Sauerb.	G5	S1
<i>Isopterygium distichaceum</i> (Mitt.) Jaeg. & Sauerb.	G4G5	S1
<i>Isothecium stoloniferum</i> Brid. <sup>11</sup>	G4G5	SH
<i>Kiaeria blyttii</i> (Schimp.) Broth.	G5	S1
<i>Kiaeria starkei</i> (Web. & Mohr) Hag.	G5	S1
<i>Lindbergia brachyptera</i> (Mitt.) Kindb.	G5	SH
<i>Meesia longiseta</i> Hedw.	G5	SH
<i>Meesia triquetra</i> (L. ex Richt.) Ångstr.	G5	S1
<i>Meesia uliginosa</i> Hedw.	G5	SH
<i>Micromitrium austinii</i> Aust.	G5?	SH
<i>Micromitrium megalosporum</i> Aust.	G5?	SH
<i>Mielichhoferia mielichhoferi</i> (Funck ex Hook.) Loeske	G3G4	S1
<i>Mnium hymenophylloides</i> Hüb. <sup>12</sup>	G3G4	S1
<i>Mnium lycopodioides</i> Schwaegr.	G5	SH
<i>Myurella julacea</i> (Schwaegr.) B.S.G.	G5	S2
<i>Orthotrichum ohioensis</i> Sull. & Lesq. ex Aust.	G5	SH
<i>Orthotrichum pusillum</i> Mitt.	G5	SH
<i>Orthotrichum sordidum</i> Sull. & Lesq. ex Aust.	G5	SH
<i>Orthotrichum stellatum</i> Brid.	G5	SH
<i>Paludella squarrosa</i> (Hedw.) Brid.	G3G5	SH
<i>Philonotis longiseta</i> (Michx.) Britton	G5	SU
<i>Philonotis mühlenbergii</i> (Schwaegr.) Brid.	GUQ	SH
<i>Physcomitrella patens</i> (Hedw.) B.S.G.	G5	SH
<i>Physcomitrium hookeri</i> Hampe	G3?	SH
<i>Physcomitrium immersum</i> Sull.	G3?	SH
<i>Pleuroidium palustre</i> (Bruch & Schimp.) B.S.G.	G4G5	SH
<i>Pogonatum brachyphyllum</i> (Michx.) P.-Beauv.	G5	SH
<i>Pogonatum dentatum</i> (Brid.) Brid.	G4G5	S1
<i>Pohlia atropurpurea</i> (Wahl.) H. Lindb.	G5	SH
<i>Pohlia carnea</i> (Schimp.) Lindb.	G5	SH
<i>Pohlia elongata</i> Hedw.	G5	S2
<i>Pohlia filiformis</i> (Dicks.) Andr.	G5	SH
<i>Pohlia lescuriana</i> (Sull.) Grout	G5	SH
<i>Pohlia prolifera</i> (Kindb. ex Limpr.) Lindb. ex Arn.	G5	S1
<i>Polytrichum longisetum</i> Brid.	G5	SH
<i>Pottia davalliana</i> (Sm. ex Drake) C. Jens <sup>13</sup>	G5	S1
<i>Pseudoleskea patens</i> (Lindb.) Kindb.	G4G5	SU
<i>Pseudoleskeella tectorum</i> (Funck ex Brid.) Kindb. ex Broth.	G4G5	SU
<i>Ptychomitrium incurvum</i> (Schwaegr.) Spruce	G5	SH
<i>Racomitrium aquaticum</i> (Brid. ex Schrad.) Brid.	GU	S1
<i>Rhacomitrium canescens</i> (Hedw.) Brid.	G5	S1S2?
<i>Rhizomnium pseudopunctatum</i> (Bruch & Schimp.) Dop.	GU	S1
<i>Schwetschkeopsis fabronia</i> (Schwaegr.) Broth.	G5	SH
<i>Scorpidium scorpidioides</i> (Hedw.) Limpr.	G5	S2
<i>Scorpidium turgescens</i> (T. Jens.) Loeske	G5	S1
<i>Seligeria calcarea</i> (Hedw.) B.S.G.	G4G5	SH
<i>Seligeria campylopoda</i> Kindb. ex Macoun & Kindb.	G4G5	S2S3?
<i>Seligeria donniana</i> (Sm.) C. Müll.	G3G5	S2S3
<i>Seligeria recurvata</i> (Hedw.) B.S.G.	G4G5	S2S3
<i>Sematophyllum adnatum</i> (Michx.) E. G. Britt.	G5	SH
<i>Sematophyllum demissum</i> (Wils.) Mitt.	G5	SH
<i>Sematophyllum marylandicum</i> (C. Müll.) E.G. Britt.	G5	SH
<i>Sphagnum andersonianum</i> Andrus	G3G5?	S1
<i>Sphagnum angermanicum</i> Melin	G5	SX
<i>Sphagnum jensenii</i> H. Lindb. <sup>14</sup>	G5	S1
<i>Sphagnum lindbergii</i> Schimp. ex Lindb.	G5	S1
<i>Sphagnum macrophyllum</i> Bernh. ex Brid.	G5	S1
<i>Sphagnum platyphyllum</i> (Lindb.) Sull. ex Warnst. <sup>15</sup>	G5	S1

Scientific Name <sup>1</sup>	Global Rank <sup>2</sup>	State Rank <sup>3</sup>
<i>Sphagnum portoricense</i> Hampe	G5	S1
<i>Sphagnum subfulvum</i> Sjörs	G5	S1
<i>Sphagnum tenellum</i> Ehrh. ex Hoffm.	G5	S2
<i>Sphagnum trinitense</i> C. Müll. <sup>16</sup>	G5	S1
<i>Splachnum ampullaceum</i> Hedw.	G3G5	S2
<i>Splachnum rubrum</i> Hedw.	G2G3	SX
<i>Syrrhopodon incompletus</i> Schwaegr.	G5	SH?
<i>Syrrhopodon texanus</i> Sull.	G5	SH
<i>Taxiphyllum taxirameum</i> (Mitt.) Fleisch.	G5	SH
<i>Tayloria serrata</i> (Hedw.) B.S.G.	G3?	SH
<i>Tetradontium brownianum</i> (Dicks.) Schwaegr.	G3?	SH
<i>Tetraplodon angustatus</i> (Hedw.) B.S.G.	G3?	S1
<i>Tetraplodon mnioides</i> (Hedw.) B.S.G.	G5	S1
<i>Thelia lescurii</i> Sull.	G5	SH
<i>Thuidium pygmaeum</i> B.S.G.	G5	S1
<i>Thuidium scitum</i> (P.-Beauv.) Aust.	G3G5	SH
<i>Tortula papillosa</i> Wils. ex Spruce	G5	S1
<i>Weissia hedwigii</i> Crum	G5	S1

<sup>1</sup> Nomenclature follows Crum and Anderson (1981) with the exception of that of *Sphagnum* which follows Andrus (1980).

<sup>2</sup> GLOBAL RANK G1 = Critically imperiled throughout its range due to extreme rarity (5 or fewer sites); G2 = Imperiled throughout its range due to rarity (6 - 20 sites); G3 = Rare or local throughout its range (21 - 100 sites); G4 = Apparently secure throughout its range (but possibly rare in parts); G5 = Demonstrably secure throughout its range (however it may be rare in certain areas); GU = Status unknown; a “?” indicates a question exists about the rank; a “Q” indicates a question exists concerning the taxonomy of the species.

<sup>3</sup> STATE RANK S1 = Critically imperiled in New York State because of extreme rarity (5 or fewer sites); S2 = Imperiled in New York State because of rarity (6 - 20 sites); S3 = Rare in New York State (usually 21 - 100 extant sites); SH = No extant sites known (verified within the past 15 years) in New York State but it may be rediscovered; SX = Apparently extirpated from New York State; SU indicated Status Unknown; a “?” indicates a question exists about the rank.

<sup>4</sup> Includes *A. muticum* var. *rufescens* (Jaeg.) Crum.

<sup>5</sup> = *Hygroamblystegium noterophilum* (Sull. & Lesq. ex Sull.) Warnst. in Ketchledge (1980).

<sup>6</sup> = *Bryum tortifolium* Funck ex Brid. in Ketchledge (1980).

<sup>7</sup> = *Rhynchostegiella compacta* (C. Müll.) Loeske in Ketchledge (1980).

<sup>8</sup> = *Schistidium agassizii* Sull. & Lesq. ex Sull. in recent literature.

<sup>9</sup> = *Coscinodon cribrosus* (Hedw.) Spruce

<sup>10</sup> = *Hygrohypnum novae-caesareae* (Aust.) Grout in Ketchledge (1980).

<sup>11</sup> = *Isoetecium eumyosuriodes* Dix. in Ketchledge (1980).

<sup>12</sup> = *Cyrtomnium hymenophylloides* (Hüb) Nyh.

<sup>13</sup> = *Pottia starkeana* ssp. *minutula* (Schleich. ex Schwaegr.) Chamberl. in Eckel (1987).

<sup>14</sup> = *Sphagnum annulatum* (Schlieph. & Warnst. ex Warnst.) Mass & Isov. ex Mass in Crum and Anderson (1981).

<sup>15</sup> = *Sphagnum subsecundum* Nees ex Strum var. *platyphyllum* (Lindb.) Card. in Crum and Anderson (1981).

<sup>16</sup> = *Sphagnum cuspidatum* Ehrh. ex Hoffm. var. *serratum* (Schlieph.) Schlieph. in Crum and Anderson (1981).

**Table 2.** Watch List for Rare Mosses of New York State: 1988

Scientific Name <sup>1</sup>	Global Rank <sup>2</sup>	State Rank <sup>3</sup>
<i>Blindia acuta</i> (Hedw.) B.S.G.	G5	S3?
<i>Brachythecium digastrum</i> C. Müll & Kindb. ex Macoun & Kindb.	G3G5	SR
<i>Brachythecium erythrorrhizon</i> B.S.G.	G4G5	SR
<i>Bruchia flexuosa</i> (Sw. ex Schwaegr.) C. Müll. <sup>4</sup>	G5	S3
<i>Bryum alpinum</i> With.	G5	SR
<i>Bryum uliginosum</i> (Brid.) B.S.G.	G5	S3?
<i>Cirriphyllum piliferum</i> (Hedw.) Grout	G5	S3
<i>Didymodon rigidulus</i> Hedw.	G4G5	S3
<i>Didymodon tophaceus</i> (Brid.) Lisa	G5	S3
<i>Ditrichum ambiguum</i> Best	G3G5	SR
<i>Drummondia prorepens</i> (Hedw.) E.G.Britt.	G5	S3
<i>Fissidens fontanus</i> (B.-Pyl.) Steud. <sup>5</sup>	G5	S3?
<i>Grimmia pilifera</i> P.-Beauv.	G4G5	S3?
<i>Grimmia unicolor</i> Hook. ex Grev.	G5	S3?
<i>Hygrohypnum montanum</i> (Lindb.) Broth.	G4G5	S3?
<i>Hylocomium umbratum</i> (Hedw.) B.S.G.	G5	S3
<i>Isopterygiopsis muelleriana</i> (Schimp.) Iwats.	G5	S3?
<i>Neckera complanata</i> (Hedw.) Hüb.	G5	SR
<i>Platydicta jungermannioides</i> (Brid.) Crum	G5	S3
<i>Platydicta minutissimum</i> (Sull. & Lesq. ex Sull.) Crum	G4	S3?
<i>Pogonatum urnigerum</i> (Hedw.) P.-Beauv.	G5	S3
<i>Pohlia drummondii</i> (C. Müll.) Andr.	G5	SR
<i>Sphagnum pylaesii</i> Brid.	G5	S3?
<i>Thuidium allenii</i> Aust.	G3G4	SR
<i>Tomenthypnum nitens</i> (Hedw.) Loeske <sup>6</sup>	G5	S3?

<sup>1</sup> Nomenclature follows Crum and Anderson (1981) with the exception of that of *Sphagnum*, which follows Andrus (1980).

<sup>2</sup> GLOBAL RANK G3 = Either very rare and local throughout its range ((21 - 100) sites); G4 = Apparently secure throughout its range (but possibly rare in parts); G5 = Demonstrably secure throughout its range (however it may be rare in certain areas); GU = Status unknown.

<sup>3</sup> STATE RANK S3 = Rare in New York State (usually 21 - 100 extant sites); SU indicated Status Unknown; SR indicates that the species is reported from New York but their is no verified specimen exists; a "?" indicates a question exists about the rank.

<sup>4</sup> Includes *Bruchia sullivantii* Aust.

<sup>5</sup> Includes *Fissidens hallianus* (Sull. & Lesq. ex Aust.) Mitt.

<sup>6</sup> Includes *Tomenthypnum nitens* var. *falcifolium* (Ren. ex Nich.) Tuom., a variety that may be rarer than the typical variety in New York state.

# **Chapter 7.**

## **MANAGEMENT LAW**



# An Overview of Natural Areas Laws

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**Abstract:** States create systems of natural areas under several different legal systems; the most common are designated natural areas laws, typically containing provisions for dedication, land acquisition and the creation of inventories and registries. Thirty-one states currently have laws labelled specifically as natural areas laws. Other states rely on general authority or provisions in other natural resources conservation laws to establish a natural areas program.

Pages 219 - 221. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Natural areas laws are important tools in a state's conservation efforts. Thirty-one states have passed natural areas laws; however, these laws are not the only options available to protect natural areas. They are joined by specific land acquisitions clauses in endangered species laws, state charter provisions, and other natural resources laws. In this article, I will cover only those laws labelled as natural areas laws, or which contain language directed mainly at natural areas, referring only sporadically to other types of legislation. I have made no attempt to describe and analyze natural areas protection systems, nor have I made an attempt to evaluate the effectiveness of the various conservation options found in the laws.

In 1977, The Nature Conservancy carried out a comprehensive analysis of natural resources laws of the states under contract with the National Park Service (The Nature Conservancy, 1977). This publication includes summaries, by state, as of July 31, 1976, including a complete discussion of natural resources protection. An update of this treatment would be useful, but it is far beyond the scope of this paper.

## TYPICAL PROVISIONS OF NATURAL AREAS LAWS

A "typical" natural areas law defines the agency to carry out the provisions of the law. Sometimes it is a new division within an existing department or even a new governmental department. The laws often designate a commission to make recommendations or oversee the implementation of the law. The laws typically describe one or more options for preserving natural areas, including dedication of natural areas, land acquisition, creation of official registries, and related conservation activities, such as maintaining inventories and establishing regulations.

The major provisions of the various laws are indicated in Table 1. Several of the laws listed replace earlier natural areas laws, such as those for Illinois, Kansas, North Carolina and Oregon. Dedication options provided by the law are indicated for both private and public lands. One column indicates whether land acquisition is authorized by the law. Finally, I have indicated laws with provisions for formal "Registry," including those that have provisions for maintaining inventories of natural areas.

### Dedication:

Dedication refers to the signing of a document, sometimes called an "instrument of dedication," restricting the future use of the land in question to a natural area or nature preserve. Most states designate dedicated lands, either private or public, as "nature preserves." However, a few states have other designations: natural area preserves (Connecticut, Louisiana, Mississippi, Wisconsin), "preserve" (Iowa), "natural and scientific preserve (Kansas), natural area (Michigan),

"nature and historical preserve" (New York), and "heritage preserve" (South Carolina).

The statutes contain provisions severely restricting the states' ability to divest state-owned lands so dedicated. Regulation or legislative restrictions likewise restrict the use of private lands so dedicated. Many statutes provide tax reductions on private land dedicated as nature preserves or natural areas.

Pearsall (1984) summarizes the fourteen natural areas laws containing provisions for dedication (Arkansas, Delaware, Illinois, Indiana, Iowa, Kansas, Kentucky, Mississippi, North Dakota, Ohio, Oregon and South Carolina). Since 1984, three additional states have passed laws authorizing dedication: Louisiana (1987), North Carolina (1985), and Wisconsin (1985).

### Land Acquisition:

Acquisition of land is authorized in most of the natural areas laws; however, several states require authorization by a commission or even from the legislature for land purchase using state money. In some states, special funds have been set up for natural areas acquisition. An example is Arkansas, which recently passed a law providing funds from a real estate transfer tax.

### Registries and Inventories:

Seventeen of the 31 states with natural areas legislation have laws establishing a Registry, variously called a "Registry of Natural Areas," "Registry of Critical Areas," "Registry of Natural and Scientific Areas," and "Registry of Fragile Areas." Some states, *e.g.* Louisiana, Mississippi and Montana, require landowner permission for listing on the registry. Others, *e.g.* Maine, require the landowner to be notified. Many other states have no landowner consent or notification requirement. Illinois directs public agencies to "recognize" registered areas in their planning processes.

Most states' natural areas laws have some provisions for survey of lands in the state or maintenance of an inventory of suitable lands. Even in states without formal authorization for inventories, such programs are most likely necessary to carry out the other provisions of the laws.

### Other Laws with Natural Areas Provisions:

Many other state laws have implications for natural areas programs. Alaska, for example, has provisions for a system of natural areas in its state charter. Massachusetts has created a natural areas program under general authority. Florida has a strong planning law, which, when coupled with a special acquisition fund, enables the state to acquire natural areas. Maryland has a land acquisition clause in its endangered species law, whereby land acquisition is facilitated through use of a special fund authorized under separate legislation. In Pennsylvania, the Wild

Resources Conservation Act establishes "Public Wild Plant Sanctuaries" on public land with significant rare plant resources.

## CONCLUSIONS

Legal authorizations for some type of natural areas programs exist in most states, the most common being designated as "natural areas laws." In my opinion, the better laws allow for acquisition and management, not just for listing on a registry. However, I recognize that various state administrative units must work within their own political framework, so the strongest law possible is rarely passed. The laws allowing for dedication of land by both private and public parties pro-

vide valuable tools for conservation of land without acquisition. Likewise, provisions for inventory, even when acquisition and dedication are not authorized, supply important tools for future conservation of plants, animals and ecosystems.

## LITERATURE CITED

- Pearsall, S. 1984. Public dedication of nature preserves. *Nat. Areas J.* 4(1): 11-23.
- The Nature Conservancy. 1977. *Preserving our Natural Heritage. Volume II. State activities.* U.S. Government Printing Office, Washington, D.C.

**Table 1.** A Summary of Provisions of Natural Areas Laws.

State <sup>a</sup>	Title of Law <sup>b</sup>	Dedication Options		Land Acquisition	Maintain Registry
		Public	Private		
AR	Environmental Quality Act of 1973, Ch. 14, Sects. 9-1401 to 9-1416	yes	yes	yes	yes
CA	Significant Natural Areas (1981), Fish & Game Code Ch. 12, Sects. 1930-1913	no	no	no	yes
CO	Colorado Natural Areas Act (1977), Sects. 36-10-101 to 36-10-103	no	no	yes	yes
CT	Natural Areas Preserves (1969), Ch. 447, Sects. 23-1 to 23-5	yes <sup>c</sup>	no	no	yes
DE	Natural Areas Preservation System (1978), Tit. 7, Ch. 73, Sects. 7301-7312	yes	yes	yes	yes
GA	Georgia Natural Areas Act (1966; amended 1972), Tit. 12 Art. 4, Sects 12-3-90 to 12-3-94	no	no	no	yes
HI	Natural Areas Reserves System (1970; amended 1988), Ch. 195, Sects. 195-1 to 195-8	no	no	yes	no
IL	Illinois Natural Areas Preservation Act (1981; amended 1985) Ch. 105, Sects. 701-726	yes	yes	yes	yes
IN	Indiana Natural Preserves Act (1967; amended 1973), Tit. 14, Ch. 5, Sects. 14-4-5-1 to 14-4-5-11	yes	yes	yes	yes
IA	State Preserves Act (1964; amended 1987)	yes	yes	no	yes
KS	Natural and Scientific Areas Law (1985), Sects. 74-6607 to 74-6609	yes	yes	no	yes
KY	Nature Preserves System (1976; amended 1978)	yes	yes	yes	yes
LA	Louisiana Natural Areas Registry (1987), Tit. 56, Ch. 8, Sects. 56:1830-1861	yes	yes	yes	yes
ME	Critical Areas Act (1974), Tit. 5, Sects. 3310-3315	no	no	no	yes
MI	Wilderness and Natural Areas Act of 1972, Mich. Stats. Ann. 13.734(1-13)	yes	no	yes	no

State <sup>a</sup>	Title of Law <sup>b</sup>	Dedication Options		Land Acquisition	Maintain Registry
		Public	Private		
MN	Scientific and Natural Areas (1969)	yes	no	yes	yes
MS	Mississippi Natural Heritage Law of 1978 Sects. 49-5-141 to 49-5-157	yes	yes	yes	yes
MT	Montana Natural Areas Act of 1974 (amended 1987), Ch. 12, Sects. 76-12-101 to 76-12-117	no	no	yes	yes
NJ	Natural Areas Act (1961), Ch. 51	no	no	yes	no
	Natural Areas Systems Act (1975), Ch. 363	no	no	yes	yes
NY	State Natural and Historical Preserves Act (1970), Art. 45, Sects. 45-0101 to 45-0117	yes	no	yes	yes
NC	Nature Preserves Act (1985), Art. 9A, Ch. 113A, Sects. 113A-164.1 to 113A-164.11	yes	yes	yes	yes
ND	Nature Preserves Act (1975), Sects. 55-11-01 to 55-11-13	yes	yes	yes	yes
OH	Natural Areas and Preserves (1970), Ch. 1517, Sects. 1517.01 to 1517.99	yes	yes	no	yes
OK	Oklahoma Biological Survey (1987), Tit. 70, Sect. 3314	no	no	no	yes
	Oklahoma State Register of Natural Areas Act (1984)	no	no	no	yes
OR	Natural Heritage Program (1979), Sects. 273.561 to 273.591	yes	yes	no	yes
PA	Wild Resource Conservation Act (1982), Ch. 104, Sects. 5301-5314	no	no	yes	no
SC	Heritage Trust Program Law (1976), Ch. 17, Sects 51-17-10 to 51-17-140	yes	yes	yes	yes
TN	Natural Areas Preservation Act of 1971, Ch. 116, Sects. 11-14-101 to 11-14-307	no	no	yes	yes
VT	Natural Areas Law (1977), Ch. 83, Sects. 2606 to 2607	no	no	yes	no
	Fragile Areas Registry (1977), Ch. 158, Sects. 6551 to 6552	no	no	no	yes
WA	Natural Areas Preserves (1972; amended 1981), Ch. 79.70, Sects. 79.70.010 to 79.70.900	no	no	yes	yes
WI	Natural Areas Law (1985), Ch. 23, Sects. 23.26 to 23.29	yes	yes	yes	yes

<sup>a</sup> No Natural areas laws *per se* in the following states: WY, AL, AK, AZ, FL, ID, MD, MA, MO, NE, NV, NH, NM, RI, SD, TX, UT, VA, WV, and WY.

<sup>b</sup> Dates in parentheses are the dates of the law's passage.

<sup>c</sup> Connecticut law does not mention the word "dedication;" however, language in the law regarding "alienation" of a natural area suggests dedication.

# The Importance of Class I Air Quality Designation in Natural Area Protection

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**Abstract:** Class I air quality areas receive the most stringent air quality protection from new sources of air pollution. In August of 1977, Congress mandated that all national parks over 5,000 acres (2,025 ha), national wilderness areas over 6,000 acres (2,430 ha) and international parks be designated Class I areas. Individual states were to decide if Class I designation was appropriate for other areas. To date, no state has redesignated an area Class I, even though the designation provides substantial air quality protection. This paper explains the Class I designation, how the designation has led to significant resource protection, and describes the redesignation process.

Pages 222 - 223. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

In the early 1970's major provisions of the Clean Air Act were the subject of several lawsuits. One purpose of the Clean Air Act is "to protect and enhance the quality of the Nation's air resources as to promote the public health and welfare and the productive capacity of its population." Although National Ambient Air Quality Standards, which are uniform minimum national standards for air quality, are intended to serve at a level that protects the public's health and welfare, Congress was concerned that the entire country would become polluted up to the maximum legal level and that there would not be areas of a "pristine" nature left. With the NAAQS in place, new industrial sources of air pollution would no longer be able to locate in historically polluted areas and they would then seek to locate in clean air areas in order to comply with the standards.

In August of 1977, the Clean Air Act was amended to "prevent significant deterioration" in areas of the United States known to have air cleaner than the Standards required. These are known as the "Prevention of Significant Deterioration (PSD)" provisions of the Clean Air Act. The Act specifies five purposes for the PSD provisions:

1. to protect public health and welfare from any actual or potential adverse effect that may reasonably be anticipated, even if the NAAQS are met;
2. to preserve, protect and enhance air quality in areas of special national or regional interest;
3. to ensure that economic growth will occur in a manner consistent with the preservation of existing clean air resources;
4. to ensure that emissions from any source in any state will not interfere with any portion of the applicable implementation plan to prevent significant deterioration for any other state; and
5. to ensure that any decision to permit increased air pollution in any area to which the PSD program applies is made only after careful evaluation of all the consequences and after adequate procedural opportunities for informed public participation in the decision-making process.

The Act established procedures to assist each state to determine the appropriate level of air quality for areas within it. The PSD provisions divided the country into three classes. In Class I areas the most stringent air quality rules apply. In Class II and Class III areas additional air pollution is allowed above ambient air quality conditions that are associated with well-planned growth. Congress specified the initial classification of lands, mandating that certain areas of special national significance become Class I areas. These include all national parks over

6,000 acres (2,430 ha), national wilderness areas over 5,000 acres (2,024 ha), national memorial parks over 5,000 acres (2,024 ha) and international parks that were in existence at the time of the passage of the amendments.

Only a few Class I areas are located in the Northeast. There are two Class I areas in the state of Maine: Acadia National Park and Moosehorn Wildlife Refuge, two in the state of New Hampshire: the Presidential Range and Dry River Wilderness Areas in the White Mountains National Forest, one in Vermont: the Lye Brook Wilderness in the Green Mountains, and one in New Jersey: the Brigantine National Wildlife Refuge. Congress designated all other areas with air quality below the standards as Class II areas and left the decision to individual states to either redesignate areas of special regional significance as Class I areas or lower the status of areas planned for intensive development down to Class III. Although there are numerous areas of the United States that would greatly benefit from Class I area protection, no state has yet redesignated any area as Class I.

The Clean Air Act defines significant deterioration by setting maximum allowable increments for the amount of additional air pollution that will be allowed over background conditions. The Act defines these allowable increments for two pollutants: total suspended particulates and sulfur dioxide. At the Environmental Protection Agency (EPA), an increment level is currently being proposed for nitrogen oxides. Table I lists the increment levels for each of these three classes.

In addition to the requirement that the allowable increments not be exceeded within a Class I area, there is a test for adverse impact that directly relates the impacts to air pollution sensitive resources of the area. Even if the amount of additional air pollution allowed above ambient air quality conditions (or the Class I increments) has not been exceeded and even if the proposed new source would not cause or contribute to concentrations of pollutants that exceed those increments, an air quality permit cannot be issued if it can be demonstrated to the satisfaction of the permitting authority that the proposed facility would have an adverse impact on a sensitive resource.

One significant part of the PSD provisions is the inclusion of the objectives of both preventing future impairment of visibility in Class I areas and remedying existing impairment due to man-made pollution. The reduction of visibility is one of the most easily perceived features of a polluted atmosphere. Research and monitoring conducted by the National Park Service show that scenic vistas are consistently perceptibly impaired by man-made pollution throughout the United States. The primary pollutant involved in most of the visibility degradation is sulfate particulate (Pitchford *et al.*, 1981; Malm *et al.*, 1985).

**Table 1.** Prevention of Significant Deterioration Allowable Increments (Maximum allowable increases in concentrations over the baseline in micrograms/m<sup>3</sup>)

Pollutant	Maximum Allowable Increase		
	Class I	Class II	Class III
<b>Particulate Matter</b>			
Annual geometric mean	5	19	37
24-hour maximum*	10	37	75
<b>Sulfur Dioxide</b>			
Annual arithmetic mean	2	20	40
24-hour maximum*	5	91	182
3-hour maximum*	25	512	700

\*Short-term maxima may be exceeded no more than once per year.

The EPA has a two-phased approach to visibility protection. Phase I addresses obvious visibility impacts that are caused by coherent plumes from point sources of the pollution. This is referred to as plume blight. Phase II will address the more complex issue of regional haze where numerous sources contribute to the problem of visibility degradation, many of which are located hundreds of kilometers from the Class I area.

States are required to amend their State Implementation Plans (SIP's), documents that outline their approach to implementing federal air quality rules, specify whether their Class I areas currently have existing impairment from plume blight and describe what controls might be imposed to correct bad situations.

The state of Vermont expanded its SIP revision, including rules on visibility, to also address the problem of regional haze. The state conducted an analysis of all existing sources in Vermont and concluded that, since they had no sources of air pollution and visibility in the Lye Brook wilderness Class I area was degraded, sources outside the region were the problem. They proposed that in order to remedy existing visibility degradation in the Vermont Class I area, sources in the Midwest had to be controlled. Thus, attempts to protect and improve regional haze conditions can have far reaching implications and may force more stringent controls on the highly industrialized Midwest.

Research conducted by the National Park Service (Malm, 1987; Malm and Molenaar, 1985) and others (Husar *et al.*, 1981) demonstrates

that rural air quality is also generally polluted. Since pollutants are transported long distances, pollutants of concern in Acadia National Park in Maine should be of concern throughout New England. The "Prevention of Significant Deterioration regulations" are important in the protection of natural areas. Although several Indian tribes have redesignated their reservation lands Class I, no state has been successful in redesignating an area as Class I to provide their natural resources the maximum level air quality protection.

The Clean Air Act provides guidelines to states for such redesignation in Section 164. This requires that the state: 1) give public notice, 2) notify federal officials if the land under consideration for redesignation includes federal lands, 3) issue analyses of the effects of redesignation, and 4) hold public hearings.

The threat of air pollution impacts to sensitive environments is a major concern to natural area managers in the Northeast United States and elsewhere. Air pollution is a regional rather than a local problem and it will require the cooperation of many states as well as assistance at the national level to resolve. States can contribute to the protection of natural areas on a smaller scale. Each state has the authority and a directive to consider areas within each state for Class I area protection, and action is badly needed.

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# The Massachusetts Wetlands Protection Act: Protecting Rare Wildlife Habitat

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**Abstract:** Recent amendments to Massachusetts' Wetlands Protection Act add wildlife habitat as the eighth interest protected under the Act. Legal protection is afforded wildlife habitat rather than populations or individual animals. New regulations promulgated as the result of these amendments offer strong protection for rare wetlands wildlife by prohibiting development projects from having any adverse effects on wetland habitat used by 105 species of vertebrate and invertebrate wildlife classified as "Endangered," "Threatened" or "Species of Special Concern" in Massachusetts. Pursuant to these new regulations, the Massachusetts Division of Fisheries and Wildlife (MDFW) has prepared Estimated Habitat Maps that delineate approximate habitat boundaries for 840 occurrences of state-listed wetlands wildlife. Developers whose proposed projects would alter wetland resource areas located on Estimated Habitat Maps must submit project plans to MDFW for review prior to applying to local conservation commissions for project approval. Project review by MDFW occurs in two stages. It is determined: 1) if a proposed project falls within the *actual* habitat of a rare wetlands wildlife species, and 2) if the project will have any short- or long-term adverse effects on the actual habitat of that species. The regulations do not protect most upland habitats used by wetlands wildlife, nor is it clear if activities in the 100-foot buffer zone can be construed as having adverse effects on habitat in adjacent wetlands. The degree of protection afforded to rare wetlands wildlife by these regulations is directly related to the amount and quality of data on occurrences of rare species contained in MDFW's Natural Heritage data base. More systematic collection of data on distributions and habitat use by rare wetlands wildlife species is necessary to allow more accurate and complete delineation of habitat boundaries.

Pages 224 - 227. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Prior to 1986, Massachusetts' Wetlands Protection Act (Massachusetts General Laws Chapter 131, Section 140) protected a total of seven wetland resource interests: 1) public and private water supply, 2) ground water supply, 3) flood control, 4) storm damage prevention, 5) prevention of pollution, 6) protection of shellfish and 7) protection of fisheries. In 1986 the Act was amended to include wildlife habitat in wetlands as an eighth interest to be protected.

Regulations implementing the new amendment were drafted by the Massachusetts Department of Environmental Quality Engineering (DEQE) with assistance from a technical advisory committee composed of representatives from the Massachusetts Department of Fisheries, Wildlife and Environmental Law Enforcement, Department of Transportation, Coastal Zone Management office, environmentalists, lawyers and representatives of the real estate industry. DEQE promulgated the new regulations on 16 October 1987, and they became effective on 1 November 1987. The new regulations include particularly strong protection for habitats of rare, state-listed, wetlands wildlife, and they constitute an important new tool for preserving habitats for rare and endangered species in Massachusetts. This paper summarizes the new wetlands regulations and the protection they provide for habitats of rare and endangered wildlife.

## THE REGULATIONS

Massachusetts' Wetlands Protection Act identifies and provides varying levels of protection for 15 different types of wetland resource areas (Table 1). These include four categories of inland wetland resource areas and eleven coastal wetland resource areas.

The Massachusetts Wetlands Protection Act was amended in 1986 to protect wetlands wildlife *habitat*, not wildlife populations or individual

animals. It defines wildlife habitat as: "those areas subject to (the Act) which, due to their plant community composition and structure, hydrologic regime, or other characteristics, provide important food, shelter, migratory or overwintering areas, or breeding areas for wildlife."

**Table 1.** Summary of types of wetland resource areas protected by the Massachusetts Wetlands Protection Act.<sup>a</sup>

### Inland Wetlands

- Banks (of streams, rivers, ponds, or lakes)
- Bordering Vegetated Wetlands (wet meadows, marshes, swamps and bogs)
- Land under Water Bodies (streams, rivers, ponds, or lakes)
- Land Subject to Flooding (areas bordering streams, rivers, ponds, or lakes; also isolated areas flooded at least once a year)

### Coastal Wetlands

- Land under the Ocean (includes land under estuaries)
- Coastal Beaches (includes tidal flats)
- Coastal Dunes
- Barrier Beaches
- Coastal Banks
- Rocky Intertidal Shore
- Salt Marshes
- Land under Salt Ponds
- Land Containing Shellfish
- Anadromous/Catadromous Fish Run

<sup>a</sup>Resource areas include a 100-ft buffer zone except for: 1) land under any water body, 2) land subject to tidal action, 3) land subject to coastal storm flowage, and 4) land subject to flooding.

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The new regulations pursuant to the Act provide two levels of protection for wildlife habitats. For more common species of wetlands

wildlife, the regulations allow small or temporary alterations of wetland habitats. Protection is much stronger, however, for habitats of rare species of wetlands wildlife. Rare wildlife is defined as "those vertebrate and invertebrate animal species officially listed as Endangered, Threatened, or of Special Concern by the Massachusetts Division of Fisheries and Wildlife (MDFW):" For these species, the regulations state that "a project that would alter a wetland resource area shall not be permitted to have adverse effects on rare species habitat".

### WHICH SPECIES ARE PROTECTED?

With the institution of the new regulations, the Massachusetts Wetlands Protection Act now protects the habitats of 105 of the 162 species of rare wetlands wildlife that are classified as "Endangered", "Threatened", or "Species of Special Concern" in Massachusetts (Natural Heritage and Endangered Species Program, 1988). Included among these 105 species of rare wetlands wildlife are 45 vertebrate and 60 invertebrate species. Of these, 28 species are classified as "Endangered", 14 as "Threatened" and 63 as "Species of Special Concern."

Unfortunately, the Act does not provide protection specifically for the habitats of plant species classified as "Endangered, Threatened," or "Species of Special Concern" by MDFW. However, a number of occurrences of these rare plants occur within the habitats of rare wetlands wildlife and, therefore, receive indirect protection.

The habitats of a wide diversity of wetlands wildlife are now protected by the Act. These include freshwater wetlands used by 14 state-listed species of birds and wetland habitats of 14 state-listed species of amphibians and reptiles. Because wetland resource areas are broadly defined to include coastal beaches and dunes, strong protection is extended to the habitats of state-listed beach-nesting birds such as the Piping Plover (*Charadrius melodus*) and Least Tern (*Sterna antillarum*). The new regulations protect the habitats of 60 species of state-listed invertebrates that occur in wetlands, including eleven species of mollusks, eight crustaceans, one bryozoan, one leech, 24 damselflies and dragonflies, two beetles and 13 species of butterflies and moths.

### THE REGULATORY PROCESS

There are generally four participants involved in the regulatory process that deals with habitats of rare wetlands wildlife: 1) project proponents, 2) town or city conservation commissions, 3) the Massachusetts Department of Environmental Quality Engineering (DEQE) and 4) the Massachusetts Division of Fisheries and Wildlife (MDFW).

Project proponents are individuals, businesses, or organizations that propose activities that would alter wetlands or adjacent buffer zones that are subject to regulation by the Wetlands Protection Act. They are required to obtain permits from the appropriate local conservation commission(s) before undertaking projects that might alter wetland resource areas.

Conservation commissions administer the Wetlands Protection Act at the local level by reviewing project proposals and then granting or denying permits for projects that would alter wetlands. Each of the 351 cities and towns in Massachusetts appoints its own volunteer conservation commission composed of three to seven local citizens. Conservation commissions make decisions within the framework of regulations promulgated by DEQE. When rare wildlife is involved, the decision of the conservation commission is largely determined by a scientific opinion provided by MDFW.

DEQE is the state agency responsible for promulgating, administering and enforcing the Wetlands Protection Act. DEQE also reviews appeals to permit decisions made by conservation commissions.

MDFW plays a scientific advisory role in the regulatory process. It determines whether a project occurs within the habitat of a state-listed species of wetlands wildlife, and, if so, whether the project will have an adverse effect on the species.

Delineating and mapping the habitats of rare species of wetlands wildlife is the responsibility of MDFW. Through its Natural Heritage and Endangered Species Program, MDFW maintains a mapped and computerized data base on the occurrences of rare species throughout Massachusetts. Prior to November 1, 1987, MDFW prepared and distributed Estimated Habitat Maps (Fig. 1) for all 840 recent occurrences of rare wetlands wildlife in the state. In January 1989 these maps were revised to include 370 additional occurrences of rare wetlands wildlife. "Recent" occurrences are those which have been documented within the past 25 years. Estimated Habitat Maps delineate the approximate boundaries of the habitat of each occurrence of a rare species, based on information contained in MDFW's Natural Heritage data base and on the opinion of its staff. Estimated Habitat Maps are plotted on 1:25,000 scale U.S.G.S. topographic maps, which are then distributed to appropriate local conservation commissions. Conservation commissions, in turn, make the maps available for review by project proponents and other interested parties. The maps have also been reduced and reproduced in an atlas format, available for sale to consulting firms, conservation organizations and interested individuals. The maps will continue to be updated annually to reflect new information that is received by MDFW.

As noted above, Estimated Habitat Maps are only approximations of the actual habitat areas used by rare wildlife species. These maps do not delineate exact locations of rare species, nor do they identify which species are present on a given map. They do, however, serve two very important purposes. First, they provide project proponents and town conservation commissions with approximate locations of rare species habitats. Second, they serve as "red flags" to insure that MDFW has an opportunity to review and comment on proposed projects that would occur within or near the actual habitat of a rare species of wetland wildlife.

Specific responsibilities of project proponents, conservation commissions and MDFW under the new regulations protecting habitat for rare wildlife are as follows:

- 1) The project proponent is required to consult the appropriate Estimated Habitat Map to determine if the proposed project will alter a wetland within the mapped Estimated Habitat of a rare species. If so, the project proponent is required to file a document called an "Appendix A" with the Natural Heritage and Endangered Species Program of MDFW. The Appendix A and supporting materials contain information on the nature and exact location of the proposed project. Submission of an Appendix A initiates a review process by the staff of the Natural Heritage and Endangered Species Program of MDFW.

- 2) After an Appendix A has been filed, MDFW must first determine if the proposed project occurs within the actual habitat of a rare species. This determination is based on information contained in the Natural Heritage data base and on the knowledge of MDFW staff. Often a site visit and review of detailed project plans are required.

- 3) If MDFW determines that the proposed project does occur within the actual habitat of a rare species, then it is required to notify the project proponent of the species that occurs at that site and its specific habitat needs. This notification may initiate a consultation or negotia-

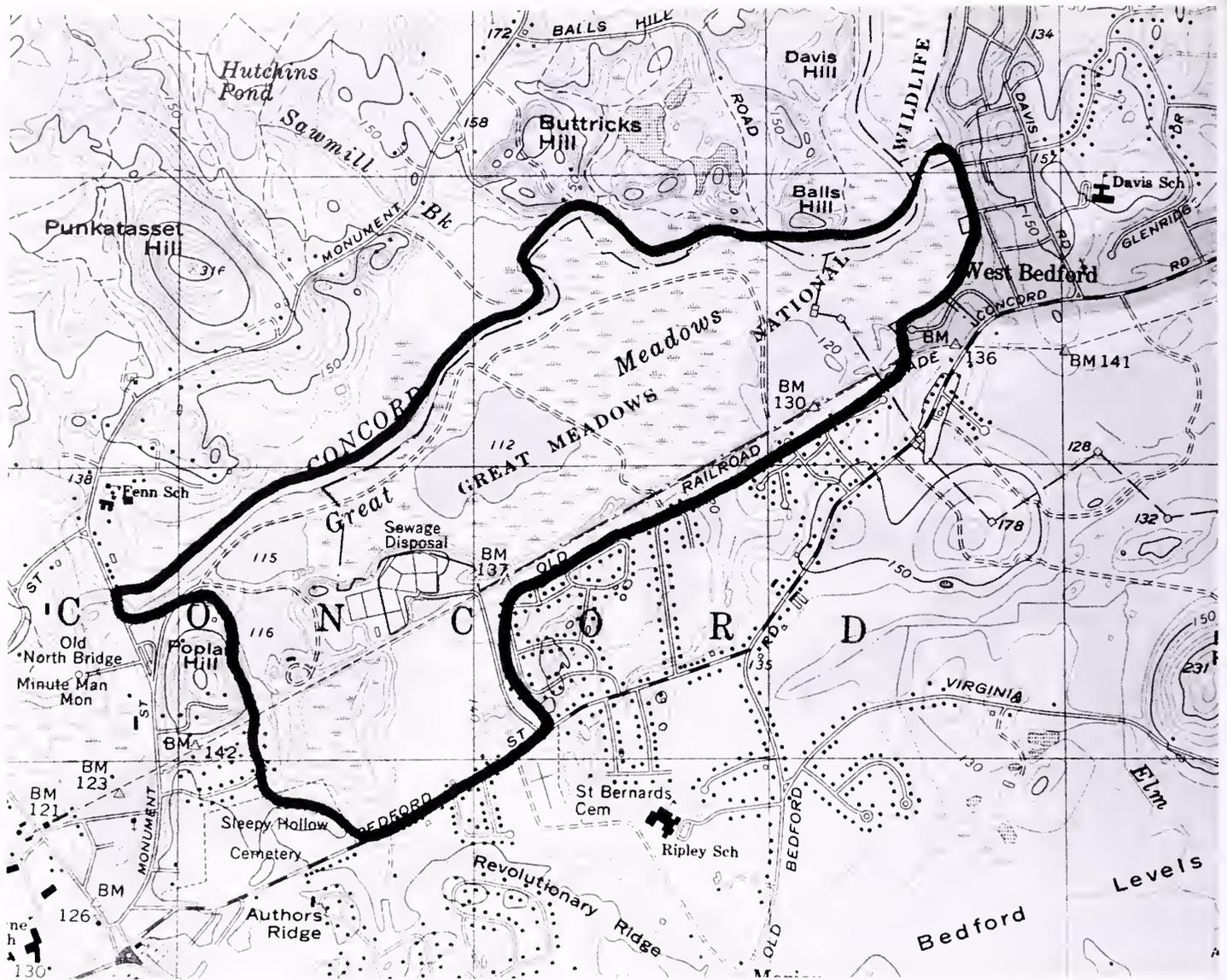


Fig. 1. Estimated Habitat Map, delineating the approximate boundaries of habitat used by one or more species of rare wetlands wildlife in Massachusetts. Prepared by the Natural Heritage and Endangered Species Program of the Massachusetts Division of Fisheries and Wildlife.

tion process, through which the project proponent may elect to modify the design or location of the project to avoid having an adverse effect on rare species habitat.

4) The final responsibility of MDFW is to determine whether the proposed project will have any short-term or long-term adverse effects on the habitat of any rare species of wetlands wildlife. This determination is based on knowledge of the species' biology and habitat requirements. A written opinion is prepared by the staff of the Natural Heritage and Endangered Species Program and is provided to the project proponent and the appropriate town conservation commission. The opinion of MDFW is presumed to be correct, although it can be rebutted by a clear demonstration of contrary information before the conservation commission.

5) Conservation commissions administer and enforce the wetlands regulations at the local level. They must decide whether or not to grant orders of condition that will allow projects to proceed. They are guided by strict performance standards contained in the regulations, and so must permit only projects that will have no adverse effects on the habitats of rare species. Again, the scientific opinion provided by MDFW is presumed to be correct.

## THE REGULATIONS AT WORK

As noted above, the new regulations that protect habitats of wetlands wildlife went into effect on 1 November 1987. In January 1988 MDFW hired a full-time wetlands biologist to process Appendix A's filed pursuant to the new regulations. Between 1 November 1987 and 1 January 1989, Appendix A's for a total of 439 projects were filed with MDFW, and determinations were completed on 433 of these. Of these 433 projects, 306 (71%), were determined by MDFW not to occur within the actual habitat of a rare species. Of the remaining 127 projects that did occur within the actual habitat of a rare species, 123 (97%) were determined unlikely to have any significant adverse effects on rare species habitat. This included 53 projects for which modifications were requested or mitigation measures carried out during the review process. Mitigation included such things as measures to control erosion and sedimentation, changes in the timing of construction, and project modifications that redirected construction activities away from wetland areas or preserved a natural vegetation buffer zone around wetlands. Finally, there were only four projects during this period that were determined to threaten adverse effects on rare species habitat if allowed to proceed.

Three examples serve to illustrate how the new regulations protect habitat of rare wetlands wildlife. The first concerns a proposal to construct a house on a site that is classified as a coastal bank and is a nesting site for Diamondback Terrapins (*Malaclemys terrapin*), a species classified as "Threatened" by MDFW. MDFW reviewed the project and issued an opinion that the project would have an adverse effect on the terrapin habitat by physically disrupting the coastal bank on which the animals nested. The town conservation commission accepted this opinion, and refused to grant an Order of Conditions to allow the house to be built.

The second example involves a proposal to build beach stairs and a cabana on a coastal beach used for nesting by Piping Plovers, a species listed as "Threatened" by both the U.S. Fish and Wildlife Service and MDFW. MDFW reviewed the project proposal and determined that the cabana would have an adverse effect on the habitat of the Piping Plover by physically altering a portion of the habitat at the site and by creating a disturbance to the birds. As a result of this determination, the project proponent withdrew his request to build the cabana, but was permitted by the local conservation commission to construct the beach stairs.

In the third example, an applicant submitted an Appendix A to MDFW that proposed to construct a road through a 7.5-acre wetland on the island of Nantucket. The wetland is inhabited by the Spotted Turtle (*Clemmys guttata*), a species listed as a "Species of Special Concern" by MDFW. MDFW reviewed the project proposal and determined that the project would have an adverse effect on Spotted Turtle habitat by altering or destroying a significant portion of the wetland. Subsequently, the project proponent elected not to proceed with the project, and sold the property to a local land conservation trust.

## DISCUSSION AND CONCLUSIONS

Although habitat loss is the most critical threat to native plants and wildlife in Massachusetts, there is currently no comprehensive law protecting populations and habitats of all rare and endangered species in the state. The new wetland regulations described in this paper do, however, provide strong protection for habitats of 105 species of rare wetlands wildlife, and, in doing so, are tantamount to an endangered species habitat protection act for 65% of the wildlife species classified as "Endangered", "Threatened" or "Species of Special Concern" in Massachusetts. We are optimistic that these regulations represent an important new tool for preserving habitats of rare and endangered wildlife in Massachusetts.

Unfortunately, these regulations do not protect all of the habitats used by rare species of wetlands wildlife. Many species, although dependent on wetlands for their survival, make significant use of upland habitats that are not protected by the Wetlands Protection Act. For example, the Act protects the aquatic habitats of the larval forms of rare dragonflies and damselflies, but it does not protect terrestrial habitats, such as forest openings, that may be used by the adults. Wood Turtles (*Clemmys insculpta*) often occur in unprotected terrestrial habitats adjacent to streams and rivers. Rare Ambystomid salamanders such as the Blue-spotted Salamander (*Ambystoma laterale*) and the Jefferson Salamander (*A. jeffersonianum*), both classified as "Species of Special Concern", represent a special example of this problem. These species require ephemeral vernal pools in the spring for mating, egg-laying and larval development, but spend the rest of the year in forested upland habitats, where they inhabit the leaf litter and upper layer of the soil. Only some vernal pools are classified as wetland

resource areas, and thus are protected under the Act. Furthermore, protection is usually not afforded adjacent upland habitats where the salamanders spend the majority of the year.

Another potential limitation of the new regulations involves the question of buffer zones. The jurisdiction of the Wetlands Protection Act extends to a 100-foot buffer zone around wetland resource areas in instances where it is determined that activities in the buffer zone will result in alteration of adjacent wetlands. Protection of habitats of rare wetlands wildlife can be extended to the buffer zone only when it is determined that activities there will adversely affect rare wildlife habitat in adjacent wetlands. It is unclear whether factors such as human disturbance can be construed to have an adverse effect on wildlife habitat. Certainly the clearing of land, construction of houses, and presence of people and pets within the buffer zone could represent disturbances that might have significant adverse effects on adjacent habitats of wetland bird species such as the American Bittern (*Botaurus lentiginosus*) or Pied-billed Grebe (*Podilymbus podiceps*). Such disturbance could be severe enough to alter the activity patterns of such birds or even cause them to abandon the wetland. However, it is unclear whether the new regulations recognize disturbance as a factor that can alter wildlife habitat, or if the jurisdiction of the Act includes activities in the buffer zone that are a source of human disturbance. This determination will, undoubtedly, be made in the future on a case-by-case basis as projects come under review by MDFW or DEQE.

Administration of the wetlands regulations protecting rare wildlife habitat is based on data contained in MDFW's Natural Heritage data base. At present, all 50 of the United States have, or are developing, mapped or computerized Natural Heritage data bases (Roush, 1985) in cooperation with The Nature Conservancy. Thus, most states have the foundation for developing their own legal protection, similar to that in Massachusetts, for the purpose of identifying and preserving habitats for rare and endangered species of wetlands wildlife.

The degree of protection that can be afforded state-wide populations of rare wetlands wildlife by these new regulations is directly related to the quantity and quality of data on occurrences of rare species recorded in MDFW's Natural Heritage data base. Occurrences must be entered into the Natural Heritage data base before they can be mapped on Estimated Habitat Maps. Later, during the project review process, MDFW staff members are both unable and unwilling to issue a determination of adverse effect on rare wildlife habitat without adequate knowledge of the distribution of the rare species and its habitat at that site. A record of a rare turtle or salamander crossing a road between two wetlands is a good clue that all or a portion of those wetlands provides habitat for rare wetlands wildlife, but such a sighting does not constitute sufficient data for determining the impacts of a construction project on the wildlife habitat within those wetlands. More intensive and systematic collection of data on distributions of rare wetlands wildlife and their habitats is necessary to allow more accurate and complete delineation of habitat boundaries and more effective use of the regulations to protect these habitats.

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## **Chapter 8.**

### **MANAGEMENT IN PRACTICE**



# Problems Encountered while Mimicking Nature in Vegetation Management: An Example from a Fire-prone Vegetation

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**Abstract:** Preservation of natural areas often requires management. Techniques that mimic natural disturbance generally seem to work best. Fire, for example, is used to manage California chaparral, a "fire-adapted" vegetation type. Prescribed burns, however, can differ from the natural fire regime in intensity, season, frequency and environmental conditions. Variation in each parameter can have considerable impact on vegetation recovery. The greatest impact is on regeneration of plant populations from dormant seeds resident in the soil, but post-fire resprouting by some species also can be considerably reduced. This paper addresses deviations of managed fires from the natural fire regime and the subsequent problems that can be encountered managing California chaparral.

Pages 231 - 234. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Protected natural areas vary in the intensity and frequency of management needed, depending on the vegetation type and how different it is from the desired composition. Areas surrounding a protected site may be highly modified, and provide a source of undesired species. Few places exist unaltered by past activities of man, like farming, grazing and logging—disturbances that may have allowed encroachment by a number of non-native species. In many areas our landscapes have become so modified that many natural processes have been drastically altered. Wildfires, for example, are limited near urban areas, in some places, due to a fire-suppression policy, but in others due to breaks in the landscape caused by development. Management may be necessary in such cases to maintain the vegetation or return to a more natural state.

Attempts to mimic natural disturbances have been quite successful as a management technique (e.g. Gill, 1977; Johnson and Bradshaw, 1979; van der Valk, 1981; van der Valk and Pederson, 1989; Mallick and Gimingham, 1985; Keddy et al., 1989). Although some plant communities are quite resilient after disturbance, there must be a careful evaluation of vegetation response and survival. No action at all is actually an active management decision, because some species are favored by this policy. Ultimately, successful vegetation management depends upon carefully maintaining community dynamics in a way that preserves species diversity, favoring some at one time, others later. The goal then becomes one of insuring that management does not result in the accidental elimination of desirable species.

Ironically, any intervention to maintain diversity favors some species at the cost of other species (Parker, 1987a; Keddy et al., 1989). This process may be exploited in the name of "enhancing" populations of rare and endangered species, since elimination of undesirable species presumably allows enhancement of native species. At the same time, many variables are not controllable (e.g. seasonal weather). Preferred species may be reduced or eliminated by such factors, and space may be opened for invasive species to become established.

A further problem comes in the guise of mimicking natural disturbances. If management is similar in characteristics to a natural disturbance, then it is assumed that the native species will respond predictably. While this is often the case, there are circumstances where the management technique is not as good a mimic as is presumed, and expectations, in some cases, may color the evaluation of the project. Because disturbances change community structure and population age classes, long-term changes in the vegetation due to large shifts in species recovery may already be in operation prior to problems being discovered.

An objective of this article is to provide examples of management problems encountered when employing large disturbance techniques believed to mimic natural processes in California chaparral. The chaparral is an evergreen shrub-dominated community that experiences severe drought each summer and is prone to, and maintained by, fire. While few other community types may offer the exact problems encountered in chaparral management, the types of errors that can be made when managing this community are common to many areas. I will discuss vegetation management from the perspective of uncovering problems prior to the implementation of a policy.

## MANAGEMENT AND THE DYNAMICS OF PLANT COMMUNITIES

Local plant communities are collections of species that vary greatly in the temporal and spatial characteristics of their population dynamics. Managing such a collection requires an understanding of individual species life histories, the influence of management on their population responses, and the longer-term interactions among the populations. Most treatments involve a disturbance of the habitat that affords a number of species an opportunity for regeneration. Development of a vegetation management policy should involve the seeking of an understanding of what to disrupt, how to disrupt, and what will regenerate afterwards.

In the case of chaparral, the dominant shrubs are burned, opening up the habitat for regeneration and reducing fuel for wildfires. Prescribed burning therefore has been believed to be the ideal way of managing chaparral vegetation, but there are some critical reassessments in progress in response to occasional problems with regeneration.

### Problems Where Words Stand for Concepts:

Many of the problems with chaparral management are based upon a misunderstanding of our use of words that seem clearly defined at first. For example, "fire-adapted" and "regeneration" actually are more properly thought of as concept-words that encompass several different characteristics. Because managers may come from a variety of backgrounds, they may inadvertently interpret these concept words literally; this becomes one of the major problems in vegetation management, and may be blamed on a failure of researchers to clearly express their meanings.

Under natural conditions, chaparral can experience an intense canopy fire and respond with considerable growth and regeneration within a year (e.g. Hanes, 1977; Keeley and Keeley, 1988). Actually, chaparral is adapted to a particular fire regime (Gill, 1975; Gill and

Groves, 1981). The community does not tolerate all fire, but is adapted to certain types of fires with a given range of intensities, that come at a particular season of the year, within a range of frequencies (Fig. 1) (Gill and Groves, 1981). Environmental conditions at the time of a fire also affect the response of the vegetation (Parker, 1987a). Chaparral may not respond well to fires out of season, or that occur too often, at too low an intensity or under moist conditions.

## FIRE REGIME COMPONENTS

### TYPE

e.g. Surface vs Canopy

### INTENSITY

### FREQUENCY

### TIMING

e.g. Dormant season vs Growing Season  
Dry season vs Wet Season

### SOIL CONDITIONS

e.g. Wet soil vs Dry Soil  
Soil Type

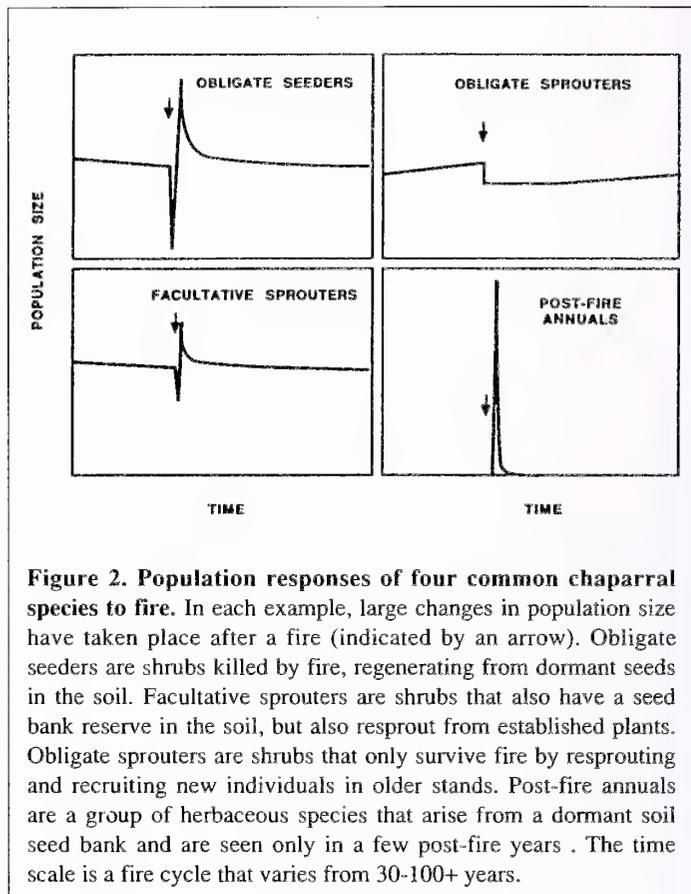
**Figure 1. Fire regime and the components that make up the characteristics to which plants adapt.** Variation in any one of the components can change the fire regime.

Similar layers of complexity exist for the various modes of regeneration found within a single community. Management practices will always interfere with some populations while promoting others, as in chaparral, where fire leads to recruitment of specific groups of species. Some chaparral species regenerate in older stands, while others regenerate only in post-fire years (Keeley and Keeley, 1988; Parker and Kelly, 1989). Chaparral life histories differ in the ways in which species respond to fire and in population changes afterwards (Fig. 2). Shrubs that resprout after fire but recruit seedlings only in older stands are termed obligate sprouters. Shrubs killed by fire and dependent upon seeds stored in the soil are termed obligate seeders. Facultative sprouters are shrubs that both resprout and recruit from dormant seed banks. Additional important groups are post-fire annuals and short-lived perennials, which are present only as dormant seeds at the time of a fire. Successful recruitment by other herbaceous perennials may be restricted to the post-fire years when sufficient light is available for growth prior to the closing of the canopy.

### Failures in Chaparral Management:

Only a few years after a fire, sprouters will again form a closed canopy if they are the dominant species. Vegetation at a site appears to regenerate, yet population sizes may be reduced by mortality in the fire or some species may have been eliminated. How is a manager to evaluate these types of possibilities? One approach is to examine the general population responses to a particular management treatment, and to determine which classes are most vulnerable to such a disturbance. For chaparral, the groups most vulnerable are those whose populations are most reduced by fire. Obligate-seeding shrubs and post-fire species lack an established population following a fire and are dependent upon recruitment from dormant seed banks the following spring (Keeley and Keeley, 1988; Parker and Kelly, 1989). These two groups include most

of the rare and endangered chaparral species, including over 40 taxa just between *Arctostaphylos* and *Ceanothus* (Parker, 1987a). When most of the sensitive species in the most vulnerable categories are found, greater care than usual will be required, and management should be aimed at the needs of those species.



**Figure 2. Population responses of four common chaparral species to fire.** In each example, large changes in population size have taken place after a fire (indicated by an arrow). Obligate seeders are shrubs killed by fire, regenerating from dormant seeds in the soil. Facultative sprouters are shrubs that also have a seed bank reserve in the soil, but also resprout from established plants. Obligate sprouters are shrubs that only survive fire by resprouting and recruiting new individuals in older stands. Post-fire annuals are a group of herbaceous species that arise from a dormant soil seed bank and are seen only in a few post-fire years. The time scale is a fire cycle that varies from 30-100+ years.

Historically, prescribed burning of chaparral was developed as a method of vegetation conversion for range improvement (Sampson, 1944; Biswell, 1974). With the advent of urban encroachment into steep, mountainous chaparral, prescribed burning is often advanced as a method of fuel reduction for safety purposes. Now that many watershed managers have incorporated preservation of species diversity as an important objective, they are being advised that prescribed burning will keep their vegetation healthy. Subsequently, prescribed burning is being used somewhat indiscriminately for three incompatible objectives, even though chaparral is adapted not to fire in general, but to a particular fire regime. In service of range development and fuel reduction goals, fires are often applied at variance with the natural fire regime, usually out-of-season under inappropriate conditions and far too frequently. Natural regeneration is disrupted and chaparral is ultimately replaced by other plant communities. To maintain the health of the community, attempts should be made to mimic the natural fire regime, so that chaparral regeneration proceeds normally.

A prescription for a burn in the chaparral should be made carefully, to make sure that the fire is controllable. Because of the amount of fuel available on steep terrain, burns are often restricted to winter when moisture and cool temperatures reduce the intensity and speed of the fire, yet obligate seeders and post-fire recruiters rely entirely on dor-

mant seed banks for regeneration. Winter burns differ in timing, intensity and other conditions from a natural fire.

Timing of fires seems to only be a problem for seed bank recruitment when fires occur late into the season (Kelly and Parker, unpublished). This has also been found to be the case in similar vegetation types in other parts of the world (e.g. Bond, 1984). Germination may fail when burns occur late in the winter at a time when soils are drying. Invasive and weedy species do establish, however, and resprouting shrubs become larger. The first season after fire, considerably more nutrients are available than in the second season. Consequently, the native vegetation does not germinate and establish from the seed bank until the second year, and undergoes considerable mortality from predation, loss to erosion and competition from established plants.

Reduction of fire intensity in prescribed burns affects species that have hard seed coats. Stimulation for germination usually requires a heat pulse in the soil. When fire intensity is reduced, heat movement in soil is greatly reduced, especially if the soils are moist; there may be insufficient heat to stimulate germination. For example, in southern California some stands are dominated by two shrubs, a facultative sprouter, *Adenostoma fasciculatum* H. & A., and an obligate seeder, *Ceanothus gregii* Gray. Burns experimentally conducted during January proved to be too low in intensity to stimulate germination in the hard-seeded *Ceanothus*, and the stands now lack this obligate seeder (Tom White, USFS, pers. comm.). Fires in this area are now set under conditions promoting greater intensities to insure recruitment from *Ceanothus*.

Recruitment is also limited in another class of species found in soil seed banks. In this group of species, seeds lack hard seed coats. Germination is cued in a variety of ways, but seeds absorb water each year when moisture is available whether or not germination has been stimulated. Prescribed burns generally occur when moisture is present in the soil to reduce fire intensity. When seeds become imbibed, however, their heat tolerance is reduced. Seeds of species able to tolerate considerable temperatures under dry conditions (as in the normal fire regime), die at temperatures as low as 70° C (Sweeney, 1956; Kelly and Parker, 1984; Parker, 1987a; Rogers and Parker, 1988). Variations in the fire regime affect other aspects of vegetation recovery as well. For example, as prescribed burns vary from the normal season of burning (fall), rates of mortality increase among established individuals of the facultative sprouter, *Adenostoma fasciculatum* (Biswell, 1974; Florence, 1985; Parker, 1987b). If fires become too frequent, populations of obligate seeders may be killed without further recruitment (Zedler et al., 1983). Increasing fire frequency can also restrict recruitment of obligate sprouters that normally occurs in older stands.

## CONCLUSIONS

Initial management treatments should be assumed to be highly experimental. Observations after deviations from the natural fire-regime in chaparral during prescribed burns indicate that recovery is not automatic but can be reduced, even in this fire-adapted vegetation. Different groups of species respond or are sensitive to different sets of conditions. Increased fire frequencies in chaparral will devastate certain species, but not significantly influence others over the short term. It is important to understand the potential response of a particular suite of species to a particular management treatment.

A necessary step in planning a management treatment is to determine the groups of species whose life history characteristics or adaptations are most vulnerable to a particular treatment, and proceed accordingly. Good examples of life history approaches are van der Valk's (1981), van der Valk and Pederson's (1989) wetland management clas-

sifications and Noble and Slatyer's (1977, 1980) "vital attributes" for species in fire-prone habitats (see Hobbs et al., 1984).

Typically, there will be certain species whose populations are more seriously reduced by a treatment, but there may also be other groups of species that will endure the same environmental "sieve." If a given treatment fails to regenerate a group of species, they may not survive to the next opportunity. For example, if fires fail to regenerate obligate seeders, these shrubs are then eliminated from the vegetation. It is fundamental to develop an understanding of species life histories and conditions important for each stage of their life cycles.

While the first step is to classify taxa into ecological response groups (like chaparral obligate seeders), it must be remembered that each species has unique life history characteristics. Because species respond differently, as in reproducing at different rates, having different mortality rates and special sensitivities to different ranges of temperature, all such eccentricities should be taken into account. Managers should avoid a uniformity of treatments and applications in order to help maintain species diversity and ecologically based groupings.

Finally, just as species are unique in their responses, so are the sites where they grow. For example, chaparral differs in its responses depending upon soil type, aspect and slope. One type of prescribed burn will not be effective for the gamut of chaparral diversity. A population on a serpentine soil site may suffer great mortality and poor recruitment, compared to plants of the same species on adjacent sandstone derived soil in the same fire (Parker, 1987b). The differences in soil conditions in this case resulted in differences in soil drying rates, nutrition and physiological status of the established plants. Thus, any community supporting the same groups of species may vary in responses when other conditions change.

Vegetation management encompasses many basic and applied disciplines. Opinions about any particular management tactic are consequently disparate. After experimentation with management techniques at a given site, there is no excuse for failing to understand responses of the community to disturbance, but until sufficient information is available, it is necessary to carry out limited and thoughtful treatments, monitored and evaluated, if valuable species are not to be lost from the association.

## ACKNOWLEDGMENTS

I would like to thank V. R. Kelly, C. Rogers, S. Hammer, D. Kelly and M. Wood for help in field work and laboratory experiments associated with chaparral. Some of the information presented here was from research supported by the Marin Municipal Water District, Corte Madera, CA, and the Rare Plant Project of the California Department of Fish and Game. I would also like to thank Jason Greenlee and an anonymous reviewer for their help in making the manuscript readable.

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# Mine Reclamation: Opportunity for Critical Habitat Development

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**Abstract:** Several wetlands, encompassing 22.4 hectares, were established on two mine sites (Guarnieri and Edwards) in western Pennsylvania at approximately the same cost that would have been incurred if the land had been returned to approximate original contour according to current regulations. A total of 34,250 wetland plants (10 species) 34 Kg of coontail (*Ceratophyllum demersum*), and 600 Kg of grass and legume seed plus 15,720 trees and shrubs representing 41 species were planted on both mine sites. Of the 833 plants (24 species) found on 22-1.0m<sup>2</sup> quadrats on the Guarnieri Mine, about 90%, (19 species) were volunteers. At the Edwards Mine, 127 individual plants (9 species) occurred on the five quadrats. Of these, approximately 40% representing seven species were volunteers in this wetland. The density of plants averaged 37/m<sup>2</sup> and 25/m<sup>2</sup> on the Guarnieri and Edwards mine sites, respectively. The wetlands provided additional habitat and wildlife diversity to the mine sites. During the first year, over 40 species of wildlife, including six species of special concern in Pennsylvania, used these wetlands and the surrounding grasslands for some phase of their life history.

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## INTRODUCTION

Throughout North America, thousands of hectares of land are disturbed annually due to surface coal mining, gravel mining, oil and gas exploration, drilling, highway construction, etc. All too often, these lands are reclaimed as monocultures or, at best, with several species of grasses and legumes. For example, Brenner (1984) calculated that of the 66 species of grasses, legumes and forbs that have been, or are currently being used for surface mine reclamation in the United States, only six are native to the northeast. Similarly, of the 52 species of shrubs used in surface mine reclamation (Vogel, 1981), only 14 are native to the northeastern coal fields. On the other hand, 41 of the 52 tree species recommended (Vogel, 1981) for the establishment of forest community and/or wildlife use are native to the northeast.

In actual practice, the majority of these mined or similarly disturbed lands are reclaimed as grasslands, which often lack the biological diversity that existed on the site prior to mining. Wetlands are a declining resource in the mid-Atlantic region of the United States (Tiner, 1987) as well as elsewhere in North America. The reclamation of surface coal mines provides us with an opportunity to establish relatively diverse wetland and terrestrial communities without a substantial increase in cost to the mine operator or developer. For over four decades, wetlands have been developed on previously mined lands, especially in the eastern (Brenner and Steiner, 1987) and midwestern (Jones et al., 1985) coal regions of the United States. Such wetlands provide habitat for a variety of wildlife species, including amphibians, reptiles, birds and mammals (Brenner and Kelly, 1981, 1982; Fowler et al., 1985; Brooks et al., 1985). Several studies have documented the use of these wetlands by nesting waterfowl and shorebirds (Brenner, 1973; Brenner and Mondok, 1979), as well as their use as resting and feeding habitat during migration (Brenner and Steiner, 1987). In addition to their value as wildlife habitats, many of these wetlands also support valuable warm water fisheries (Brenner, 1983).

Although restored wetlands can be used extensively by wildlife, there is often little, if any, planning for their integration into the overall reclamation of a site. The primary objective of this paper is to describe and discuss how different types of wetlands may be integrated into the overall reclamation of mined lands as well as to outline what additional costs may be incurred by coal companies. For over a decade, I have advocated the development of wetlands on mined lands (Brenner,

1973, 1983) including special areas designed for rare and endangered species (Brenner, 1985, 1986). In this paper, the construction and establishment of wetland vegetation and its use by wildlife will be discussed.

## METHODS

Prior to the actual construction of the wetlands discussed here, a 10-month planning evaluation and permitting process was completed to obtain necessary permits. An example of cooperation between the public and private sector, this project involved the Pennsylvania Department of Environmental Resources, Pennsylvania Fish Commission, Pennsylvania Game Commission, U.S. Fish and Wildlife Service, U.S. Soil Conservation Service, U.S. Army Corps of Engineers, Lawrence County Conservation District, Lawrence County Planning Commission, and the Western Pennsylvania Conservancy.

Approximately 22.4 ha (56 acres) of wetlands were established on two mine sites in northwestern Pennsylvania in 1987 as part of a mitigation and reclamation plan developed by Adobe Mining, Inc. On the site, hereafter referred to as the Guarnieri Mine, seven palustrine and one deep-water habitat totaling 19.2 ha were established on a 73.6 ha mine site to replace approximately 4.8 ha of palustrine wetlands associated with an adjacent active mine. The palustrine wetlands consisted of a sedimentation pond, a deep-water habitat and six other areas constructed during site reclamation. Water level control devices were installed on four of the palustrine wetlands, the sedimentation pond and the deep water habitat, so that water levels could be controlled for the management of aquatic vegetation. All water courses on the site were diverted into the wetland complex. Six of the wetlands were designed with emergency spillways, to allow flood waters to flow into the deep water habitat, which could then overflow into the sedimentation pond. The pond drained into a natural wetland complex owned and managed by the Pennsylvania Game Commission. A 0.2 ha island was also constructed approximately in the center of the deep water habitat to provide additional waterfowl nesting habitat.

On a second site, the Edwards Mine, three palustrine wetlands totaling 3.2 ha were constructed to replace a 1.8 ha palustrine wetland destroyed during mining of the site. As with the Guarnieri Mine, one wetland was a converted sedimentation basin in addition to two more

palustrine habitats constructed during reclamation. All water courses were diverted into these wetlands so that adequate water levels could be maintained.

One 1.4 ha palustrine wetland was constructed with irregular shorelines and five islands to increase the amount of edge, and thereby provide additional feeding and nesting areas for waterfowl. At both mine sites, non-wetland areas were planted with trees and shrubs interspersed among grasses and legumes to provide additional habitats for upland wildlife species.

### Construction:

Two different procedures were used for the construction and establishment of wetlands on these sites. At the Guarneri Mine, two wetlands sites received soils and sediment materials salvaged from natural palustrine wetlands that were to be destroyed by mining. Once the upper 60 cm of material was removed and stockpiled at the site, the lower clay and muck deposits could be removed, transported, and spread in the newly constructed wetland. Within five working days, the construction and transportation phase of the project was completed, and water was released into the sites. These areas did not receive supplemental plantings of wetland species. This project was scheduled to be completed either in late fall or early spring, but, due to the excessive delays in the permitting process, it was not completed until late April.

**Table 1.** Types and quantities of wetland plants planted or seeded on the Guarneri and Edwards Mines in 1987.

SPECIES	NUMBER PLANTED/LB. SEEDED	
	Guarneri Mine	Edwards Mine
Sago Pondweed ( <i>Potamogeton pectinatus</i> )	1500	1500
Wild Celery ( <i>Valisneria americana</i> )	1500	1500
Arrowhead ( <i>Sagittaria</i> sp.)	1500	1500
Duck Potato ( <i>Sagittaria rigida</i> )	1500	1500
Burreed ( <i>Sparganium eurycarpum</i> )	2500	2000
River Bulrush ( <i>Scirpus fluviatilis</i> )	2500	2000
Sweet Flag ( <i>Acorus calamus</i> )	2500	2000
Nodding Smartweed ( <i>Polygonum lapathifolium</i> )	2500	2000
Pickereel Weed ( <i>Pontederia cordata</i> )	2000	2000
Buttonbush ( <i>Cephalanthus occidentalis</i> )	150	100
Coontail ( <i>Ceratophyllum demersum</i> )	23 Kg	11 Kg
Switchgrass ( <i>Panicum virgatum</i> )	45 Kg	95 Kg
Reed Canary Grass ( <i>Phalaris arundinacea</i> )	114 Kg	68 Kg
Barnyard Grass ( <i>Echinochloa crusgalli</i> )	114 Kg	91 Kg
Lespedeza Mixture ( <i>Lespedeza</i> spp.)	36 Kg	36 Kg

The remaining six wetland sites received a total of 18,150 individual plants, (10 spp.) plus 23 Kg of coontail (*Ceratophyllum demersum*) and 218 Kg of grass and legume seeds. In addition, a total of 6,950 trees and shrubs, representing 17 species, were planted either adjacent to the wetlands or in upland areas, depending on site conditions (Table 2). The total cost of supplemental planting was \$9,842, which included \$7,326 and \$2,516 respectively, for plant material and labor. Reclamation costs were approximately the same as would have been incurred if the land had been returned to approximate original contour (AOC) as required by current regulations.

Prior to mining the Edwards Mine, soils and sediments were removed from a natural palustrine wetland and transported to a 1.4 ha wetland constructed on the same site. The same procedure was followed as at the Guarneri Mine for the removal, storage and transporta-

tion of materials between the natural and man-made systems except that scrapers rather than trucks were used to transport the material. Due to intermittent rainfall in mid-August, when this project was initiated, water from the active pit (pH 7.0-7.2) was pumped into the site as soon as all material was in place. The time lapse from the start of the project until there was sufficient water to maintain the system was approximately ten days. A total of 6100 individual wetland plants, 11 Kg of coontail, plus 7,450 trees and shrubs and 314 Kg of seeds, representing 37 species, have been planted on the site (Tables 1 and 2). The total cost of the establishment of vegetation at this entire 3.2 ha wetland complex was approximately \$7,226, of which \$5,488 and \$1,738 were for materials and labor, respectively. Reclamation costs to the company were approximately the same as would have been incurred if the land was returned to AOC as required by current regulations.

**Table 2.** Types and quantities of trees and shrubs planted on the Guarneri and Edwards Mines in 1987.

SPECIES	NUMBER PLANTED	
	Guarneri	Edwards
Red Oak ( <i>Quercus rubra</i> )	4300	300
Pin Oak ( <i>Quercus palustris</i> )	00	200
Bur Oak ( <i>Quercus macrocarpa</i> )	200	100
White Oak ( <i>Quercus alba</i> )	100	—
Swamp White Oak ( <i>Quercus bicolor</i> )	200	—
Scrub Oak ( <i>Quercus ilicifolia</i> )	400	—
Sawtooth Oak ( <i>Quercus acutissima</i> )	—	300
Washington Hawthorne ( <i>Crataegus phaenopyrum</i> )	500	700
American Plum ( <i>Prunus americana</i> )	150	100
American Crabapple ( <i>Malus coronaria</i> )	100	300
White Ash ( <i>Fraxinus americana</i> )	—	100
Silky Dogwood ( <i>Cornus amomum</i> )	900	1200
Gray Dogwood ( <i>Cornus racemosa</i> )	150	100
Buckthorn ( <i>Rhamnus cathartica</i> )	200	100
European Black Alder ( <i>Alnus glutinosa</i> )	2500	1500
Norway Spruce ( <i>Picea abies</i> )	250	500
Black Spruce ( <i>Picea mariana</i> )	100	50
Scotch Pine ( <i>Pinus sylvestris</i> )	100	350
White Spruce ( <i>Picea glauca</i> )	300	—
Austrian Pine ( <i>Pinus nigra</i> )	—	50
Tartarian Honeysuckle ( <i>Lonicera tatarica</i> )	—	200
Black Locust ( <i>Robinia pseudo-acacia</i> )	—	200
American Bittersweet ( <i>Celastrus scandens</i> )	—	200
Japanese Flowering Crabapple ( <i>Malus</i> sp.)	—	300
Sargent Crabapple ( <i>Malus</i> sp.)	—	300
Red Toringo Crabapple ( <i>Malus sieboldii</i> )	—	300

To provide additional nesting sites for wildlife, six and four wood duck (*Aix sponsa*) nesting boxes along with 17 and seven blue bird (*Sialia sialis*) boxes were placed on the Guarneri and Edwards Mines, respectively.

### Vegetation and Wildlife Assessments:

To assess the natural revegetation process of wetland species on the Guarneri Mine, 22m<sup>2</sup> quadrats were established on five wetlands, three of which received supplemental planting and two which received wetland soils and plant material from the adjacent mine site; 12m<sup>2</sup> quadrats were established on the Edwards Mine. On each visit to these sites, the numbers of the different species of wildlife observed were recorded.

## RESULTS

### Wetland Revegetation:

At both mines there was good survival and growth of vegetation on both the relocated wetlands as well as those receiving supplemental plantings of wetland species. Of 833 individual plants found on the quadrats (not including *Nitella* sp. and *Lemna minor*), 752 (or 90%) were volunteer species. *Nitella* and *Lemna* had a density of approximately 15/cm<sup>2</sup>. Nineteen of the 24 species identified on these wetlands were the result of volunteer invasions and/or generated from seeds or root stocks present in the materials transported from the

**Table 3.** Plant species found on quadrats on wetland sites on the Guarnieri and Edwards Mines during September and October 1987.

### NUMBER OF INDIVIDUALS

SPECIES	Guarnieri	Edwards
Arrow Tearthumb ( <i>Polygonum sagittatum</i> )	45	0
Water Smartweed ( <i>Polygonum amphibium</i> )	4	0
Spikerush ( <i>Eleocharis obtusa</i> )	118	8
Cockspur Grass ( <i>Echinochloa muricata</i> )	130	2
Bur-marigold ( <i>Bidens frondosa</i> )	115	19
Panic Grass ( <i>Panicum capillare</i> )	212	16
Bur-reed ( <i>Sparganium eurycarpum</i> )	45	3
Rice Cut Grass ( <i>Leersia oryzoides</i> )	5	0
Goldenrod ( <i>Solidago canadensis</i> )	23	0
Sedge ( <i>Cyperus strigosus</i> )	10	0
Shepherd's Purse ( <i>Capsella bursa-pastoris</i> )	7	0
Cup Grass ( <i>Eriochloa contracta</i> )	31	0
Red Clover ( <i>Trifolium pratense</i> )*	21	35
White Clover ( <i>Trifolium repens</i> )*	2	26
Cattail ( <i>Typha latifolia</i> )	47	2
Ragweed ( <i>Ambrosia artemisiifolia</i> )	0	16
Arrowhead ( <i>Sagittaria</i> sp.)*	4	0
Sago Pondweed ( <i>Potamogeton pectinatus</i> )	4	0
False Loosestrife ( <i>Ludwigia</i> sp.)	3	0
Birdsfoot Trefoil ( <i>Lotus corniculatus</i> )*	5	0
Pokeweed ( <i>Phytolacca americana</i> )	1	0
Foxtail ( <i>Setaria glauca</i> )	1	0
<i>Nitella</i> ( <i>Nitella</i> sp.)	15/cm <sup>2</sup>	10/cm <sup>2</sup>
Duckweed ( <i>Lemna minor</i> )	74/cm <sup>2</sup>	0
TOTAL	752	127

\*Planted during reclamation

adjacent mine site. The number of species/m<sup>2</sup> and the density of plants/m<sup>2</sup> on those wetlands that received transplanted wetland material was similar to those receiving supplemental plantings. Between eight and 13 different species occurred on the individual quadrats receiving supplemental plantings, compared to the eight or nine species that occurred on the transplanted sites. The average number of plants was 36 (32-41) and 38 (23-53) individuals/m<sup>2</sup> for areas that received supplemental plantings and those that received wetland soils, respectively. The principal species occurring on all wetlands on the Guarnieri Mine were: *Echinochloa muricata*, *Eleocharis obtusa*, *Bidens frondosa*, *Panicum capillare* and *Polygonum sagittatum* (Table 3). Sensitive fern (*Onoclea sensibilis*) and skunk cabbage (*Symplocarpus foetidus*) along with rushes (*Juncus* sp.), and sedges (*Carex* sp.) were observed on the transplanted wetlands, but they did not occur in the randomly selected quadrats. Throughout the trans-

**Table 4.** Birds observed at the Guarnieri and Edwards Mines during the spring and summer months of 1987.

FAMILY	SPECIES	Guarnieri	Edwards
Ciconiiformes	Great Blue Heron ( <i>Ardea herodias</i> )*	X	X
	Green Heron ( <i>Butorides striatus</i> )	X	X
Anseriformes	Tundra Swan ( <i>Cygnus colombianus</i> )	X	
	Canada Goose ( <i>Branta canadensis</i> )	X	X
	Mallard ( <i>Anas platyrhynchos</i> )	X	X
	Black Duck ( <i>Anas rubripes</i> )	X	
	Blue-winged Teal ( <i>Anas discors</i> )	X	
	Gadwall ( <i>Anas strepera</i> )	X	
	Wood Duck ( <i>Aix sponsa</i> )	X	X
	Ring-necked Duck ( <i>Aythya collaris</i> )	X	
	Greater Scaup ( <i>Aythya marila</i> )	X	
	Bufflehead ( <i>Bucephala albeola</i> )	X	
Podicipediformes	Ruddy Duck ( <i>Oxyura jamaicensis</i> )	X	
	Hooded Merganser ( <i>Lophodytes cucullatus</i> )	X	
Falconiformes	Pied-billed Grebe ( <i>Podilymbus podiceps</i> )	X	
	Red-tailed Hawk ( <i>Buteo jamaicensis</i> )	X	
Gruiformes	American Kestrel ( <i>Falco sparverius</i> )	X	
	Northern Harrier ( <i>Circus cyaneus</i> )*	X	X
	American Coot ( <i>Fulica americana</i> )	X	
Charadriiformes	King Rail ( <i>Rallus elegans</i> )*		X
	Herring Gull ( <i>Larus argentatus</i> )	X	
Columbiformes	Killdeer ( <i>Charadrius vociferus</i> )	X	X
	Spotted Sandpiper ( <i>Actitis macularia</i> )	X	X
	Upland Plover ( <i>Bartramia longicauda</i> )*	X	
	American Woodcock ( <i>Scolopax minor</i> )	X	
Coraciiformes	Willet ( <i>Catoptrophorus semipalmatus</i> )	X	
	Mourning Dove ( <i>Zenaidura macroura</i> )	X	X
Passeriformes	Belted Kingfisher ( <i>Ceryle alcyon</i> )	X	
	Red-winged Blackbird ( <i>Agelaius phoeniceus</i> )	X	X
	Eastern Meadowlark ( <i>Sturnella magna</i> )	X	
	Bobolink ( <i>Dolichonyx oryzivorus</i> )*	X	
	Song Sparrow ( <i>Meiospiza melodia</i> )	X	X
	Field Sparrow ( <i>Spizella pusilla</i> )	X	X
	Vesper Sparrow ( <i>Pooecetes gramineus</i> )*	X	X
	Chipping Sparrow ( <i>Spizella passerina</i> )	X	X
	Common Grackle ( <i>Quiscalus quiscula</i> )	X	
	Barn Swallow ( <i>Hirundo rustica</i> )	X	X
Tree Swallow ( <i>Tachycineta bicolor</i> )	X	X	

\*Listed as species of special concern in Pennsylvania.

planted wetlands, numerous dogwoods (*Cornus* sp.), black willow (*Salix nigra*) and buttonbush (*Cephalanthus occidentalis*) were observed sprouting from rootstocks present in the soil. In the wetlands that received supplemental plantings of wetland species, plants in the shallow and intermediate depth zones became established within two months of planting and those in the deep zone were beginning to become established at the end of that period. The success of these species, however, cannot be firmly decided until the end of the second growing season.

At the Edwards Mine the response of vegetation on the site that received the transplanted material was not as rapid as it was at the Guarnieri Mine. Natural revegetation by wetland plants did not occur until the following spring and summer months; however, the responses of those species that did regenerate naturally, as well as the supplemental species in the shallow water zone, resulted in excellent survival and growth.

In contrast to the Guarnieri Mine, only nine species were present in the five quadrats. Of those, two species *Trifolium pratense* and *Trifolium repens* probably invaded from adjacent upland areas. A total of 127 individual plants were found in this wetland, and 40% of these were volunteers. Between four and six species were on each individual quadrat with an average density of 25 (22-29) individuals/m<sup>2</sup>.

This wetland has dense grass and legume stands immediately adjacent to the relocated wetland. That dense buffer zone and elevated water levels on the transplanted wetland materials were undoubtedly factors contributing to a reduced number of volunteer species invading the area.

#### Wildlife Use:

Over 30 species of birds, including five species of special concern in Pennsylvania (Genoways and Brenner, 1985), were observed using wetlands and adjacent uplands on the Guarnieri site during the first year (Table 4). At least five broods of Canada geese (*Branta canadensis*), three mallard (*Anas platyrhynchos*) broods, and two wood duck (*Aix sponsa*) broods were observed using this wetland complex. On separate visits during the summer months between 100-300 Canada geese and flocks of 100 ducks, including mallards, black ducks (*Anas rubripes*), blue-winged teal (*Anas discors*), and wood ducks, along with approximately 20 spotted sandpipers (*Actitis macularis*) and three great blue herons (*Ardea herodias*) and two green herons (*Butorides striatus*) were observed feeding on these wetlands and in adjacent uplands. Two of the six wood duck nesting boxes and 13 of the 17 blue bird boxes were used by nesting tree swallows the first year. A total of 64 tree swallows (*Tachycineta bicolor*) and 21 wood duck fledglings were produced.

The use of the Edwards Mine by wildlife was not as great as on the Guarnieri Mine, with only 20 species of birds, including three species of special concern (Genoways and Brenner, 1985) observed, even though the site was visited frequently (Table 4). These wetlands were used for feeding on a regular basis by 25-50 Canada geese and 15-25 ducks including mallards, blue-winged teals, and wood ducks. Twenty tree swallow fledglings were produced from four of the seven blue bird nest boxes set out at the Edwards Mine during the first year.

In addition, three breeding pairs of bobolink (*Dolichonyx oryzivorus*), a species of special concern in Pennsylvania, were observed on surrounding grasslands at both mines. In addition to birds, the areas were also used by several mammal species, including muskrat (*Ondatra zibethicus*), mink (*Mustela vison*) raccoon (*Procyon lotor*) and white-tailed deer (*Odocoileus virginianus*).

## SUMMARY

On both the Guarnieri and Edwards Mine sites, man-made and reconstructed wetlands provided additional habitat diversity. The responses of both planted and spontaneous vegetation and the use of these areas by wildlife will continue to be studied over the next several years to determine the feasibility and economic benefits of wetland relocation in more sites and/or across the overall region, but prospects at present seem excellent for further use of the techniques employed in this study.

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# Ecological Planning and Resource Management: A Necessary Partnership for Small Lake Restoration

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**Abstract:** Successful ecosystem management requires a balance between public concerns, governmental policies and assessment of the possible environmental consequences of management activities. This approach necessitates a thorough and progressive examination of field data before an actual management/restoration technique can be chosen. Critical reassessment following management is also required, to determine whether or not restoration goals have been met. In the Honeoye Lake Restoration Program, this basic philosophy has been applied to small lake management. Progress occurred each year, culminating in the summer of 1987 with the selection of weed bed harvesting as the major restoration technique applied to the lake. Other activities have included biweekly water quality monitoring, the formation of a lake association and working with unanticipated problems. These activities have provided information essential to a better understanding of lake ecology as well as a mechanism for the dispersal of this knowledge. The type of ecosystem management of small lakes described here also provides a generic model that may be applicable to other types of natural areas.

Pages 239 - 243. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Small freshwater lakes often support diverse and productive plant communities. As in non-aquatic associations, the composition and dynamics of these plant communities are influenced by physical and biotic environmental factors, successional tendencies and human impacts (Barnes and Mann, 1980). Eutrophication of lakes, a process that increases nutrient supply, results naturally from soil erosion, decay and the build-up of organic matter, but it can be accelerated significantly by man's activities such as shoreline cottage development, watershed deforestation, farming and absent or inadequate sewage treatment (Oglesby et al., 1975). These watershed activities collectively comprise the external loading factors, and partially account for troublesome periodic algal blooms, dense submerged weed beds and high coliform bacteria counts in a lake ecosystem.

Internal nutrient loading results from the organic sediments that have historically accumulated in the basin. Man also impacts small lakes through the introduction of plant and animal species, thermal discharges and the deposition of hazardous substances. The combined effect of such human activities has reduced the quality of many important lake resources. Whether it be for recreation, potable water or simply aesthetics, the affected lakes deserve protection and demand the best possible management and restoration programs.

## DESCRIPTION OF THE ENVIRONMENTAL PLAN

In the case of Honeoye Lake, an environmental plan was established following the hierarchic framework recommended by Westman (1985). The long-term program goal was to enhance the multiple-use quality of lake resources through nutrient reduction. This concept became policy in the narrative of the grant that ultimately sponsored all phases of the restoration program. The policy statement mandated a holistic approach to lake restoration... one that would clearly distinguish symptoms from causes and address the desirability of nutrient reduction through proper watershed management.

Four strategies were identified to achieve this policy. The first was to monitor existing lake water quality and pertinent watershed features. Such descriptive information is essential in understanding the ecological variability of small lakes from year to year and to establish a base line. The second strategy was to assess the degree of resource degrada-

tion that had already occurred. Critical problem areas (i.e., dense weed beds) had to be inventoried throughout the growing season to determine the suitability of potential restoration techniques. The third strategy reflected the subjective nature of judgments about degradation. For example, dense weed beds were considered a problem if they occurred adjacent to cottage development, but they might not be considered of concern if adjacent to undeveloped forested wetlands. In addition, weed beds might be a nuisance to the swimmer, but valued by the fisherman as a nursery area for aquatic life.

Management of a small lake must be sensitive to diverse sets of values among property owners, visitors and agencies involved in lake dynamics, and act, in a balanced way, to satisfy most resource users most of the time. A watershed questionnaire was developed to survey the public's perceptions and attitudes toward the Honeoye Lake resource. The final and perhaps most important strategy was to involve the public in the restoration program. General education through town meetings and active participation in a newly formed lake association could then allow a resident to play a role in the planning and implementation process.

## METHODS

To fulfill long-term program goals, a three-year management schedule was adopted. The tactics employed the first year included a scientific literature search for background information and in-lake, fall biomass sampling of the littoral weed bed communities. Dennis and Isom (1984) provide a review of macrophyte inventory methods and data interpretation. For Honeoye Lake, 100 inventory stations were equally grouped among 20 transects. For each alphabetically coded transect (Figure 1), a preshrunk mooring line was anchored by grappling hook at the shore, and the remaining line was drawn out toward the center of the lake and held in position by a heavy navy anchor. The line was kept afloat by large boat fenders and a mast buoy; these also served to mark the precise location of each inventory station. The first station was located only ten feet from the shoreline and the remaining four were equally spaced from the first at 100 foot intervals. Wind drift was minimal despite the total transect line being 410 feet in length. It was anticipated that, with this large number of inventory stations, most of the lake's weed bed variety would be sampled. At each station, standing crop biomass within a weighted 1/4 square meter quadrat

frame was clipped at substrate level and later sorted by species for drying and weight determination.

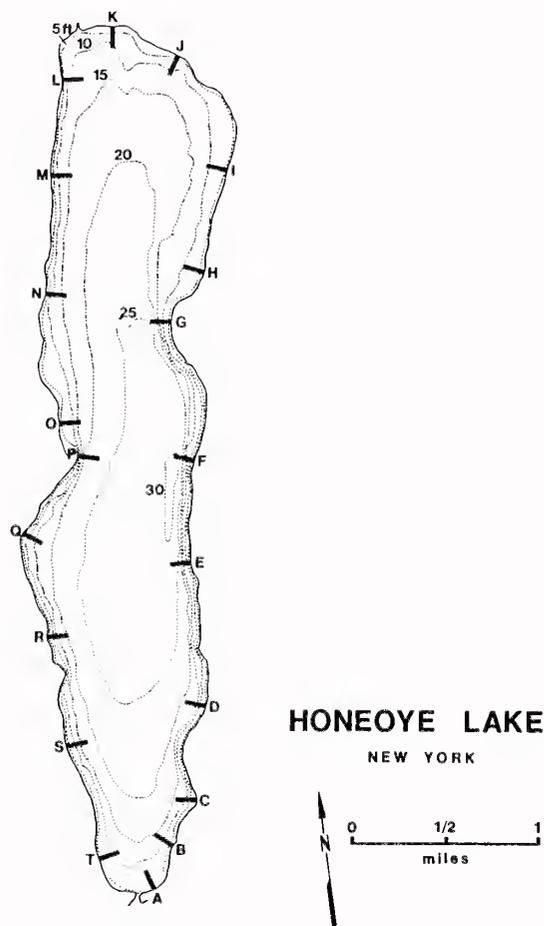


Figure 1. - Location of weedbed inventory transects.

Concurrent environmental measurements at each station included water depth and water transparency. Substrate samples were taken at each station, air dried, sieved and subjected to several analyses. Soil pH was determined, using the saturated paste method of Peech (1965). Particle size distribution was determined by separation in soil sedimentation tubes. Organic matter content was estimated by loss on ignition at 500 degrees centigrade in a muffle furnace (Wilde et al., 1964). Nitrogen, phosphorus and potassium levels were estimated with a LaMotte soil test kit.

Four indices of plant community structure were determined, based on fall, standing-crop biomass data. Community richness (s) represents the total number of plant species found in the quadrat. Community diversity was estimated with the H' index (Shannon and Weaver, 1948). Community equitability was analyzed with the J' index of Pielou (1966) and its complement, community dominance, was calculated using the concentration of dominance index suggested by Odum (1971). The relationship between fall standing-crop biomass, water, substrate, and community structure was tested by simple, linear regression. Correlation coefficients (r) were calculated according to Mendenhall (1971).

## RESULTS AND DISCUSSION

A summary of total dry weight fall standing-crop biomass for each

inventory station is presented in Table 1. Values exceeding 200 grams per square meter may lead to weed bed complaints, especially if they occur in shallow water, but transect averages give a more generalized picture of plant biomass within the lake. These averages, illustrated in Figure 2, clearly indicate that weed bed density varies considerably along the lake shore of Honeoye. Variation in the growth of components of submerged aquatic communities involves environmental, competitive and successional interactions (Barnes and Mann, 1980). Simple linear correlations were calculated between fall standing-crop biomass and fourteen characteristics of the vegetation at inventory stations (Table 2). Positive correlations were found between fall standing-crop biomass and richness, textural class, organic matter content and silt content. Fall standing-crop biomass was negatively correlated with sand content, pH and substrate phosphorus level. These correlations suggest that, as silt and detritus accumulate, substrate conditions improve weed bed productivity. Because detritus will decompose to organic acids, a negative relationship with pH is observed. Under acidifying substrate pH and summer anoxia conditions, bound phosphorus is released to the water column where it readily supports prolific weed bed and algal growth. If phosphorus stays mineralized in the sediment, it is not immediately available for plant growth and a negative correlation is detected. Multiple regression analysis reveals that these factors account for 30.9% of the variation observed in weed bed biomass.

Table 1. Total dry weight fall standing crop, in grams per square meter, for each inventory station.

Transect	Inventory Station					Average	
	1	2	3	4	5	All	Vegetated
A	239	206	1373	166	733	543	543
B	317	510	311	248	136	304	304
C	184	1112	615	56	1	394	394
D	11	33	0	0	0	9	22
E	1	0	0	0	0	1	1
F	114	2	0	0	0	25	63
G	0	6	0	0	0	1	6
H	0	22	0	0	0	4	22
I	47	28	51	216	196	108	108
J	2	47	127	0	52	46	57
K	2	22	8	16	84	26	26
L	7	28	10	9	0	11	14
M	0	1	0	1	0	1	1
N	100	15	0	0	0	23	58
O	3	0	0	1	2	1	2
P	0	41	0	0	0	8	41
Q	0	0	0	0	0	0	0
R	3	265	0	0	0	54	134
S	0	226	50	0	0	55	138
T	152	783	90	363	176	313	313

Factors worthy of future investigation include the effects of annual drawdown and wave action, the significance of herbivory, dissolved nutrient levels, sedimentation rates, species life histories and the annual heat budget. Further discussion is provided in Gilman (1985). Information gathering, data analysis and interpretation should be considered among the highest priorities in any restoration program, since a basic understanding of local natural resources is a necessary precursor to successful management activity.

## SECOND YEAR STRATEGY AT HONEOYE

With sufficient background information gathered, the tactics for the second year involved a watershed questionnaire, some limited use of aquatic vegetation control techniques, an examination of possible long-term funding opportunities, and more detailed mapping of the occurrence of curly pondweed, *Potamogeton crispus* L., a non-native, early spring dominant of the macrophyte communities.

**Table 2.** Simple linear correlation coefficients ( $r$ ) between fall standing crop biomass and characteristics of the vegetated inventory stations in Honeoye Lake.

Biological characteristics	$r$
community richness	0.322*
community diversity ( $H'$ )	0.089
community evenness ( $J'$ )	-0.045
concentration of dominance	-0.045
Water regime	
water depth	-0.232
Substrate factors	
textural class	0.400**
% organic matter	0.308*
% sand	-0.434**
% silt	0.387**
% clay	-0.089
active acidity (pH)	-0.541**
nitrogen concentration	-0.032
phosphorus concentration	-0.316*
potassium concentration	0.077

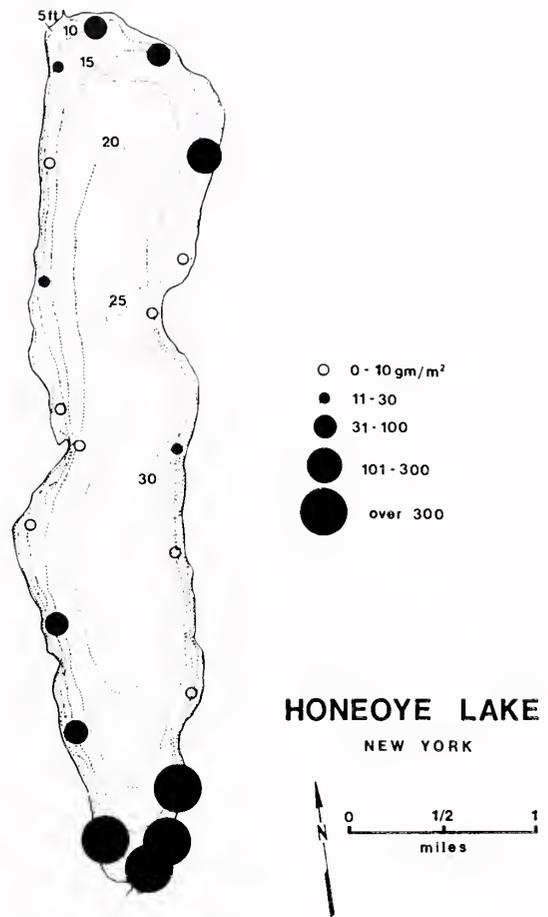
\*  $p=0.05$ ; \*\*  $p=0.01$

Questionnaire results provided valuable information on residential features, water consumption, waste-water disposal facilities and land use practices. The public's general impressions on lake problems provided a list of priorities for action that was later utilized by the newly formed lake association. The public's impressions on the causes of lake problems also served to illustrate the degree of public education that would be needed to gain program acceptance. For example, the leading cause of lake problems was perceived to be the use of pesticides and fertilizers. Little agricultural land exists within the watershed, yet it appears that the public still desires to use the familiar scapegoat of farmland runoff.

The second most frequent response was that lake problems were due to individual citizen abuse. The public's tendency to place the blame on someone else is often unfortunate, since all resource users contribute to the problems and all bear some level of responsibility. This premise must be presented to the public and at least partially accepted before a sound ecological management plan with local support and cooperation, can be achieved.

Selected third among the causes for lake problems were the natural lake conditions themselves. This is indeed correct, and it is hoped that most watershed residents and lake users will realize the limitations placed on lake restoration due to factors like shallowness of the lake, warmth of the water during the growing season and the high nutrient levels. A discouraging response was the low ranking of past pollution as a cause of modern lake problems. Many apparently believe that past sewage contamination quickly flows downstream and out of a lake.

Although Honeoye Lake has a relatively short hydraulic retention time, nutrients appear to reside in the bottom sediments for many years, and past abuses will continue to haunt present resource users.



**Figure 2.** - Average fall standing-crop biomass within each transect of Honeoye Lake. The average is based on all five inventory stations, whether or not they support vegetation.

The final portion of the questionnaire dealt with potential program alternatives. The desirability of a coliform testing program, a fish stocking program, and a weed management program were evaluated along with different avenues of funding: state sponsorship, countywide support, watershed district or private lake association support. The former, requiring the least local monies, was of course the overwhelming selection of the respondents, but they also favored the formation of a lake association to facilitate working with the state agency.

Mechanical harvesting experiments and benthic shading experiments were also completed in the second year of the program. Both weed bed control techniques were successful, but both left some ecological questions unanswered. Harvesting reduced biomass immediately, with a 25% recovery one month later; however, a rapid recovery occurred the next year, and one third of the experimental areas actually showed a slight increase in biomass compared to adjacent control plots. This raises several interesting questions about the management technique. Did harvesting actually stimulate growth one year later, or could these data be explained by differences in the growing season from one year to the next? Were the same species present? Were the more troublesome weeds proliferating with greater success? What

other variables may have contributed to this response? Were the control areas identical to the experimental areas? What was the spatial patterning of the weed bed communities? Did land use changes occur that might influence weed bed productivity? These questions clearly identify areas where more research is needed.

Some tentative conclusions can be made at this point in the ongoing research program. The composition of the harvested weed bed areas did not change significantly. Both richness and relative abundance (based on standing-crop biomass) were similar in control and harvested areas. Good historical data on macrophyte productivity are not available, but some year-to-year variability is to be expected. Lake users often remark about a particularly good growth year for a certain plant. The experimental harvesting study period may transcend the effects of such natural variability. Ultimate biomass reduction of weed bed communities following harvesting has taken many years in some instances (Myers, 1983), and overnight results are not to be expected. Patchiness in the distribution of macrophytes within weed beds is often observed, and a larger sample size is needed to determine its influence on the regrowth data.

Trials of benthic shading eliminated weed growth in small areas, but proved both expensive and labor-intensive. In addition, such alteration of the substrate must impact other, non-target organisms. Did it interfere with bass spawning, for instance? Did it significantly reduce cover for forage fish and invertebrates? Correspondingly, did it improve feeding opportunities for established predators in the lake? The posing of these questions underlines the need for further scientific investigation to accompany management/restoration techniques. The potential impact of benthic shading must be assessed relative to the per cent of bottom covered and availability of suitable habitat elsewhere for displaced organisms. The littoral zone of Honeoye Lake is extensive, reaching depths of 17 feet (see Fig. 1). Our benthic shading experiment covered two 500 foot square areas, only a fraction of one per cent of the littoral zone. Siltation was evident on the shade film applied by the end of the growing season, and some colonization by invertebrates was also observed. Any such accumulated sediment would have to be removed to maintain the effectiveness of the shade film. As expected, no spawning activity was observed on the shade film. Further discussion is provided in Gilman (1986).

Three general approaches to weed bed management are possible and practical in small lake restoration (Moore and Thornton, 1988). The selection of techniques belonging to any of these three approaches should be based on lake usage and public concerns, governmental policies and the possible environmental consequences of the management activity. The first approach, chemical control techniques, was eliminated from consideration due to the New York State's reluctance to approve the necessary permits. In addition, potential conflicts over riparian rights of the public, including many who use the lake as a water source, needed to be avoided if the lake restoration program was to have public acceptance and longevity.

Biological control techniques typify the second approach for weed bed management. Introduction of herbivorous fish like the sterile grass carp has yielded good results in the southeastern United States, but is currently against the law in New York. The density of natural herbivores (e.g. the snail, *Viviparus georgianus*) seems more than adequate to exert grazing pressure on submerged plants.

The remaining approach uses mechanical techniques to reduce nuisance vegetation. Dredging of nutrient-rich substrates requires an Army Corps of Engineers permit, and may, depending on contaminants, necessitate proper disposal of by-products as hazardous waste. Costs may then become prohibitive. Nutrient diversion, through pro-

viding sewers for perimeter cottages has already been accomplished at Honeoye Lake. Benthic shading is effective, but best suited to small-scale applications, and it does not reduce the nutrient budget. Harvesting realizes the immediate, desirable goal of removing plants from the upper water column where water-based recreation activities occur. With removal of harvested biomass from the lake, some reduction in recycled nutrients will take place. Ongoing research should indicate whether this biomass removal has any long-term effect on nutrient budget reduction, or if it is more than compensated by other loading factors.

### THIRD YEAR PROGRESS AND FUTURE PLANS AT HONEOYE

Progress made during the third year of the lake restoration program included intensive weed bed harvesting, monitoring of water chemistry and detection of possible sources of bacterial contamination. All three activities provided short-term improvements in the multiple-use capacity of Honeoye Lake. Harvesting successfully removed vegetation without any of the drawbacks associated with the use of chemical herbicides or the introduction of new organisms to the water. Just over 400 wet tons of vegetation were removed from the deeper section of the littoral zone, where harvesting proceeds more efficiently. It was estimated that 5-10% of the lake's weed bed communities were affected by this management, once in midsummer and again in early fall. Problems associated with harvesting were minor (*i.e.*, escape of plant fragments, accidental capture of juvenile fish), and through continued research and program modification these can be reduced. For the year to come, a small boat equipped with a bow-mounted rake will collect most plant fragments that escape from the harvester. The juvenile fish captured are predominately sunfish (85.6%) and belong to the young-of-the-year or first-year cohort. Insignificant numbers are captured relative to their estimated density within the weed bed communities. Trap netting is planned to assess the use and value of the weed beds to adult fish.

Water chemistry analysis revealed how the lake is intimately tied to events within the watershed. Nutrient pulses occurred following precipitation events that undoubtedly caused runoff. Similar findings have been reported by Stewart and Markello (1974) for other nearby lakes. Water clarity temporarily decreased with siltation caused by runoff as well as wind-generated mixing. Overall, water clarity, measured as secchi disk transparency, has improved during the first three years of the lake restoration program. Bacterial contamination "hot spots" were associated with older cottages in high-density land use. Action has now been taken to curtail raw sewage from entering the lake. A discussion of the continued diversification of the lake restoration program is provided in Gilman and Stone (1987).

### RECOMMENDATIONS BASED ON THE CURRENT STUDY

An integrated approach to small lake restoration based on a sound understanding of the dynamic nature of lake ecosystems is recommended. Baseline data collection is essential prior to, and following, management activities. Some management activities should be favored under certain conditions that will vary among small lakes. Most management activities will produce new questions about lake ecology, and, therefore, direct research into areas for future investigation. As a result, success must be judged within both short-term and long-term frameworks. The apparent trends in Honeoye Lake discussed here may become somewhat tentative as more research data accumulate. Due to

this uncertainty, no single, quick cure for the environmental problems of a lake should be expected.

Public participation is critical in assessment of resource use, misuse and recovery. A local sense of accomplishment can be achieved if restoration activities are funneled through a local lake association.

Small lake restoration should be holistic. Often what is observed as a lake problem should instead be thought of as a symptom, with the real causes of the problem to be found elsewhere in the lake's watershed. Lake restoration, therefore, should utilize a wide variety of techniques, employed at several levels. A recent compendium by Moore and Thorton (1988) provides an excellent and thorough review of such techniques currently in use.

#### ACKNOWLEDGMENT

Field research on Honeoye Lake was sponsored by the New York State Legislature through a local community assistance grant.

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# Some Effects of Beaver Flooding on Rare Swamp Birch (*Betula pumila* L.) Vegetation in Connecticut: 1984 to 1987

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**Abstract:** The response of a regionally rare vegetation type to beaver flooding was studied in the Beeslick Preserve in northwestern Connecticut. Water levels and other factors in a community dominated by *Betula pumila* L. were monitored during and after two episodes of beaver flooding (spring, 1984 and spring-summer, 1985). Plant species richness in 15 permanent plots declined from an average of 16.3 in 1984 to 12.8 in 1986. Two groups of plants experienced the most dramatic declines: 1) members of the Asteraceae, and 2) various species of low growth habit. Woody species in general showed severe signs of stress such as chlorosis and stem die-back. *Carex lasiocarpa* Ehrh. and *Peltandra virginica* (L.) Schott increased somewhat during the study period. As of 1987, frequency and cover of most species trended toward pre-flood values. The average number of species per plot increased, some species missing since 1984 returned, and woody species showed fewer signs of stress by the end of the study. Though the community recovered from two short periods of flooding, sustained flooding might eliminate rare species, damage peat mats, and allow exotics to invade. Future management of this area will include removal of beaver dams and continued monitoring of water levels and vegetation parameters, with a goal of restoring plant vigor and species richness to pre-flood levels.

Pages 244 - 247. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Effective management of natural vegetation requires an understanding of community dynamics (Niering, 1987). Consequently, formulation of vegetation management objectives and the acquisition of baseline ecological data have become essential for land stewards. The present study examines short-term responses and trends in species richness at a swamp birch (*Betula pumila* L.) wetland recently subjected to two episodes of beaver (*Castor canadensis* Kuhl) flooding. Though a great deal has been learned about beaver ecology (Hodgdon & Larson, 1980) and beaver effects on stream ecosystems (Avery, 1983; Naiman *et al.*, 1986), relatively little is known about beaver impacts at natural areas. Hutchison *et al.* (1986) evaluated beaver-related impacts at a botanically significant area in Illinois, listed nine detrimental and nine positive impacts, and stressed the need for clear policies regarding natural areas where beavers create an impact.

The purpose of this paper is to document and evaluate beaver-related impacts on a Connecticut natural area. Our goals were to understand the influence of beaver on vegetation at the site, study the response of the vegetation following flooding, and prescribe appropriate management.

## STUDY AREA AND SITE HISTORY

The study was conducted at Beeslick Preserve, located at Beeslick Pond in Salisbury, Litchfield County, Connecticut, at 41° 52' N latitude, 73° 25' W longitude. Elevation is 249 m and a humid continental climate prevails, with average yearly precipitation ranging from 112 to 132 cm (Gonick *et al.*, 1970). The underlying bedrock is the Stockbridge Formation, a calcite-cemented dolomitic marble (Gates, 1979). The soil is classified as peat and muck (Gonick *et al.*, 1970).

Beeslick Pond and its associated wetlands are topographically situated in a headwater position and receive abundant seepage as well as runoff. Wetland vegetation at the site is diverse, and includes a rich fen

and a calcareous seepage swamp that are considered exemplary occurrences in the state (Kenneth Metzler, pers. comm.).

The present study was confined to the Dwarf Birch Zone (Lyons, 1978), or as classified by Caljouw (1982), the *Betula pumila*-*Acer rubrum*-*Carex aquatilis* plant community. Tall deciduous shrubs, 1.5-3.0 m in height, dominate this vegetation. The site is owned and managed by The Nature Conservancy, and it harbors one of the richest assemblages of rare plants in the State. Rarities include *Carex schweinitzii* Dewey ex Schwein., *Carex prairea* Dewey ex Wood, *Carex crawei* Dewey ex Torr., *Betula pumila* L., *Salix candida* Flugge ex Willd., *Salix serissima* (Bailey) Fern., and *Calamagrostis stricta* (Timm) Koeler (Connecticut Natural Diversity Data Base, 1985).

The outlet of Beeslick Pond flows through a culvert which in early May, 1984 was found to be plugged by a beaver dam. The dam raised the water level approximately 20 cm at the site of the *Betula pumila* stands and caused water to drain across a farm road rather than through the culvert. The dam was removed, and by June 1984, normal water levels existed at the wetland once again. Between September, 1984 and May, 1985 beavers again plugged the culvert, and the wetland vegetation remained flooded until late July, 1985, at which time the dam was removed and normal water levels subsequently returned. Since July 1985, no beaver activity has been noted at Beeslick.

## STUDY DESIGN AND METHODS

Fifteen permanent, 2 x 2 m plots were established along a straight transect extending from the wetland-upland edge through the *Betula pumila* vegetation toward the pond. The transect starting point and the first plot were randomly selected. From the first plot, subsequent plots were spaced regularly at 10 m intervals along the transect, a procedure which facilitated plot relocation. The coverage of each plant species was estimated visually and recorded, using the following scale: (R) solitary and less than 5% cover; (+) few and less than 5% cover; (1) numerous and less than 5% cover; (2) 5-25% cover; (3) 26-50% cover; (4) 51-75% cover; and (5) 76-100% cover. Subsequent data analyses

converted these cover classes to per cent cover values such that R = 0.5%, + = 1%, 1 = 3%, 2 = 15%, 3 = 37%, and 5 = 87%. Water depth, or depth to water table, was measured using a meter stick at each plot during sampling. Plots were sampled on similar dates each July (1984-1987) to help standardize growth condition of the vegetation. In 1985 vegetation data were not collected because deep water from beaver flooding prevented accurate sampling at inundated sites. To minimize bias, previous years' cover-class data were not examined at the time of sampling. The senior author generated all cover-class data to avoid person-to-person variability.

### Results and Discussion

In 1984, standing water was a centimeter or two deep in most plots, but during the flooding episode of 1985 water depths rose 15 to 18 cm higher (Figure 1). Water depth measurements at plots 11 through 15 were not taken in 1985 because the peat mat had become separated from bottom sediments and floated above the normal surface level, thereby distorting water depth measurements. Water levels in 1986 were high, probably resulting from abundant rainfall earlier in the year. In contrast to 1986, the water table remained entirely below the surface in 1987.

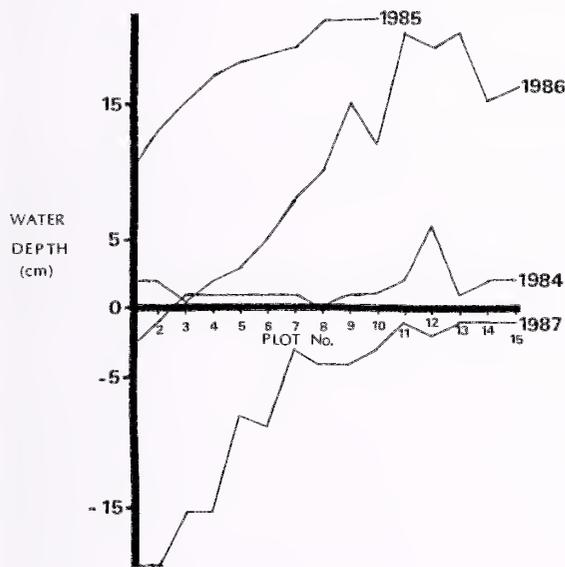


Fig. 1. Late July water level measurements at 15 permanent plots in *Betula pumila* vegetation, Beeslick Preserve, Salisbury, Connecticut, 1984 to 1987.

Table 1 presents frequency and average per cent cover for all plant species detected in the plots from 1984 to 1987. The five most important species characteristic of the vegetation type, dominating in terms of frequency and average per cent cover, were *Betula pumila*, *Thelypteris palustris* Schott, *Salix candida* Flugge ex Willd., *Carex stricta* Lam., and *Typha latifolia* L. Less important, but also characteristic of this association in Connecticut, were *Carex aquatilis* Wahl., *Potentilla fruticosa* L., *Campanula uliginosa* Rydb., *Carex prairea* Dewey, *Salix serissima* (Bailey) Fern., and *Calamagrostis stricta* (Timm) Koel. *Sphagnum* species do not occur in this vegetation type, but they dominate the moss stratum in *Cladium mariscoides* (Muhl.) Torr. vegetation about 100 m away.

Plant species richness declined from 1984 to 1986, dropping from an average of 16.3 species per plot in 1984 to 12.8 in 1986 (significant at  $P < 0.001$ , t-test). Accompanying this decline in richness were apparent symptoms of severe flooding stress to woody species. In July,

*Betula pumila* showed stem die-back and chlorotic leaves, *Potentilla fruticosa* had few flower buds, and the leaves of *Acer rubrum* L. were red.

Coincident with the decline in species richness from 1984 to 1986 was a decline in average cover values during this period; 29 species declined in cover while only nine species increased. *Aster puniceus* L., *Solidago purshii* Porter, *Eupatorium perfoliatum* L., and *Eupatorium maculatum* L. experienced a dramatic decline in frequency and cover (Table 1). A suite of low stature plants including *Carex interior* Bailey, *Hydrocotyle americana* L., *Proserpinaca palustris* L., *Scutellaria galericulata* L., *Viola* sp., *Lycopus uniflorus* Michx., *Triadenum virginicum* (L.) Raf., and *Campanula uliginosa*, also declined during this period (Table 1). The summed importance value (maximum equals 100) for these species dropped from 5.8 in 1984 to 1.8 in 1986.

*Carex lasiocarpa* Ehrh. and *Peltandra virginica* (L.) Schott were the only species to increase somewhat from 1984 to 1986 (Table 1). Their summed importance value increased from 5.1 to 9.1 during this period. The *Carex* may have increased in response to higher water levels (Gates, 1942). Frequency and coverage data from 1987 indicated a recovery trend in the vegetation (Table 1). Plant species richness increased to an average of 15.7 species per plot, approaching the 1984 baseline value of 16.3. From 1986 to 1987, 25 species increased in cover value while only 10 species had lower figures (Table 1). A few species present in 1984 but not 1986, including *Campanula uliginosa*, *Calamagrostis stricta*, and *Boehmeria cylindrica* (L.) Sw., reappeared in the plots in 1987. Also, *Betula pumila* showed lush green foliage and vigorous sprout growth.

### MANAGEMENT CONSIDERATIONS

Management goals were established to improve the condition of the wetland vegetation at Beeslick Preserve to 1984 conditions, and eliminate future flooding stress. We did not know what constituted a "normal" or optimal condition, therefore the 1984 data served as baseline information out of practical necessity, rather than certainty that these data represented optimal conditions for the vegetation.

To measure success in achieving this goal, we monitored five variables: water depth, species richness, vigor of red maple, vigor of swamp birch, and the reappearance of *Calopogon tuberosus* (L.) BSP. The management objectives were stated as follows:

- 1) Summer water depth rises no more than 2.0 cm in the study plots, similar to 1984 baseline data.
- 2) Vascular plant species richness is restored to 1984 levels, about 16 species per plot.
- 3) *Acer rubrum* should not show red foliage in summer.
- 4) *Betula pumila* should produce abundant fruit and lush foliage.
- 5) *Calopogon tuberosus*, seen in 1984, but not in 1985 appears on the nearby fen mat.

In 1987 we measured our success at achieving these objectives. Water depth was below 2.0 cm. Plant species richness approached 16 species per plot, but certain species last seen in 1984, such as *Onoclea sensibilis* L., *Solidago purshii* Porter, *Eupatorium perfoliatum*, *Scutellaria galericulata*, *Bromus ciliatus* L., and *Hydrocotyle americana*, remained absent. Red maple still showed considerable early red foliage. The foliage of swamp birch was green and lush, especially on fast-growing basal sprouts, but fruit production remained sparse. A single *Calopogon tuberosus* plant was observed on the fen mat in 1987, where a few had been present in 1984.

**Table 1.** Species frequency and average per cent cover in a *Betula pumila* wetland at Beeslick Preserve, Salisbury, Connecticut, 1984 to 1987. Data from 15, 2 x 2 m permanent plots.

SPECIES	YEAR:	FREQUENCY (%)			AVE. % COVER		
		84	86	87	84	86	87
<i>Betula pumila</i> L. <sup>a</sup>		86	86	86	36	22	18
<i>Thelypteris palustris</i> Schott		100	100	100	25	13	34
<i>Salix candida</i> Flugge ex Willd.		80	80	80	19	19	25
<i>Carex stricta</i> Lam.		40	46	46	40	40	30
<i>Typha latifolia</i> L.		93	93	100	10	9	12
<i>Cornus sericea</i> L.		60	53	46	15	12	17
<i>Alnus serrulata</i> (Dry.) Willd.		53	60	66	13	8	7
<i>Equisetum fluviatile</i> L.		66	73	73	8	4	4
<i>Acer rubrum</i> L.		86	60	73	3	8	5
<i>Spiraea latifolia</i> (Ait.) Borkh.		60	46	66	9	13	14
<i>Lycopus uniflorus</i> Michx.		80	20	46	1	1	1
<i>Peltandra virginica</i> (L.) Schott		46	46	46	7	14	16
<i>Carex aquatilis</i> Wahl.		40	53	53	8	5	4
<i>Triadenum virginicum</i> (L.) Raf.		60	40	60	4	1	1
<i>Galium</i> sp.		53	26	46	1	1	1
<i>Lysimachia terrestris</i> (L.) BSP.		53	60	86	1	1	1
<i>Lysimachia thyrsiflora</i> L.		53	26	46	1	1	1
<i>Carex lasiocarpa</i> Ehrh.		33	40	60	7	11	6
<i>Potentilla fruticosa</i> L.		40	40	40	3	2	2
<i>Eupatorium maculatum</i> L.		40	6	6	3	5	3
<i>Cicuta bulbifera</i> L.		46	20	46	1	1	1
<i>Calamagrostis canadensis</i> (Michx.) Beauv.		46	53	46	3	2	4
<i>Campanula uliginosa</i> Rydb.		46	0	26	1	0	1
<i>Aster puniceus</i> L.		33	6	6	1	1	3
<i>Onoclea sensibilis</i> L.		13	0	0	8	0	0
<i>Carex prairea</i> Dewey ex Wood		13	6	13	8	1	1
<i>Polygonum amphibium</i> L.		20	33	26	1	1	2
<i>Solidago purshii</i> Porter		20	0	0	1	0	0
<i>Symplocarpus foetidus</i> (L.) Salisb.		20	6	13	1	1	3
<i>Scirpus acutus</i> Muhl. ex Bigel.		13	13	13	2	2	2
<i>Salix serissima</i> (Bailey) Fern.		13	13	13	1	9	3
<i>Eupatorium perfoliatum</i> L.		13	0	0	5	0	0
<i>Salix bebbiana</i> Sarg.		6	6	6	1	5	1
<i>Salix discolor</i> Muhl.		6	13	13	1	9	20
<i>Toxicodendron radicans</i> (L.) Kuntze		6	6	0	1	5	0
<i>Calamagrostis stricta</i> (Timm) Koeler		6	0	13	1	0	1
<i>Carex diandra</i> Schrank		6	6	0	1	1	0
<i>Carex interior</i> Bailey		6	0	0	1	0	0
<i>Hydrocotyle americana</i> L.		6	0	0	1	0	0
<i>Proserpinaca palustris</i> L.		6	0	0	1	0	0
<i>Scutellaria galericulata</i> L.		6	0	0	1	0	0
<i>Viola</i> sp.		6	0	0	1	0	0
<i>Toxicodendron vernix</i> (L.) Kuntze		6	0	0	5	0	0
<i>Bromus ciliatus</i> L.		6	0	0	5	0	0
<i>Torreyochloa pallida</i> (Torr.) Church		6	6	20	5	5	1
<i>Carex</i> sp. (ovales)		6	0	0	5	0	0
<i>Boehmeria cylindrica</i> (L.) Sw.		6	0	6	5	0	1
<i>Asclepias incarnata</i> L.		6	6	13	5	5	2
<i>Carex</i> sp.		6	0	6	5	0	5
<i>Rumex</i> sp.		0	6	13	0	5	5
Poaceae sp.		0	13	0	0	5	0
<i>Sagittaria latifolia</i> Willd.		0	6	20	0	5	2
<i>Solidago canadensis</i> L.		0	0	6	0	0	1
<i>Equisetum arvense</i> L.		0	0	13	0	0	1
<i>Carex comosa</i> Boott		0	0	6	0	0	1

<sup>a</sup> Nomenclature follows Mitchell (1986).

Flooding stress to this community not only jeopardized the existence of a diverse flora with rare species, but it also made the wetland vulnerable to invasion by exotics. In particular, *Lythrum salicaria* L., which inhabits old canal banks at Beeslick, is capable of rapid spread under flooding-drawdown regimes (Rawinski, 1982).

Flooding also jeopardized the peat mat because, the mat was, for a time, influenced more by pond water than by groundwater. A peat mat at one Massachusetts fen was destroyed by sustained flooding from a man-made dam. Mats of vegetation at that site floated on the surface for several years until they disintegrated (Hubert Olsen, pers. comm.)

### CONCLUSIONS

Two episodes of flooding at Beeslick resulted in relatively short-term declines in species richness and plant vigor. As of 1987, two years after the last flooding, the vegetation appeared to be recovering. Continued monitoring of the study plots is needed. Sustained periods of flooding will likely lead to invasion by exotics, lower species richness, and disintegration of peat mats. At Beeslick, we support our initial decision to remove beaver dams that threatened swamp birch and other rare plants at the site. Beaver flooding apparently constitutes a threat to this type of vegetation in the Northeast, and managers should carefully monitor or eliminate beaver related impacts on sensitive wetlands.

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# Management Strategies for Increasing Habitat and Species Diversity in an Urban National Park

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**Abstract:** Gateway National Recreation Area comprises 10,522 hectares located in New York City and adjacent counties of New York and New Jersey. Gateway properties range from relatively intact natural landforms to the dredge spoil and landfill that underlie most of the upland habitats. Since its establishment in 1972, the National Park Service has been implementing a policy of restoring and enhancing these altered and once degraded habitats and the flora and fauna that occupy and utilize them. Specific management activities include active and passive revegetation of impacted sites, restoration of grassland habitats utilized by regionally rare grassland-dependent bird species, creation of freshwater habitats, and a transplant program to restore populations of native amphibians and reptiles. Through these activities, the diversity of native biota will be restored, preserved and made accessible to millions of people in the New York Metropolitan area who visit this unit of the National Park System.

Pages 248 - 250. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Gateway National Recreation Area consists of 10,522 ha, distributed among four counties in the states of New York and New Jersey. Located within and adjacent to New York City, Gateway lands range from relatively intact natural land-forms to the dredge spoil and landfills that underlie most of the upland habitats. Typical upland habitats include dunes dominated by beachgrass (*Ammophila breviligulata* Fern.), mixed grasslands, thickets of bayberry (*Myrica pensylvanica* Loisel.), early successional woodlands of grey birch (*Betula populifolia* Marsh.), black cherry (*Prunus serotina* Ehrh.), poplars (*Populus* spp. L.), and American holly (*Ilex opaca* Ait.). On more disturbed sites, generally associated with landfill, monocultures of common reed [*Phragmites australis* (Cav.) Steud.] and mugwort (*Artemisia vulgaris* L.) predominate.

With the establishment of Gateway in 1972, the National Park Service adopted a policy of "restoring" and enhancing the altered and heavily impacted habitats it had acquired. A number of strategies are now being pursued, dependent upon the specifics of the sites. Two developments have guided the strategies: the establishment of a greenhouse and nursery at Floyd Bennett Field, Gateway's Headquarters, enabling the propagation of native plant species not available commercially, and the institution of a habitat management plan that defines management goals for a particular site, and in turn guides its restoration.

At some sites, "restoration" consists of eliminating disturbance factors and allowing succession to revegetate the site. At other sites, a more active approach is required. Barriers to exclude vehicles are erected, and native species of grasses, shrubs, and trees are planted. To date, major efforts have included the planting of 300,000 culms of American beachgrass to revegetate a beach nourishment project, the planting of 10,000 culms of salt marsh cordgrass (*Spartina alterniflora* L.) to restore a salt marsh and control erosion, and the planting of several hundred trees and shrubs in the past five years. Over 30 common coastal species have been planted. Other species, such as butterfly weed (*Asclepias tuberosa* L.) and butterfly bush (*Buddleja davidii* Franch.) have been planted to improve habitat for lepidopterans and aid in their census.

## COASTAL ECOSYSTEM RESTORATION

At the Sandy Hook Unit of the Park, a natural sand spit exists where shoreline dynamics have separated the tip area (a spit by geological

definition) from the mainland many times over the last several hundred years. Considerable effort on the part of federal and state agencies has gone into evaluating alternatives for solving the potential breaching of the "critical zone," where erosion was leading to the separation of the spit from the mainland. A single route to the tip is the only means the U.S. Coast Guard, the National Oceanographic and Atmospheric Administration, New Jersey Marine Science Consortium, and American Littoral Society have to reach their facilities by road, as well as being the only access for over two million beach users annually. A special Congressional appropriation of \$12 million, to be reimbursed to the Federal Government by charging beach user fees, was allotted to help restore the original 1954 Sandy Hook shoreline profile.

The plan called for some three million cubic meters of sand, dredged from Sandy Hook and Ambrose Channels, to be hydraulically dumped into the critical zone, beginning in March 1983. In addition to the sand material, an attempt at dune construction and stabilization would be made along a 2,414 meter strip of beach, once the original shoreline was reestablished. Three hundred thousand culms of beachgrass were purchased, planted and fertilized over a two month period beginning November 1983. Aerial photographs of the entire planted area were made and used in conjunction with an on-site ground evaluation of the survivorship of the culms.

The establishment process was surprisingly simple to carry out, since the grass was so well adapted to growth under strand conditions. Few species are tolerant of the stresses associated with the beach environment, where they must be able to survive sand blasting, burial, salt spray, saltwater flooding, drought, heat and low nutrient supply. A perennial grass such as beachgrass is well adapted, since it is a vigorous, cool-season dune grass, and grows in dense clumps spreading laterally by rhizomes. It is easy to propagate, harvest and store, and is readily available from commercial nurseries. The grass is extremely easy to transplant; it establishes, grows rapidly and begins trapping sand by the middle of the first growing season. The rhizomes may spread as much as 3 to 4 meters per year while accumulating as much as 1.2 meters of sand in a single growing season.

The work force for this project consisted of the Sandy Hook maintenance staff equipped with a tractor and agricultural transplanter and 50 volunteers. The volunteers included representatives of the National Park Service, National Oceanographic and Atmospheric Administration, United States Coast Guard, local Sierra, Audubon and garden clubs, Boy Scouts, camping clubs, and others. The planting

**Table 1.** Population status of amphibians and reptiles released on Ruler's Bar Hassock Island, Jamaica Bay Wildlife Refuge, New York, NY., 1980 - 1987.

Species Released	Year	No. of Individuals	Overwinter Survival	Breeding Records	Established
Spring peeper	80-83	58 adult 3600 larvae	Yes	Innumerable	Yes
Gray tree frog	1987	1000 larvae	Yes	a	a
Green frog	85-87	130 adult 212 larvae	Yes	a	a
Spotted salamander	1987	14,000 embryos	a	a	a
Redback salamander	83-86	361 juvenile 1443 adult	Yes	12 offspring recorded	a
Northern brown snake	80-84	23 juvenile 49 adult	Yes	42 offspring recorded 83-84	Yes
Smooth green snake	81-86	17 juvenile 64 adult	Yes	10 offspring recorded	a
Eastern hognose snake	84-85	21 hatchling 4 adult	Yes	a	a
Eastern milk snake	84-87	19 juvenile 13 adult	Yes	1 offspring recorded	a
Black racer	85-87	6 juvenile 18 adult	Yes	25 offspring recorded	a
Snapping turtle	83-87	320 hatchling 12 juvenile 38 adult	Yes	3 offspring recorded	a
Eastern painted turtle	82-87	28 juvenile 361 adult	Yes	8 offspring recorded	a
Eastern box turtle	80-86	12 juvenile 183 adult	Yes	6 offspring recorded	a

a = Insufficient elapsed time or data to determine.

went quickly, totaling approximately 150 rows of about 1200 plants installed. To encourage establishment, 10-10-10 fertilizer was applied two days after planting in the fall of 1983 and again in the spring of 1984.

As the new dune system continues to take shape, powered by natural processes, other plant species will be introduced both by natural invasion and with the help of park employees. To do this, Gateway has begun a nursery system with the prime objective of growing coastal plant species that often are unavailable from commercial sources. A nursery site at Sandy Hook and one at Floyd Bennett Field in Brooklyn, along with a 7.3 x 12.2 meter free-standing greenhouse, will provide transplants of such plants as dusty miller (*Artemisia stelleriana* Bess.), seaside goldenrod (*Solidago sempervirens* L.), bayberry, lance-leaved coreopsis (*Coreopsis lanceolata* L.), prickly pear (*Opuntia humifusa* Raf.), beach plum (*Prunus maritima* Marsh.), and eastern red cedar (*Juniperus virginiana* L.).

The dune planting exercise at Sandy Hook is not intended to stop erosional processes. What is anticipated is that the grass will aid in stabilizing a new dune system, reduce the rate of erosion in the critical zone, increase habitat diversity thereby providing for additional passive recreational opportunities, and continue to maintain the connection that provides vehicle access to Sandy Hook.

### GRASSLAND MANAGEMENT

While the majority of habitat restoration efforts have focused on plantings, a number of other projects are also underway. At Floyd

Bennett Field in Brooklyn, 56.6 of the site's 579 hectares have been designated as managed grasslands, and the Park Service, working cooperatively with the New York City Audubon Society and the Seatuck Research Program of the Cornell Laboratory of Ornithology, has begun restoring grassland habitat there. This site was originally a series of salt marsh islands, filled in by dredge spoil and rubble to create New York City's first municipal airport, and later a naval air station (Blakemore, 1979; Black, 1981). Through plantings and natural colonization, communities of mixed grassland, common reed, shrub thickets of bayberry and winged sumac (*Rhus copallina* L.), and scattered stands of black cherry, grey birch, and cottonwood (*Populus deltoides* Marsh.) have developed (Lent et al., 1985).

From Gateway's beginnings in 1972, Floyd Bennett Field had become known for supporting populations of breeding and wintering grassland birds. Since grassland habitat in the northeastern United States has declined dramatically, as evidenced locally by the loss to development of the Hempstead Plains (Stalter, 1981), a number of grassland birds are listed by many states as Endangered, Threatened, or of Special Concern. The possibility of losing regionally unique grassland birds at Floyd Bennett Field, due to successional changes, led to research into bird-habitat relationships. As a result of this research, a "grassland indicator species" the grasshopper sparrow (*Ammodramus saviannarum*) was identified, and prime habitat delineated (Lent et al., 1985). A resulting habitat management scheme for Floyd Bennett Field provides for a number of successional stage habitats, thereby maintaining greater overall species diversity on-site, and also contributing to greater species diversity regionally. Grassland restoration began in

1985, with Audubon Society volunteers and Park Service staff removing trees and shrubs from the designated grasslands. Pending studies on other techniques such as fire, these grasslands will be maintained through biennial mowing.

Monitoring of bird usage has indicated a slight increase in grasshopper sparrow abundance, as well as its return to restored areas that had been recently abandoned. Other New York State-listed grassland species that utilize these grasslands for feeding, winter roosts, and migratory stopovers include: northern harrier (*Circus cyaneus*), common barn owl (*Tyto alba*), short-eared owl (*Asio flammeus*) and upland sandpiper (*Bartramia longicauda*).

### FRESHWATER WETLANDS CREATION

Through deposition of dredge spoil and other land filling processes in the Gateway area, existing wetlands were destroyed, and few new ones created. Since these habitats were part of the original landscape and were extremely important contributors to habitat and species diversity, freshwater wetlands are now being recreated. The first ponds were actually created in 1953, when the City of New York managed the Jamaica Bay Wildlife Refuge. Two ponds (18.2 and 40.5 ha) were created by diking off sections of salt marsh with dredge spoil. These two ponds, though much fresher than adjacent Jamaica Bay, are still slightly brackish, and do not represent true freshwater ecosystems.

More recently at the Jamaica Bay Wildlife Refuge, a 0.16 hectare pond, and several smaller ponds were created by bulldozer and with hand tools. Upon excavation, these ponds were planted with emergent and submergent aquatic vegetation, obtained locally and through commercial sources. Planting not only accelerates the development of vegetation in the ponds, but also provides an infusion of invertebrates into the newly created system. At Floyd Bennett Field, a joint project of the National Park Service and the New York State Department of Environmental Conservation, funded through a "Return-A-Gift" grant for wildlife enhancement, will create another 0.8 hectare pond in similar fashion.

### AMPHIBIAN AND REPTILE RESTORATION

In an urban area such as Gateway, habitat restoration and management can only go so far in increasing animal species diversity. Urbanization creates many dispersal barriers to wildlife (Campbell, 1974), and, while avian and insect diversity at Gateway are high, non-volant taxa such as amphibians, reptiles and mammals are impoverished, relative to the area's habitat diversity and original fauna (Cook and Pinnock, 1987). In order to restore at Gateway an animal community more representative of the original fauna of the New York City region, and help preserve populations of locally declining species, a program of amphibian and reptile transplants was begun. Most of the

releases have been made at the Jamaica Bay Wildlife Refuge, using local animals, often collected from sites scheduled for development. Since 1980, individuals of 13 species, collected as eggs, larvae, neonates or adults have been released. Results to date indicate that two species are definitely established, and nearly all are surviving and beginning to reproduce (Table 1). Based on findings following this pilot program, amphibian and reptile population restoration has begun Gateway-wide.

### CONCLUSION

As a result of habitat restoration activities, the National Park Service is enhancing the altered and once degraded habitat at Gateway National Recreation Area (Tanacredi, 1987). By restoring habitats, and maintaining habitat diversity, greater wildlife utilization is foreseen for the future. In the heavily urbanized New York City area, the existence of such wildlife habitat is important not only for the animals it supports, but also for the millions of people who have an opportunity to recreate in this Unit of the National Park System.

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# Species Management for the Closed-lip Penstemon

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**Abstract:** The closed-lip penstemon (*Penstemon personatus* Keck) has a limited range, mainly within the Plumas National Forest, with a few isolated locations in Lassen National Forest. The species is currently on the U.S. Forest Service Sensitive Plant Species List, and is a candidate species for Endangered or Threatened status by the U.S. Fish and Wildlife Service. In each of its locations, intensive timber harvest activities are proposed which could significantly impact the species. A species management guide has been developed for the long term conservation and enhancement of the closed-lip Penstemon that will prevent the need for listing it as Threatened or Endangered. The guide provides a process for the management of *Penstemon personatus* that develops regulated silvicultural prescriptions to maintain viable population levels, and it establishes protected population areas in order to maintain the geographic and genetic variability of the species.

Pages 251 - 254. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The National Forest Management Act and Forest Service policy require that Forest Service lands be managed to maintain viable populations of all native plant and animal species. A viable population consists of the number of individuals necessary to perpetuate their existence in natural, genetically stable, self-sustaining populations. Such populations need to be adequately distributed throughout their range to ensure genetic diversity.

Besides those species listed as Threatened or Endangered under the Endangered Species Act, the Forest Service has also recognized the need to apply special management direction to the rare flora and fauna on the lands it administers. Species recognized by the Forest Service as needing special consideration are designated "sensitive" by the Regional Forester. The management objective for these designated species is to ensure continued viability throughout their range on National Forest lands.

*Penstemon personatus* Keck is currently on the Region 5 (Pacific Southwest) Sensitive Plant List and is a candidate species for Threatened or Endangered status by the U.S. Fish and Wildlife Service. This species has a limited range, and is found mainly with the Plumas National Forest with a few isolated locations in the Lassen National Forest; it was known from only four locations prior to 1979. Since 1979, additional locations have been found in the Plumas National Forest and vicinity. In each of these locations, intensive timber harvest activities are proposed which could significantly impact the species. A species management guide has been developed that outlines a management plan for long-term conservation and enhancement of the closed-lip penstemon which will prevent the need for listing the species as threatened or endangered at the federal level.

## BIOLOGICAL INFORMATION

The field work to locate and investigate this species was begun in 1979 by the author. In subsequent years, other Forest Service employees have also assisted in the endeavor.

This species was originally named and described in 1936 by D.D. Keck from a 1900 collection by J.B. Leiberger from Flea Valley, Butte County, California. The closed-lip penstemon differs from all other penstemons occurring in this area of California by the closed corolla tube with the densely bearded inner surface near the tip and the very short staminode (Niehaus, 1977; see Fig. 1). Keck (1936) considered its nearest relative to be *Penstemon nemorosus* (Dougl.) Trautv., which

occurs in the Cascade Range. They are similar in the short upper lip and well extended lower lip of the corolla as well as the short staminode. Keck (1936) named a new section, *Cryptostemon*, to include this species and call attention to the stamens not only being included in the throat, but also hidden by the closing of the orifice of the corolla; also, the brevity of the staminode is unmatched within the genus. Thus, *Penstemon personatus* has characteristics that are uncommon to the genus and which make the species unique.

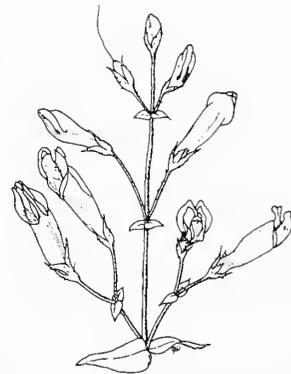


Figure 1. Inflorescence of *Penstemon personatus*<sup>1</sup>

Although *P. personatus* had been described taxonomically in the 1930's, not very much was known about the species biology or ecology prior to 1979. An article in the California Native Plant Society journal, *Fremontia*, on penstemons in California by Kenneth and Robin Lodewick (1983) said that the species was probably extinct, but a subsequent letter was submitted by this author to *Fremontia*, indicating that the species was still extant in California.

*Penstemon personatus* is presently known to occur in eleven populations, comprising approximately 5,500 acres of the Plumas National Forest and the southwestern portion of the Lassen National Forest (see Figure 2). The majority of sites for this species are located near Bucks Lake, approximately 20 miles west of Quincy, California. In each area, plants are scattered in patches rather than continuous populations.

The type locality for this species at Flea Valley, Butte County, California, has been visited several times since the original collection, by the author and many others, but the penstemon has not been relocated there.

<sup>1</sup> Illustration by Judy Wheeler

The elevation of *Penstemon personatus* populations ranges from 4,600-6,200 feet, with the majority of the locations above 5,000 feet. The species has been found in mixed conifer forest, montane chaparral and red fir forest. It occurs under a wide variety of light intensities from full sun to dense shade in each of the known locations.

By contrast, plants in full sun reach heights of up to three feet and have leaves that are larger, thicker and darkly pigmented with purple pigment present, particularly on the undersides (Meyer, 1982). They almost invariably bloom (90%) and have more blooms per plant, usually 20 or more.

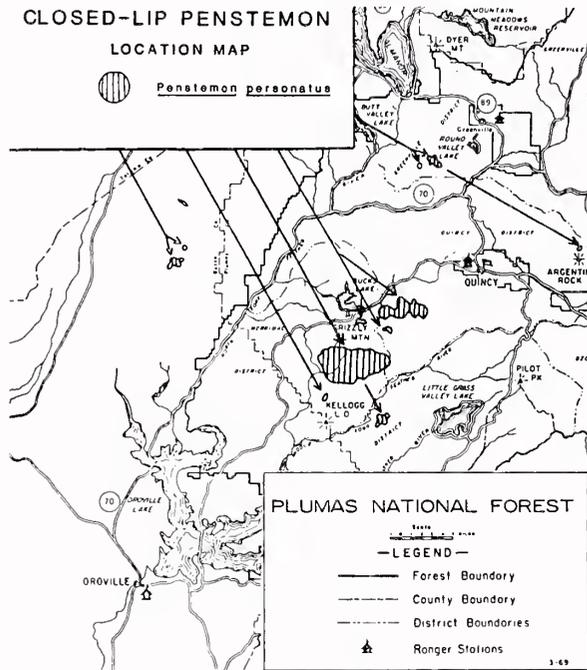


Figure 2. Known Distribution of the Closed-lip Penstemon

Soils are mainly metavolcanic in origin, but the plants are found occasionally on substrates associated with granitic or sedimentary rocks. In all cases the soils are well developed, with significant organic content. The species has not been found on poorly developed soils or rock substrates. Road cuts and fills provide a minimal amount of habitat for the species as a result of these sites collecting organic debris.

The majority of the plants are located on north-facing slopes, while plants may occasionally be found in areas with a west or east-facing aspect. The only areas where the species has been located on a south facing slope are in the population south of Bucks Lake near Mt. Ararat and Bear Creek. The plants are most often found on a slope greater than 20%; however, vigorous populations have been located in flat areas (Meyer, 1982). The species appears to prefer saddles and the lips of ridges, downslope to the drainage. The plants rarely extend over the ridge top to a south facing slope.

The closed-lip penstemon is a perennial plant that grows in clumps from an underground rootstock. Each year the plant sprouts a new shoot and leaves from a bud about one inch beneath the soil surface. Excavation of a patch of the species determined that roots 6 inches below the soil surface intermingled and sometimes branched to link separate shoots, which suggests that vegetative reproduction is occurring.

The species appears to have morphological responses related to light availability (Meyer, 1982). Meyer (1982) indicated that under a closed canopy (60% cover or more) the plant has thin delicate pale green leaves occasionally with some purple pigment on the underside and is usually less than 12 inches tall. The species rarely flowers (3%) in a closed canopy and (if it flowers) it has eight to ten flowers per plant.

## EFFECTS OF LAND DISTURBANCE

*Penstemon personatus* seems to tolerate disturbance that opens the canopy and increases available light. However, the plants do not appear to tolerate heavy ground disturbance, especially activities that cause soil compaction or soil displacement. Effects of past logging operations on the species have been observed at each site. The majority of the timber harvest has been uneven-aged management but some even-aged management has also occurred. The exact extent and abundance of *P. personatus* prior to these activities is not known.

The species has tolerated timber harvest activities such as the selective removal of scattered large trees, partial cutting of the canopy, reduction of the canopy by 30% and various thinning operations. For each silvicultural prescription, a limited amount of ground disturbances has occurred which has opened up the canopy. Such activities actually appear to be ideal to maintain the closed-lip penstemon populations and they probably create situations similar to the natural conditions that occurred prior to human activities. The disturbances increase the amount of light reaching the forest floor, which apparently stimulates the penstemon to reproduce, although they do not drastically alter the microhabitat or discourage the germination of competing weedy species.

By contrast, *Penstemon personatus* appears not to tolerate certain intensive forest management practices that have occurred in the past. These were mainly tractor-logged tracts south of Bucks Lake left as large clearcut areas and probably brush raked. Each of these units is now devoid of all shade producing species except for a few small patches of scattered trees. The penstemon found there is located near the remaining trees or under brush. The species is more abundant on the edges of the old land units in the forested areas. In the clear-cut areas, the penstemon is very scattered and sparsely populated. On south facing slopes south of Bucks Lake, the plant was not found in the clear cut areas, but was located in isolated pockets of trees or in adjacent forested areas. The species does not seem to tolerate heavy ground disturbance. The penstemon is not invading areas that were cut over 15 years ago, or it is doing so at a very slow rate.

In areas that have had some type of timber harvest, the penstemon has been found in logging slash, growing under natural fir regeneration and on the edges of skid roads. The species has not been found in the wheel ruts of skid roads, but it is occasionally found in the grassy center track. This suggests that the plants do not tolerate soil compaction.

The penstemon appears to tolerate a cool burn, such as a spring broadcast burn, but not a hot pile burn. The penstemon was observed only outside the area where the handpile had burned hot enough to destroy the larger logs. In another handpile, in which the larger logs didn't burn, the penstemon was present. Thus, in most cases, pile burning is too hot for the species to be able to survive, but it may survive a cooler burn.

The species has been located after springtime broadcast burns in several units. In units east of Bucks Lake, the penstemon was abundant after the burn, perhaps in response to the fire. To determine whether the species will continue to flourish in these units, it is currently being monitored there.

*Penstemon personatus* does not appear to tolerate soil displacement

or compaction, due to its shallow root system, nor does it tolerate herbicidal spraying. In the area south of Bucks Lake a few widely scattered penstemons were located on the edges of a young plantation that had been brush-raked, but they were not found in the central portion of the affected area. In another plantation south of Bucks Lake, the species only occurred in pockets of the forest adjacent to, but not within, the plantation itself. The plantation was about 15-20 years old and had been brush-raked into wind rows and sprayed with herbicide. In the same vicinity, the penstemon was not found in an area of brush that had been sprayed with herbicide but was located in the surrounding forest. From these samples, it is apparent that site preparation which displaces the soil and spraying with herbicides significantly reduce the number of plants found in a given area.

## INTERIM MANAGEMENT

All known population locations for *P. personatus* occur in areas where timber harvest and management are a primary activity. Although the species appears to tolerate certain timber management activities, its numbers have been consistently reduced by intensive timber management in the past. In impacted areas, individual plants are usually extremely scattered and reduced in numbers compared with surrounding forests. While there are no immediate threats jeopardizing the continued existence of *Penstemon personatus*, intensive timber management may eventually significantly impact the species. The most threatening current activities for the closed-lip penstemon include those that displace or compact the soil, increase the number of competing weedy species, create hot burns or apply herbicides directly to the plants.

When timber sales were being proposed in the early 1980's an interim management prescription was developed. This prescription was intended to modify the proposed sales and lessen the impact of even-aged management. The interim management prescription directed that no more than 10% of the locations of the penstemon species would be impacted through intensive even-aged timber harvesting techniques nor would the combination of even-aged and uneven-aged timber management impact more than 25% in that ten year planning cycle. The interim management prescription directed that these levels not be exceeded until the effect of timber harvest activities on the species was assessed.

Since the extent and abundance of the penstemon prior to the existing disturbances is not known, it has been hard to determine exactly what the species can tolerate. Thus, more information is needed on the effects of different types of timber harvest activities on the species. Recently, monitoring activities have been established to assess the impacts of the various timber and post harvest treatments on the species for a five year period.

Different types of monitoring plots have been used, depending upon the type of proposed treatment. Modified range transects were established for a salvage logging sale, and these have been monitored every five years. A 100 ft transect was established and sampled at 10 ft intervals with a 3 ft square sampling plot. Numbers of plants, records of spatial distribution and verifying photos were taken of each plot. For a broadcast burn, the presence of the penstemon before and after the burn was determined with fixed mil acre plots.

To assess herbicide application, a grid of 50 mil acre fixed plots was established with each plot being 66 feet apart. Numbers of plants and photos of each plot are recorded each year. For uneven and even aged timber management units, a grid of mil acre plots was established with plots 50 feet apart. A datum point outside the unit was used, along with

bearing and distance to the first plot, to avoid any bias from timber harvest activities. The presence or absence of the species at each plot is recorded each year. Data from each type of monitoring plot were gathered prior to the treatments to provide baseline information.

Control plots have also been established to determine whether population fluctuations are due to the treatment or to natural fluctuations in the species. From the various monitoring schemes, data are being gathered to determine the types and amounts of disturbance *P. personatus* can tolerate. Once this information is obtained, more precise recommendations will be given for future timber harvest activities specific to each of the species locations.

## SPECIES MANAGEMENT GUIDE

The interim management prescription predisposed dealing with the penstemons on a sale-by-sale basis without a knowledge of the overall impact to the species. Long-term management for the closed-lip penstemon was needed, so a species management guide was developed to compile known information and establish a long-term management plan for the species. The guide was developed in 1982 and revised in 1984 and 1986. It was signed by the Forest Supervisor of the Plumas National Forest in 1987.

The species management guide documented the agreed conservation strategy for the closed-lip penstemon as follows:

1. Develop regulated silvicultural prescriptions that maintain viable population levels in species locations.
2. Establish protected population areas within the occupied habitat to maintain the geographic and genetic variety within the species.

Regulated silvicultural prescriptions will be developed to maintain viable population levels in species locations for timber harvest, fuels treatment, site preparation and release. These will be developed by timber, fuel and biological specialists through the interdisciplinary team process. The regulated prescription may apply throughout the species habitat except in protected populations.

To maintain geographic and genetic variability within the species, protected population areas will be established within the currently occupied habitat of the penstemon. Management activities that enhance or do not affect the characteristic for which the protected population location was chosen may occur.

In developing reserve areas, the recommendations for Bullfrog Ravine (southwest of Round Valley Lake), Argentine Rock and Hartman Bar Ridge (south of the Middle Fork of the Feather River) populations will be more conservative and restrictive, since these are the known outlying populations on National Forest lands within the range. These populations are also not as large as populations south and east of Bucks Lake. Historically, smaller populations have shown a higher probability of extirpation (Wilcox, 1986). By maintaining the Bullfrog Ravine, Argentine Rock and Hartman Bar Ridge locations, the genetic diversity of the species will probably be maintained. As the results of ongoing monitoring are disclosed, the species management guide will also be updated accordingly.

## CONCLUSIONS

Development of a species management guide with long-term management strategies for the closed-lip penstemon will hopefully ensure the preservation of the species. Monitoring responses to different types of disturbances will also provide a growing body of information about

the species biology and ecology of the plants, which will ensure sound, long-term management for *Penstemon personatus*.

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# Ecology and Management of a Timber Rattlesnake (*Crotalus horridus* L.) Population in South-Central New York State

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**Abstract:** Timber rattlesnake use of a communal denning area in south central New York State was studied from 1983-1986. Shale outcrops and talus slopes were abundant, and provided the primary source of cover for the snakes. Snake use of rock cover during the spring, summer and fall was significantly non-random. Most apparently suitable cover was never used, some was occasionally used and a few sites were heavily used. In addition, rattlesnake use of outcrops varied significantly between years. Some outcrops that had no rattlesnakes one year had many sightings the next. Habitat use also varied with size-class. Large rattlesnakes, especially gravid females, were more sedentary than small rattlesnakes and were resighted significantly more often. Recommendations for appropriate denning area management techniques include: the construction of rock piles, spot-thinning at basking sites and posting and security patrols. Because rattlesnake habitat use patterns are so variable, long-term and repeated observations are necessary for accurate population monitoring.

Pages 255 - 261. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Populations of the timber rattlesnake (*Crotalus horridus* Linnaeus) have been reduced or extirpated by anthropogenic disturbance in many areas. In New York State, the decline of the timber rattlesnake has warranted its listing as a threatened species by the New York State Department of Environmental Conservation. Between April 1983 and October 1986, a field study of a timber rattlesnake population on a Nature Conservancy preserve in south central New York was conducted. The goals of this study were to document the basic ecology of the timber rattlesnake at the site and to recommend management techniques.

## STUDY SITE

The study site was located in Chemung County, New York on the Appalachian Plateau. The denning area was located in shale outcrops and talus along the crest of steep, south-facing bluffs that overlook the Chemung River. Forest cover consisted primarily of oaks and hickories. Forest cover on the level uplands adjacent to the bluffs consisted of oak-pine or beech, mixed northern hardwood stands. Gypsy moth (*Lymantria dispar*) defoliation of trees around the denning area was observed for the first time in 1980, with increasing severity through 1983. Some crown mortality of oaks resulted. Defoliation was not as severe away from the denning area. After 1983, the gypsy moth population crashed, and from 1984-1986, no tree in the area suffered more than 5% defoliation.

## METHODS

The denning area was patrolled several times a month, from April through October, during each year of the study. The search pattern consisted of walking slowly through the denning area and stopping at rock outcrops, piles, slides and crevices to check for timber rattlesnakes. The date, weather conditions, time, duration and area of each search were recorded. Whenever a rattlesnake was seen, its location relative to one of 13 numbered reference stakes or known rock feature was recorded. The time of observation, air temperature, approximate length of the snake, color phase, position, reproductive state (if discernible) and behavior upon approach were also noted. It was often possible to identify repeat sightings of individual snakes, particularly gravid females, because many individuals were sedentary and had identifiable markings. Brief security checks (less than 1/2 hour) of gravid females nearing parturition were not included in the study results except where

specifically noted. The location and number of road-killed rattlesnakes on perimeter public roads or on neighboring lands were also recorded. In addition to the weather and temperature data gathered in the field, monthly weather summaries for the area were obtained from the National Climatic Data Center of the National Oceanic and Atmospheric Administration.

Sound mast on the forest floor in the denning area was sampled each year by count quadrat (Smith, 1974) in mid-July to estimate residual mast from the previous year, and between September 28 and October 2, to estimate the current year's production. A 0.25 m<sup>2</sup> quadrat was randomly laid down four times within 10 m of each reference stake. Sound mast inside the quadrat only, both above and below the leaf litter, was counted and recorded by species.

Small mammal populations were also sampled in the denning area during the fourth week of each September. Fifty Sherman live traps were set along the line of reference stakes. In 1983, the traps were set about 10 m apart, between stakes one and six. In all other years, the traps were set about 15 m apart, between stakes one and thirteen. The traps were baited with apple slices, checked daily and rebaited as necessary. The traps were operated for six or seven nights each year. Captured mammals were recorded by trap number and species, then released unmarked if alive. Weather data (e.g., moon phase, cloud cover, wind, rain and minimum temperature) for the trap-night were also recorded to note unusual weather patterns that might affect trap success.

Mast production in the denning area was tested for non-randomness between years with a Chi-square goodness-of-fit test. Small mammal trapping success rates were tested by one-way analysis of variance (ANOVA) with arcsine transformation, followed by the Student-Neuman-Kuels (SNK) multiple range test, to determine significant differences in the trapping success rate between years (Scheffler, 1980). Three different tests were used to analyze snake observation data. They were ANOVA with SNK, Chi-square goodness-of-fit and Student's t-test (Scheffler, 1980). Interactions between some site components (e.g., mast production and mammal trap success) were tested with correlation analysis. Monthly snake sighting frequencies based on less than five hours of search (six instances) were not included in statistical analyses. All tests were conducted at the 0.05 level of significance.

## RESULTS

There was significant variation in annual mast production in the denning area ( $X^2=31.85$ ,  $p<.005$ , 3 d.f.). Mast production in 1983 was

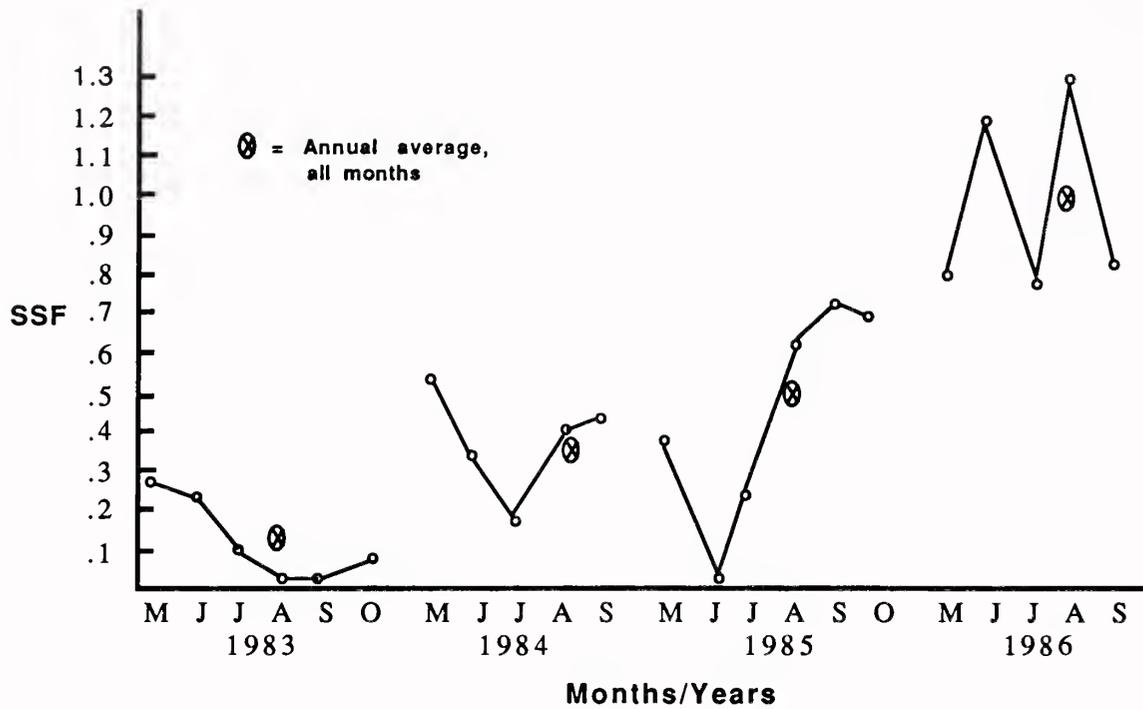


Figure 1. Monthly snake sighting frequency, 1983-1986.

measured at 0.62 nuts/m<sup>2</sup> and consisted entirely of chestnut oak (*Quercus prinus*). In 1984, production was 0.08 nuts/m<sup>2</sup>, including both hickories and oaks. Mast production increased sharply in 1985 to 2.2 nuts/m<sup>2</sup>. Most of this material (80-90%) was from chestnut oak. In 1986, production declined to 1.4 nuts/m<sup>2</sup>. Northern red oak (*Quercus rubra*), white oak (*Quercus alba*) and the hickories produced similar amounts of mast in 1986. There was no apparent relationship between annual mast production and either temperature or rainfall during the growing season.

There was significant variation in the small mammal trapping success rate in the denning area between years ( $F=15.42, p<.01, 23 \text{ d.f.}$ ). From 1983-86, it was 3.6%, 1.9%, 24.5%, and 7.0% respectively. There was a significant, positive correlation ( $r=.959, 2 \text{ d.f.}, p<.05$ ) between annual mast production and the small mammal trapping suc-

cess rate in the denning area.

The small mammal most frequently trapped (>80% of all captures) in all years was the white-footed mouse (*Peromyscus leucopus*). Other species included shrews (*Blarina brevicauda* and *Sorex*), meadow voles (*Microtus pennsylvanicus*), red squirrels (*Tamiasciurus hudsonicus*) and chipmunks (*Tamias striatus*).

Approximately 328 man-hours were spent searching the denning area, resulting in 134 timber rattlesnake observations (excluding snakes <1 year old) for a mean snake sighting frequency (SSF) of 0.41 rattlesnakes/hour. The earliest annual observation was April 22 and the latest annual observation was October 30. The SSF varied greatly between months (Figure 1), but there was no significant difference between the pooled monthly means (e.g., all Junes vs. all Augusts) ( $F=.33, 16 \text{ d.f.}, p>.05$ ). There was a difference in the SSF between

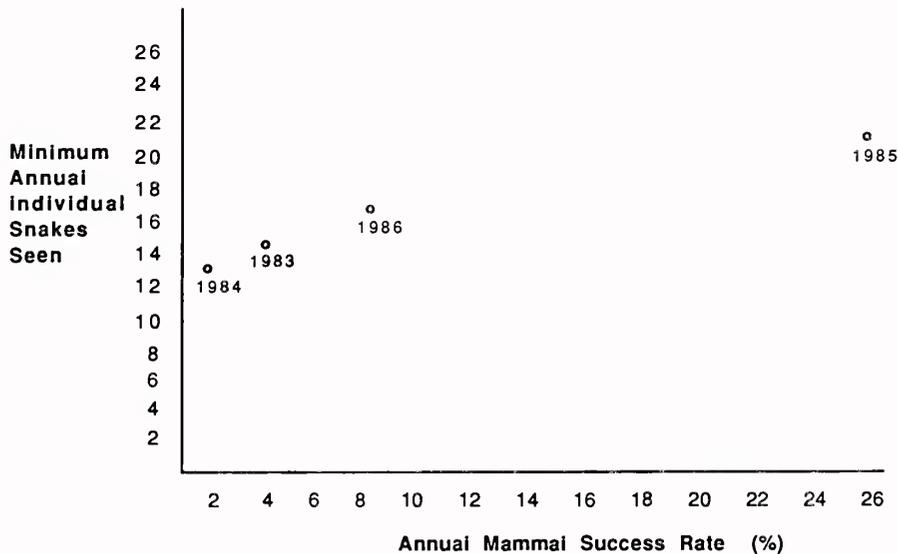


Figure 2. Relationship between annual minimum number of snakes observed and the small mammal trapping success rate.

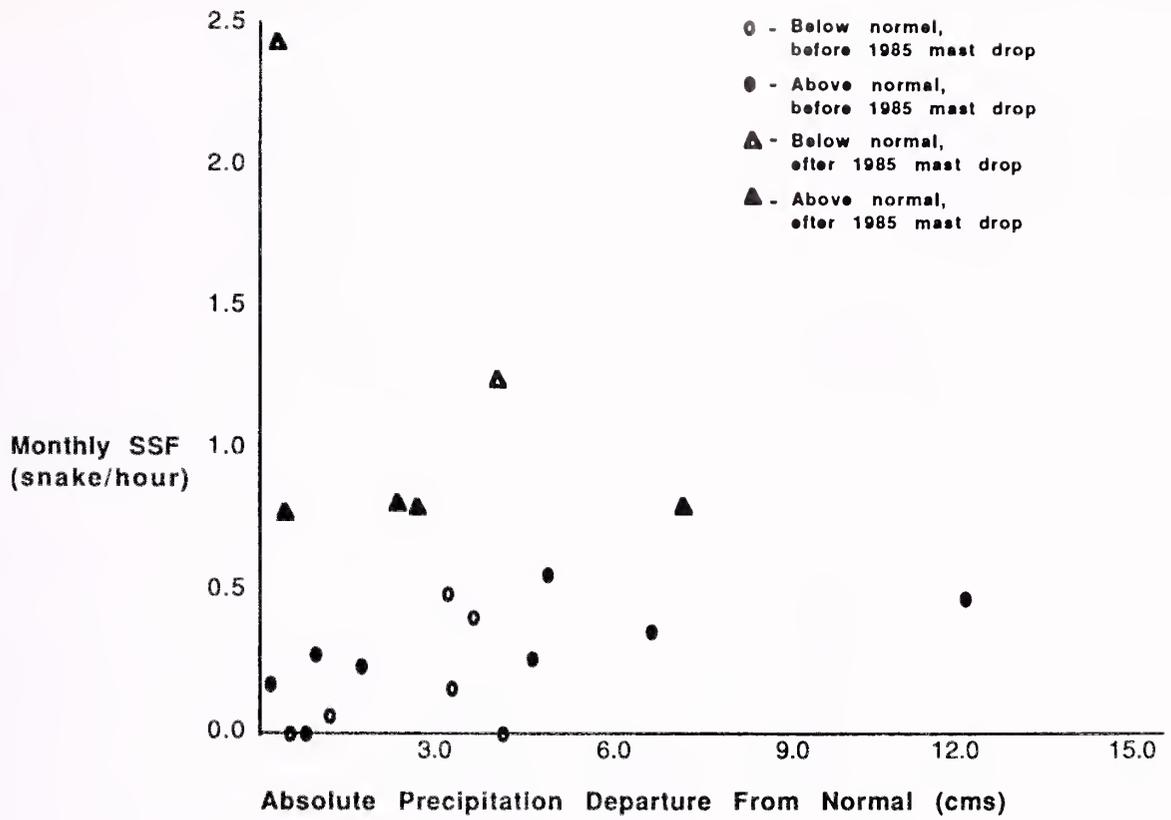


Figure 3. Relationship between absolute monthly precipitation departure from normal and the monthly snake sighting frequency.

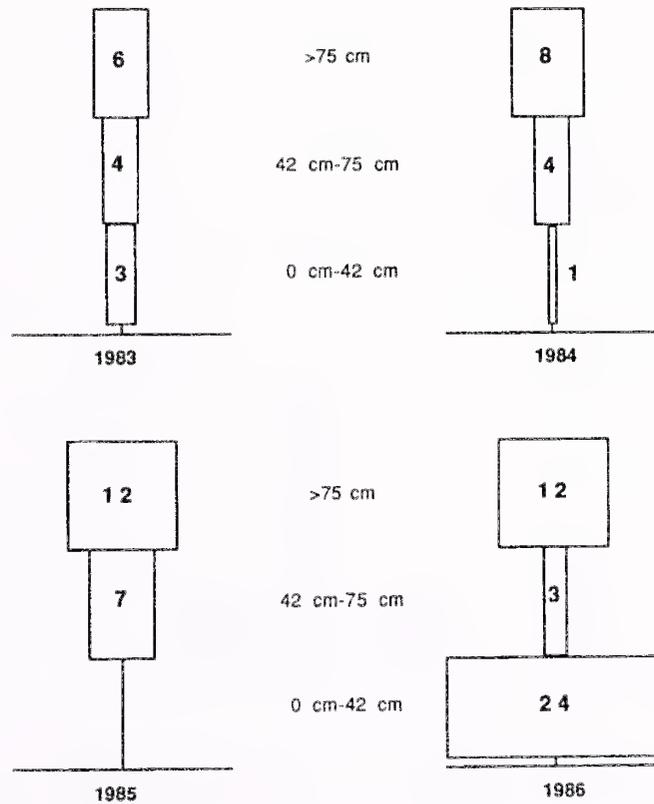


Figure 4. Annual size class breakdown of known, individual snakes.

years. The 1986 SSF's were higher than all other years ( $F=6.16$ , 19 d.f.,  $p<.01$ ). More precisely, the SSF was significantly higher ( $t=5.71$ , 18 d.f.,  $p<.001$ ) after the 1985 mast drop and the recovery of the small mammal population. This increase is partially confounded by repeat sightings of gravid snakes in 1986, however, even discounting these observations, the SSF remained high.

There was less variation in the minimum number of individual snakes seen each year. This number varied randomly about the mean ( $X^2=1.72$ , 3 d.f.,  $p>.05$ ), from 1983-1986, being 15, 14, 21 and 17 snakes, respectively. However, there was a significant correlation ( $r=.978$ , 2 d.f.,  $p<.05$ ) between the minimum number of individual snakes seen each year and the corresponding annual small mammal trapping success rate (Figure 2).

There was no correlation between the SSF and the absolute monthly temperature departure from mean ( $r=.127$ , 13 d.f.,  $p>.05$ ) before or after the 1985 mast drop. There was, however, a significant positive correlation between the SSF and the absolute monthly precipitation departure from mean ( $r=.542$ , 13 d.f.,  $p<.05$ ), before the 1985 mast drop. Months that had higher or lower than normal amounts of rain had higher SSF's (Figure 3). After the 1985 mast drop, the SSF was consistently high (Figure 1). There was no apparent correlation between temperature and rainfall departure from norm, nor was there any trend in precipitation over time.

The predominant color phase of the timber rattlesnake population was yellow (4-year mean, 68%). The largest rattlesnakes seen during this study were about 117 cm. long (three individuals). Most adults' lengths were from 91-107 cm. Adult rattlesnakes were observed significantly more often than juveniles or immatures ( $X^2=27.6$ , 2 d.f.,  $p<.005$ ), except during August and September, 1986 (Figure 4), when three separate litters were observed. No precise determination of the actual rattlesnake population size could be made, however the minimum population size at the end of 1986 was 41 (24 newborns; 17 juveniles and adults).

The number of gravid rattlesnakes seen each year, including 1982, varied from random ( $X^2=10.38$ , 4 d.f.,  $p<.05$ ) and their production of young was also significantly non-random ( $X^2=72.2$ , 4 d.f.,  $p<.005$ ). Although the numbers of newborns observed were approximate, 1982 and 1986 were peak years for reproduction (Table 1). Only one litter was observed in intervening years.

**Table 1.** Number of gravid rattlesnakes & newborns seen each year.

	1982	1983	1984	1985	1986
Number of Gravid Females	5	0	1*	0	3
Number of Young Observed	35	0	1(8*)	0	24

\* Assumed

The timber rattlesnakes did not use all 13 denning area segments equally (Figure 5). The distribution of all sightings from 1983-1986 was significantly non-random ( $X^2=21.03$ , 12 d.f.,  $p<.005$ ). Rattlesnakes were seen almost exclusively within 2 m of a rockpile, crevice or ledge, and they also seemed to exhibit preferences for certain rock features. Most cover that seemed suitable was never used; several sites were occasionally used and a few sites were heavily used. For example, one small outcrop area, less than 30 m in circumference, yielded 32% of all observations.

Habitat use varied between size classes. Adult rattlesnakes (>75 cm) frequently spent many days or weeks at a single rock outcrop, and many

individuals were observed repeatedly. Smaller snakes (<75 cm) were not as sedentary as adults, and they were almost never re-sighted. In fact, only four repeat observations of smaller snakes (excluding newborns) occurred from 1983-1986. Rattlesnake use of rock features and shade within the denning area depended on ground temperature in full sun. There was a significant difference ( $t=3.85$ , 26 d.f.,  $p<.001$ ) in unshaded ground temperature for basking and sheltered snakes. When the ground temperature in the sun was greater than 27.8°C, the rattlesnakes tended to be found either under rock ledges or in full shade. When the ground temperature in the sun was less than 27.2°C, rattlesnakes tended to bask in full sunlight. Twelve rattlesnakes were reported killed (seven confirmed) during the study, both on roads and nearby farms.

## DISCUSSION

### Gypsy Moth Defoliation, Mast Production and Small Mammal Populations:

Xeric oak-hickory forests are highly vulnerable to repeated gypsy moth defoliation, especially if they exist in areas subject to disturbances such as fire, wind-throw or erosion (Houston and Valentine, 1985). Repeated defoliation can cause mast failure (Palmer, 1984) and oak mortality, which leads to replacement by hickories or other species (Ganser *et al.*, 1983). Because the denning area forest possesses all these risk factors, severe defoliation will probably recur. As a result, oak mast production will probably decrease at the site.

Oak mast is a primary food of small mammals, including white-footed mice (Hamilton, 1941; Verme, 1957; Whitaker, 1966; Goodrum *et al.*, 1971; Batzli 1977). Since food supply regulates small mammal populations (Jameson, 1953; Bendell, 1959; Fordham, 1971; Smith, 1971; Nixon *et al.*, 1975), repeated gypsy moth defoliation can indirectly cause a decline in small mammals, the timber rattlesnakes' primary prey. This appears to have occurred at the denning area.

### Rattlesnake Observations:

The rattlesnake dens at the study area are typical, in that they are located on a steep slope with a discontinuous forest canopy. Within this habitat mosaic, we did not observe any clear patterns of snake aggregation and dispersal between spring, summer and fall (see Figure 1).

Rattlesnake use of a denning area may be affected by weather patterns. There was a weak, but significant, correlation between the monthly SSF and precipitation departure from normal, before the 1985 mast drop, yet, the exact relationship is inexplicable. The SSF increased with both unusually wet months and unusually dry months.

The only clear pattern involving the number of rattlesnakes seen in the denning area was the correlation between rattlesnakes sighted and small mammal trapping success. However, this correlation has been disputed by other rattlesnake experts in informal discussions. We did not measure small mammal populations directly, but only inferred them from trapping success rates. Statistical comparison of ratios, such as success rates and sighting frequency is risky and may yield misleading results, especially when there is large variation in the denominator (Green, 1979). In addition, we found that our chances of observing rattlesnakes were biased by daily temperature and reproductive condition. Therefore, the correlation between trapping success and snake sightings may only be used to suggest that such a link might exist.

### Rattlesnake Morphology, Population Structure and Size:

The distribution of color morphs in the rattlesnake population was not unusual. Other workers have noted a preponderance of yellow individuals (Martin, 1986) or a rough balance with non-yellow

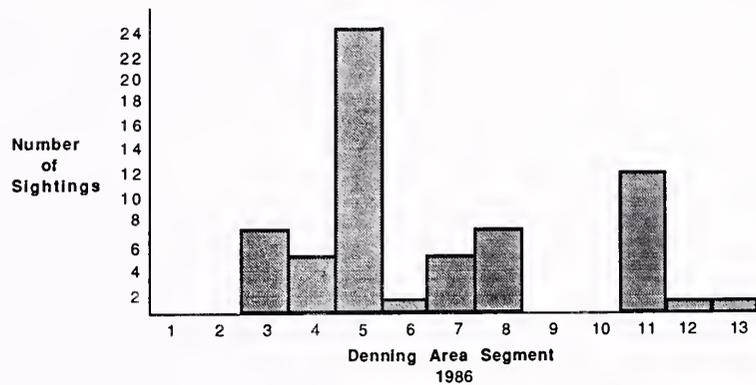
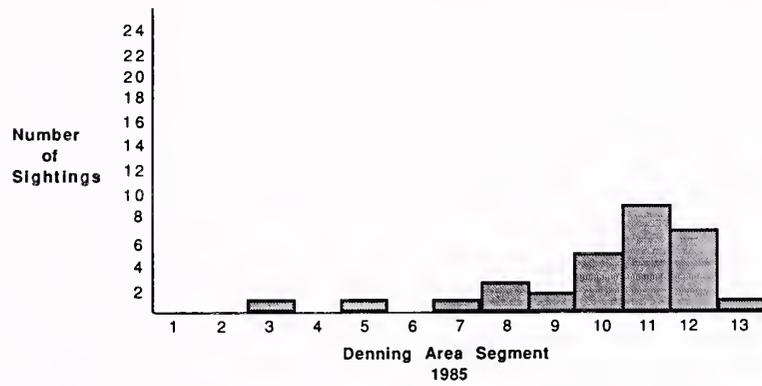
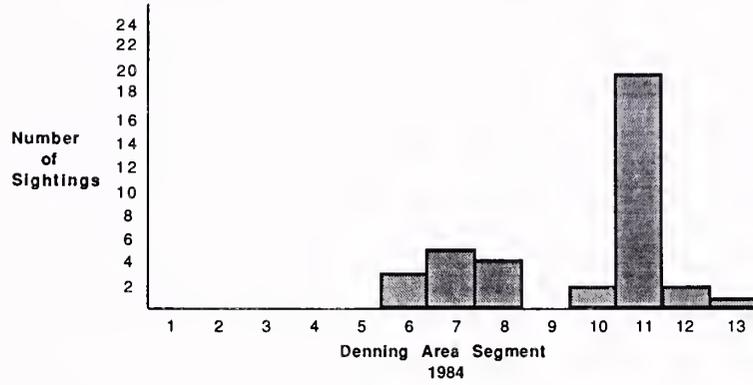
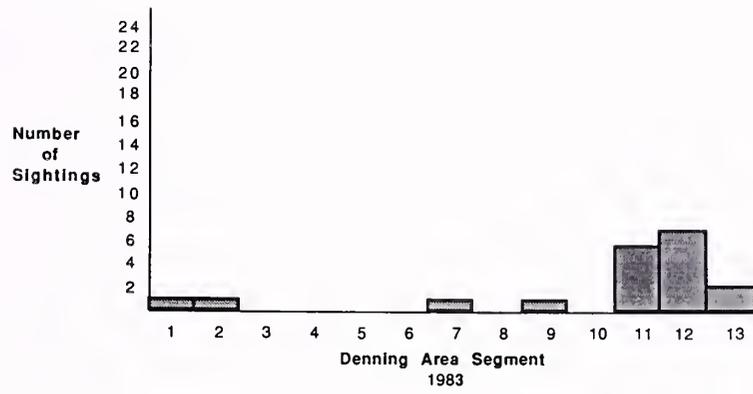


Figure 5. Distribution of all sightings, by year (except those <1 year old).

snakes. The maximum sizes reported in this study (117 cm) are also typical. Maximum sizes for other timber rattlesnake populations were 124 cm (Martin, 1986) and 114 cm (Galligan and Dunson, 1979). The size class distribution observed is also typical. Galligan and Dunson (1979) reported a preponderance of adult snakes and a scarcity of smaller ones. In an earlier study of rattlesnake dens in Chemung County, Axtell (1947) reported that most of the snakes seen were between 89 and 114 centimeters long. This inverted population structure is normal, because the timber rattlesnake is a k-selected species, with high newborn mortality (Martin, 1984), low reproductive capacity (Gibbons, 1972; W. H. Martin, unpubl. data) and long lifespan (Martin, 1986).

#### **Rattlesnake Reproduction:**

Adult female timber rattlesnakes typically breed every three or four years (Martin, 1984). The frequency of reproduction is affected by the females' ability to store body fat (Keenlyne, 1972), which is influenced by environmental factors such as climate and food supply, so variations in these factors lead to a skewed reproductive output by the rattlesnake population. The specific reason for a skewed breeding cycle at our study site is unknown; however, observed fluctuation in the snakes' food supply is an obvious possibility. Because the behavior of gravid females allows them to be observed repeatedly, the SSF during years of peak reproduction may be much higher and bear little relation to the number of rattlesnakes actually inhabiting the area. Because gravid female rattlesnakes do not eat (Keenlyne, 1972), but rely on stored fat, low mammal populations in the denning area should not affect the birth rate.

#### **Rattlesnake Habitat Use:**

Gravid female rattlesnakes were the most sedentary individuals observed in the denning area. They could be located easily and repeatedly once their den was known. Throughout the study, other adult snakes also spent considerable time at single dens, although it is not known why some adults were sedentary and others moved on after a single sighting. Dens with resident snakes could be identified by the matted and compressed nature of the leaf litter at the den entrance.

It is also not known why young snakes used the denning area differently than the adults. The difference was not due to a sighting bias against smaller snakes. The den sites were very open areas, and basking rattlesnakes, with the exception of newborns, were easily seen. Because we deliberately and routinely looked into crevices and underneath ledges, concealed snakes were located regardless of size. In fact, we often observed only a head or tail of a snake and were unable to determine size. It is possible that young snakes either used unobtrusive dens that were never located, or they were more mobile and actually spent proportionately less time in the denning area than adults. Whatever the explanation, the fact remains that smaller rattlesnakes almost never spent an appreciable amount of time at the dens commonly used by adults.

Although some rattlesnakes were killed on the preserve each year, the significance of this mortality cannot be assessed, since we have no precise data on the actual rattlesnake population size nor do we know exactly how many were killed each year (additional rattlesnakes were probably killed which we did not learn about). We cannot, however, assume that the population can withstand this added mortality simply because it has endured for over 150 years. Several nearby, isolated rattlesnake populations are known to have been extirpated since 1940.

## **MANAGEMENT RECOMMENDATIONS**

Gypsy moth defoliation should be considered an undesirable component of rattlesnake denning area ecology. Adverse impacts include the forest damage noted here and in other cited studies, probable impacts upon mammal populations and their potential effect upon rattlesnake habitat use. Specifically, rattlesnake managers should consider small mammal abundance in the denning area to be a possible factor affecting rattlesnake use of the habitat. Management practices that increase small mammal populations may also increase rattlesnake populations. A small mammal trapping program may help explain unexpected variation in rattlesnake sighting frequencies.

However, rattlesnake denning areas within the range of the gypsy moth should not be managed to create oak monocultures. Although oak forests produce large quantities of mast for small mammals, severe defoliation could result in an undesirable loss of forest cover. A mixed-species forest in the denning area will have greater resistance to defoliation (Houston and Valentine, 1985) and will provide a more stable food supply. In addition, a variety of plant communities should be maintained near the denning area to provide a number of small mammal habitats. A diverse small mammal community could provide a more stable and reliable food supply and could minimize long-distance movements of rattlesnakes (and hence reduce mortality on roads and farmlands). Road and farm mortality should be monitored to help assess the rattlesnake population's health. Because rattlesnakes reproduce so slowly, even small additive mortality could affect population size.

Forest cover in rattlesnake denning areas should be thinned whenever it exceeds 75%. This will help ensure that den entrances receive adequate sunlight. When trees are felled in the denning area, care should be taken to avoid damage to standing trees, and slash should be removed or lopped close to the ground. These precautions, recommended by the U.S. Forest Service (Houston and Valentine, 1985), will minimize the number of gypsy moth egg-laying sites.

Because rattlesnakes routinely use man-made rockpiles, pile construction is recommended, especially where natural outcrops or talus are scarce. Because rattlesnakes use their denning area so variably, long-term, repeated observations are needed to assure accurate population monitoring. The potential for a few sedentary snakes to be sighted repeatedly and to create an impression of rattlesnake abundance should be kept in mind. A marking program, or even photographs and detailed descriptions of individual snakes could reduce uncertainty over actual rattlesnake numbers.

Anthropogenic disturbance to denning areas should be minimized, especially during years of high reproduction and should be patrolled regularly to protect gestating females.

## **ACKNOWLEDGMENTS**

The Nature Conservancy provided funds to cover all expenses incurred as part of this research. M. Richmond of the Cornell University Cooperative Wildlife Research Unit provided small mammal live traps and advice on trapping techniques. G. Spak conducted related vegetation studies on the preserve and assisted in data collection. W. Brown of Skidmore College provided literature and advice on rattlesnake study methods and A. Bishop did the word processing. This project could not have been conducted without the extraordinary support of Dr. Arthur Smith, Jr. Dr. Smith, who donated the land to The Nature Conservancy, serves as resident manager and manages his adjacent property as a refuge as well. He compiled most of the rattlesnake observations and conducted most of the live-trapping. He provided

transportation in and out of the study area and shared his great knowledge of the rattlesnakes. The assistance of all these people is gratefully acknowledged.

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# Creation and Management of Artificial Foraging Habitat for Wood Storks

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**Abstract:** The United States population of Wood Storks (*Mycteria americana*) has decreased, and in 1984 the species was included on the endangered species list by the U.S. Fish and Wildlife Service. The decrease has been attributed largely to loss of foraging habitat. In 1985, the Department of Energy created a 14-ha artificial stork foraging habitat at the site of Kathwood Lake on the National Audubon Society's Silver Bluff Plantation Sanctuary in Jackson, South Carolina. Four ponds were constructed, stocked with aquatic prey and managed specifically for Wood Storks. This report covers stork use of the ponds during their first two years. Recorded maxima reached 97 and 151 storks in 1986 and 1987, respectively. The ponds can sustain storks at this foraging intensity for about two months.

Pages 262 - 267. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Wood Storks (*Mycteria americana*) have decreased in numbers in the United States over the past 50 years (Ogden and Patty, 1981), although there has been some disagreement as to the magnitude of the population decline (Kushlan and Frohring, 1986). In 1984, the U.S. Fish and Wildlife Service listed the U.S. population of Wood Storks as endangered (Bentzein, 1984). The decrease in their numbers has been attributed primarily to loss of habitat, particularly that which is important during the breeding season (U.S. Fish and Wildlife Service, 1986). Large areas in southern Florida have been severely affected by changes in hydrologic regimes resulting from water management practices (Kushlan *et al.*, 1975). The drainage of ponds and other water management practices have affected habitat in other parts of the storks' range as well.

In 1985, artificial foraging ponds were created at the site of Kathwood Lake on the National Audubon Society's Silver Bluff Plantation Sanctuary in Jackson, west-central South Carolina. These ponds have been managed specifically for storks. Wood Storks have foraged in artificial habitats for many years. They feed in abandoned rice fields along the coast of South Carolina, and they visit fish hatchery ponds when the water levels are lowered. While such areas present good foraging habitat for storks, they are not managed specifically to provide optimal habitat for the birds.

The National Audubon Society made an early attempt to provide artificial stork foraging habitat between 1969 and 1978 near the Corkscrew Swamp Sanctuary stork colony in southern Florida (J. Hansen, pers. comm.). The purpose of the project was to investigate the feasibility of providing fish in artificial ponds for storks to feed upon. Eleven ponds, with a total area of less than 13 ha, were built and stocked with fish. A major problem encountered was difficulty in maintaining water levels in the ponds due to water loss through a porous soil base. The ponds were not gravity-fed, but maintained by pumping. It was encouraging that Wood Storks and other wading birds did feed in the ponds when they were drawn down, but the experiment was abandoned in 1978.

The Kathwood ponds were designed specifically to provide foraging habitat for storks. The lake bed was divided into four impoundments and stocked with aquatic prey. Management of the ponds included not only making the ponds available to storks, but stocking and managing the ponds for the greatest reproduction of the fish introduced into them. Because other species of wading birds also eat fish, it was necessary to provide sufficient food for other waders as well, in order to insure sufficient food for the storks. The new ponds were built during

the summer and fall of 1985, and first made available to storks in 1986. Details of management for the aquatic prey are discussed elsewhere (Coulter, 1987). The history of the ponds and results of their first year of use were presented by Coulter *et al.* (1987). I summarize here the overall management scheme and present results from 1986 and 1987.

The design of the ponds and development of management plans has been a joint effort of the U.S. Department of Energy, E.I. Du Pont de Nemours and Company, the U.S. Fish and Wildlife Service, the National Audubon Society, and the Savannah River Ecology Laboratory. The Audubon Society is responsible for the day-to-day operations of the ponds and the Savannah River Ecology Laboratory is responsible for management and research.

## METHODS

### Pond Design:

The Kathwood foraging ponds were created by modifying Kathwood Lake. This was an artificial lake, created in 1850, when a 14-ha depression was diked and water was diverted from Hollow Creek by way of a canal. To create the foraging ponds, the lake bed was divided into four impoundments, separated by internal levees (Fig. 1). Pond 1 is about 4.9 ha; ponds 2 through 4 are 4.6, 4.8 and 1.9 ha respectively. A water control structure in the diversion canal from Hollow Creek governs the amount of water entering the pond system, and, thus, controls the potential water levels in the ponds. Water enters pond 1 from Hollow Creek, flows successively through ponds 2, 3 and 4, and returns to the creek from pond 4. The water level in each pond can be raised and lowered independently. Pond 1 can only be lowered partially, but the water is shallow in this pond, and it is available to storks throughout the year. Ponds 2, 3, and 4, when full, are deepest (about 1.5 m) in central channels, but they are shallower (about 0.5 m) at the dikes. Depressions were dug into the bottom of pond 4 so that, when the pond is lowered, the prey are further concentrated. Pond 3 has an irregularly shaped bottom, while the bottom of pond 2 slopes gradually from the northwest to its deepest part southeast of the adjacent railroad bridge. The bottom of pond 1 was not changed, and the aquatic vegetation in that pond has been retained.

### Pond Management:

Pond management is designed to provide optimum aquatic prey and water levels conducive to foraging by storks. During much of the

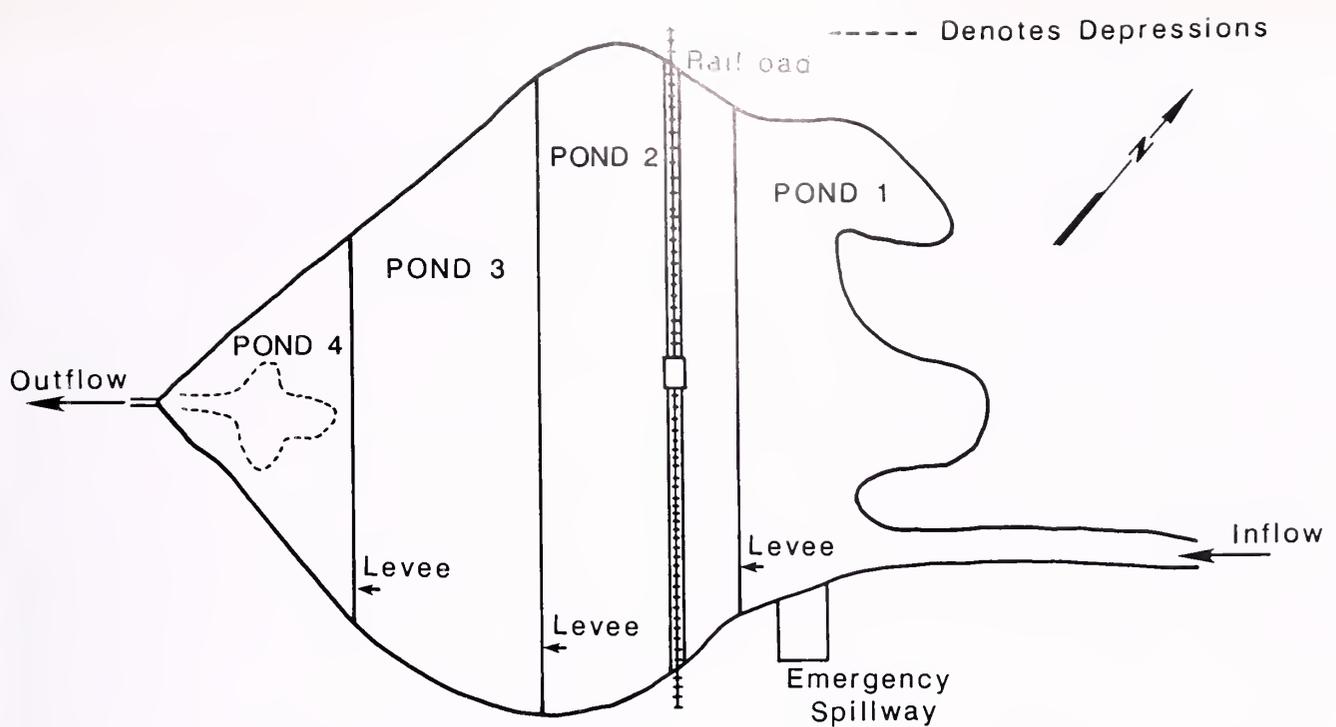


Figure 1. The design of the Kathwood foraging ponds.

year, the ponds should remain full, with water quality managed for the fish. It is hoped that the fish will reproduce well, and that this will lead to high densities of food for the storks. Because fish in high densities do not grow well, the high densities will hopefully keep the fish from growing to sizes too large for the storks to handle. During the breeding season and post-breeding dispersal period, the ponds will be lowered and made available to the storks. Details of the water chemistry management are dealt with elsewhere (Coulter, 1987). The prey are made available to storks by lowering the water level to an appropriate depth (less than 50 cm) for the foraging birds. At the same time, the prey become more concentrated as the water level is lowered. In order to attract storks to the ponds, stork decoys are used during initial draw-downs. While one pond is lowered and available to the storks, other ponds can remain full, as water conditions are managed for optimal reproduction of the fish.

In 1986 and 1987, water levels in the ponds were decreased to between 15 and 30 cm to make prey easily available to storks. Fish were stocked into pond 1 before the 1986 stork nesting season, but not in 1987. Pond 1 was not manipulated, but was kept in a relatively natural state, and the water level remained low. That pond was always available to storks.

#### Fish Stocking:

The ponds were stocked with fish that are preferred by storks in the wild, but also with species that can live compatibly. Bluegill sunfish (*Lepomis macrochirus*) were chosen, because sunfish were common in regurgitation samples from storks of the nearby Birdsville stork colony (Coulter, 1986a; 1986b; 1986c). Bluegill are also available at most commercial and government fish hatcheries. Catfish (*Ictalurus* sp.) are also frequently eaten by the storks. Although brown bullheads (*I. nebulosus*) are not readily available from fish hatcheries, they were chosen for stocking, because their eggs are less vulnerable to predation than the eggs of other catfish. Bullfrog tadpoles (*Rana catesbeiana*) were available at a nearby hatchery and these were added to the ponds as well. We did not record the stocking rate for the tadpoles. Finally, sterile grass carp (*Ctenopharyngodon idella*) were stocked to help control

aquatic weeds. These carp were too large to be eaten by the storks. Fish were initially stocked into the ponds between November, 1985, and April, 1986, at rates of about 20,000 fingerling bluegill, 5,000 fingerling brown bullheads, and 12 mature grass carp per ha (Table 1). The exception, pond 1, which was not actively managed, received only 2,500 bluegill and 250 brown bullheads per ha, and no grass carp. In order to increase the chances of storks foraging in the ponds, we decided to make pond 4 particularly attractive to storks. We stocked bluegill into the pond at a rate of 53,000 per ha.

Table 1. Numbers of bluegill, brown bullheads and sterile grass carp stocked into the Kathwood foraging ponds in 1986 and 1987.

YEAR	SPECIES	Total numbers of fish stocked			
		Pond 1	Pond 2	Pond 3	Pond 4
1986	Bluegill	12,000	91,200	94,400	100,000
	Brown bullhead	1,200	22,800	23,600	9,600
	Grass carp	0	57	59	24
1987	Bluegill			36,000	14,000

During the fall of 1986, after the ponds had been filled with water, we sampled for fish and consulted with Claude Boyd, a fish specialist, to determine whether additional fish stocking would be necessary to prepare for the 1987 season. We decided to add 7,500 bluegill per ha into ponds 3 and 4. These fish were added to the ponds in early January, 1987 (Table 1). Pond 2, which was available to storks for only a brief period in 1986, did not require any fish.

#### Censuses:

Wood Storks and other wading birds were counted from the ground at roughly two-hour intervals throughout the day. On 18 days in 1986 and 25 days in 1987 we counted the numbers of adult and hatching-year storks.

## Fish Densities:

Fish densities were determined with a meter-square throw trap (Kushlan, 1974; 1981), and this sampling technique provided density measures comparable to the figures we obtained at other foraging sites. In two of the four ponds (ponds 3 and 4), it was only possible to sample fish when the water level was low; when sampling fish, every effort was made to avoid disturbing the storks. In 1986, fish were only sampled when no storks were present at the ponds, and sampling was discontinued when storks arrived at the ponds. Because fish were sampled primarily in the lowered ponds, sampling was only possible on a few days. We discovered in 1986 that the storks fed primarily in the early morning and late afternoon, and either rested in the area or left to roost elsewhere during other hours. In 1987, we sampled prey density about once a week after the storks had finished foraging during the morning. If storks were resting in the area, they usually left, but the disturbance did not appear to inhibit them from returning to feed later during the day.

## RESULTS

### 1986 Seasonal Use:

In 1986, the storks at the nearby Birdsville colony had a very successful breeding season, in part because prey was locally abundant. At foraging sites used by storks, the average prey density was  $21.8 \pm 54.1$  (s.d.) per  $m^2$ , which was greater than densities recorded in 1984 and

1985 (Coulter, 1987). We did not expect the storks from the colony to visit the Kathwood ponds until the colony began to disperse in late July. In anticipation, we began lowering pond 2 on the third of July, and placed decoys in ponds 1 and 2 (Table 2). We observed other wading bird species but no Wood Storks.

On July 14, we began lowering pond 4, and a few days later moved the decoys to that pond. On July 30, we saw four juvenile storks in pond 4 (Fig. 2). The numbers increased rapidly and on August 10, 97 storks were observed. Due to predation, the fish densities dropped from 14.7 bluegill per  $m^2$  on 1 August to 3.4 bluegill per  $m^2$  on August 12. As prey density decreased, the numbers of storks and other wading birds also declined. On August 14, we counted only eight storks and 22 other wading birds.

It seemed time to fill pond 4 and make pond 3 available. In order to avoid oxygen and temperature stress to the remaining fish in pond 4, we wished to fill that pond quickly by emptying pond 3 into pond 4. We felt that it was best to have one pond always available to the birds, and chose to lower pond 2 before manipulating ponds 3 and 4. Pond 2 was lowered on August 8, and by August 15, storks were foraging in ponds 1 and 2.

We began lowering pond 3 and filling pond 4 on August 18. Storks began foraging in pond 3 on August 22. Their numbers increased again, and we counted 54 storks on August 30, and 87 storks on September 18. Fish densities dropped from 25.8 bluegill per  $m^2$  on September 9 to 15.7 bluegill per  $m^2$  on September 23. The densities were still sufficiently high at that time to support many foraging birds.

We felt that, by late September, it was no longer necessary to feed

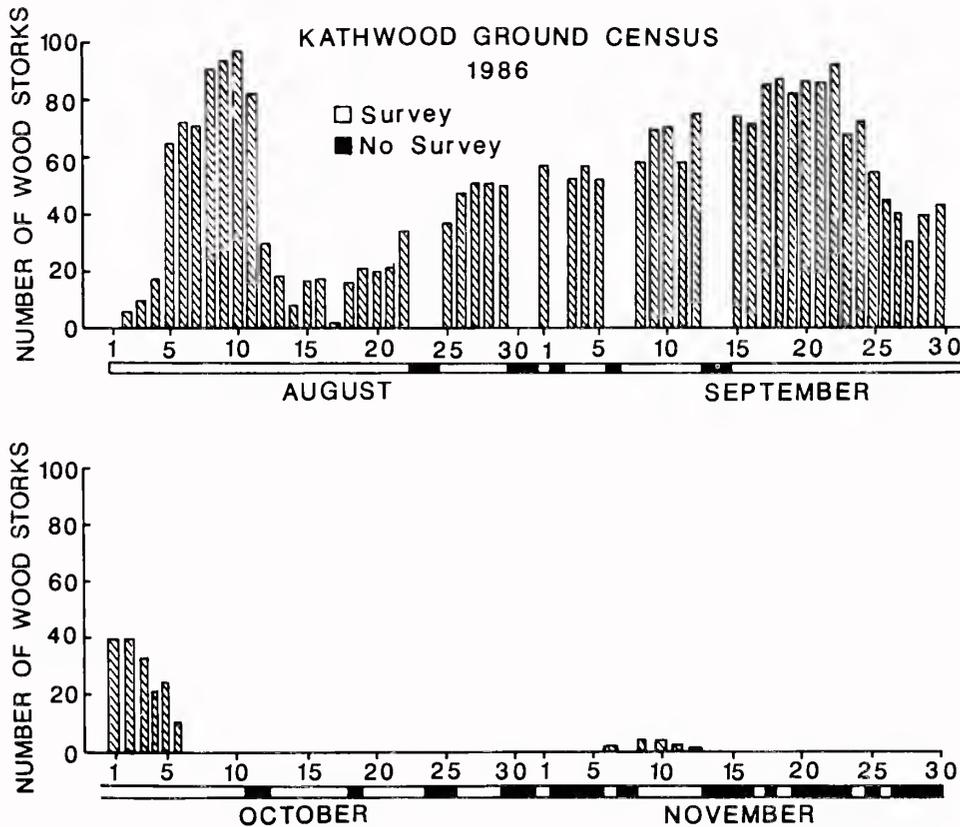


Figure 2. Maximum numbers of Wood Storks observed during ground censuses at the Kathwood foraging ponds during 1986.

the storks, because many storks would be dispersing naturally from the area. Pond 3 was filled, and, to gain additional experience, we lowered pond 2 for a few days. We began lowering pond 2 on September 25, and began filling pond 3 on October 1. Storks were foraging in pond 2 by October 26. We began filling pond 2 on October 6.

**Table 2.** Overview of management and observations at the Kathwood foraging ponds in 1986.

POND 2	POND 3	POND 4
3 July began lowering pond 2		
7 July put decoys into ponds 1 and 1		
9 July began filling pond 2		14 July began lowering pond 4
		24 July moved decoys into pond 4
		30 July 4 juvenile storks
		1 August fish density, 14.7/m <sup>2</sup>
		6 August fish density, 8.3/m <sup>2</sup>
8 August began lowering pond 2 fish density, 9.8/m <sup>2</sup>		12 August fish density, 3.4/m <sup>2</sup>
15 August storks began foraging in ponds 1 and 2	18 August began lowering pond 3	
21 August 21 storks foraging in pond 2		
22 August began filling pond 2	22 August storks foraging in pond 3	
	9 September fish density, 25.8/m <sup>2</sup>	
	23 September fish density, 15.7/m <sup>2</sup>	
25 September] began lowering pond 2		
26 September storks foraging in pond 2		
	1 October began filling pond 3	
6 October fish density/ 12.2/m <sup>2</sup> began filling pond 2		

#### 1987 Seasonal Use:

In 1987, the storks again had a very successful breeding season, although the prey densities at local foraging sites were lower than in the preceding year. The average density was 8.42 individuals per m<sup>2</sup>. As in 1986, we began lowering pond 2 on July 6, in anticipation of colony dispersal in late July. On July 8, we saw eight storks in the pond (Fig. 3).

On July 13, we began lowering pond 4. On July 14, we recorded an average density of 212.0 bluegill per m<sup>2</sup> in that pond, and observed storks foraging. The numbers of storks increased to 105 by August 2, but the birds had depleted the fish to 17.0 per m<sup>2</sup> by then.

We began lowering pond 3 on August 6, and on the next day began filling pond 4. We felt that we could encourage the birds into pond 3 without lowering pond 2, as we had done in 1986. The transition took longer than we anticipated, and storks left the pond. We saw no storks at the ponds from August 6 to 8. On August 10, we saw a single stork. The number of storks increased to 124 on August 29. Prey densities decreased from an average of 74.6 on 7 August to 32.3 on September 3. On that day we began lowering pond 2 and filling pond 3. The numbers of storks continued to increase to a high of 151 on September 16. By that day, the prey density in the pond had decreased to 1.1 bluegill per m<sup>2</sup>. It seemed that there were few fish left in the ponds, and that the birds would soon be dispersing naturally from the area, so we began filling pond 2 on September 21.

#### Ages of Storks:

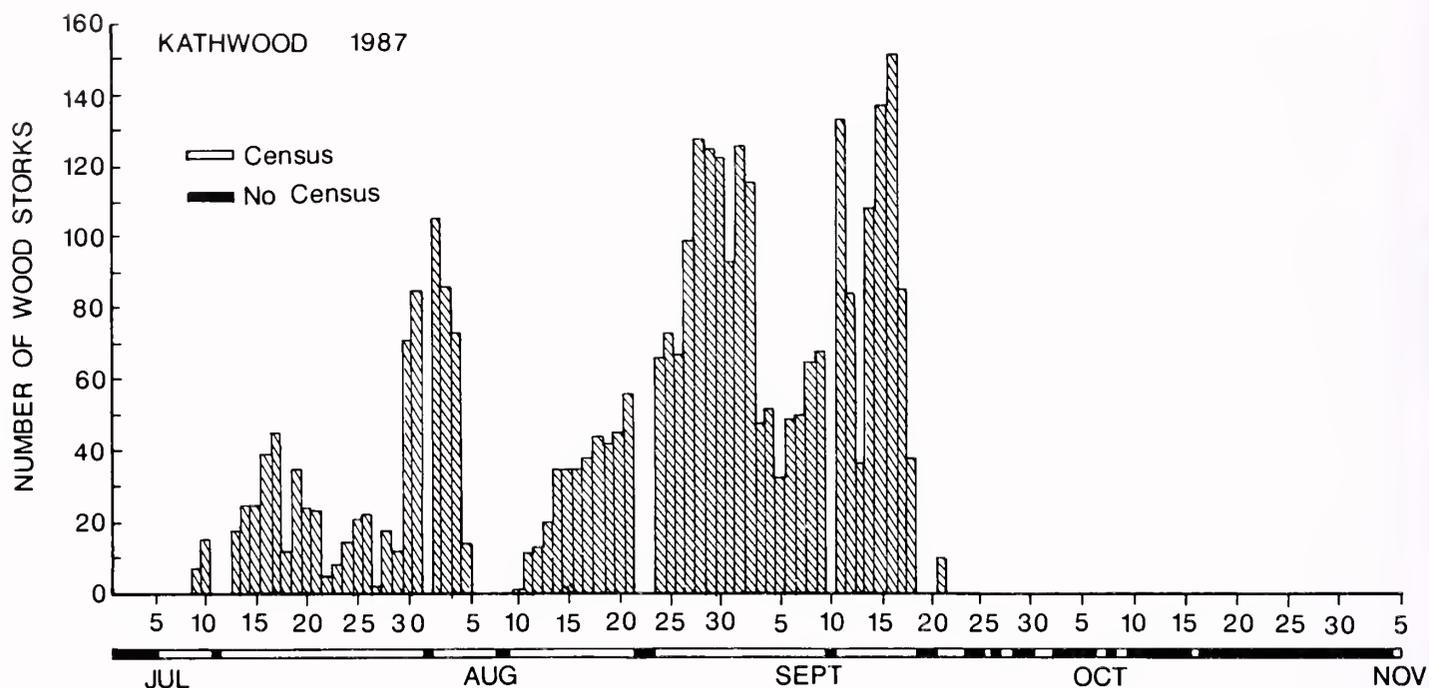
The first four storks to visit the ponds in 1986 were hatch-year birds. During that summer, 89% of the storks were birds of the year. On August 27, we counted 17 adult storks among 51 birds—the greatest number of adults seen at the ponds that year. In 1987, 83% of the storks were hatch-year birds. We counted a maximum of 36 adults among 119 storks on August 28.

#### Other Wading Birds:

In addition to Wood Storks, we observed other wading bird species including Great Egrets (*Casmerodius albus*), Snowy Egrets (*Egretta thula*), Cattle Egrets (*Bubulcus ibis*), Great Blue Herons (*Ardea herodias*), Little Blue Herons (*Egretta caerulea*), Green-backed Herons (*Butorides striatus*), and White Ibis (*Eudocimus albus*). Among our sightings of 8,204 and 7,838 wading birds made in 1986 and 1987, 47% and 58% were of species other than storks. At least one other wading bird species was recorded during every day that storks were observed. The numbers of these other species fluctuated during the seasons. Great Egrets and Great Blue Herons were recorded regularly, and Great Egrets were by far the most abundant wading bird. We counted yearly maxima of 122 of this species on September 23, 1986, and 111 on August 28, 1987. In 1986 we noted large numbers of Little Blue Herons, and recorded 65 on September 25. During the next year we saw few birds of this species, and counted a maximum of only 22 birds on July 17. We recorded fewer of the other species, except for Cattle Egrets that fed on insects along the dikes rather than on fish in the ponds.

#### DISCUSSION

The patterns of numbers of storks at the ponds in 1986 and 1987 suggest that the artificial ponds do indeed present appropriate foraging habitat for these birds. In 1986, the storks foraged there for about two months. In late September, we filled the ponds inducing the birds to disperse at a time that they would disperse naturally from the area.



**Figure 3.** Maximum numbers of Wood Storks observed during ground censuses at the Kathwood foraging ponds during 1987.

There was still an ample population of fish in pond 2 at this time. In 1987, the birds depleted prey in the ponds by late September, after about two months feeding, partly because greater numbers of storks visited the ponds during the second year. We feel that the ponds can sustain the storks at this level of foraging intensity for about a two month period, and we should count on this in future management plans.

The presence of large numbers of other species of wading birds meant that they also consumed substantial numbers of fish. In order to supply adequate fish for the storks, it will be necessary to supply food for the other species as well. We hope that the ponds can become largely self-sustaining, requiring minimum maintenance. We hope also that only a minimum stocking effort will be required each year. The fish in pond 2 were largely depleted in 1987, and needed restocking. In both years the other ponds had many large fish that could replenish the fish populations. Many fry have also been caught in the ponds, indicating that the fish have successfully reproduced. During the first few years of the project, the ponds will continue to be restocked as a precautionary measure, but it is hoped that, in the long term, stocking can be largely reduced.

#### ACKNOWLEDGMENTS

I am grateful to the Savannah River Ecology Laboratory for a tremendous amount of support throughout this work. I appreciate the continued support of the U.S. Department of Energy, in particular, W.E. Wisenbaker, A.B. Gould, Jr., and J.R. Jansen. I have been lucky to have worked with a dedicated, enthusiastic and hard-working crew: A.L. Bryan, Jr., S.L. Coe, F.C. Depkin, L.C. Huff, M.A. Rubega, D.J. Stangohr, N.K. Tshipoura, J.M. Walsh, and B.E. Young. It has been a pleasure to work with D.M. Connelly and P. Koehler of the National Audubon Society. I thank the other members of the Kathwood Technical Working Group for their advice: D.M. Connelly, A.B.

Gould, Jr., H.E. Mackey, Jr., W.D. McCort, N.A. Murdock, and J.C. Ogden. C. Boyd has worked with us throughout the program and contributed significantly to the fisheries management plans. I also wish to thank M.H. Smith, J.W. Gibbons, and W.D. McCort for their full support. M.L. Detchemendy, J.M. Novak, and S. Gettier spent long hours solving our computer problems. J. Coleman and L. Orebaugh drew the figures. Finally, I would like to thank the many staff of the Savannah River Ecology Laboratory who contributed in numerous ways. This research was supported by the U.S. Department of Energy, Savannah River Operations contract DE-AC09-76SROO-819 with the University of Georgia, Institute of Ecology, Savannah River Ecology Laboratory.

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**Table 3.** Overview of management and observations at the Kathwood foraging ponds in 1987.

	POND 2	POND 3	POND 4
6	July began lowering pond 2		
8	July fish density, 1.4/m <sup>2</sup>		
9	July 7 storks in pond 2		
12	July No storks in pond 2		
14	July began filling pond 2		
			13 July began lowering pond 4
			14 July 25 storks in pond 4 fish density, 212/m <sup>2</sup>
			21 July 23 storks in pond 4 fish density, 57.2/m <sup>2</sup>
			30 July lowered pond 4 an additional 5 inches 70 storks in pond 4 through August 4
			2 August 105 storks in pond 4
		6 August began lowering pond 3	
		10 August fish density, 74.6/m <sup>2</sup>	
		11 August 11 storks in pond 3	
		3 September fish density, 32.3/m <sup>2</sup>	
3	September began lowering pond 2		
4	September 52 storks in pond 2		
8	September fish density, 15.8/m <sup>2</sup>		
16	September fish density, 1.1/m <sup>2</sup> 151 storks in pond 2		
21	September began filling pond 2		
			7 August began filling pond 4



**Chapter 9.**  
**STEWARDSHIP**



# Prescribed Burning and Mowing of Coastal Heathlands and Grasslands in Massachusetts

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**Abstract:** Experimental plots were established on Nantucket Island, Massachusetts in 1982 to investigate methods for managing rare, grassy, coastal heathlands. Three techniques: biennial burning, mowing in the summer, and burning in the spring, were tested. Compared to control plots, the cover of shrub species declined following each treatment, while the frequency of graminoids and other herbaceous species increased, especially in the summer burn and mow plots. The results of this study suggest several management options for preserving this ecosystem, and these are discussed below.

Pages 271 - 275. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Two islands in southeastern Massachusetts, Nantucket and Martha's Vineyard, contain some of the largest remnants of North American coastal heathlands and sandplain grasslands (Harshberger, 1914; Tiffney and Eveleigh, 1985). These vegetation associations include a continuum of communities that occur within a few miles of the coast from Cape Cod, Massachusetts to Long Island, New York. Dominant species include varying proportions of low shrubs [*Gaylussacia baccata* (Wang.) K. Koch., *Vaccinium angustifolium* Ait., *Myrica pensylvanica* Loisel.], graminoids [*Schizachyrium scoparium* (Michx.) Nash, and *Carex pensylvanica* Lam.], and other herbs. These communities regarded as globally rare (Rawinski, 1984; Godfrey and Alpert, 1985), and several studies are focusing on their protection and management. Many plant and animal species that are regionally threatened occur in these habitats, such as the short-eared owl (*Asio flammeus*), regal fritillary butterfly (*Speyeria idalia*), bushy rockrose [*Helianthemum dumosum* (Bickn.) Fern.], and sandplain blue-eyed grass (*Sisyrinchium arenicola* Bickn.) (Mass. Natural Heritage Program, 1982). Other species, such as the globally rare sandplain gerardia (*Agalinis acuta* Pennell), were once abundant (Taylor, 1923), but may have declined in the absence of disturbance from grazing or burning (Caljouw et al., 1988).

Extensive building of subdivisions and summer homes has eliminated large areas of this coastal vegetation; the sixty thousand acre Hempstead Plains in New York are virtually gone (Conrad, 1935; Cain et al., 1937). In the remaining parcels, encroachment of taller, woody species, particularly scrub oak (*Quercus ilicifolia* Wang.) and pitch pine (*Pinus rigida* Mill.), as well as sumac (*Rhus* spp.) has resulted in the loss of many additional acres.

Paleoecological studies indicate that much of this succession represents the gradual reestablishment of native species following the cessation of sheep grazing in the 19th century (Dunwiddie 1986, 1987, 1989). Many of the extensive grasslands and heathlands that were found in these areas for several hundred years appear to have been created when Europeans cleared large areas of land for agriculture. In pre-Colonial times, this vegetation probably occurred only in a narrow band along the coast where wind and salt-spray suppressed the oak-dominated forests.

Because much of the native range of this vegetation has been lost, aggressive conservation efforts must be made if rare species are to be

protected in the habitat that still exists. In addition to land acquisition by conservation organizations to prevent losses to real estate development, active management is necessary if the encroachment of taller woody plants into heathlands is to be averted. We present here the results of six years of investigations designed to develop and evaluate the effectiveness of different methods for maintaining and restoring coastal heathlands and grasslands.

Both prescribed burning and mowing were examined in this study as potential management options. Historical records indicate that the study areas burned occasionally (by accidental or deliberate ignitions), usually between March and May. Daubenmire (1968) and Wright and Bailey (1982) cite several studies suggesting that burns during the growing season may be particularly effective in limiting woody plants. Therefore, we decided to compare the effects of spring burns, when the vegetation is dormant, with summer treatments of burning and mowing.

## STUDY LOCATION AND DESIGN

Initial studies of sandplain grassland management were begun on Nantucket in 1982 by the Massachusetts Audubon Society in Ram Pasture, an area of grassy heath along the south shore of the island (Fig. 1). Four 50 x 50 m square plots were established to evaluate changes in floristic composition and structure induced by three different treatments, which included spring burning, summer burning and summer mowing (Zaremba et al., 1983). The plots were separated from each other by 10 m wide mown borders. Baseline data on the percent frequency and cover of all plant species were collected from thirty 1 m segments along three randomly located line transects within each plot. Maximum height of dominant species (>5% cover) at each 1 m segment was also recorded.

Prior to treatment, the vegetation in each plot was similar. Each was dominated by *Schizachyrium scoparium* (37-48% cover), *Gaylussacia baccata* (15-18%), and various shrubs (29-51%). The number of species ranged from 36 to 43, of which the vast majority were native perennials or biennials. *Rumex acetosella* L. was the only introduced species frequently encountered in the plots.

One plot was burned in April, 1983, a second in August, 1983, and a third mowed in August, 1983. A fourth plot served as a control. We have repeated the same treatments on a biennial basis since that time, burning or mowing the respective plots in 1985 and again in 1987.

## RESULTS AND DISCUSSION

The focus of this work is primarily on the management of the heathland community as a whole, rather than on particular species. In Figures 2-5, the taxa are grouped according to growth-form (shrubs, graminoids, other herbs). This permits rapid evaluation from a community perspective of the relative success of different treatments for controlling shrubs or enhancing herbaceous growth. Annual monitoring has allowed us to separate year-to-year fluctuations in growth from longer-term trends resulting from the experimental treatments. By comparing only the most recent year's data (1988) with the pre-treatment values in Figures 2-5, we have tried to emphasize trends that have developed from the accumulated effects of repeated treatments.

By using height, cover, and frequency to describe the vegetation in this study, different aspects of plant responses to treatments are emphasized. In mature plants, the height of many herbaceous species is often directly related to the vigor of the individuals, whereas height may reflect age more directly in woody plants. Per cent cover incorporates both the size and form of a plant, and often relates directly to its dominance in the vegetation. Changes in this value may variously reflect changes in vigor, age or structure of the plants. Per cent frequency reveals more about the ubiquitousness of a species in an area. Recruitment of individuals or their disappearance from a population are more likely to be reflected by changes in this parameter.

In the control plot (Figures 2 and 3), the dominant vegetation has shifted from grasses to shrubs in less than six years. The cover and frequency of graminoids and other herbs have generally declined during this period, as shrub values increased. Shrub heights have similarly increased 18-33%, whereas heights of herbaceous species show little consistent change. These data indicate that, in the absence of management, shrubs have increased in size, density and area.

The increase in shrub cover is particularly striking when compared with previous decades. Analyses of aerial photographs from Nantucket indicate that between 1938 and 1975, the relative cover of clonal shrubs [*Gaylussacia baccata*, *Vaccinium angustifolium*, *Myrica pennsylvanica* and *Aronia arbutifolia* (L.) Pers.] increased from 21% to about 30%. This compares with the 16% increase in relative cover of these taxa (plus an additional 14% increase in other shrubs, including species of *Rosa* and *Rubus*) measured in this study between 1983 and 1988. Methodological differences may account for some of this apparent increase in rate of change, but it is clear that shrub encroachment is accelerating in Ram Pasture, and that this is occurring at the expense of the herbaceous plants.

The effects of the experimental treatments were measured by evaluating changes in per cent cover and frequency of shrubs, graminoids and herbs between 1983 and 1988 in each plot. The corresponding values from the control plot were then subtracted, and the differences plotted (Figures 4 and 5). This correction procedure therefore accounts for those changes that would have occurred had no treatments been applied. All three treatments achieved the desired objectives of reducing shrub expansion and promoting graminoids and herbs. As measured by per cent cover, August treatments (burning and mowing) appeared almost equally effective in limiting shrubs and enhancing grasses. Burning in April had less impact on shrubs and did not enhance graminoid growth as much. The per cent frequency values showed similar trends. These patterns were manifested after the first treatment, and have become increasingly evident with repeated applications.

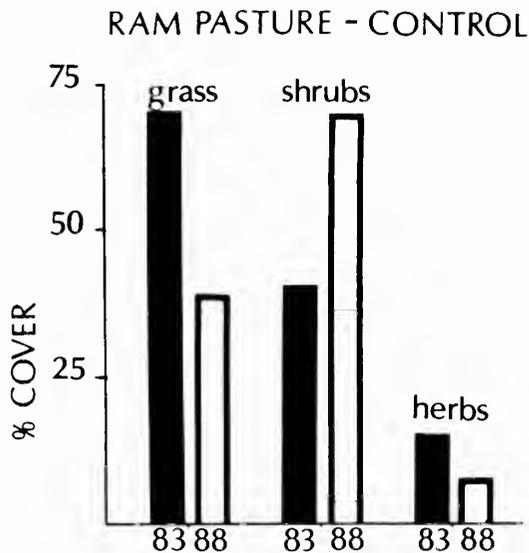
Height measurements of shrubs also indicated differences between the treatments similar to the patterns found in cover and frequency data. Shrub heights have progressively declined after each treatment.



**Figure 1.** Location of study site on Nantucket.

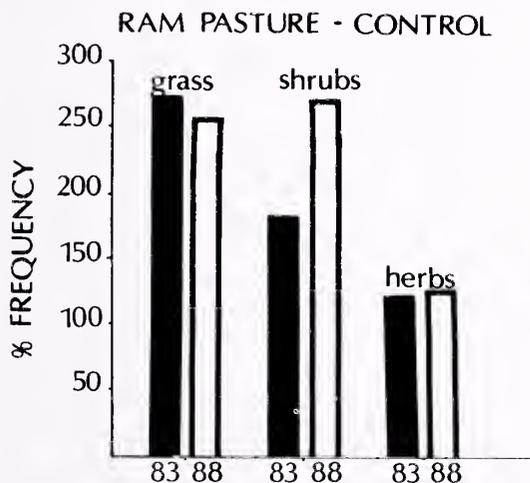
Sampling transects have been permanently marked such that annual monitoring can be carried out in the same locations. Each plot is monitored at the same time each year to minimize interannual variability in the data due to seasonal differences.

In 1986, these studies were expanded to include larger areas of heathlands of similar composition on Nantucket and Martha's Vineyard islands. However, due to the shorter duration of those studies, vegetation trends are less clear than in the Ram Pasture plots. Therefore, only data from the latter will be presented here.



**Figure 2.** Per cent cover of graminoids, shrubs and herbs in the control plot in Ram Pasture for 1983 and 1988.

When compared to their pre-treatment levels, the greatest reductions are seen in the August burn (31-70%) and mow plots (38-53%), with lesser declines in the April burn plot (0-31%). Trends are less clear from measurements of herbaceous taxa that have considerable annual variability in height. *Schizachyrium scoparium* was the only graminoid with a sufficient number of height measurements to permit meaningful comparisons among plots. Height increases in this species have been consistently greatest in the August burn and mow plots; in 1988, values were 35 and 48% larger, respectively, than in 1983. In the April burn plot, 1988 heights were only 15% greater.

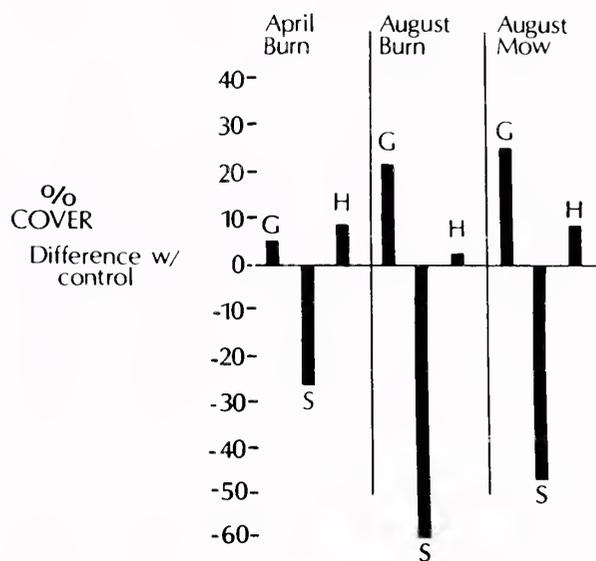


**Figure 3.** Per cent frequency of graminoids, shrubs and herbs in the control plot in Ram Pasture for 1983 and 1988.

Species-by-species comparisons of changes in cover and frequency for major taxa in the Ram Pasture plots (Table 1) highlight the variable responses of individual species to the different treatments and illustrate some of the problems in attempting simple characterization of species as “increasers” or “decreasers” (cf. Anderson and Bailey, 1980). For example, *S. scoparium* may be significantly affected according to one measure (cover or height) but not another (frequency). Other parameters not recorded in this study, such as flowering and seed production, root growth and susceptibility to browsing, may also vary considerably, and have important implications to the long-term status of a species in a community. Furthermore, the timing, frequency and intensity of treatments must be considered in determining species’ fire susceptibility. Some are strongly affected by the season of treatment, such as decreasing in the August burn and mow plots, but increasing in the April burn and control plots (e.g. *Rosa carolina* L.). Repeated annual or biennial burns may eliminate a species over time (some of the shrubs, for example), whereas a single burn may result in their stimulated growth (e.g. *Rubus strigosus* Michx., cited by Anderson and Bailey, 1979).

Many factors contribute to the response of species to fire, and, as already noted, the number of variables that can produce very different changes in a single species is considerable. A review of the available literature reveals many examples of opposite responses of a species to fires in different areas. For example, *S. scoparium* has variously been reported as being severely harmed by a winter burn (Schripsema, 1977), and as having either increased (Schripsema, 1977), remained the same (Anderson et al., 1970) or decreased (McMurphy and Anderson, 1965) in cover or yield after spring burns. Thus it may be

difficult to extrapolate apparent trends in species listed in Table 1 to other locations or to other treatment regimes.



**Figure 4.** Changes in per cent cover for graminoids (G), shrubs (S), and other herbs (H) between 1983 and 1988. Values for the changes measured in the control plot have been subtracted (see text for further explanation). Each of the three plots (April burn, August burn, and August mow) were treated three times during this period.

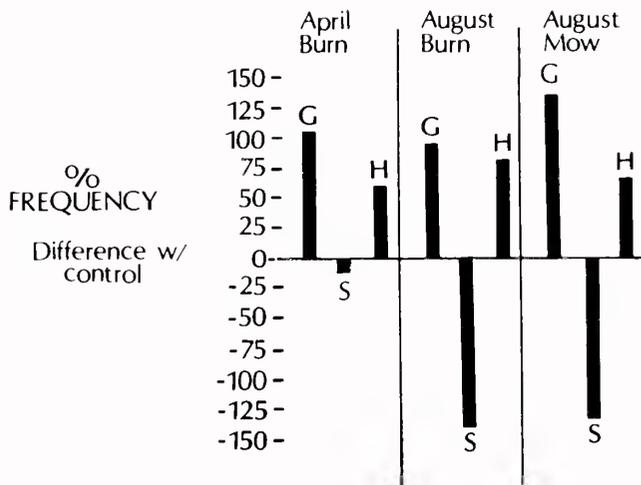
No other studies of the effects of fire on coastal heathland vegetation in the Northeast have been published. Niering and colleagues have described the effects of repeated spring burns on *S. scoparium*-dominated grasslands in Connecticut (Niering et al., 1970; Niering and Dreyer, 1986). After twelve burns since 1968, Niering concluded that spring burning is effective in maintaining grasslands, noting an increase in vigor and flowering in *S. scoparium* and other herbs. Our results (Figures 4 and 5, Table 1) generally support the contention that biennial spring burning can be an effective grassland maintenance tool, but they also indicate that summer treatments (burning or mowing) may be even more effective. Even after three burns, however, our data do not suggest increases in clonal species such as *Gaylussacia baccata* and *Myrica pensylvanica* or overall species richness, as observed by Niering (unpublished data).

Not all species’ responses coincided with management objectives. For example, *S. scoparium* is now taller and at least as frequent as it was prior to treatment, but its cover values have universally declined, a trend opposite to what Niering observed in Connecticut. This is particularly evident in the April burn plot, where the spread of *Rosa carolina* and *Rubus hispidus* L. is also of some concern.

These results suggest that prescribed burning and mowing may be effective methods for maintaining the diverse herbaceous composition of grassy coastal heathlands. Additional studies are in progress to examine alternative treatments (fall burning, herbicide use), as well as to provide more details on other ecological effects, including impacts on rare species, nutrient cycling, wildlife and invertebrate populations.

In addition to heathland maintenance, these studies have suggested methods for restoring heath vegetation in areas where it has largely been eliminated by succession. Combinations of treatments are now

being tested that may prove more effective in removing large shrubs and opening habitats for shorter, less competitive taxa.



**Figure 5.** Changes in per cent frequency for graminoids (G), shrubs (S), and other herbs (H) between 1983 and 1988. Values for the changes measured in the control plot have been subtracted (see text for further explanation).

#### ACKNOWLEDGMENTS

We would like to thank the Nantucket Conservation Foundation for logistical support and permission to use the Ram Pasture site for this study, Robert Zaremba for initially setting up the plots used in this study, William Patterson III of the University of Massachusetts, Amherst, the fire departments of the Towns of Nantucket and Edgartown for help in conducting the prescribed burns, and the many volunteers who have assisted with burns and data collection.

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**Table 1.** Cover (C) and frequency (F) of dominant species in Ram Pasture plots. 1983 data are pre-treatment; 1988 data are after treatments in 1983, 1985, and 1987.

SPECIES	CONTROL		AUG BURN		AUG MOW		APR BURN	
	1983 C/F	1988 C/F	1983 C/F	1988 C/F	983 C/F	1988 C/F	1983 C/F	1988 C/F
<i>Agrostis</i> spp.	1/30	<1/17	2/20	1/33	1/34	1/45	2/37	<1/17
<i>Arctostaphylos uva-ursi</i>			5/27	2/27				
<i>Aronia arbutifolia</i>			2/7	1/7	3/24	2/17	4/17	3/27
<i>Aster dumosus</i>	7/57	3/50	2/50	2/67	4/59	4/79	3/40	2/67
<i>Aster paternus</i>	<1/7	<1/10	12/43	11/60	<1/10	2/10	2/17	4/30
<i>Carex pensyl.lumbellata</i>	15/57	11/63	2/30	5/67	2/24	4/65	2/23	4/67
<i>Danthonia spicata</i>	<1/17	<1/3	<1/23	<1/20			<1/7	<1/7
<i>Festuca ovina</i>	13/63	4/80	6/50	3/57	6/55	3/93	14/77	3/90
<i>Gaylussacia baccata</i>	15/30	23/37	16/37	10/37	17/62	7/66	18/43	19/50
<i>Myrica pensylvanica</i>	2/27	5/27	9/50	3/23	5/28	1/7	4/33	5/30
<i>Toxicodendron radicans</i>					4/28	2/24		
<i>Rosa carolina</i>	4/43	9/77	3/37	1/23	5/59	2/41	2/40	2/60
<i>Rubus hispidus</i>	12/57	19/77	22/60	12/53	12/66	10/79	8/47	8/70
<i>Rumex acetosella</i>			1/17	1/23	<1/14	1/38		
<i>Schizachyrium scoparium</i>	40/97	19/90	37/87	23/93	43/100	33/100	49/90	26/100
<i>Solidago tenuif./gramin.</i>	0/0	<1/10			2/21	1/28	0/0	<1/7
<i>Vaccinium angustifolium</i>	5/17	10/27	4/17	8/17	2/31	18/41	15/47	19/57
<b>Total No. of Species</b>	25	29	34	37	24	32	30	33

# Inventory, Acquisition and Management of the Human-modified Landscape of the "Punkhorn"

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**Abstract:** Structural (descriptive) and functional (relational) attributes of abiotic, biotic, and cultural components of the Punkhorn landscape were identified, hierarchically mapped and interrelated for purposes of acquisition and management, using a highly modified form of the Abiotic-Biotic-Cultural (A-B-C) resource survey/planning methodology. The Punkhorn constituted the largest unprotected, extant natural area on Cape Cod, Massachusetts at the time of this study. A significant remnant biota was identified in the Punkhorn. The relative abundance of "area-sensitive" and "forest-interior" birds attested to the unfragmented quality of the Punkhorn landscape. Atlantic coastal plain pond shores were the single most significant habitat type for rare flora. However, state-listed species were also found associated with roadsides, forest openings, settlement patches, sandplain grasslands, wet meadows and some types of scrub-shrub swamp. Seventeen plant species listed by the Massachusetts Natural Heritage Program were identified at 38 sites in the study area, of which 31 sites constituted new records. Several species have their greatest global abundance in southeastern Massachusetts. A preliminary survey identified two state-listed invertebrates associated with pond shore and quaking bog wetland habitat types in the Punkhorn. Areas of the Punkhorn are also significant for their fundamentally undisturbed archaeological record. Archaeological evidence has confirmed human presence for at least 8,000 years. Successful acquisition of the Punkhorn was responsive to the historical pattern of European land use and tenure.

Pages 276 - 289. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

The Punkhorn, with a land area of over 400 hectares (1,000+ ac.), was the largest unprotected, extant natural area on Cape Cod, Massachusetts when this study began in 1985 (Figure 1). The term 'Punkhorn' has complex etymological roots, but is most probably derived from English and "has traditionally carried subtle connotations of deprecation and ridicule, a sense...of being 'out in the sticks'" (Finch, 1983). Until recently, this cultural isolation obviated development initiatives for the Punkhorn. With development of the Punkhorn imminent, far-sighted individuals, in Brewster, Massachusetts, especially the noted nature essayist Finch, persuaded the town to pass a two year zoning moratorium to allow time to study and determine the future of this natural area.

Development pressures on Cape Cod have become so extreme in recent years that a committee, lead by former U.S. Senator Tsongas, recently proposed a non-binding construction moratorium for all of Cape Cod. It was overwhelmingly approved by Cape Cod's voters. Current Massachusetts legislative initiatives to regulate and dampen the rate of growth on Cape Cod have included various land bank formulas and the establishment of a regional planning entity under the aegis of the Cape Cod Planning and Economic Commission (CCPEDC).

State agencies were canvassed by the Town of Brewster Land Acquisition Committee (LAC) for general information on the Punkhorn study area freshwater fisheries and wildlife potential (Division of Fisheries and Wildlife), acid rain monitoring data (Division of Marine Fisheries) and forest resource capability (Department of Environmental Management or DEM). The study area corresponded to the southwest Brewster Rural-Residential planning zone. This approach proved inadequate to provide environmental information sufficient to support even the initial proposal of a 80+ hectare (200+ ac.) park, situated along 2.8 kilometers (1.75 mi.) of Great Pond (lake) shoreline. Consequently, the author was retained in June, 1986 by the Town of Brewster LAC as Environmental Coordinator for this project. Meanwhile, several key land parcels had been acquired by the Town in 1985, with the assistance of state land

acquisition (Self-Help) funding.

A landscape perspective, as contrasted with an ecosystem view, was explicitly pursued as an integrative tool in this study. Ecosystems are often selected for study based on their homogeneity (Godron and Forman, 1983), yet, as Karr and Freemark (1983) have noted, variation in species and population characteristics, such as distribution and abundance, while a burden to early ecologists in their search for temporally and spatially homogeneous communities, may significantly enhance ecological integrity. Therefore, a landscape perspective may be distinctly contrasted with an ecosystem outlook, since landscapes

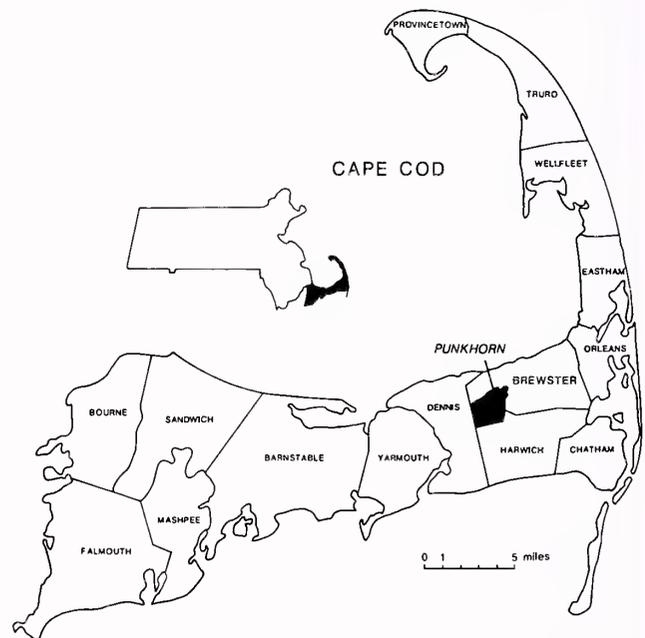


Figure 1. Cape Cod and the Punkhorn

are almost without exception heterogeneous (Godron and Forman, 1983). Godron and Forman (1983), for example, defined a landscape as "a kilometers-wide area where a cluster of interacting stands or ecosystems is repeated in similar form... The boundary of a landscape encloses the area subject to a common overall disturbance regime including both human and natural disturbances, and normally a common geomorphology." For the purposes of this study, the boundaries of the Punkhorn landscape were considered to be the transition zones between areas where natural and human processes were predominant to those places characterized by aggregated human settlement (settlement patches), with their associated infrastructural features.

The decline in effective size and increase in isolation of the Punkhorn remnant natural area, along with the intrusion of competing land uses, has been documented by aerial photography over the past fifty years. This paper considers the development, modification and implementation of an abiotic, biotic and cultural (A-B-C) inventory and planning methodology in the Punkhorn natural area. The hierarchical maps and results of surveys conducted in the Punkhorn are discussed. Implications for planning and management of this and other natural areas in human-modified landscapes are also considered.

**A-B-C Methodology:**

A philosophical and technical basis for resource conservation and landscape planning has been articulated by Dorney (1989), among others. The principles of reverence for land, life and diversity form an ethical triad basic to ecology, and should be fundamental to sound environmental management of natural areas and resources (Leopold, 1949; Dorney, 1977, 1989). Technical guidelines for development of resource survey and planning methodologies may be derived from such philosophical considerations (Dorney, 1978; Dorney and Hoffman, 1979; Bastedo, 1986; Dorney, 1989) (Table 1), and their employment has lead to development of an Abiotic-Biotic-Cultural (A-B-C) resource survey and planning methodology (Figure 2). This approach, first proposed by Dorney (1976), classifies the environment, dividing it into abiotic (non-living), biotic (living) and cultural (human) components. Subsequently, an A-B-C methodology was developed at the University of Waterloo for study of Environmentally Significant Areas in the Yukon and Northwest Territories, Canada (Bastedo, 1982; Hans-Bastedo, 1983; Bastedo *et al.*, 1984; Smith, 1984; Grigoriew *et al.*, 1985; Bastedo,

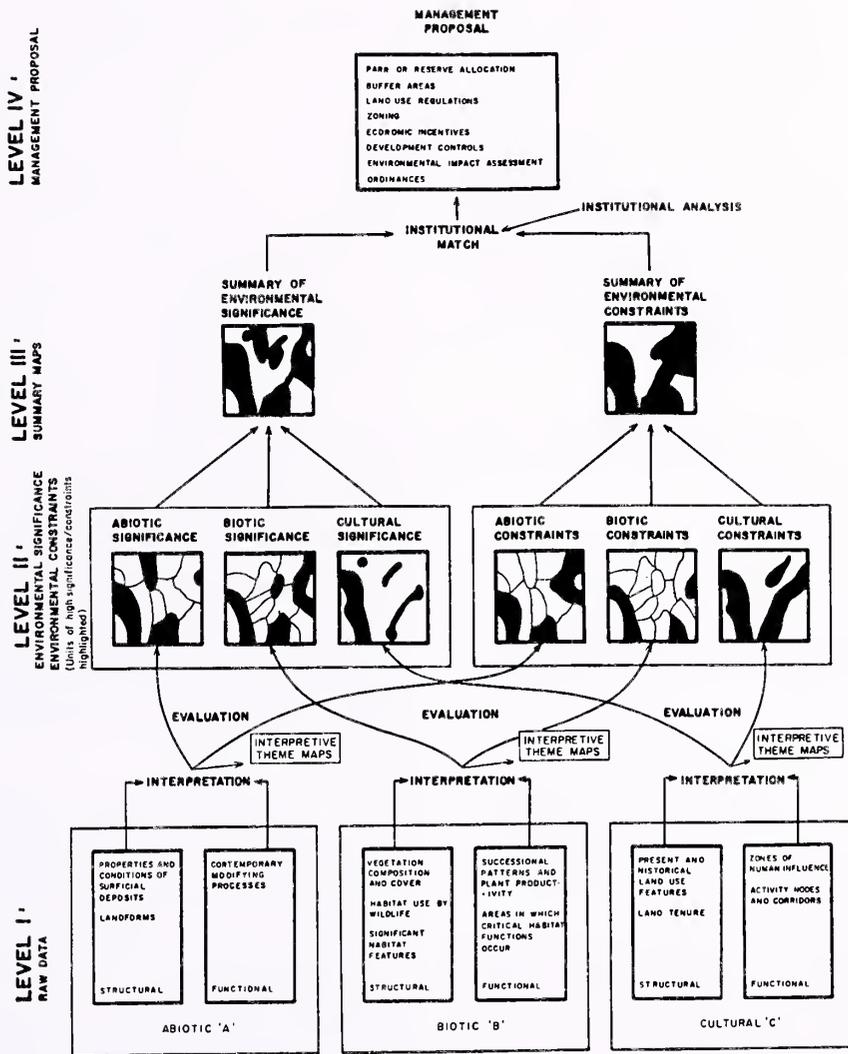


Figure 2. Conceptual model of the A-B-C resource survey and planning methodology (adapted from Bastedo, 1986)

1986; Smith *et al.*, 1986). The method allows the structural (descriptive) and functional (relational) attributes of a landscape to be identified and their significant and sensitive aspects to be hierarchically mapped and interrelated for purposes of planning, acquisition, and management.

Bastedo (1986), in examining the relative merits of inventory and planning methodologies, identified three types of resource survey integration: logistical (integration of field and write-up activities), cartographic and ecological (the combining of raw or interpreted data to identify "ecological" units through interrelationships among resource categories). "Ecological," in the A-B-C usage, applies both to biophysical and human ecological processes. An A-B-C methodology encompasses all three types of integration identified by Bastedo (1986). The use of comprehensive, objective inventory and planning methodologies is essential if conservation efforts are to compete with agriculture, forestry, recreation, urban and industrial development for diminishing lands and resources (Best, 1981; Margules and Usher, 1981; Hellyer, 1985).

While the original A-B-C methodology was developed for surveys of comparatively pristine areas in Canada's north, the general concepts and principles (Table 1, Figure 2) were considered applicable to resource surveys in almost all environments at any scale (Bastedo *et al.*, 1984; Bastedo, 1986). Therefore I decided to modify and develop abiotic, biotic and cultural field sheets, criteria and mapping techniques for purposes of this study. The extensively revised Punkhorn A-B-C methodology uses components of all 20 generic classes of abiotic, biotic, cultural, planning and management criteria assessed by Smith and Theberge (1986) from 22 selected natural areas evaluation systems. The current work was the first application of an A-B-C approach at such a fine geographical scale and in an intensely human-modified landscape context. Subsequently the Waterloo form of this methodology has been applied to an Ontario Provincial Park (Swinson and Grieg, 1987). The development and implementation of an A-B-C methodology at a relatively fine scale of map resolution for a Cape Cod natural area supports the contention of general applicability.

**Table 1.** Technical Guidelines for Development of Resource Survey and Planning Methodologies (adapted from Dorney, 1978; Dorney and Hoffman, 1979; Bastedo, 1986).

Guideline	Description
Data Variables	inclusion of structural (descriptive) and functional (relational) environmental attributes;
Classification	distinct yet relatable methods of delineating and mapping abiotic, biotic, and cultural land units;
Data Acquisition	replicable and generalizable results; sampling intensity sufficient to capture range of ecological diversity;
Data Interpretation	transforming raw data to reflect critical ecological features and processes;
Evaluation	combining interpreted data to differentiate relative ecological values and sensitivities of land units;
Prescription	management recommendations for land units of high ecological significance, boundaries and buffer areas;
Communication	organizing results to be meaningful to park, reserve or environmental agencies, the general public, and other interested parties

Abiotic, biotic and cultural landscape attributes of sample sites (see Forman and Godron, 1986; Iverson, 1988) were recorded on A-B-C field sheets. Abiotic attributes sampled included: exposure type, aspect, slope, site position, ecological moisture regime, landform type, current landform processes (see Walmsley *et al.*, 1980); surficial hydrological descriptors, wetlands classification, water regime, and special modifiers (*cf.* Cowardin *et al.*, 1979).

Biotic attributes sampled, in addition to vegetative descriptors mentioned in the next section, included: general community physiognomy, evidence of fire periodicity, presence of significant species or floral assemblages, probable direction of succession, estimated seral stage, spatial pattern, natural processes influencing vegetation, resilience and ecological fragility. Wildlife indicators were noted, including droppings, tracks, sightings, hearings, evidence of nesting/denning and feeding behavior. Wildlife habitat features noted included escape cover, forage plants, ecotones, resting cover, deer yarding, nesting and migratory bird behavior. Evidence of anthropogenic and natural processes affecting vegetation or wildlife were also recorded.

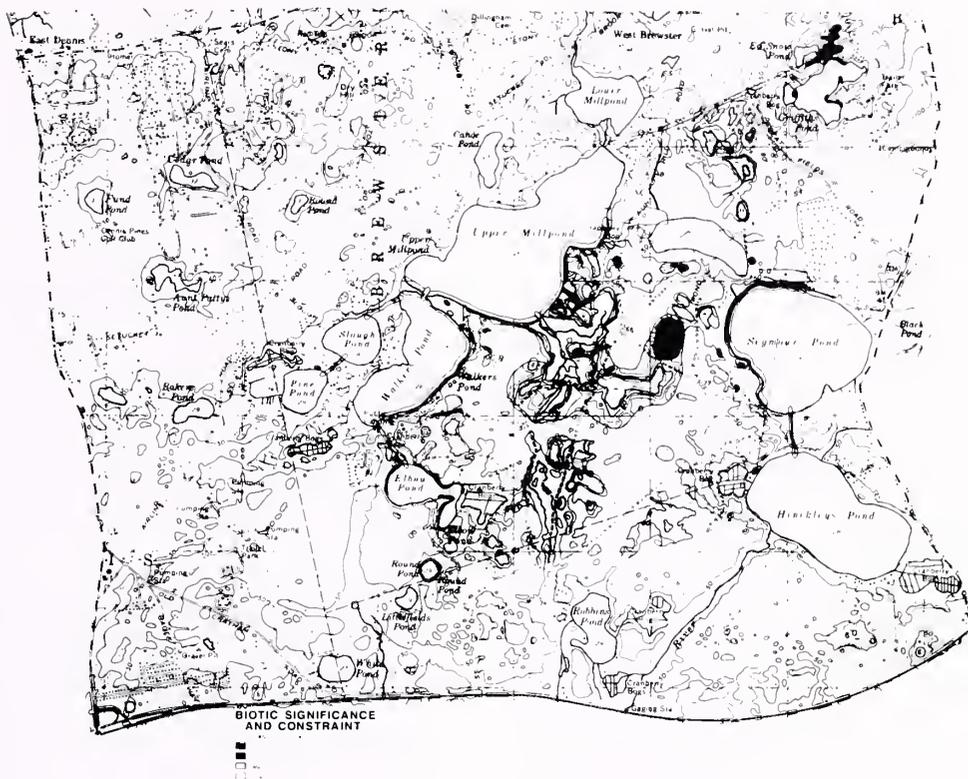
Cultural attributes sampled for forestry, included evidence of cutting, planting, probable purpose, location and extent. Agriculture data were noted for cranberry bogs, water control structures, evidence of maintenance and mosquito control techniques (*i.e.* ditching, aerosurf, bacteri). Information collected for transportation corridors included location, width, estimated frequency of use, nodes with other corridors, intrusion effects (*e.g.* dumping and weedy species), exposure effects (*e.g.* bank instability and desiccation), edge effects and evidence of use (*e.g.* regeneration). Hydro corridor intrusion was noted by type of location (*e.g.* private, municipal), width, nodes with other corridors, effects of intrusion, exposure or edge, evidence of use and management. Settlement patches were described by location, type (*e.g.* subdivision, permanent residence, cottage), status of use (*e.g.* historic, recent, current, potential), septic disposal, proximity to wetland, size of patch, proximity to other patch types, effects of intrusion, exposure or edge and management approach (*e.g.* naturalistic or human-modified). Aggregate/fill extraction sites were described by location, size, characteristic slope, exposure effects, seral stage and probable time of abandonment. Education/nature, interpretation/recreation uses and capability were noted for scenic vistas, natural features, hiking, running, bicycling, horse-riding and motorbike/ATV trails, swimming, boating, hunting, fishing, windsurfing and effects of intrusion, edge and evidence of use, with additional general cultural observations.

Environmental specialists were contacted to provide particular studies or services, including hydrological mapping, soils mapping and surveys of rare plants, breeding birds, migratory waterfowl, salamander vernal ponds, rare dragonflies/damselflies and archaeological sites. Land appraisal and legal services became increasingly important as the acquisition process developed (Palmer, 1989).

## VEGETATION

Vegetation types were delineated using Fall, 1984 1:25,000 infrared photography. Extensive field sampling, particularly focusing on wetland types, further clarified the typology. An effort was made to sample homogeneous vegetative community types. Problems arose in distinguishing "boundaries," seral stages, topo-edaphic transitions, effects of past land use practices and other factors contributing to the heterogeneous landscape mosaic of the Punkhorn (see Forman and Godron, 1986). Ecotones proved the most problematic and were under-represented in field sampling.

Vegetation types were delineated in the field, based on relative



**Figure 3.** Biotic Significance and Constraint (Level II) Map

- \* Unique Significance/Constraint -(Heritage Species; Unique and Restricted Community Types);
- \* High Significance/Constraint -(Regionally and locally significant plants; Significant Vertebrate and Invertebrate Habitat; Wetlands - depending on ecological integrity, landscape context of patch, and degree of human intrusion);
- \* Medium Significance/Constraint - (Upland forest and wetlands depending on ecological ...);
- \* Low Significance/Constraint - (Patches with high levels of recent, current, or imminent human intrusion within &/or proximal to the patch)

abundance (% cover) of species in different strata, providing a means of characterizing vegetation composition, structure and relative abundance. Vegetation types provided a structural basis for delineating habitat types and for assessing the relative effects of anthropogenic and natural disturbance.

Forest samples were taken, using a forester's 10 basal area factor wedge prism, a Suunto clinometer, tree corer and diameter breast height (DBH) tape. Descriptive and inferential measures calculated from forest data for each sample plot and extrapolated per acre by upland forest type included: basal area (square feet); observed and B-level number of trees; arithmetic mean, standard deviation and coefficient of variation of DBH (Zar, 1974) by tree species or species group. For instance, members of the Black Oak complex of *Quercus rubra* L. - *Q. velutina* Lam. - *Q. coccinea* are often indistinguishable, owing to extreme levels of hybridization of these species in southeastern Massachusetts (J. Clements, DEM Regional Forester, pers. comm., 1986). Also recorded were: observed age; representative height, and, for one property as a condition of sale, estimated potential yield for firewood (cords/acre). Since Clements (*ibid.*) was unable to identify site indices for pitch pine (*Pinus rigida* Mill.) dominated forest types on Cape Cod, the B-level was estimated only for oak-dominated forest types (OA; OP) based on the closest equivalent site indices.

## RESULTS AND DISCUSSION

### A-B-C Methodology:

Table 2 identifies four levels of A-B-C maps and some of the primary mapped data and criteria developed for the Punkhorn. This mapping was executed in final cartographic form at a scale of 1:7,200 (1":600'), to be consistent with other town-wide and regional mapping efforts underway on Cape Cod. The original format of the A-B-C methodology integrated abiotic, biotic and cultural data into a single significance map for each category (Bastedo, 1982); however, the revised Punkhorn methodology required development and integration of discrete sets of abiotic, biotic and cultural significance and constraint maps (Figure 2) (Bastedo, 1986). The current study found numerous procedural, definitional, and philosophical problems with the separation of significance and constraint into discrete categories. Therefore, in this study, categories of significance and constraint were effectively subsumed at the Level II mapping stage for biotic and cultural features. Abiotic significance was not delineated because of the ubiquity of significant abiotic features for the Punkhorn (e.g. ground water recharge areas). Level III summary maps of ecological significance and constraint, overlaying abiotic, biotic and cultural attributes, were not explicitly produced.

Level III maps are derivative and may be readily generated through overlaying any combination of Level I, II or IV maps. Consistent with the revised form of the A-B-C methodology (Figure 2) two Level IV Environmental Management maps were produced. Figure 3 illustrates the highest level biotic map produced—that of biotic significance and constraint. In terms of biotic data for the Punkhorn, significance and constraint were considered to be analogous, biotically significant elements (*cf.* Jenkins, 1981) or landscape attributes imposing constraint.

## VEGETATION

Table 3 categorizes the vegetation types delineated within the Punkhorn. Topo-edaphic factors accounted for a substantial amount of the observed forest pattern in the Punkhorn. Punkhorn forests are typical of moderate to excessively drained sites on the Atlantic Coastal Plain, such as in the New Jersey Pine Barrens (Forman and Boerner, 1981). An initial working hypothesis was that the pattern of pitch pine dominated forests in the Punkhorn landscape was an artifact of fire history. While the role of fire in the New Jersey Pine Barrens is well documented (*e.g.* Forman and Boerner, 1981), no ecological or anecdotal evidence of catastrophic fire in the Punkhorn was uncovered in the course of this study; however, some evidence of localized burning was observed. The competitive advantage of pitch pine over white and black oak propagules in more poorly drained sites, such as dry kettleholes and valley bottom deposits, might substantially account for the observed pattern of pitch pine-dominated forest. Pitch pine propagules tend to become established in adjacent forest types, accounting for the proximity of pine-oak (PO) and oak-pine (OP) associations to pitch pine (PP) stands. The oak (OA) forest type is the most distant from the densest pitch pine-dominated communities, offering further support for this hypothesis.

**Table 2. Punkhorn A-B-C Maps and Primary Mapped Data**

<b>ABIOTIC MAPS</b>	
<b>Level I</b>	
<b>Surficial Geology of the Punkhorn</b>	
<b>Soil Types, Drainage, Slope and Suitability for Agriculture</b>	
<b>Ground Water Recharge Areas</b>	
	* Surface runoff watersheds
	* Ground water recharge areas
	* Ground water table and direction of flow
	* Monitoring wells
	* Ground penetrating radar transects
<b>Future Town Well Sites and Zones of Contribution</b>	
	* Punkhorn proposed well sites - north and south
	* Harwich and Dennis well sites
	* Zone 1 of contribution (400' radius)
	* Zone 2 of contribution (simulated 180 day pump test)
<b>Level II</b>	
<b>Abiotic Constraint</b>	
	* <b>Areas of Abiotic Constraint</b>
	- Hazard lands ( <i>e.g.</i> steep slopes, floodplains)
	- Areas sensitive to contamination and degradation
<b>BIOTIC MAPS</b>	
<b>Level I</b>	
<b>Vegetation Types</b> [See Table 3]	
<b>Significant Vascular Plants</b>	
	* Heritage (state listed) species
	* Locally and regionally significant species

## Some Vertebrate and Invertebrate Habitat

### Birds

- \* Emlen transects (June - July 1986)
- \* Breeding bird 'hot spots' 1986 census
- \* Fish eating/pondshore-wetland species
- \* Landscape boundary
- \* Migratory waterfowl

### Fish

- \* Herring migration corridors

### Mammals

- \* Fox dens (inactive)
- \* Fox dens (recently active)

### Dragonflies/Damselflies

- \* Heritage (state listed) species
- \* Significant habitat

### Amphibians/Reptiles

- \* Spotted salamander vernal ponds
- \* Reptile Habitat

## Level II

### Biotic Significance and Constraint

- \* **Unique Significance/Constraint**  
(Heritage species; unique and restricted community types)
- \* **High Significance/Constraint**  
(Regionally and locally significant plants; significant vertebrate and invertebrate habitat; wetlands - depending on ecological integrity, landscape context of patch, and degree of human intrusion)
- \* **Medium Significance/Constraint**  
(Upland forest and wetlands depending on ecological ...)
- \* **Low Significance/Constraint**  
(Patches with high levels of recent, current, or imminent human intrusion within &/or proximal to the patch)

## CULTURAL MAPS

### Level I

#### Archaeological Sensitivity and Significance

- \* known prehistoric sites
- \* native use pre-contact
- \* native use post-contact
- \* early European historical use (to 1694)

#### Landscape Pattern in 1938

- \* active cranberry bogs
- \* abandoned cranberry bogs
- \* anthropogenic disturbance patches
- \* settlement patches, nodes, and corridors
- \* natural features

#### Landscape Pattern in 1952

[see typology for 1938]

#### Landscape Pattern in 1971

[see typology for 1938]

#### Landscape Pattern in 1986

[see typology for 1938]

#### Recreational Potential and Use

- \* trails (existing and proposed)
- \* hiking and nature appreciation
- \* interpretative
- \* equestrian
- \* running
- \* skiing
- \* mountain/off-road bicycling
- \* picnic areas

- \* Town landings
- \* windsurfing, boating
- \* skating ponds
- \* scenic vistas
- \* swimming access areas
- \* parking areas
- \* cultural artifacts

## Level II

### Cultural Significance and Constraint

- \* **Areas of Cultural Significance**
  - known and potential archaeological sites
  - known historical sites
  - areas of local importance
- \* **Areas of Cultural Constraint**
  - potential archaeological sites
  - areas of land use conflicts (e.g. settlement patches; well zones of influence; traditional uses, such as hunting)

## Level IV

### Environmental Management of the Punkhorn - Land Tenure

- \* town-owned lands
- \* lands recommended for acquisition and easement
- \* acquired house/cottage structures

### Environmental Management of the Punkhorn - Monitoring

- \* monitoring sites
  - significant flora
  - significant fauna
  - 200' zone for fox dens around wetlands
  - vegetation succession
  - habitation patch intrusion
  - groundwater and surface waters
  - replicable photographic sites

Canopy gaps are prime factors in forest heterogeneity, allowing for intrusion and establishment of disturbed-site species, shade-intolerant species, species indicative of other vegetation types or seral stages and 'release' of subcanopy species already in place. In the Punkhorn, release of species such as Gray Birch (*Betula populifolia* Marsh.) allows them attain canopy height. Punkhorn bird species utilizing openings created by canopy gaps include cedar waxwings and northern orioles (Table 5). Canopy gaps, created by natural disturbance (e.g. mortality and windthrow) or human activity (e.g. cutting and roadway abandonment) in some instances have an abundant graminoid layer in early gap-phase regeneration in the Punkhorn. Bear oak (*Quercus ilicifolia* Wang.), a 'scrub' oak, is a common species generally present only in gaps and on disturbed forest edges in the Punkhorn.

The size, shape and orientation of openings influence microclimate. A thousand square meter gap may be needed to regenerate several intolerant species, whereas smaller gaps may be large enough for regeneration of some species typical of northern New England (Runkle, 1983). In the Punkhorn, large gaps created by anthropogenic disturbance, such as settlement patches or parking areas, tend to persist, impeding woodland succession.

Five wetland forest types were identified in the Punkhorn, dominated by red maple (*Acer rubrum* L.) and/or Tupelo (*Nyssa sylvatica* Marsh.). Almost all wetland forests in the Punkhorn have experienced intense human-induced disturbance. These communities differ radically in their disturbance history, hydrology, pattern, structure and landscape context (as in proximal community types, land uses, etc.).

Shrub wetland types of the Punkhorn were classified into three broad categories based on species composition, % cover, and community struc-

ture. These "types" should not be viewed as discontinuous or discrete samples; rather, they are samples on one or more successional gradients.

A significant wetland type, a quaking bog complex, is unusual on Cape Cod, given geological history, geomorphology, and soils of the area. The present vegetation structure and composition of Punkhorn wetlands depends on past conversion to cranberry bogs, time since abandonment, subsequent anthropogenic and natural disturbance history and both present and historical hydrology. Thus, the pattern of wetland types in the Punkhorn is an expression of human presence and ongoing impact on this landscape.

## RARE PLANTS AND THEIR HABITATS

In 1985 and 1986, 38 sites for 17 plant species listed as are by the Massachusetts Natural Heritage Program were identified in a moratorium study area (southwest Rural-Residential zone). Of these, 31 sites constituting new records (LeBlond, 1985; 1987)(Table 4). Five rare species with historical records in the study area are apparently extirpated.

Table 3. Vegetation Types of the Punkhorn

### Upland Types

#### \* Upland Forest

- OA (oak)
- OP (oak/pitch pine)
- PO (pitch pine/oak)
- PP (pitch pine)

#### \* Non-Forested Upland

- PC Power Corridor Vegetation
- ED Early Successional Disturbance
- RD Roadside Disturbance
- SP Settlement Patches
- FO Forest Openings

### Wetland Types

#### \* Wetland Forest

- RM-1 Red Maple 1
- RM-2 Red Maple 2
- RM-3 Red Maple 3
- TP-1 Tupelo/Red Maple 1
- TP-2 Tupelo/Red Maple 2

#### \* Scrub-Shrubland

- SS low shrubland
- MS medium shrubland
- TS tall shrubland
- QB-1 Quaking Bog 1
- QB-2 Quaking Bog 2

#### \* Wet Meadow

- WM-1 *Drosera* (Sundew) meadow
- WM-2 *Platanthera* (Orchis) meadow

#### \* Wetland Graminoid

- GE Graminoid Emergent
- WD Wetland Disturbance Patches

#### \* Wetland Habitat

- FE Floating Emergent
- CB Working Cranberry Bog
- CP Coastal Pondshore Presettlement

#### \* Deep Water Habitat

- OW Open Water
- LA Lacustrine (lake)

**Table 4.** Habitats of Punkhorn Study Area Rare Vascular Flora. Habitat types are as listed in Table 3 except for the sandplain grassland (SG)\* and water willow (WW).\*\* NWI Status refers to listing on the current Regional list of Wetland Plant Species (adapted from LeBlond, 1987; Reed, 1988)

	NWI Status	Habitats						
		Uplands			Wetlands			
		FO/RD	FO/SP	SG	CP	WM	TP	WW
<b>Species of Upland Habitats</b>								
<i>Asclepias tuberosa</i>			x					
<i>Dichanthelium ovale</i>	(FACU)	x						
<i>Helianthemum dumosum</i>				x				
<i>Lespedeza stuevei</i>		x	x					
<i>Lupinus perennis</i>			x					
<i>Quercus stellata</i>		x	x					
<i>Spiranthes tuberosa</i>			x	x				
<b>Species of Wetland Habitats</b>								
<i>Bidens discoidea</i>	(FACW)							x
<i>Ophioglossum vulgatum</i>	(FACW)					x	x	
<i>Drosera filiformis</i>	(OBL)				x			
<i>Linum striatum</i>	(FACW)				x			
<i>Polygonum puritanorum</i>					x			
<i>Sabatia kennedyana</i>	(OBL)				x			
<i>Sagittaria teres</i>					x			
<i>Stachys hyssopifolia</i>	(FACW+)				x			
<i>Utricularia resupinata</i>	(OBL)				x			
<i>Utricularia biflora</i>	(OBL)				x			

FACU - Facultative Upland - generally occurs in non-wetlands 67%-99%, occurs in wetlands an estimated 1%-33%; FACW - Facultative Wetland - typically occurs 67%-99% in wetlands, but occasionally in non-wetlands; OBL - Obligate Wetland - occurs almost always (estimated probability >99%), under natural conditions in wetlands. Species without an NWI Status do not appear on the current Northeast regional list.

\*Sandplain grasslands (SG) are one of the most significant and threatened habitats types in southeastern Massachusetts. However, only degraded remnant examples along a power corridor are proximal to the boundaries of the Punkhorn landscape, therefore this type is not listed in Table 3.

\*\*The water willow (*Decodon verticillatus*) dominated habitat is classified as a type of scrub-shrubland (MS) in Table 3.

Atlantic coastal plain pond shores were the single most significant habitat type for rare plants in the study area, including 17 sites for eight species. Elbow Pond, despite increasing settlement pressure and two adjacent active cranberry bogs, continues to be an exceptional example of a nationally significant Atlantic Coastal Plain pond type of southeastern Massachusetts (LeBlond, 1987). Such habitats are particularly vulnerable to degradation from development-related factors including recreational use, non-point source runoff from roads, parking areas and trails. Chemical inputs from cranberry bogs, including aerial spraying, also pose a threat to the biota of this significant habitat. Pond shore habitats of this type are restricted globally, occurring most frequently in southeastern Massachusetts.

Two globally restricted species identified in the Punkhorn area were Plymouth gentian (*Sabatia kennedyana* Fern.), and *Sagittaria teres* S.Wats. Bushy rockrose [*Helianthemum dumosum* (Bickn.) Fern.] is also globally restricted and reported from Cape Cod, but no sites for it were found within the Punkhorn landscape.

Two species listed by the Heritage Program, adder's tongue (*Ophioglossum vulgatum* L.) and a bladderwort (*Utricularia biflora* Lam.) are known from fewer than eight sites in Massachusetts, and Punkhorn records are the only ones known for Cape Cod. Heritage-listed species located in Punkhorn upland habitats of roadsides (RD),

forest openings (FO), and clearings (FO/SP) were *Dichanthelium ovale* var. *addisonii*, bush clover (*Lespedeza stuevei* Michx.), Wild Lupine (*Lupinus perennis* L.), and post oak (*Quercus stellata* Wang.).

Species of local and regional significance were also identified in the Punkhorn, including over 200 stems of white fringed orchis (*Platanthera blephariglottis*) occurring in a wet meadow with adder's tongue. Other uncommon species found in the Punkhorn included shining clubmoss (*Lycopodium lucidulum* Michx.), trailing bush-clover (*Lespedeza procumbens* Michx.), blazing star (*Liatris borealis* Nutt.), tickseed-sunflower (*Bidens coronata* var. *brachyodonta* Fern.), a bulrush (*Scirpus rubricosus* Fern.), grass pink [*Calopogon pulchellus* (Salisb.) R.Br.], and quite abundant pink lady's slipper (*Cypripedium acaule* Ait.).

## BIRDS

An 'intact' breeding bird assemblage was identified using seven Emlen, variable-width transects over two sampling periods of two days each in late June and early July, 1986 (Emlen, 1971, 1977; Brush, 1987a). The relative abundance of bird species recorded indicated that censusing probably adequately sampled the breeding birds of the Punkhorn. The observed density for all birds (150 individuals/40 hectares) is characteristic of dry, sandy, pine-oak woods in northeastern

North America. Breeding densities in the Pine Barrens in similar habitats have been found to vary from 80 to 210 individuals/40 hectares (Brush and Stiles, 1986; Brush, 1987b). The four most common species were black-capped chickadee, rufous-sided towhee, ovenbird, and pine warbler, representing 65% of the bird community by density. The next three most common species (gray catbird, blue jay and common yellowthroat), typical of wetland habitats, represented 13% of the bird community. The other 26 species observed during these two survey intervals, are characteristic of mixed habitats, such as wetlands and dry woods, dense forest and roadsides, and this group comprised 22% of the bird community. Table 5 summarizes the neotropical migratory strategy, forest interior habitat preference and habitat use of 18 species observed during this survey of the Punkhorn.

**Table 5. Neotropical migratory strategy, forest interior habitat preference, and observed habitat use (June-July, 1986) of selected Punkhorn birds.** Species are classified according to their degree of forest interior habitat preference, as determined by Whitcomb et al. (1981), except for Brown Creeper which was defined by Askins et al. (1987). Species with a question mark were not classified. Neotropical migratory strategy is as defined by Whitcomb et al. (1981) and was determined using Godfrey (1966). Numbers refer to individual birds. Observed habitat use and abundance was adapted from Brush (1987a).

Species	Upland Forest	Observed Habitat Use		
		OA/OP (nw)	Wetland	%Wet
Pine Warbler	(FI)	8	0	0
Ovenbird	(FI-NEO)	6	1	25
Eastern Wood-peewee	(IE-NEO)	4	0	0
Great Crested Flycatcher	(IE-NEO)	3	0	0
Hermit Thrush	(FI-NEO)	4	0	0
Scarlet Tanager	(FI-NEO)	2	0	0
Brown Creeper	(FI)	1	1	6
Northern Oriole	(ES-NEO)	1	0	50
White-breasted Nuthatch	(FI)	1	0	0
Hairy Woodpecker	(FI)	1	0	0
Black-and-White Warbler	(FI-NEO)	0	3	100
Northern Parula Warbler	(IE-NEO)	0	0	100
Nashville Warbler	(?-NEO)	0	0	100
Red-breasted Nuthatch	(?)	0	0	100
Yellow Warbler	(?-NEO)	0	0	100
Cedar Waxwing	(ES)	0	0	100
American Goldfinch	(FE)	0	0	100
Common Yellowthroat	(IE-NEO)	0	0	12

OA/OP refers to oak-dominated forest types near wetlands.  
 FI - "forest interior specialists nest only within the interior of the forest and tend to avoid edge habitats"  
 IE - "interior-edge generalists may have territories located entirely within the forest, but also can utilize forest edge such species often are able to integrate several isolated patches of habitat into a single territory"  
 ES - "edge species organize their territories primarily or exclusively at the border of forest..although such species may occupy territories in forest interior, major gaps, such as those generated by large treefalls, are required. Most territories of such species are organized along wood margins, hedgerows, or similar habitat"  
 FE - "field-edge species may nest at the margins of forest or even in the forest interior; but require fields or other open habitat for foraging"

Palustrine-forested and scrub-shrub wetland types with an upland oak dominated "buffer" forest, were strongly associated with the breeding presence of some significant neotropicals, such as northern parulas and Nashville warblers, and they also exhibited high densities of other breeding birds.

Some bird species may move between wetland and upland habitats as food resources change during the breeding season (Brush, 1985). Northern parulas are a Species of Special Concern, and Nashville warblers had not been confirmed as a breeder on Cape Cod for the past decade (Nikula, 1985). Species, such as pine warbler, hermit thrush, eastern wood-peewee, ovenbird, great crested flycatcher and scarlet tanager were found mainly in dry oak or pitch pine forest well-removed from wetlands.

The presence and relative abundance of birds representative of area-sensitive and forest-interior conditions, such as ovenbird, pine warbler, black and white warbler, eastern wood-peewee, scarlet tanager, black-billed and yellow-billed cuckoo, great crested flycatcher, northern parula warbler, red-eyed vireo and whip-poor-will, attested to the unfragmented quality of the Punkhorn landscape. These forest-interior and area-sensitive species are extremely sensitive to habitat fragmentation and the resulting intrusion of conditions characterizing forest edges (e.g. Whitcomb et al., 1976; Whitcomb, 1977; Whitcomb et al., 1981; Robbins, 1979; Robbins, 1980; Robbins, 1984; Robbins et al., 1986; Noss, 1987; Askins et al., 1987). These conditions preclude parasitism and competition from other birds, hunting, disturbance from cats and dogs, increased human access to breeding areas, creation of edge habitats and other disruptive influences.

The relative absence of species such as starlings, (that compete with native cavity nesters), grackles (nest robbers) and brown-headed cowbirds (brood parasites that appear to be significant factors in the decline of insectivorous birds such as warblers and vireos) attests to the 'integrity' of the Punkhorn landscape for breeding birds. Nest predators and parasites abound in areas characterized by comparatively high levels of human intrusion and disturbance (see Whitcomb et al., 1981).

The foregoing observations bolstered support for efforts to preserve extensive areas encompassing the relatively unfragmented upland and wetland habitat "mosaic" of the Punkhorn.

The recurring presence of 'summit predators' over several seasons, such as the bald eagle, a species listed as Endangered in Massachusetts, and osprey, a Species of Special Concern, suggested a relatively intact food chain structure. Furthermore, the presence of species such as great blue heron, Green Heron and Kingfisher, was directly related to the Stony Brook herring (alewife) run, one of the largest in Massachusetts. Seymour's Pond, bordering the eastern boundary of the Punkhorn, is also a herring pond, connected to the Herring River system.

Migratory waterfowl censuses were made of Punkhorn ponds over four years (1983-1986) (Nikula, Cape Cod Bird Club, pers. comm., 1986). Large numbers of scaup, buffleheads, mergansers and canvasbacks, among others, were found, particularly during the fall migration and over the winter.

**INVERTEBRATES**

A preliminary survey identified two state-listed invertebrates associated with pond shore (CP) and quaking bog (QB) wetland habitat types of the Punkhorn. The barren bluet damselfly (*Enallagma recurvatum*) had not been collected in New England in 33 years and was believed extirpated. This species is globally restricted, occurring only on Cape

Cod, Long Island, and in New Jersey. In 1986, this species was rediscovered at five ponds on Cape Cod, including two records for this elusive species on Elbow Pond in the Punkhorn (Carpenter, 1986). Elbow Pond is an exceptional Coastal Plain pond exhibiting a high diversity of Odonate species. The Barren Bluet Damselfly's status was subsequently altered from Extirpated to Endangered.

A second species, the Long Legged Green Darner (*Anax longipes*), a Species of Special Concern, was sampled at an unusual quaking bog complex in the Punkhorn (Carpenter, Cape Cod Museum of Natural History, pers. comm., 1987).

The habitat of these species is extremely sensitive to degradation from development-related activities and factors such as recreational use, habitat alteration and conversion, water drawdown, non-point source runoff, and other factors causing a decline in water quality.

## CULTURAL FACTORS

Archaeological evidence has confirmed at least the seasonal presence of humans in the Punkhorn for approximately the past 8,000 years, in certain areas that are known as fundamentally undisturbed archaeological sites. Given the rapid rate of development and concomitant loss of archaeological resources on Cape Cod, such protected resources will undoubtedly increase in importance. Areas of prehistoric and early historic archaeological sensitivity were delineated in this study, based on ecological and human settlement factors (Dunford, 1985; Dunford, 1987). The crucial site-selection variables identified by Dunford (1987) were: "...fairly level, well drained, sandy soil; proximity to fresh water—within 300 meters; proximity to ecotones or to highly productive ecosystems such as tidal rivers, estuaries, marshes, rivers or ponds. Cultural variables that might have played a structuring role in site location are proximity and access to communication and travel networks, and centrality of location for group interaction."

The early human inhabitants of the area were closely tied to marine and freshwater resources. Significant biological resources such as the herring runs, appear to have been important seasonally to these Middle Archaic (8,000-5,000 B.P.) hunters and gatherers. Use of the lands and waters continued through the late Archaic (5,000-2,000 years B.P.) and Woodland (2,000-500 B.P.) Periods, with settlements developing that were associated with highly productive, typically wetland and deep-water ecosystems. Late Woodland economies were characterized by intensive coastal adaptations and swidden (slash and burn) agriculture, as rivers and ponds continued to function as important areas for resource procurement (Dunford, 1985).

As Europeans began acquiring land in the Punkhorn about 300 years ago, native settlement and resource use became concentrated around the Punkhorn wetlands complex and the shoreline of Seymour's Pond. The Native American society was eventually disrupted, and the last Saquatucket tribe member died in 1813 (Dunford, 1985).

Early maps indicate that the Punkhorn remained hinterland to the active economic life of the white community. Maps dated 1709-30, 1831, and the U.S. Geological Survey (USGS) map of 1893, show a developing refinement in mapping of the Punkhorn region; however, the 1893 USGS map had little topographic detail and retained significant inaccuracies copied from earlier maps, including the placement of the Punkhorn wetland complex decidedly west of its true location. The prevalence and persistence of such mapping inaccuracies suggests the degree to which the Punkhorn region remained a little-known, remote area to all but the residents who utilized its abundant natural and aesthetic resources, primarily on a seasonal basis.

By 1831, maps were made showing the presence of principal roads

in the area (e.g. East and West Gate Roads) and possibly the extent of forest cover. The area between the East and West Gate was apparently used as a sheep common during the 19th century. Although the Punkhorn area was apparently never used extensively for settlement, several hunting camps were established along pondshores, some continuing to be used seasonally until the last few years.

With the advent of aerial photography in the 20th century, an empirical record of the geography of the Punkhorn landscape has become available. Landscape pattern maps were prepared for 1938, 1952, 1971 and 1985, describing changes in natural features, settlement patches and nodes, development corridors, cranberry bogs and anthropogenic disturbance patches typically associated with settlements and agriculture (Table 2). The sequential photographic record chronicles the Punkhorn's transformation into a remnant natural area over the past half century.

Anthropogenic disturbance patches were found to be clustered around sites of active cranberry bogs and settlement. Patch shape tended to be more regular and less complex than that of natural features, such as vegetation types. This finding agrees with observations of Krummel *et al.* (1987) that the fractal dimension (*cf.* Mandelbrot, 1977, 1983), a measure of complexity of patch perimeter, is lower for disturbed patches. That is to say disturbed patches tend to have more regular perimeters. Quantitative measures of such observations could be made, including calculation of the fractal dimension (*cf.* Iverson, 1988), perimeter to area ratio, an Index of Geometric Intactness [*i.e.* a unitless measure of the degree to which a patch or nature reserve perimeter deviates from a circular or "idealized" form, independently proposed by Hellyer (1985) and Duffey (1974)], or a related measure of limnological shoreline development, applied to measure the shape of urban parks (Faeth and Kane, 1978).

The Stony Brook herring run remained an important fishery event for white settlers, and it is still a significant social event and tourist attraction in Brewster. As settlement density increased on Cape Cod, the intensity of use of other natural resources, such as the forests for boat building, salt works, try-works for whale oil, turpentine production, firewood for glass factories and timber for building construction, also increased. Harvesting of wood occurred in the Punkhorn, while industrial and commercial applications occurred elsewhere. Thus, the original forests, as described by early settlers, were significantly altered.

The "Provincelands," as characterized in Mourt's Relations (1623; cited by Schall, 1982) was "all wooded with oaks, pines, sassafras, juniper, birch, holly, vines, some ash, walnut; the wood for the most part open and without underwood, fit either to go or ride in." The relationship between European settlers and the land has substantially shaped the nature of the Punkhorn landscape. Palustrine wetlands were converted to cranberry bogs and upland forest was extensively harvested. With the advent of more profitable, less demanding sources of employment on Cape Cod in recent decades reversion and ecological succession of most harvested areas has occurred.

## ACQUISITION AND MANAGEMENT

The disturbance factors that maintained open upland habitats on Cape Cod prior to European settlement included fire (natural and human-initiated), agriculture, forestry, windthrow and wildlife grazing (LeBlond, 1987). Management will be needed to simulate the effects of historic disturbance regimes and to limit, for example, the intrusion of woody succession into readily accessible, open upland habitats.

Upland and wetland areas of the Punkhorn have numerous walking, running, mountain biking, horseback and cross-country ski trails.

Water-based recreation, including fishing, boating and windsurfing, are attractive features for the local population as well as visiting city-dwellers. Several seasonal houses have been acquired that may be suitable for interpretive or research centers. It is clear that conflicts exist between recreational users and significant habitats and species of the Punkhorn. The future role of hunting has proven particularly problematic pitting 'conservationists' against 'sportsmen,' evidenced by the 1988 Brewster land acquisition program having been held ransom by elements of both groups.

Two proposed municipal well-fields in the Brewster Punkhorn as well as well-fields in the bordering towns of Dennis and Harwich are perceived as increasingly significant resources for local Cape towns, though their use may have adverse impacts on wetland habitats and the biota of the Punkhorn. Construction of pipelines along Punkhorn roads may impact some of the significant roadside flora as well. Although summer drawdowns may provide additional habitat for lower pond-margin flora, sustained drawdowns would not mimic the historical pattern of inter-year variability in pond levels, possibly threatening sensitive species and habitats. Recent USGS groundwater mapping, using ground-penetrating radar, may assist in monitoring water levels (Johnson and Davis, 1988).

Management decisions for the Punkhorn landscape must cross artificial (non-biological) political and administrative boundaries, involving adjacent towns with diverse, sometimes competing, groups and interests within Brewster, if protective means are to be effective over the long term (Margules and Usher, 1981). For example, recreational uses can threaten rare plant populations through overuse, casual or intentional sampling, vandalism or other means. Portions of the only Cape Cod population of *Bidens coronata* var. *brachyodonta* were removed by a four-wheel drive vehicle that became stuck on a Punkhorn access road that also leads to a *Drosera filiformis* wet meadow. Despite posting and erection of barriers, this road and sensitive habitats like it continue to be used by ATVs.

The Town has established a Punkhorn Advisory Committee composed of members of the Conservation Commission, Water Department and Board of Selectmen, all of whom have management authority over discrete portions of the Punkhorn landscape. Such parceling of management authority is unfortunate and, tends to impede development and implementation of effective protective strategies. Nevertheless, townspeople have explicitly rejected a recent proposal to unify management of most of the Punkhorn under the Conservation Commission.

Designation of Sensitive Faunal and Floral Habitat Zones and limiting some types of seasonal access to particular areas will help protect sensitive elements and natural diversity in the Punkhorn (Table 2). Establishment of floral and faunal monitoring programs and replicable photographic sites will allow assessment of the effectiveness of management and protection efforts; however, once a natural area is perceived as "protected" there might be less willingness to expend funds for future research, monitoring, or management. Given the past involvement of Cape Cod researchers, the Cape Cod Museum of Natural History and prominent townspeople, this will hopefully not be the case.

For purposes of acquisition, current land values were determined, utilizing an appraisal methodology developed in response to conditions characteristic of natural areas such as the Punkhorn (Palmer, 1989). Although the political process of acquisition became more dependent on social and cultural factors than on biological ones, an almost 400 hectare (1,000 ac.) tract was acquired and assembled, to become known as the Punkhorn Parklands. Land was primarily acquired fee-

simple for an average value of \$10,000/acre and through the "owner-unknown" process. The owner-unknown process is unusual, if not unique, to the cluttered land title records of Cape Cod, in which some isolated parcels have no demonstrated owner and thus may be acquired by Towns for processing costs.

Considerable effort has been expended to draft site-specific conservation easements for some parcels, but negotiations were uniformly unsuccessful and these parcels were ultimately taken by eminent domain. Some key, unacquired Punkhorn parcels, including those bordering Elbow Pond have been recommended for easement.

A pre-approved subdivision segment between the "northern" and "southern" Punkhorn was not acquired, due to the wildly inflated value of subdivided land on Cape Cod. A negotiated agreement maintained two recreational travel corridors between these portions of the Punkhorn. The fragmentation resulting from this subdivision, while still considerable, was somewhat lessened by the unique biological and physical characteristics of the 'northern' and 'southern' Punkhorn areas.

Settlement patches within and proximal to the Punkhorn will, among other negative effects, increase anthropogenic disturbance, providing sources of weedy propagules and allowing access by dogs and cats who will undoubtedly impact bird populations, such as those of ground nesting neotropicals. Use of a portion of the seemingly vast Punkhorn landscape for affordable housing has also been proposed, partially to make the acquisition and preservation process more politically palatable to certain townspeople. Although such patches fragment natural areas, involvement of these individuals in the management process may mitigate some of the effects.

Environmental management of the Punkhorn Parklands and adjacent lands, in a manner sensitive to abiotic, biotic, and cultural significance and constraint, will be essential to long-term viability of this natural area and its significant habitats and rare species (see Hellyer, 1985). As Finch (1983) in his essay *Punkhorn* observed, "The landscape ...seen in this way becomes after a while a curious mixture of past and present, of man-made and earth-made. In a way, the archaeologist and the ecologist have similar points of view, for both see all of the earth's various features attached to one another at some point, either in space or time."

**Note:** This paper was written by Greg Hellyer in his private capacity. No official support or endorsement by the Environmental Protection Agency or any other agency of the Federal Government is intended or should be inferred.

## ACKNOWLEDGMENTS

The Town of Brewster Land Acquisition Committee supported this research. Bob Finch, Dave Palmer and Paul Wightman deserve special mention. My mentor, the late Prof. Bob Dorney, was instrumental in my natural areas work and his influence continues to guide my thinking about landscapes. This study is dedicated to his memory. Herb Heidt of Mapworks went far beyond the call of duty in his cartographic contribution, producing the maps for this report and paper. P.G.R. Smith, J. White and an anonymous reviewer commented on earlier versions of this paper. My chief at EPA, Doug Thompson, assisted with my attendance at this conference and in my preparation of this paper.

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# Northern Wild Monkshood in New York: Practical Problems in Resource Management

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**Abstract:** Northern Wild Monkshood (*Aconitum noveboracense*), a plant species federally listed as Threatened, is discussed with regard to its occurrence in New York State and its significance to management priorities. A land manager's role requires special consideration of human impacts on such rare plant species on a mosaic of public and private lands. Present activities protecting and supporting the occurrence of northern monkshood in New York are discussed, and some research and educational needs are recommended from the land manager's point of view.

Pages 288 - 290. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

"The more we looked, the more monkshood we found." These encouraging words lead a 1986 report (Birmingham, 1986). Since then, even more monkshood plants have been found, and now some 4000 individuals have been counted in New York State (Birmingham, 1987). The number is impressive, especially considering that, prior to 1978, only a few hundred plants were known to exist.

Northern Wild Monkshood, *Aconitum noveboracense* Gray, is both federally listed and state listed as Threatened in New York State (Mitchell and Sheviak, 1981). It is known to occur only in New York, Wisconsin, Iowa and Ohio. Few other federally listed plant species span such a wide geographic range, yet these plants are quite rare except in Iowa.

Northern wild monkshood is a beautiful plant, often with several stems arising from a group of tubers. The height is variable. New York plants average under a meter in height, but an individual may occasionally reach 2 m (Birmingham, 1986). Leaves are smooth, with deeply cut, toothed margins. Stems and leaves are glabrous. The most striking feature of the plant is the blue to deep violet flower (rarely white) with a distinctive helmet resembling a monk's hood concealing the upper two sepals, hence the name. Seeds are borne in a follicle, reminding one observer of a cat's claw (Birmingham, 1986). Flowers and fruits may be borne on the plant at the same time, with flowering occurring from mid-June to September. Many flowers hang out over streams, and the tiny seeds often fall in the water, a possible dispersal mechanism (Birmingham, 1987).

Monkshood in New York is a higher elevation plant, most plentiful above 2,500 ft (Birmingham, 1987). It prefers soil kept moist and cold by subterranean ground water percolation (Mitchell, 1982; Read and Hale, 1981). Mountain streambanks, the edges of springs, cold river banks and upwelling cool ground water locales are among the very few locations where it can be found. Many aspects of the plant's biology are still being investigated. Among them, propagation is especially perplexing. Though the plants produce copious seeds, J. K. Dean's germination tests in 1980 failed (Mitchell, 1982). Others since then, given the right stratification conditions, obtained relatively high germination rates. Seedling recruitment observed in the field is often small (Cook and Dixon, 1986; Birmingham, 1987). The most successful propagation was reported by Read and Hale (1981) through lateral tuber division.

Through ignorance, carelessness, or deliberate misdemeanor, human activity can impact the plant. Monkshood inhabits environments attractive to human activities, so land managers should develop and implement policies to decrease the number of impacts and enhance survival

whenever possible. Professional land managers face several questions crucial to the protection of monkshood:

- 1) What are the major sources of negative impacts?
- 2) How do you deal with a threatened species on publicly owned lands?
- 3) What can be done to protect monkshoods on private lands?

The New York State Department of Environmental Conservation (DEC) is responsible for a number of habitats favored by the northern monkshood. State lands, part of the Forest Preserve in the Catskill Park, provide protection to all plants. Article XIV of the New York State Constitution insures special protection to all the natural features of the Forest Preserve: "The lands of the State, now owned or hereafter acquired, constituting the forest preserve as now fixed by law, shall be forever kept as wild forest lands. They shall not be leased, sold, or exchanged, or be taken by any corporation, public or private, nor shall the timber thereon be sold, removed, or destroyed." (Shaffer, 1984, 1985).

Our current and major protection efforts involve education and persuasion of visitors to the Forest Preserve to respect the plants in their natural surroundings, as well as the enforcement of land-use regulations on State lands. However, we are actively seeking to address the following management problems, on both our State lands and nearby private properties.

### Deer Herd:

The whitetailed deer population in the Catskills is above historic levels (Birmingham, 1987). Deer find monkshood an attractive food (Birmingham, 1986), and the plants appear heavily grazed when accessible to deer. All aerial parts of the plants are eaten (Birmingham, 1986, 1987). Observers report 86% to 90% of the monkshood plants in a given locale were browsed by mid-September (Mitchell, 1982; Cook and Dixon, 1986). Lower elevation plants are also more heavily browsed than higher elevation plants (Cook and Dixon, 1986).

Here, as with some other species, we are unsure of the total effect of browsing of monkshood populations. Deer also browse competing plant species, and thus may improve conditions for monkshood. Some stream bank populations are infrequently visited or disturbed by deer, though we have observed browsed monkshood in similar habitats nearby. We believe, in general, that lower deer populations would benefit monkshood.

DEC would be willing to permit researchers to install fencing or exclosures of known monkshood locations to examine the effects of eliminating browse, but such experiments may prove less beneficial than we would hope if other species proliferate in the protected area,

and compete successfully with monkshood. By fencing an area, we would heighten the interest of the curious hiker or hunter, as well, who might want to know what was so special as to be fenced in. Enclosure experiments may be more practical and less likely to be disturbed on private lands.

### **Recreational Impacts:**

Much of the known range of monkshood in New York State is visited by anglers, hunters, hikers, campers and other recreationalists. Much of this recreation is water based, with fishing being a major attraction in the Catskills, along world-famous trout streams. Several private fishing clubs own and maintain many miles of trout stream for their member's enjoyment, and streamside trails are heavily used by the anglers moving from one spot to another. Monkshoods are found directly adjacent to the paths in many cases, occasionally within inches of the treadway itself. The fishing clubs designate parking areas, and also limit the number of permits out on any stretch of the stream at any one time. Having the streams on private hands, rather than open to the general fishing public is undoubtedly a protective policy beneficial to the monkshood.

The clubs install various stream improvement devices, like rock dams, deflectors and rip-rap. These have the potential to affect monkshood sites, but DEC can request protection via the mechanism of our stream disturbance process. The Endangered Species act does not provide any specific protection to plants on private property. However, DEC extends protection to threatened plants via the authority that it exercises in denying or issuing permits. Identification of locations of protected plants and implementing measures to insure that they are not impacted are requirements of DEC permit applications and issuance.

The Forest Preserve is a favorite destination for hikers and campers, many of whom are drawn to watercourses for the tranquil setting, murmuring brooks, level campsites and the water source itself. The DEC currently prohibits, via regulation, camping "within one hundred fifty feet of any road, trail, spring, stream, pond or other body of water except at camping areas designated by the department" (6 NYCRR 190.3). Enforcement of this regulation is being stepped up all over the Forest Preserve, especially in areas known to harbor monkshoods. In one instance, a large plant at a station known since 1886, was trampled by careless campers (Mitchell, 1982), and other illegal campsites have also been reported to damage monkshood (Birmingham, 1986).

The DEC is in the process of preparing Unit Management Plans (UMPs) for Forest Preserve areas. The Draft Recovery Plan for Northern Monkshood by Read and Hale (1981), and the Recovery Plan and Status of Northern Monkshood in New York State, by Mitchell (1982) are both consulted in preparing UMPs.

The UMPs for the portion of the preserve harboring monkshoods will give special attention to the plant, its rarity, and need for protection. Known illegal campsites and fire rings will be removed as quickly as possible in these areas. Camping along the major stream will be strictly regulated, and only two sites, each with no known monkshood, will be designated for camping along its banks (Balsam Lake Mountain Draft Unit Management Plan, 1988).

The largest populations of monkshoods are found on one mountain and its drainages. This largely privately owned mountain is, by design, kept as a trailless peak to give hikers a challenge and an opportunity to find their own way to the top. As a result, monkshood plants may be trampled. In this special case, DEC has contacted landowners and hiking groups to protect the plant, but simple education of hikers may not suffice. The Catskill 3500 Club included a notice in their publication, requesting that hikers avoid water courses in hiking to the top of this

peak (*The Catskill Canister*, 1988). An ultimate option would be the outright purchase of the mountain with funds available from the 1986 Environmental Quality Bond Act.

Overall, the prime habitats of the monkshood are in the higher and more remote areas away from the most popular and heavily used fishing and hiking areas. Keeping "dispersed recreationists" (that is, people who do not stay on the hiking trails) away from the monkshood, wherever it is found, will continue to be a major objective in management efforts on both State and private lands.

### **Road and Trail Maintenance:**

County roads, town roads, private roads, and private trails all adjoin monkshood habitats. Sparsity of monkshood has been noted along lower elevation stretches of roadside streams, where herbicides are used. This may be partly an elevation preference, with monkshoods more poorly adapted to lowland situations, since some monkshoods are found along higher elevation roads where herbicides are also used. Even away from roads, there are often long gaps between individuals or colonies of monkshood. Why these gaps occur is not well known, but the phenomenon is under study by ecologists and population biologists interested in the rarity of the plant species (Birmingham, 1986). The county and State highway departments may be contacted if necessary, and their cooperation sought in avoiding impacts on locations of the plant resulting from vegetation control practices. At various places along the roads, rocks, dirt and gravel are dumped over the banks to fill washouts, and sometimes the loose fill lands within a few feet of monkshood plants; however, little impact is likely in most instances. Working with highway departments will no doubt be a most challenging task. Major projects for bridges or road widening require compliance with New York State's Environmental Quality Review program, which will give the DEC ample opportunity to include protective or mitigation measures for monkshood sites, especially if State or federal funds are involved.

DEC will direct special attention to State-owned trails and stream crossings in the Forest Preserve. We can route trails away from monkshood plants and make sure bridges and fords are at locations which avoid the plant. We will do more of this as we become aware of more monkshood sites.

### **Resource Utilization:**

Virtually 100% of the known range of northern monkshood in New York has been subject to natural or human disturbance over the past century. Both nature and man have impacted, and still affect, the plant and its environment, yet monkshood continues to survive human activities (Birmingham, 1986).

Monkshood should receive minimum future disturbance. New York State, through various programs, including Best Management Practices for Silviculture, Cooperative Consultant Forester and Cooperative Timber Harvester Programs, recommends care in choosing crossings and leaving uncut buffer zones along streams.

Foresters lay out access roads and designate stream crossings on logging or forest management sites, and they are becoming more and more sensitive to recognizing and avoiding locations of rare plants. Forest Rangers and Conservation Officers investigate potential violations, so law enforcement officers include information about plant protection practices when speaking to individuals and groups nowadays. Conservation professionals, knowledgeable concerning permit requirements, also instruct permit applicants in implementing procedures to protect threatened and endangered plants.

Collecting plants for research, another form of resource utilization,

is regulated in New York. No plants may be removed from State land except under permit. New York also has a Protected Plant Law which provides fines (fairly nominal ones, unfortunately) for anyone who would "knowingly pick, pluck, sever, remove, damage...or carry away a plant listed as protected without the consent of the landowner." (McKinney's, 1988). As of June 1989 the list will include monkshood, along with a number of other listed State rarities. With consent, any plant, including the monkshood, can be removed or otherwise disturbed on private lands.

#### Research Needs:

As ongoing research continues, we expect to have a thorough inventory of all actual and potential typical habitats for *Aconitum* (Birmingham, 1986). Propagation of the plants remains a challenge, but we stand ready to assist in experimental or practical outplantings in the wild, as well as seed dispersal and deer exclosure tests.

The DEC is supporting research in several basic areas of the plant's habitat, genetic diversity and population biology. Through a multi-year contract with Cornell Plantations of Cornell University, we hope to learn techniques to conserve the plant through changes in State land use. Funding for this research is provided by the U. S. Fish and Wildlife Service under the Endangered Species Act of 1973. In addition, we seek recommendations on educational programs and advice on potential agreements with other State agencies. We will be especially interested in efforts to reestablish and transplant monkshood. Finally, we will consider additional rule-making or regulation to protect the plant. We are determined to implement the best management practices for the monkshood, and we will review all findings as soon as they become available.

#### Public Education:

In public dealings concerning the monkshood, land managers are deliberately vague about exact locations of the plant. This adds somewhat to the difficulty in securing public support for what might seem an invisible entity. Professional managers understand the need to obfuscate the extent, numbers, locations and specific habitat of the plants and we hope the public will appreciate this as an attempt to protect the species. We simply can't police the populations all the time.

The northern wild monkshood, a rare plant made all the more diffi-

cult to protect by it's delicate beauty and unusual habitat preferences, remains a challenge for land managers. More research is necessary, and we are confident that it will provide direction for enhanced protection and propagation. The monkshood is an example of a serious test of resource management and wise stewardship of our natural areas, and the New York State Department of Environmental Conservation hopes to meet that challenge with excellent results.

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## **Chapter 10.**

### **AGENCY AND PUBLIC INVOLVEMENT**



# Researchers, Agency Attitudes and Wilderness

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**Abstract:** While mention is frequently made of the need for communications between researchers and land managers, and for incorporation of research results in management activities, relationships of researchers with the lands they use and agencies with which they interact have been largely unexplored. As a follow-up to research that found significant differences between research use of U.S. Forest Service (USFS) and National Park Service (NPS) Wildernesses, a questionnaire was distributed and completed by researchers who have used such lands. Researchers perceived the NPS as being more encouraging to research, while they perceived the USFS as being more indifferent. Agency funding, interest, and cooperation are relatively important to researchers working in Wildernesses. Travel time to study areas and area size ranked as relatively unimportant. Personnel involved in managing 13 USFS and NPS Wildernesses were also questioned. They hold a variety of views concerning the importance of, and problems associated with, research in their areas.

Pages 293 - 297. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

One of the reasons frequently cited for the protection of natural areas is their use for scientific or research purposes (*e.g.*, Myers, 1983; Stankey, 1982; Brower, 1960; Hendee *et al.*, 1978). While these areas can indeed be of great value to the advancement of science, the place of research activities in natural areas remains a poorly understood subject.

Generalizations about research in various types of natural areas were made in an early report about research use of natural areas (AAAS, 1963). Since then, the 1980s have seen researchers giving more attention to scientific-research use of reserved areas, particularly National Parks and Biosphere Reserves (*e.g.*, Wright and Hayward, 1985; McCrone *et al.*, 1982a, 1982b; Johnson and Bratton, 1978; Mack *et al.*, 1983; Turner and Gregg, 1983). The studies by Mack *et al.* (1983) and Turner and Gregg (1983) were based on surveys of reserve personnel; they rated reserves on research use and availability of baseline data. According to Mack *et al.*, who investigated NPS areas only, the NPS has been emphasizing short-term data acquisition for planning and management purposes. Turner and Gregg studied USFS, NPS, and other reserves, but considered 'observational' (*e.g.*, NPS areas) and 'experimental' (*e.g.*, USFS) reserves as two different categories, because "the main interest and purpose of the administrative agency has a strong influence on the use of Biosphere Reserves." One recent study (Butler, 1986; Butler and Roberts, 1986) confirmed differences in research levels and topic emphases between administering agencies in Wilderness-containing National Park areas and Forest Service Wildernesses, with the NPS areas receiving higher levels of research use.

## PURPOSE AND METHODS

Both agency personnel and researchers hold perceptions that may affect the use of an individual natural area for research or scientific purposes. Perceptions of managers and recreational users have been of some concern to researchers interested in the management of wilderness (Stankey and Lucas, 1984; Lucas, 1985; Lucas, 1964; Peterson, 1974; Hendee and Harris, 1970), but the perceptions of research users have not been given attention. To find out more about researcher perceptions and attitudes about wilderness research, a sample of primary authors having reported research based in Forest Service Wildernesses or National Park areas was selected for a survey conducted in 1985-86. The 58 respondents (66% return) came from a variety of backgrounds,

including forestry, archeology, urban planning, animal ecology, recreation, geography, geology, and others. They were employed mostly by universities and government agencies.

The second part of this research assessed perceptions of agency personnel. Their attitudes affect research use of an area through their direct or indirect encouragement/discouragement of researchers. Questionnaires similar to those sent to researchers were sent to a very small sample of Park Service Superintendents and Forest Service Supervisors, stratified according to identification as low/no, medium, or high research areas (1970-80), to gain some indication of their views concerning how research fits into wilderness. The units sampled were chosen from a larger sample of 75 areas investigated as to research levels (Butler, 1986). Responses to the questionnaire were received from six NPS areas and seven USFS areas (100% response rate).

## RESULTS

The results are presented in two forms: quantitatively (in tables) and qualitatively, with summarizations of major points and illustrative short quotes from the respondents. While the tabular data are informative, the verbal expressions of respondent perceptions add depth to understanding of researcher/agency relations not possible with quantitative data alone.

### Researcher Perceptions:

Researchers were asked if they thought the USFS and NPS tend to encourage or discourage research in Wildernesses and National Parks (Table 1). The largest number of responses were that research is encouraged to strongly encouraged by the USFS and the NPS. Of those respondents expressing an opinion, 60% saw the Forest Service as mildly to strongly encouraging research in Wilderness areas. This proportion is 77% for National Park areas (in general) and 78% for National Park Wildernesses. For both agencies, about 12% of the respondents thought research is discouraged. The Forest Service was seen as being neither discouraging nor encouraging more often than the Park Service. Overall, the Park Service was seen as somewhat more encouraging than the Forest Service.

Funding was often mentioned as a means of encouragement, as were other agency contributions, such as personnel time, logistical support, and use of facilities and equipment. Verbal encouragement and agency solicitation of research, permission and permits, and personnel interest

were also mentioned as means of encouragement. However, some researchers cited some of the same factors as discouraging. Funding and access are major concerns, seen as limitations to research in both NPS and USFS areas:

- “Reduced budgets”
- “Hassle on permits”
- “Not given access... ”
- “Difficulty in obtaining permits in certain areas in National Parks”
- “NPS permits must be obtained, an informal verbal ‘screening’ process seems to be in effect”
- “Outside agency researchers are actively excluded from working in National Park[s]”
- “National Forest administrators often are less than fully cooperative, and sometimes put up special requirements that can be discouraging to research people. Funding for National Forest Wilderness research from the USFS Branch of Research. . . is often difficult to obtain by agency personnel and others.”
- “Discouragement by FS—do it elsewhere if at all possible, extremely restrictive application of regulations, bureaucratic stalls, no assistance of any kind.”

**Table 1.** Researcher Perceptions of Agency Attitudes Toward Research<sup>1</sup>

	USFS Wilderness	NPS Areas	NPS Wilderness
Strongly encouraged	6	16	12
Mildly encouraged	14	21	17
Neither encouraged nor discouraged	9	5	3
Mildly discouraged	3	5	3
Strongly discouraged	1	1	2
No opinion	21	4	15
Total response	54	52	52

<sup>1</sup>Answers to the following questions:

“Do you think that research in *USFS Wilderness* areas is encouraged, discouraged, or neither encouraged nor discouraged by *Forest Service* personnel?”

“Do you think that research in *National Parks* areas is encouraged, discouraged, or neither encouraged nor discouraged by *Park Service* personnel?”

“Do you think that research in *National Park Wilderness* areas is encouraged, discouraged, or neither encouraged nor discouraged by *Park Service* personnel?”

Choices given for each question were as shown in the table.

By no means were all the comments made by researchers negative, however; many were quite positive:

- “My requests for logistical support were never denied. Never prohibited from doing what was necessary to gather data” [regarding NPS]
- “funding [and] excellent cooperation (logistics, facilities, etc.)”
- “Professionals in the U.S.F.S. helpful in (virtually) all aspects of our work and in almost every case, once they understood our purpose, they have bent over backwards to make the work pleasant and safer.”

A number of researchers expressed both points of view, listing ways in which research is encouraged, and ways in which it is discouraged. Researchers displayed a variety of views as to attitudes of personnel. Many cited interest on the part of personnel as a reason for encouragement:

- “many of the personnel believe in the value of knowledge”
- “personal interest and enthusiasm by NPS and FS personnel”
- “apparently genuine interest by USFS Forest research senior scientists and NPS senior staff”

On the other side, there were also criticisms of personnel attitudes:

- “researchers are sometimes considered a ‘nuisance’”
- “some administrators in the USFS have been less than enthusiastic about Wilderness in general, and some research administrators are wary of Wilderness research because it often becomes embroiled in controversy.”
- “local officials view research as a pain in the neck, irrelevant, and potentially upsetting to the current order of things.”

The researcher who contributed the last comment above was also critical of some researchers, characterizing them as “arrogant and demanding,” however. Researchers who expressed the most extreme views of the agencies may have come into contact with units at the extremes. A view that was expressed several times, perhaps by researchers with wider experience, was that agency attitudes toward research vary from area to area, with the attitudes of personnel. This was clearly stated by one researcher, who said: “The attitudes of Park and USFS personnel varies from unit to unit. Attitudes are certainly colored by mandate of Congress, purpose of research, and personal feelings of such personnel.”

When asked why they thought research was encouraged or discouraged, two common responses were the interest (or lack of it) of agency personnel and the need for information for management purposes. Researchers cited the usefulness of information to management, but cynicism appeared in a few comments; research, according to one respondent, “gives [the] Superintendent, Naturalist, or manager something to put in his or her reports.” Researcher perceptions appear to be in agreement with the Mack *et al.* (1983) observation that NPS emphasis has recently been on research for planning and management purposes.

Researchers were asked to rank several conditions in terms of their importance to research in Wildernesses and National Parks (Table 2). Resources of an area are of primary importance to the researchers; they go first where they can find the subjects in which they are interested. Of secondary importance is agency interest in the research, very closely followed by agency cooperation and funding. Area size and travel time were ranked as least important overall.

A number of respondents had additional comments on Wilderness and National Parks research. One theme found in researchers’ comments was that of Wilderness and Park value to science. Base-line data and wildernesses as “benchmarks” for comparison to altered environments were mentioned. Wildernesses, as substantially unmodified areas where natural processes can be observed, are important to researchers, and there is a feeling among some that the potential for research use of wilderness areas is not being met:

- “I believe that National Parks represent a relatively untapped resource in terms of areas where biological (and other) research can be conducted.”

Research is seen as a primary purpose of National Parks and Wilderness areas among a number of researchers:

- “I figure that the research effort in these areas is one of the top priorities and justifications for their existence.”
- “Certain representative ecosystem areas within the NWPS

should be designated for research, and research should be the #1 priority use.”

—“Interestingly, wilderness is only wilderness when humanity is essentially excluded. ...it would seem foolish for humans (in significant numbers) to hope to utilize wilderness for anything except research and as resource reservoirs. ...”

Other comments touched on funding concerns, agency/university cooperation, and restrictions on research activities and equipment.

**Table 2.** Researcher Rankings of the Importance of Area Characteristics to their Wilderness and National Park Research<sup>1</sup>

	Very Important	Somewhat Important	Slightly Important	Not at all Important
	N (Row %)	N (Row %)	N (Row %)	N (Row %)
Area size	11 (20.0)	11 (20.0)	12 (21.8)	7 (12.7)
Travel time to area	5 (9.3)	15 (27.8)	12 (22.2)	10 (18.5)
Resources of area (study subjects)	47 (83.9)	7 (12.5)	0 (0.0)	1 (1.8)
Agency interest in project	19 (36.5)	23 (44.2)	6 (11.5)	2 (3.8)
Agency cooperation	17 (32.1)	19 (35.8)	11 (20.8)	5 (9.4)
Agency funding of research	26 (47.3)	11 (20.0)	8 (14.5)	5 (9.1)

<sup>1</sup>The questionnaire item gave respondents the choices shown, and was worded as follows:

“Indicate how important each of the following items have been to your Wilderness and National Park research.”

Spaces were also provided for respondents to add and rate other items affecting their research.

### Agency Personnel Perceptions:

Agency personnel were asked whether they would consider the total number of research projects conducted in their specific Wilderness, as compared to other Wildernesses, to be above average, average, or below average. A majority seemed to regard their Wildernesses as comparatively low research use areas; seven areas were seen as below average. Although four areas have had relatively high research use (Table 3), only two of the respondents saw their Wilderness as above average.

The respondent for Everglades NP stated that research there is “[f]ar above average, due to national recognition of importance and sensitivity of Everglades resources.” Though not mentioned by the respondent, the location of a “park-based research center with excellent laboratory, computer, and library support” at Everglades (Agee *et al.*, 1983) is

likely to have had an effect on the amount of research in the area as well. A respondent for the BWCA listed three reasons for high research use there: it is “the most used wilderness in the system,” has easier access—by canoe—than other Wildernesses, and “is well known to college professors and grad students.”

Those who thought their areas received below average use gave a number of reasons for the low use. These included: “logistics,” funding, requirements governing research activities, low recreation use, “extremely rugged and steep terrain,” remoteness, “lack of attractions (especially lakes),” fear of grizzly bears, small size, and a “sere environment.” One USFS official said that other Wildernesses are closer to population centers and universities and are more “famous” than his area.

**Table 3.** USFS and NPS areas surveyed.

Research use Area	Identified	Agency
Boundary Waters Canoe Area (BWCA)	High	USFS
Everglades	High	NPS
Guadalupe Mountains	High	NPS
Isle Royale	High	NPS
Ansel Adams (previously Minarets)	Low/Medium	USFS
Haleakala	Low/Medium	NPS
Lassen Volcanic	Low/Medium	NPS
North Absaroka	Low/Medium	USFS
Black Canyon of the Gunnison	None	NPS
Gates of the Mountains	None	USFS
Gearhart Mountain	None	USFS
Mount Zirkel	None	USFS

Agency personnel were also asked to what extent their areas encourage research. None of the area personnel felt that research is entirely discouraged in their areas, but two did give indications that research can be encouraged or discouraged, depending on the situation:

—“[r]esearch is encouraged if it relates to management or understanding of park resources. In practice, most studies meet these criteria... Research will not be allowed, however, if it is judged to be detrimental to park resources.” (NPS Research Biologist)

—Research is “[e]ncouraged if pertinent to wilderness management. Discouraged if it could be done equally well outside the wilderness.” (USFS Specialist)

Most USFS personnel replied that research is neither encouraged nor discouraged in their areas (Table 4). Park Service personnel were more likely to say that research is encouraged in their areas. It appears that managers in areas with low past research use tend not to have strong attitudes toward research; three of the four respondents from areas with no identified research use stated that research is ‘neither encouraged nor discouraged.’ Personnel in areas with higher past levels of research use tend to see research as encouraged. Whether research has followed encouragement, or encouragement has followed research, is not known.

Those personnel who said that research is encouraged in their areas all cited the usefulness of research to management. Three of the four mentioned academic institutions in their descriptions of how research is encouraged. Agency personnel may, for example, seek researchers from universities and pursue cooperative agreements. As mentioned by

personnel, research may also be encouraged through direct funding, assistance in support functions and orientation, and provision of work space and accommodations—means of encouragement that were also often cited by researchers.

**Table 4.** Encouragement of Researchers as seen by Agency Personnel.

Response— Research is:	Total	By Agency		By Area Research Use		
		NPS	USFS	High	Low-Medium	None
Discouraged	0	0	0	0	0	0
Neither	6	1	5	1	2	3
Both	2	1	1	1	1	0
Encouraged	5	4	1	2	2	1

One respondent said that more research would be of no benefit to management of the Wilderness. The respondent stated that “[b]udgets are so low, we couldn’t fund or implement any suggestions. We know how to manage and what to do—we just don’t have [the] budget.” For another area, the question asking if more research would be of benefit was answered both “yes” and “no.” There was concern that “unnecessary research in some cases aggravates congestion.” All other personnel said that more research would be useful. A variety of study topics were mentioned as potentially useful to the areas.

Two USFS personnel had additional comments as to the place of research in Wilderness; they emphasized research as long as no damage to the Wilderness value would be done:

—“I feel research in wilderness areas is a legitimate use—no doubt about it. But not if the intent is to ‘advertize’ wilderness in general or a particular wilderness area. Visitor impact within [the area] is presently very manageable; I’d hate to see that change drastically.”

—“Wilderness research is appropriate so long as such activities do not impair the basic values for which these areas were designated.

Three additional NPS comments emphasized a need for more research, more management/researcher cooperation, and more management-oriented research.

## CONCLUSIONS

From this survey, generalizations concerning attitudes throughout the agencies cannot be made, but some insight into views that exist among personnel and researchers was gained. Personnel generally reflected either indifference or a positive view toward research. Although some researchers have found perceptions of wilderness managers and recreational users at some variance (Lucas, 1964; Peterson, 1974; Hendee and Harris, 1970), this study shows fairly good agreement between manager and researcher perceptions. Matching researcher perceptions, USFS personnel more often saw research as neither encouraged nor discouraged in their Wildernesses, and NPS personnel more often said that they encourage research. Those who said research is encouraged in their areas all mentioned the usefulness of research to management as an underlying reason, again coinciding with researcher perceptions.

Means of encouraging research mentioned by agency respondents and researchers included: funding, granting of permission and permits

for research activities, provision of work space and accommodations, equipment use, personnel time, verbal encouragement, research solicitation, and cooperative agreements. Some of these items can be of little cost to the agency, such as provision of work space, accommodations, and simple verbal encouragement, but these can make a difference to potential researchers. While funding is often mentioned as a major concern, researchers can easily be given more encouragement in other ways. The resources of an area are of primary importance to the researchers; agency interest in the research is of secondary importance, very closely followed by agency cooperation and funding.

More study into how the attitudes and perceptions of agency personnel and of researchers themselves affect research could be informative. There is a common perception of variation in the attitudes of agency personnel between units of the same agency. To what extent do these attitudes have effects on research use; how much do they vary between areas, and how much do they vary from official policy? Do researchers develop preferences for one agency or type of area over another? Why and how might this affect the use of different areas? Future agency policies and activities could benefit from such information, so that actions could be taken to create more consistent research management throughout a given system of reserved areas.

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# The Role of Local Government in Habitat Protection: A Long Island Perspective

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**Abstract:** Local government, through implementation of various land use controls, has the opportunity to protect and preserve significant flora and fauna. Preservation requires the effective interaction of scientists, managers, land owners, special interest groups, as well as politicians and local government staff. Political and economic interests must blend with scientific and management concerns to provide a Habitat Protection Plan which is effective. Implementation of such a plan is discussed here.

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## INTRODUCTION

Municipalities are, or should be, on the forefront of determining land use and the future of a given locale. They are placed in the middle of many conflicting concerns, and many times the situation is highly charged with emotion and legal ramifications. Within that atmosphere, the municipality is expected to make decisions that are just, while protecting the environment, the rights of the general public, property owners and the future of the community.

The planning process can produce a feasible way to manage a municipality's natural resources. A Habitat Protection Plan should be one of the key components of any community's land use planning. The protection plan not only provides the basis for environmentally sound land use planning, but it also provides a resource in decision-making for officials, staff members, land owners, and the general public. The document should provide all members of the community with the resources to make informed, rational decisions concerning land use changes.

Pressure to develop the land on Long Island is very real. The population of Suffolk County townships has jumped markedly in the last decade, and the Town of Brookhaven has felt the greatest development pressure of the townships. This point is emphasized in the Generic Environmental Impact Statement (GEIS) prepared by the Town of Brookhaven for the implementation of the 1987 Land Use Plan (Town of Brookhaven, 1988). The GEIS states that "on a county wide basis, 75 per cent of Suffolk County's population growth between 1970 and 1980 occurred within the Town of Brookhaven. Between 1970 and 1980 Brookhaven's population increased from approximately 245,000 to approximately 365,000 representing a growth of almost 50 per cent."

Rapid development has obviously resulted in tremendous pressure on existing significant habitats. The major challenge to local officials is to attempt to protect these habitats through intelligent land use decisions. While working within their legal authority and without violating property owner rights, local government professionals have the responsibility to identify and protect the significant natural resources existing in their localities.

The Town of Brookhaven is a unique municipality. Over the last few years, the Town's Division of Environmental Protection (D.E.P.) has been fortunate to add staff members with various levels of specialized expertise relating to habitat protection. The duty of the D.E.P. staff is to protect and preserve natural resources within the Town to the maxi-

imum extent practicable. This task is not easy, especially since the Town of Brookhaven is the largest township in the United States (Town of Brookhaven, 1989). However, this paper intends to propose the mechanism to develop a Habitat Protection Plan and discuss strategies that have been successfully utilized within the Town of Brookhaven.

## DISCUSSION

As recently as the late 1960's, certain local governments on Long Island viewed their coastal and forest resources in a way that promoted exploitation, not conservation (Hiecoff, 1980). The exploitation varied through the years, following development trends dependent on the economic atmosphere.

Long Island is presently undergoing a period of prosperity, and with prosperity comes increased residential, commercial and industrial development. Existing natural areas are rapidly diminishing. The remaining land is not only more valuable as habitat for resident wildlife species, but also for development. Since the value of such land is high on both ends of the spectrum, land-use decisions become critical.

With any land-use decision, there are many groups with special interests in the outcome, and most groups feel their opinions are right and most important. People involved may include: special interest groups, property owners, project applicants, civic organizations, conservation groups, unions, economic development groups, local public officials, or various professionals.

The knowledge of most non-technical parties concerning habitat protection is usually limited. Private developers many times do not consider the natural environment as a complete resource. Generally, the overriding concern is maximizing profits, not habitat protection.

Members of the public may or may not appreciate their natural resources; however, the common adage "not in my backyard", usually applies as the rule. This personal point of view usually keeps the public from looking at the overall picture, and they get involved only when directly affected. Thus, public sentiment many times hinders appropriate environmental planning processes.

A factor that is sometimes overlooked is public perception of the wildlife resource. Many times a species or particular habitat becomes a celebrity in its own right. For example, the Long Island Pine Barrens in most people's minds should be protected at all costs as a habitat. Yet there exist other habitats such as breeding areas for terns (*Sterna* spp.),

black skimmer (*Rynchops niger*), and piping plover (*Charadrius melodus*) of equal or greater importance. Monies and protection may be available for Pine Barrens preservation but not for other important habitats because of favorable press influencing the views of the general public. Education of the public can enlighten all on the complexity and importance of our sensitive species and their associated habitats. In addition to education, however, local government's role, should provide the land use controls to protect and regulate the future development around sensitive areas.

### **A Habitat Protection Plan:**

The feasibility of a Habitat Protection Plan should be determined in the early stages of design. The plan, no matter how badly needed in a given locale, might be destined to fail if certain components are not taken into consideration during the initial developmental stages. Of major importance are the specifics of the plan and their clear demonstration to all parties involved. This is particularly critical because support can easily be lost if people are misled. All of those concerned must have a clear understanding of the values of the plan and how it relates to sound land-use management decisions. The following are considerations when proposing the implementation of a Habitat Protection Plan.

Initially, support for the plan must be gained from local officials. Local government officials need to have an understanding of the importance of a protection plan. The assurance that a balance between development and protection will be accomplished must be emphasized. Positive aspects of the plan from the local government's point of view should also be stressed.

Once the favor of the local official(s) is gained, background material should be made available in an effort to promote the plan. This might include information pertaining to specific sensitive habitats, unique plant communities and geologic features that should be protected. Maps and other graphics could emphasize specific areas that would benefit from the implementation of the plan. The next important step is gaining support of special interest groups. These would include community groups, civic associations, Audubon Society chapters, The Nature Conservancy and other related organizations. Many times it is these groups who hold the key to public acceptance.

Meetings should be held to explain the program. Discussions of the imminent need to protect habitats may solicit a positive response. The active members of each group can then promote the plan directly to its general membership.

Education is also needed on the professional level. Lawyers, teachers, builders, scientists and planners all can help in the promotion of the plan. It is important to present the plan in layman's terms in order to assure complete understanding. Not everyone has a background in conservation, but many will be able to understand, once the benefits of a protection plan are explained. Through interaction, liaisons can be developed to encourage the plan and involve colleagues and other staff members. Again, the greater the awareness, the easier the plan can be implemented.

Of course, the ability to implement the plan will ultimately be dependent on funding availability. This varies with the municipality's commitment to wise use and protection of its natural resources and its capability to allot monies for new programs.

## **INVENTORY**

Inventory is the backbone of any Habitat Protection Plan. The ideal situation is to have the staff expertise to undertake the identification

and evaluation of wildlife habitats and plant communities within a local government's borders. The Town of Brookhaven is very fortunate to employ a staff with a wide range of expertise. Specialization areas include wildlife management, environmental planning, environmental science, botany, wetlands management and bay management. With this diversity of backgrounds, the capability exists to undertake a town-wide inventory. This is an almost unique situation for a local municipality, because most do not have the funding to support such a large and varied staff.

The Town of Brookhaven has in the past relied upon information from other agencies (state, county, and federal) and the records of special interest groups, such as the Audubon Society, The Nature Conservancy, and the National Heritage Foundation, in attempting to assess the value of wildlife habitats and plant communities. Although adequate information is available concerning certain wildlife species (e.g., the osprey, *Pandion haliaetus*) and special plant communities (e.g., the Pine Barrens), specific information is needed on many others. While staff at the Town of Brookhaven has the expertise to conduct the studies required to document the information needed, time constraints limit the amount and type of information that can be recorded.

The implementation of the New York State Environmental Quality Review Act (SEQRA) Law (see page 302) has placed the major constraint on Town staff members. The high rate of development on Long Island in the past few years has tremendously increased the volume of work involving environmental review. This high rate of development has, in turn, increased the need for Natural Resources Inventories. These inventories can aid the Town in the following ways: (1) Provide knowledge of value in SEQR review; (2) Assess impacts of habitat fragmentation; (3) Provide baseline information to monitor changes in wildlife species diversity in the future; and (4) Identify areas in need of protection.

The Town of Brookhaven has been successful in collecting information on wildlife species populations by utilizing methods that do not require a great amount of staff time. The following is a discussion of inventories that have been successfully completed and also those planned for the future.

The Town of Brookhaven D.E.P., in cooperation with the Long Island Colonial Waterbirds Association has been conducting summer surveys of waterbird colonies on Town and Suffolk County properties. The main techniques used are direct (adult) count and nest counts. This is easily accomplished with even a limited staff because: (1) Waterbird colonies within the Town of Brookhaven are few (eight in number); and (2) each colony requires only two counts per breeding season. The surveys take approximately four man-days. The Town feels this to be a critical part of the program because population data are collected and proper protection is afforded to, the Federally endangered piping plover and roseate tern (*Sterna dougallii*), New York State endangered least tern (*Sterna antillarum*) and New York State threatened common tern (*Sterna hirundo*). In addition, data are recorded on species that tend to associate with the above species because of similar habitat requirements. These additional species include black skimmer, American oystercatcher (*Haematopus palliatus*), glossy ibis (*Plegadis falcinellus*), great egret (*Casmerodius albus*) and snowy egret (*Egretta thula*).

Winter waterfowl surveys of ponds, bays and harbors within the Town of Brookhaven have also been conducted to establish a yearly population index. These surveys are conducted once during the winter season and require four man-days to complete. These surveys are particularly important in providing population information on declining species, such as the American black duck (*Anas rubripes*).

The Town of Brookhaven D.E.P. has also undertaken late winter-

early spring inventories of vernal ponds to document the presence of the New York State endangered tiger salamander (*Ambystoma tigrinum*). Through these surveys newly documented breeding sites have been recorded, adding to information already gathered in a New York State Department of Environmental Conservation report (Cryan, 1984). The existing and newly gathered information regarding the tiger salamander has been extremely valuable in the protection of this species from development pressure. The Town of Brookhaven has had great success in protecting known aquatic and upland tiger salamander habitat by clustering development away from these sensitive areas.

The fact that tiger salamander surveys must take place at night makes the scheduling of staff time more difficult. This has been eased in the past by the provision of overtime funds and by staff members volunteering their own personal time. The staff has found the value of this endangered species work worth the donation of time and extra effort in the interest of preserving this species and its habitat. In addition, during the course of the surveys, important information may be gathered on other species associated with vernal pond habitat [e.g. various reptiles and amphibians, American woodcock (*Scolopax minor*) and fisheries resources].

Major concern over the fragmentation of important wildlife habitat in the Town of Brookhaven has been a driving force in the implementation of bird surveys on Town-owned property. The main thrust of the studies is to survey various terrestrial habitats from grasslands to forests in an attempt to collect valuable baseline data on species populations inhabiting representative Long Island plant communities. These surveys will provide the following information of value to the Town: (1) Baseline data on Town property to aid in protection and management programs; and (2) Data for determining what species may be present on properties slated for development.

One technique to be used in conducting the bird surveys is the variable circular plot method (Reynolds *et al.*, 1980). This method has been utilized successfully by Suffolk County (unpublished data) with limited manpower. The Town is currently preparing to survey five properties using two staff members. These properties will be surveyed twice a month (1-2 hours per survey) for two breeding seasons. These surveys will require a maximum of 20 hours a month in staff time, and will be partially funded by overtime.

In the interest of gathering specific information on rare and unique plant communities, the Town of Brookhaven plans to begin an evaluation of kettle holes in the northern portion of the Town. Kettle holes have been well-documented as containing remnant and unusual plant communities (Greller, 1977; Greller *et al.*, 1978). The study of kettle holes has been targeted because many have either already been impacted by development or are in danger of being destroyed. Most of these kettles occur in a small section of the Town, so surveys can be conducted in a well-defined area. Each kettle hole will be evaluated by recording all species present. The location of each kettle hole will be noted in order to be able to insure protection to these unusual geologic features in the future. Once field surveys have been completed, the information gathered should be added to the background material. The sum total of compiled data should then be incorporated into a scaled, base map. The map should list the occurrence of all species that have either federal or New York State status as Endangered or Threatened. Species designated of special concern by the New York State Department of Environmental Conservation should also be included. In the Town of Brookhaven we have also compiled a list of Town-wide species of special concern (Town of Brookhaven, 1987a). This list includes species that are unusual within the Town and whose habitats should be considered when development is imminent. Now, through

the use of background material and on-site investigation, the significant species have been identified within the area in question. The habitat needs for the identified flora and fauna now can be incorporated into development proposals.

## THE PLAN

To help promote a Habitat Protection Plan, oral presentations should be given during the course of its development and implementation for a number of reasons:

1. **PRESENTATIONS GIVE STAFF A FEELING OF ACCOMPLISHMENT.**
2. **PROBLEMS AND MISDIRECTION MAY BE DETECTED EARLY.**
3. **PRESENTATIONS KEEP THE PUBLIC INFORMED ABOUT THE PLAN.**
4. **PRESENTATIONS PROVIDE POLITICIANS WITH GOOD PUBLIC RELATIONS OPPORTUNITIES AND MAY BE USED TO ENCOURAGE SUPPORT.**
5. **MILESTONES AND GOALS FOR THE FUTURE OF A HABITAT PROTECTION PLAN CAN BE EMPHASIZED.**

Information should be made available to all interested parties. Materials may include graphs, maps or booklets that are easily read and reproducible.

To aid future planning for areas identified as significant habitats, overall land use patterns need to be addressed. The municipality cannot properly protect or preserve habitats if the patterns of development have not been analyzed. Maps depicting existing conditions, existing zoning and ownership of parcels prove very helpful. In addition, land use trends and future use patterns should be mapped out. Some or all of this information may help to eliminate mistakes in protecting habitats.

Existing plans and procedures should be considered when assembling the plan. In this way it can be related to plans that have already received acceptance from the public and private sectors. The protection plan is best when presented in a form that is easily understood where the application is clear.

## IMPLEMENTATION

Many factors affect the success of a habitat protection plan. Of major importance are achievements accomplished through the implementation of the SEQR process. Wise choices must be made during the process in order to afford adequate habitat protection, and success will not only be dependent on decisions made by the reviewing staff member, but upon choices made by local decision-making bodies (*i.e.*, Town Boards, Planning Boards, etc.). In addition, it is critical that public support is gained through a well organized public education process.

A major plus in the education process is to assure the public that they will be able to use their land, and that the plan will not necessarily mean a stop to development, but will define an acceptable way to utilize the land while also preserving significant habitats. If the public understands and appreciates the goals of the protection plan, then they will recognize the fact that development is feasible in conjunction with proper steps to protect critical habitats.

Economics will, of course, limit how the plan will be executed, as the cost of implementation of the plan will only be a fraction of the total cost. Once implementation has been accomplished, the cost to

administer the Habitat Protection Plan must be considered. How much will this Plan cost the Town to implement and carry out? How much will this cost the citizens of the Town in taxes? All of the economic ramifications need to be identified and thought out well in advance, or funding may not be obtainable.

Many times plans are drawn up and guidelines established, only to have their enforcement fall apart. If the municipality is serious about protecting its sensitive habitats, a commitment must be made to follow recommendations and findings of a protection plan closely. A trained staff should have the responsibility to utilize the Habitat Protection Plan when reviewing projects, and the plan should provide a mechanism by which significant habitats can be preserved, while allowing for carefully planned development.

### Strategies:

A key question arising for the municipality is: what methods might be best used to accomplish the goals of the Habitat Protection Plan? In the Town of Brookhaven, many techniques have been employed to protect significant habitats. The foremost concern should be to provide protection to the environment while not violating property owners' rights, since the municipality is in jeopardy of law suits and potential liability. It is usually easier to implement the following strategies prior to submission of development plans for a given piece of property. If this is not possible, the strategies to be considered should be thoroughly outlined in a comprehensive document so future plans will be reviewed accordingly.

**Acquisition** provides total control over preservation. This is the best option; however, it is expensive and funds are usually limited. This technique can be utilized most effectively as a last resort, while other less expensive mechanisms may achieve similar protection of critical areas of concern. Acquisitions should especially be considered when more than one governmental agency is interested in preserving an area. Joint acquisition is often a desirable step, since funding by one authority is usually not feasible. The Town of Brookhaven has recently used acquisition to protect eight millions dollars worth of environmentally sensitive land.

**Zoning** is the most commonly employed form of land use control in the United States. Local governments have jurisdiction over regulating land-use type and minimum lot size within their boundaries (Town of Brookhaven, 1987f). To protect habitats with significant flora and fauna, it may be desirable to implement certain zoning changes.

Two benefits may be achieved from zoning changes: (1) Development pressure may be reduced in sensitive habitat areas. An example of this is the upzoning of parcels from one dwelling unit per acre to one dwelling unit per two acres. (2) Inappropriate land uses may be defined and prohibited where protection is desired. Inappropriate land uses may include the rezoning of industrially and commercially zoned parcels to residential zoning, which allows at least partial preservation. The Town of Brookhaven is in the process of implementing numerous rezonings to accomplish future protection of natural resources. The following is a brief summary of each:

A. Commercial Rezoning - The change of commercially zoned property to residentially zoned property on approximately 1,250 acres of land.

B. Industrial Rezoning - The transfer of industrial zoning from inappropriate areas to more suitable locations. This action will change approximately 6,580 acres of land from industrial use to residential use.

C. Residential Rezoning - The change of approximately 95,000 acres of high density residential land to medium-density residential land (one dwelling unit per acre).

D. Large Lot Rezoning - The change of approximately 23,000 acres of medium density residential land to low density residential land (one dwelling unit per five acres and one dwelling unit per ten acres).

E. Stream Corridor Rezoning - The change of approximately 660 acres of high and medium density residential land to medium-low density residentially zoned land (one dwelling unit per two acres).

**Clustering** applies Section 281 of the New York State General Municipal Law which allows for "clustering" of allowable lot number (yield) onto only a portion of the total parcel (Town of Brookhaven Code, 1987e). Remaining land may then be preserved as open space. Such a land use technique provides opportunity to relocate or transfer development from a sensitive area within a project site to other areas more appropriate for development. In addition, this alternative may also reduce the cost of development. An illustration of this potential is a 100-acre site zoned residential for one dwelling unit per acre. Approximately 80 homes could be built, meeting the existing zoning restrictions and providing interconnected roadways. Clustering the same 80 homes on half-acre plots would require less roadway in addition to providing 50 acres to remain undisturbed. If attached clustering is desirable, development may only occur on ten acres while preserving 90 acres of sensitive habitat. Many times, in our experience, developers have dedicated lands that have been preserved through clustering to the Town, County, State, or Federal Government. This is an added benefit, since the property can then be managed to protect the existing resources into the future.

While no legislation exists in New York State regarding *Transfer of Development Rights* (TDR), the Town of Brookhaven and other municipalities have used a modified form of the Section 281 clustering to achieve similar results. The TDR Program essentially allows for the "clustering" of development rights from one parcel to another (Town of Brookhaven, 1987e). This method of land use allows for the preservation of large contiguous areas. This is beneficial when managing for species that require large tracts of land such as mammals, various bird species (especially raptors), ambystomid salamanders, etc.

Special incentives can be utilized to help promote a TDR Program. For example, an area appropriate for ultimate preservation would be designated a **Sending Zone**. The area would be zoned one dwelling unit per ten acres; however, if the owners chose to transfer development rights, they would be given five development rights for the same ten acres. A designated **Receiving Zone** would be then established where development is appropriate. Zoning here would be applied at higher than normal densities, with bonuses to those who purchase development rights from Sending Zones. A market would then be created for the buying and selling of development rights as commodities, with an ultimate goal of preserving large contiguous tracts of open space, while centering development within areas that have the infrastructure capabilities to accommodate more intensive uses. The problem the Town of Brookhaven has faced with TDR is locating receiving zones. Higher density development is often opposed by the people now living in the specified area. However, the Town is looking into TDR on a limited scale to preserve agricultural land and the associated habitats.

**Overlay Districts** (or Overlay Zoning) provide a method utilized to identify areas of special concern. Within a delineated area, constraints may be associated with any development. Examples of Overlay Districts include sole source aquifer recharge areas, flood plains, areas of steep slope, etc.

**Covenants** have long been in use prior to the zoning of lands. They basically limit the use of a particular piece of property. Covenants have been applied to everything from the restricting the nationality of occupants on a parcel (now illegal) to mandating where a person can put

out a clothes line. With regard to wildlife protection, covenants may be employed as a mechanism to protect a unique habitat where other methods fail.

Covenants can only be placed on a piece of property by the property owner. However, once in place, a municipality can require that the covenant specify that it cannot be changed without the expressed permission of the municipality requiring it. Covenant use is usually associated with a condition of approval. For example, if a significant habitat has been identified on a large lot during the review of a proposed subdivision plan, a covenant may be mandated by the municipality requiring the owner to restrict the use of this lot. In this instance the lot may have a covenant that allows no further subdivision and dictates where construction can take place. Other examples of covenants include required buffers, specified location of structures, maximum lot clearance, protection of steep slope areas, etc. The Town of Brookhaven has employed the use of restrictive covenants to protect habitats on literally thousands of projects.

**Special Legislation** to protect areas of concern may be enacted at many levels of government, and such legislation may or may not be accompanied by funding mechanisms to accomplish regulation. Often funds are unfortunately limited to the point where a municipality cannot implement enacted legislation.

In New York State, municipalities have the ability to enact local laws. These fall within the jurisdiction of what is known as "Home Rule." If properly developed and implemented, local laws have the ability to be more restrictive than New York State law allows. The following are examples illustrating laws that have been adopted and implemented by the Town of Brookhaven:

As with other townships on Long Island, Brookhaven is unique within New York State concerning wetlands regulation. The New York State Freshwater Wetlands Act of 1975 states that there are certain criteria which must be met before a local government can assume regulatory jurisdiction over their wetlands (Freshwater Wetlands Act 1975, 6 NYCRR 665).

The Town of Brookhaven has met these criteria by adopting their own wetlands ordinance (Town of Brookhaven, 1987d). Several townships have protected their wetlands through patents set up by the King of England. In our case, these patents establish a Board of Trustees and mandate the protection of resources for the good of the freeholders (residents) of the Town of Brookhaven (*Townsend v. Trustees of Freeholders*, 1904). Court decisions have reaffirmed the legality of such patents, which predate both the United States of America and New York State (Heikoff, 1980). The ability of municipalities to protect their resources is extremely important. Because state and federal agencies are usually interested in the protection of larger habitats that are of regional or statewide importance, some significant habitats may be overlooked.

Flood plains are usually not considered significant habitat areas; however, significant habitats often do occur within them. Most flood plain ordinances only address the setting of structures at a height that will protect against destruction by flooding (Town of Brookhaven, 1987b); however, when permitting activity within these zones, preservation of the natural environment should also be considered.

Tree preservation ordinances have different intents, depending on the problems facing a particular municipality. Within the Town of Brookhaven, the focus of the Tree Preservation Ordinance determines allowable percentages of tree clearing on a parcel, areas to be left undisturbed to protect wildlife habitats, prevention of erosion and aesthetic impacts of a proposed land use project (Town of Brookhaven, 1987c). Many rare and endangered plants have been preserved within the Town by the use of this ordinance.

In order to implement such local laws, our Habitat Protection Plan is used not only by the staff to review projects efficiently, but it is also utilized by the property owner/developer to identify problem situations and/or constraints on property. As a resource, it affords both the reviewing agency and developer critical information concerning sensitive habitats.

**Trade-offs** - During the review and permit process, it often becomes apparent that total preservation of a given habitat is not always possible. In a case like this, a trade-off may be an effective mechanism to reclaim lost habitat on another site. This has become quite common in the reclamation of wetlands habitat. The problem with reclamation of habitats is that unless the operation is monitored closely, the ecological community often fails to reestablish itself successfully.

One mechanism to assure that a reclamation project is carried out correctly is to bond the particular activity. This means that the municipality secures sufficient funds from the developer to correct or undertake the entire project if the developer does not implement the plan as proposed. In considering this strategy, all impacts and alternatives, including economic and legal ramifications, must be thoroughly investigated before an agreement is reached.

**A Management Agreement** can be an effective way to maintain desirable habitats that, under normal conditions, would be lost to natural succession. This technique has been utilized to preserve grassland and meadow habitats within the Town of Brookhaven.

The Town of Brookhaven D.E.P. has compiled a list of general policy standards to which all procedures are subject. These standards have been developed to provide general guidelines for acceptable land development. Examples of policy standards include maintenance of specific setbacks from bluffs, revegetation of native or "native-compatible" plant species, incorporation of wildlife corridors, nitrogen discharge limits for sanitary systems, etc.

*The State Environmental Quality Review Act (SEQRA)* is, without question, the most valuable tool for assessing potential impacts from proposed land use changes (New York State, 1978). The New York State Department of Environmental Conservation has recently revised the (6NYCRR 617) SEQRA rules and regulations (NYS-DEC, 1987).

The use of SEQR concerning the protection of significant habitats could be the subject of a paper in itself. Every person or group involved with land-use decisions in New York State has recognized SEQR's importance. This includes not only those who wish to protect the environment, but also those who seek to utilize it. Therefore, it is imperative that the local agencies use this law in the manner it was intended, *i.e.*, as an information gathering tool, not as a weapon. A misuse could result in the loss of resources and significant impairment to wise land-use goals.

SEQR should be implemented at the earliest possible time within any planning process. In this way the municipality is able to require a developer to consider existing natural resources on a project site. As the SEQR process continues, particularly if an Environmental Impact Statement is required, the alternatives and mitigation measures become extremely valuable to decision-makers. A Habitat Protection Plan can accurately provide background information necessary to address the sites in question.

## SUMMARY

A Habitat Protection Plan provides a mechanism by which information can be easily summarized and distributed to all parties involved in land-use activities within a municipality. The plan should be a matter of public record and identify constraints on a given piece of property.

The document provides the basis not only for land-use decisions, but also serves as an invaluable resource in defending legal challenges of approved actions by a municipality.

Developers who wish to move their project through the planning process have the ability to reference a Habitat Protection Plan and be abreast of problem situations associated with the land parcel in question. The plan allows the developer opportunity to avoid significant habitats or incorporate sufficient mitigation to reduce potential impacts. If information given out in such a plan is ignored, the local government staff then has appropriate backup to recommend the necessary mitigation.

A Habitat Protection Plan should be one of the key components of any community's land use planning. The plan not only provides the basis for environmentally sound land use planning, but also provides a common ground for decision-making for officials, staff members, land owners and the general public.

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# Interagency Cooperation Using The Landowner Contact Approach

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**Abstract:** An informal discussion is presented on a principle that has proved most successful when contacting owners of natural areas and suggestions are made on methods of supplying information in an acceptable manner. The approach is equally appropriate when dealing with managers in natural resource agencies and industries. Information and justification for natural areas preservation should be presented to them in such a way that they will want to pass the facts along to their managers and colleagues as appropriate. It is desirable to plant a natural-areas concept, "a seed", then go back and "water" that seed so that it will "grow" into a recognition and appreciation of natural areas by persons working within an agency or industry. People often remember only a few of the things told to them, so points presented to them early-on should be chosen carefully and explained clearly.

Pages 304 - 305. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## LANDOWNER CONTACT WITH PRIVATE OWNERS

The underlying philosophy of landowner contact (LOC) in the natural areas field is that of supplying information to the owner of a natural area in an acceptable manner. The immediate objective of the first LOC is usually to establish a positive relationship with the owner, so that you can maintain contact and monitor the condition of the natural area. You must constantly be aware that the landowner is generally unfamiliar with (and sometimes hostile to) the natural-areas concept and, in particular, and has many demands on his or her time; you are asking for a part of that valuable time.

Many landowners will be immediately impressed by the concept of a natural area and the fact that *they own one*. Their ability to protect that area will vary with their financial situation, their need to use that area and possibly their educational background. Don't come off as a bureaucrat (even if you are). Be patient and show genuine interest in the landowners' situation and desires. Be willing to listen to their interests, such as their family, the weather and how the crops are doing.

One thing to keep in mind is the distinction between a "registry" meeting and an "acquisition" project. If you are proposing acquisition, and you convince the owner that the area is unique or otherwise special, then he or she may expect a higher than fair market value for that tract. Make sure landowners understand that their natural areas are among a few areas left, and, if they decide not to sell, then you will approach another owner of a similar natural community.

## AGENCY AND INDUSTRY OWNERS

Public agencies and private industrial owners must be made aware of the significance of the natural areas under their management. If they own several natural areas, then their role as a multiple tract owner makes recognition of natural areas under their control even more important. Working with such an agency or industrial owner should be considered a long-term commitment, so strive to appreciate their constraints.

In the majority of cases, when approaching agency or industrial owners of natural areas, you are not interested in acquiring the property. Your interest concerns "in-house" protection or recognition of the area(s) and in some sort of management agreement. A management agreement will require expenditure of time and dollars for needed management (e.g., fencing, prescribed burning, controlled access) — time and dollars that currently are being spent on other resources.

When you approach an agency or company, you will be dealing with a representative or manager. This person is the equivalent of a landowner and should be treated as such. One of the most important things to keep in mind is that the manager you are dealing with may have no concept of the significance of natural areas to his or her organization. Hopefully, your first contact with a manager will be informal. Your objective should be to get to know that person, not to apply undue pressure to have them protect their natural area(s). Keep in mind that managers have other demands, deadlines to meet, and resources to manage. Managers must set priorities, and suddenly you are asking for a prominent place on their agenda!

At the first meeting, time is better spent presenting clearly defined facts rather than trying to convince a manager that natural areas should be given immediate consideration. Remember that the manager is on "company" time, and will more than likely tow the company line during your early meetings. You need to get beyond that, and to accomplish this requires getting to know that person *as a person* rather than talking all business when you first meet. Have a cup of coffee, a cola, or a beer after your meeting. Try to schedule your meeting near lunch or the end of the day. Your objective is to have the manager want to have contact with you again, or, at the very least, not want to avoid you. When an issue comes up that would benefit from your input, it is important that the manager feels comfortable contacting you to get your opinion.

A central tenet to understand is that people often remember only a few of the things you tell them when you meet. So your natural-area facts should be short and to the point. In other words, plant a good "seed."

If it is a good "seed" or set of ideas, the manager will think about the natural-areas concept *after* you leave, and it may begin to take "root." If this happens, your first and most important objective has been accomplished; you have instilled an awareness about natural areas within the agency or company visited. If you charge in full-blown, however, demanding for protection of a natural area, you will most likely be branded as just another environmental extremist, and the natural-areas concept will be shelved, right along with all other environmental issues. Don't allow that to happen. Natural areas are a critical resource, and they deserve primary consideration; their protection and management is defensible, even setting aside aesthetics.

When providing more "water" (facts) to the manager, provide information in a format that the manager will want to pass along to his or her supervisor. Offer information that will make the manager seem responsible and efficient when presenting to the supervisor. Remember that in order for your information to go any further than the manager, it

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needs to be reasonably defensible to upper levels of management. This requires having concise and accurate information. Don't add information that is speculative. Sharing information that makes everyone look good helps the manager and helps to foster a good relationship between you and that contact.

The amount of "water" and how often it should be applied will depend on the personalities encountered and the urgency of the project. Most likely, your goal of protecting and helping manage the other natural areas under a manager's jurisdiction will be met only if you establish such a positive working relationship. Too little water and the seed never germinates; too much water and you drown the seed.

As the natural-areas "plant" gets bigger, it can take more water — more information, more requests, etc. If everything goes well, you will eventually have produced a lasting awareness of natural areas within the agency or company concerned. Sometimes, however, you are faced with a situation where a natural area is in immediate jeopardy and

there is simply no time to work to develop a relationship. In this case, you will have to feed information more rapidly and hope that the agency or company doesn't choke on the volume. Give them the facts, one right after the other, but only one spoonful at a time. Force feeding often doesn't work.

## CONCLUSIONS

The main points of this discussion are to 1) get behind the agency or industry mask so that you come to know the manager as a person, 2) provide clear and concise information that is easily passed along, 3) do not expect immediate acceptance of the natural-areas concept, and 4) create an awareness of the need to protect natural areas, emphasizing their value to that particular agency or industry. Agency and industrial managers have many demands on their time, and they will require time and patience when considering your proposals.

# Participation by Local Cooperators: Management of Nuttall Oak, Rare in Illinois

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**Abstract:** Considerable opportunity exists at the local level for natural area professionals to establish mutually beneficial relationships with regional or local arboreta, private horticultural gardens, specialty plant nurseries, and similar allied entities. This paper provides an overview of some of these opportunities, with a focus on the management of the Nuttall oak (*Quercus nuttallii* Palmer), a species rare in Illinois. Topics discussed include public awareness, technical assistance, species recovery plans, establishment of secondary populations, research opportunities, and the ethics and implications of field collecting. Resource guides for locating potential cooperators also are discussed, and a case study is reviewed that illustrates possibilities for state, municipal and private interaction, centered upon a unique population of an Illinois Endangered Species.

Pages 306 - 309. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Terms such as public participation, local involvement, and volunteer program stimulate both positive and negative responses from natural-area professionals. Apprehensions may develop, based on visions of action for its own sake, even if that action might be inappropriate ("doing something, even if it's wrong!"). Many opportunities to enhance conservation programs may be lost due to the reluctance of overworked professionals to delegate responsibilities to ready, willing, and somewhat able volunteers. Local organizations and interested citizens can contribute much to natural heritage management, for instance. From the perspective of a conservation agency employee with a personal involvement in the natural heritage programs, I will present an example of such an opportunity and a viewpoint concerning volunteer contributions to rare species management programs.

## THE ROLE OF LOCAL COOPERATORS

There are many opportunities to involve volunteers and local cooperators in rare plant management. This is true particularly for locally significant species, ecotypes and significant taxa below the species level. The loss of biological diversity is globally epidemic, and substantial efforts directed beyond the most critical areas of species extinction would quickly drain the resources of governmental programs. Even the capabilities of the Center for Plant Conservation (CPC), with its 19 affiliated institutions (Falk and McMahon, 1988), would be stretched if the CPC had not identified specific priorities for its involvement. These priorities emphasize threatened, species-rich areas of Hawaii and the Southern and Western United States (Falk, 1988). Locally significant species and ecotypes not addressed by such programs could benefit from interim involvement of local cooperators.

While the possibilities for such involvement in basic *in situ* conservation work have been recognized widely and pursued ingeniously (Brown, 1989), the growing acceptance of *ex situ* activities as valuable components of conservation opens a new opportunity spectrum (Falk, 1987; Ambrose, 1987; Center Notes, 1987). The basic Natural Heritage goal, that of preservation of biological diversity (Gibbons, 1987), is best attained using *in situ* protection as a means (one of several), not an end. Complementary efforts in *ex situ* conservation, properly integrated into a comprehensive plan, frequently are useful or necessary to achieve this goal (Sharik, *et al.*, 1988; Moore, 1988; Olwell, 1988; see also Falk and McMahan, 1988).

## AN ILLINOIS CASE HISTORY

A master plan resource inventory was conducted by the author in 1972 at the Horseshoe Lake State Conservation Area, a waterfowl refuge located in Alexander County in Southern Illinois. It revealed Illinois' first known natural occurrence of Nuttall oak (*Quercus nuttallii* Palmer). A large, isolated individual, later determined to be a National Co-Champion Tree by the American Forestry Association (AFA), was identified on an island in the 2000-acre lake within the Conservation Area boundaries. This tree was twenty feet in circumference and scored 351 points on the AFA scale.

Further investigation confirmed its identification and established it as a species not formerly reported in Illinois. A subsequent search located several smaller mature individuals on the same island, existing as canopy intermediates and codominants in an old growth forest that later would be legally dedicated as Horseshoe Lake Nature Preserve. One putative representative of the species has also been reported more recently along the nearby Cache River (West, 1988), so the known natural population of Nuttall oak in Southern Illinois now consists of about a dozen individuals. A few more plants probably exist, but not many, considering the extensive human use of suitable Nuttall Oak reproductive habitat in Illinois and the failure of numerous biological inventories to detect additional individuals.

Nuttall oak shares many morphological and habitat characteristics with pin oak (*Quercus palustris* Muenchh.) (USDA, 1965), which is native and common in Illinois. Had the size of the large tree not attracted my attention, this species, new to the state, might have gone unrecognized. Even within its well-established range, where it commonly forms pure stands of high commercial value, Nuttall oak was not distinguished taxonomically as a species until 1927 (Little, 1953).

On the Element Ranking Scale developed by The Nature Conservancy (Morse, 1987), Nuttall Oak is assigned a G-5 global rank (secure) but an S-1 state rank in Illinois, reflecting its status as an extremely rare Illinois species, placed on the Illinois Endangered Species List (Illinois Administrative Code, 1984). Nonetheless, state ranking is not the primary source of interest in preservation of this recently discovered population. The unique attribute of Illinois Nuttall oak is its provenance and status as an allopatric outlier, occurring more than thirty miles north of the northernmost previously published latitudinal natural limit of a species otherwise characterized by high frequency saturation within its range and habitat (USDA, 1965; Miller and Lamb, 1985). It is not clear how this disjunct population became established at Horseshoe Lake. Is it an advance colonizing position of

an expanding species range, a relict position of a retreating range, or perhaps the product of a stochastic event unrelated to general range dynamics? Regardless of its origin at this location, it is important to consider the potential to isolate genetic alleles which, fixed in the local population by directional selection pressure and drift, may contribute to its ability to establish and survive beyond the climatic fringe of the species (Solbrig, 1970).

Nuttall Oak is a shade intolerant, acidophilic, fast-growing, early-maturing, seral inhabitant of alluvial clay soils; it exhibits a tendency to decline rapidly with advancing age (USDA, 1965; Miller and Lamb, 1985; Kennedy and Johnson, 1984). Considering its tolerance of low pH and flooding, its rapid growth, its early (five year) sexual maturity (USDA, 1965; USDA, 1974) and potential wildlife food value of its small (104/pound) fruits (USDA, 1965), the recently discovered natural population as far north as Southern Illinois, the population seems likely to have direct management potential beyond speculation on its contribution to biological diversity.

An immediate concern associated with the Illinois population is time linked. The Horseshoe Lake trees, although legally protected, are perilously few in number. No natural recruitment has been observed, and suitable habitat for successful germination and growth of seedlings is not present. The old trees exhibit obvious decline, and, since they are adapted to colonizing open, disturbed areas, they cannot successfully reproduce in the protected, closed canopy forest of the nature preserve or in the adjacent cultivated fields. I have observed only one tree (the National Co-Champion) to be relatively fruitful, and that occurs sporadically, as is typical of the species (USDA, 1965; USDA, 1974). Its occasional fecundity is probably due to its large crown, free from the competition which shades the smaller trees in the area. The frequency of this tree's mast years can be expected to decrease with its advancing age, however, and the future of this tree and its offspring, without human intervention, is in doubt.

Due to its significance being manifested below the species level, this Illinois population has not yet received priority for evaluation, research, or management. Several questions remain open:

1. Is the population genetically pure, distinct, and large enough to be significant and manageable?
2. What is the danger of hybridization with associated *Quercus* subgenus *Erythrobalanus* species contaminating any open-pollinated progeny?
3. Conversely, what is the risk of a deleterious increase in homozygosity due to the probability of selfing in this small population? How seriously might inbreeding depress the vigor of, or result in the expression of, lethal genes in the progeny?
4. Would attempts at genetic enrichment from core populations be likely to swamp any adaptive clinal shift, thereby reducing any specific adaptation of the progeny to Illinois?
5. What are the actual adaptive advantages of the Illinois population in terms of management for forestry, wildlife, horticulture or other purposes?

While these and similar questions remain unanswered, the Nuttall oaks continue to age. Citizen involvement is the primary force presently active in the preservation of the Illinois trees. Recognition as a National Co-Champion has secured enough interest in the largest tree to attract a restorative pruning by a local arborist volunteer, with the encouragement and approval of the conservation department (Illinois Dept. of Conservation, 1983). This pruning to removed dead wood and strengthened the crown is an attempt to help the old tree survive a few more storms than it might otherwise.

A seedling was germinated and established as a documented accession at Starhill Forest Arboretum in Menard County, Illinois. I have developed this Arboretum near Petersburg (American Horticulture Society, 1982; National Register, 1987) and, as the first person to identify Nuttall oak in Illinois, I have an obvious personal interest in the species. However, Starhill Forest is more than 200 miles north of Horseshoe Lake. Since the subject population is already beyond its published northern limit at Horseshoe Lake, Starhill Forest may not serve a major role in *ex situ* conservation other than as a hardiness test location.

A proposed recovery plan, recommending on-site enhancement of the Horseshoe Lake population, was suggested by K. Andrew West, District Heritage Biologist for the Department of Conservation. Direct seeding in a nature preserve buffer area was considered; Nuttall oak is reported to be one of the few hardwood species adapted to this technique (Johnson and Krinard, 1987; Miller and Lamb, 1985). However, predictable pressures from predation and competition would result in unacceptable losses in this situation, so outplanting of seedlings or whips was selected as a method presenting less risk to the relatively few propagules that might be available. Several private nurseries and a municipal park district have volunteered to propagate acorns from the Illinois trees for this program.

Even if sufficient seedlings can be propagated, loss of heterozygosity due to selfing may still be a problem (Wright, 1976; Namkoong, 1979). Some of the seedlings therefore may be used as understock for an attempted grafting of scionwood from the parent trees. Oaks are notoriously difficult to graft, and grafting techniques have not been published for this species; however, we have completed several interspecific grafts within *Quercus* subgenus *Lepidobalanus* at Starhill Forest, and Pin Oak (subgenus *Erythrobalanus*) is grafted commercially onto its own seedlings (Dirr, 1977). Since Nuttall oak and Pin oak are very close taxonomically, morphologically, ecologically, and (presumably) horticulturally, grafting success with one suggests a feasible approach for the other. If this can be accomplished, several of the young oaks to be planted at Horseshoe Lake may be grafted in an attempt to preserve the full genetic potential of the old parent trees.

Additional seedlings, if available, could be used to establish archival plantations at other secure locations within the likely adaptive range of the provenance. It is also anticipated that future research activities may require still more plants for winter hardiness trial distribution, common-garden phenology and growth comparisons, and other studies. The trees could be found suitable for wildlife, forestry, acid mine spoil reclamation, or other purposes in Southern Illinois, and orchards might be established for seed production or breeding.

Perhaps none of these economic potentials will materialize, and Nuttall oak will remain an obscure botanical curiosity in Illinois. This would be sufficient in itself, but excellent possibilities might be lost without timely intervention of citizens interested in propagating and rejuvenating the Illinois population from the senescing, significantly fruiting individual.

## PRECAUTIONS AND LIMITATIONS

Beyond preliminary planning and the matching of tasks to talents, the first step in successful integration of volunteers and local cooperators into *ex situ* aspects of a plant conservation agenda involves a review of the parameters within which a conservation team must operate. The popular admonition to be "part of the solution, not part of the problem" is appropriate in all public participation.

*Ex situ* conservation includes acquisition of propagules. To ensure

that this is done properly, all who are involved should follow a series of ground rules which I call "Ethics of Collecting", as follows:

1. Obtain landowner permission;
2. Acquire any necessary regulatory permits;
3. Do nothing to deplete or destabilize the source population, unless its loss is imminent, due to other influences. This precaution applies to the careless introduction or dissemination of pathogens or competitive exotics as much as to the overharvest of the target species;
4. Harvest from the wild population minimum viable propagules that are intended to be preserved *in situ*, being sure to collect enough material from different individuals, clones, or mating groups to ensure a relatively complete representation of the genetic diversity available;
5. In rescue-collecting (where subsequent *in situ* preservation is not anticipated), carefully remove as much material as possible, together with any associated mycorrhizae if present;
6. To assure optimum probability for successful *ex situ* reestablishment, use well planned, state-of-the-art horticultural techniques during the most practical season;
7. Learn enough about the autecology and physiological requirements of the target species to facilitate subsequent cultivation and propagation;
8. Do not operate in a manner which may encourage others to engage in unauthorized collecting, and use prudent restraint in revealing sensitive locations.

### VALUES OF *EX SITU* PROGRAMS

Artificial secondary populations, or plantations, have many potential uses. Among the most apparent and immediate possibilities are the alleviation of risk of local extirpation (due to disturbance to the wild population) and the increase in simple numbers of extant individuals. Placing specimens in labelled display groupings also serves an important public awareness function, if thoughtfully designed in a strategic setting (Jones-Roe, 1986; Brumback, 1987).

Properly managed plantations usually can be increased to provide surplus (and if necessary, expendable) research material which could not be acquired in the field without risk to wild populations. External pressures of pathology, herbivory, parasitism, competition, allelopathy, and edaphic or climatic conditions all can be explored more freely under controlled conditions in an experimental, non-critical plantation, conveniently located for observation and manipulation. Accessibility also fosters documentation of important phenological observations, which can be useful for *in situ* aspects of the program. Such observations can answer questions such as:

1. When are fruits ready for harvest or dispersal?
2. When are ephemeral herbaceous species most visible for field census?
3. When is foliage sufficiently senescent to allow nonselective systemic herbicide overspray treatment of competing, intermingled exotics?

Selective herbicide or phytotoxic pesticide tolerances and spray calibrations can be tried on test plantations before field application. Life cycles (*e.g.* blooming age of monocarpic perennials) also may be confirmed through observation of propagated individuals for which complete accession histories are known; such data are seldom retrievable with precision in spontaneous wild populations.

Accelerated reproduction (achievable asexually from stock plants by

cuttings and tissue culture) and genetic enhancement through controlled pollination may be used respectively to increase the quantity and local adaptation or heterozygosity of collected material. Obvious paths then lead back to *in situ* alternatives. Reintroduction into former habitat and introduction into potentially suitable substitute habitat become possible, if desired (Moore, 1988; Olwell, 1988). Inadequate natural recruitment can be reinforced, using selected material of local provenance (Falk and McMahan, 1988). Relict self-incompatible mating groups can be restored to breeding levels through introduction of cross-compatible groups from plantations established from other populations (DeMauro, 1988).

Such projects, and the subsequent stewardship often required, all can benefit from the selective assistance of local volunteers. Public involvement also can be applied to less technical activities. The most obvious, and certainly among the most useful, of these activities are fund raising, promotion, and political activism, none of which require botanical or horticultural training. Commercial opportunities should also not be ignored. Surplus plants may be distributed to discourage unauthorized collecting, or used in fund-raising plant sales.

### RESOURCE GROUPS

Participation in some of the previously mentioned activities might involve several of the following categories of local groups:

1. Arboreta and botanic gardens, if available;
2. Local natural heritage groups, organized to work on specific natural heritage projects;
3. Schools and colleges, with benefits of staff participation, meeting space, horticultural facilities, class projects, student volunteers, and extension services;
4. Scouts, garden clubs, Sierra chapters, and Audubon chapters;
5. Native plant societies and local experts, to provide technical assistance and information for the data base;
6. Park, conservation, and forest preserve districts, which can offer staff participation, facilities, newsletters, and possibly a land base;
7. Soil and water conservation districts, with their countywide rural landowner newsletter networks;
8. Utility providers, whose management can be encouraged to practice habitat management within rights of way;
9. Nurseries, capable of assisting with propagation and technical expertise;
10. Businesses and business associations, to sponsor activities and donate equipment and supplies.

Resource guides for locating most of these categories are available. [Examples of these guides were distributed to participants in the Public Involvement session of the 1988 Natural Areas Conference]. Incentives to encourage participation can occur at three levels: fee contracts, special privileges, such as visitation passes or sharing of surplus propagated plants, or simply media recognition, civic pride, good will, sense of achievement, and sincere appreciation which every skilled manager of volunteer programs has employed to accomplish the impossible.

### CONCLUSIONS

In terms of evoking a reaction from the listener, "public participation, local involvement," and "volunteer assistance" are terms that share a common element with such terms as "prescribed burning" (the

key word here being "prescribed," meaning intentional and planned, with predictable and desirable consequences, in a controlled situation). One wants immediately to know how the process will be kept from getting out of hand. Volunteers are available to assist in various ways, but their efforts require planning, supervision, and common sense. People and organizations have an inherent need to contribute to posterity, and one of the most satisfying and useful acts they can perform is to help prevent the extinction of elements of our natural environment. By allowing them to participate in such programs, we can employ an underestimated force to work in behalf of the preservation of biological diversity.

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# Maximizing the Effectiveness of a Natural Areas Program in Influencing Public Policy: The Maine Experience

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**Abstract:** Information on unusually significant natural features can play a pivotal role in shaping public policy; however, the existence of the data does not in and of itself guarantee an action upon it. While accuracy of information is fundamentally important, a variety of other factors, including the reputation of the organization that produced the information, the relationships between its authors and policy makers, and how the information was developed, are also major determinants of its impact.

Pages 310 - 312. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

## INTRODUCTION

Natural areas information influences public policy in several important areas. These include the content of state legislation, the content of executive branch positions and plans, acquisition priorities, and decisions on the management of lands and waters.

## EFFECTIVENESS OF NATURAL AREAS INFORMATION IN MAINE

In Maine, natural areas information has been very effectively used to influence the state's public policy. Natural areas information was a critical component in developing the state's nationally recognized rivers policy, which established that 1,100 miles of Maine's most outstanding rivers would be off-limits to the construction of new dams. This single act gave permanent protection to 39 of the State's most significant natural areas. Protection for important natural areas has been built directly into the management plans for some of the state's most important public lands, and protecting important natural areas will undoubtedly be a consideration in deciding which areas to buy with Maine's recently approved public lands bond issue.

## MAINE'S PROGRAM

Maine's natural areas effort, known as the Critical Areas Program, was established by the Maine Legislature in 1974 to identify unusually significant natural features of statewide importance. The program has compiled a register of over 600 areas with exceptional botanical, zoological, ecological, geological and scenic value. Direction for the program comes from the Critical Areas Advisory Board, a group of citizens appointed by the Governor. The program is also responsible for compiling the State's official list of endangered and threatened plants and working with landowners to encourage conservation of critical areas.

## FACTORS THAT CONTRIBUTE TO SUCCESS

There are a number of reasons for the effectiveness of natural areas information in Maine, beyond the fact that a broad constituency exists for protecting important natural resources. Factors that have contributed to the success of Maine's program are discussed in detail below, and include the following:

- 1) **A clearly-defined mission** aimed at identifying natural features of statewide significance;
- 2) **A strategic location** within the policy development arm of state government;
- 3) **Key actors** who understand the program's role and bring credibility to it;
- 4) **A focused program** that anticipates the information needed for emerging policy issues;
- 5) **Credibility** from thorough and systematic research on the relative significance of natural areas;
- 6) **Effective linkages with other programs** and organizations with similar objectives;
- 7) **Visible political support** for natural areas protection as an important state function; and
- 8) **Stability in program funding and management.**

## Mission:

Maine's program has benefited from having a clearly defined mission that focuses on identification, rather than control, of natural areas and covers the full spectrum of natural area types, from botanical to geological. The principal mission of Maine's Critical Areas Program is defined in its enabling legislation as the identification of "unusual natural features of statewide significance." This definition is important because it has both established a clear focus for the program, and kept it out of the regulatory business.

This mandate has proven to be of appropriate breadth for an effective program. Rather than concentrating on a single type of natural area, *e.g.* rare plant habitats, Maine's program has worked with a wide variety of natural features from the habitats of rare fish to the state's most significant waterfalls and rapids. Recently, the program has begun the identification of scenic areas. This breadth of coverage makes the program relevant to a broad cross-section of the public. At the same time, Maine has avoided defining large areas of the state, *e.g.* an entire river corridor, as a critical area. Thus, the confusion that can result from mixing regional planning concerns with natural area protection has been avoided.

Finally, while the program does work with landowners to encourage conservation, its mission does not include controlling land use in natural areas. Thus, the program can do its identification work objectively without being buffeted by the politics of regulation.

**Location in State Government:**

Maine's Critical Areas Program is located in the State Planning Office, part of the Executive Branch of State government, and the hub of public policy development in Maine. This has advantages: first, it gives Maine's program direct access to the state's policy makers, and second, as the State Planning Office has no land use regulatory function, it isolates the program from these pressures. As stated previously, this independence is of critical importance in assuring that assessments of significant areas are done objectively.

**Choice of Key Actors:**

Maine's program has succeeded in part because the persons on the Advisory Board and staff have understood the program's role relative to statewide natural resource issues and how public policy is developed. These key actors have guided the program through its formative years, and, because of their long-standing interests in these issues, they have brought credibility to its results.

**Focus:**

Information that becomes available today, but was needed yesterday, represents a lost opportunity. To avoid such losses and maximize the impact of natural areas information in forming public policy, means anticipating events—in essence, predicting what the political issues of the day will be, one or two years beforehand, and focusing the program to meet these needs.

Maine has focused its program by establishing yearly objectives. In the first few years, these objectives involved developing a planning report process for comparing all areas of the same type across the state and building the register to a respectable number. As the program matured, objectives focused on developing the natural areas information that was needed to resolve emerging public policy issues. For example, in the late 1970's and early 1980's a variety of efforts were undertaken to identify important, riverine natural areas. These included rare plant habitats, waterfalls, gorges and white water rapids. This work was undertaken in anticipation of disputes over hydropower proposals. Thus, when work on developing a state policy regarding river conservation and development began, inventories were already available, and these proved crucial in preparing a timely, informed analysis. This work resulted in Maine's precedent-setting Rivers Policy mentioned earlier.

**Credibility:**

It goes without saying that natural areas information should be factually accurate. Beyond this, the information should have further attributes if it is to have maximum effect. Maine's experience has been that information on the importance of particular areas has the greatest credibility if it is developed systematically and emphasizes comparisons with other areas having similar features. In this regard, the Maine effort has been organized around topics which are researched statewide. For example, reports have been prepared on topics as diverse as alpine tundra and fossil localities. The principal benefit of these reports is that all areas known to have a particular type of feature in the state can be systematically compared.

The objectivity of these comparisons is also important. On complex topics, Maine's program has even gone to the extent of developing explicit, numerical rating schemes to evaluate areas. These numerical methods take into account the variety of factors that contribute to the value of each area and rate the areas according to a cumulative numerical score. Such rating systems must obviously be put together with care to reflect as accurately as possible the diversity of values to be

considered. However, when properly done, these ranking methods can be both accurate and a highly effective tool for making clear to decision-makers and the public exactly how areas are evaluated, and why one is more significant than another. This approach is important, as it eliminates some of the mystery perceived in natural area evaluation and thus builds confidence in the process. In addition to making methodologies explicit, an agency should be sure that all information is supported by experts in the particular subject area under consideration.

**Links to Other Programs:**

It is extremely important for an organization generating information on natural areas to have effective links with related governmental and non-governmental efforts. One critical link is with experts who have natural areas information. For example, we have found that many experts at our State University and scientists affiliated with various natural history societies can provide a wealth of information on certain topics quickly and at modest costs. Without their help, the many topics Maine has addressed would have required tens of thousands of dollars and many years to research independently.

Links with conservation organizations like The Nature Conservancy, The Maine Coast Heritage Trust, The National Natural Landmarks Program, and others have stimulated thought and collaborative efforts, provided support for funding and mechanisms for non-state acquisition of important natural areas. Such alliances may avoid what can be extremely damaging political charges of duplication of efforts. The Critical Areas Program works very closely with The Nature Conservancy's Heritage Program in Maine to better protect highly ranked natural elements. Persons involved with natural areas inventories should maintain links that keep them abreast of emerging public policy issues which might require natural areas information or benefit from a natural areas perspective.

**Visible Support:**

Maine's program has benefited from support that it has received from governors, legislators, conservation organizations and the public. For example, the fact that Maine's last two governors, one a Democrat and one a Republican, have held yearly ceremonies to make critical areas awards to deserving citizens and corporations, has increased the program's stature as a legitimate part of state government.

**Stability:**

Maine's program has never been allocated a great deal of money. Early budgets were approximately \$30,000 and present budgets have increased to \$90,000. However, the program has persisted, maintained a stable core of supporters, moved from soft money to hard dollars and accomplished a great deal for a modest yearly outlay. In contrast with the present program, early efforts to develop information on natural areas in Maine resulted in a tome of information, but no ongoing effort to refine it or answer questions about its content. Lack of continuity and interpretation impaired its utility. It has proven more beneficial to have a stable, dependable program than a boom or bust effort with little coherence.

**HOW NATURAL AREAS INFORMATION HAS INFLUENCED PUBLIC PROGRAMS IN MAINE RIVERS POLICY**

The premier example of how natural areas information has influenced state policy is Maine's Rivers Policy. Aware that the unprecedented levels of hydropower development that were to follow the ener-

gy crisis would create conflict, members of Maine's Critical Areas Program began researching river-related natural area topics several years before the Rivers Policy initiative began. Planning reports were prepared on riverine, rare plant habitats, waterfalls, gorges and white water rapids. These reports evaluated all the areas known to possess such features in the state and ranked them according to their relative significance.

In the case of rare plants, waterfalls and gorges, the process of reaching conclusions about relative significance was quite straightforward. However, in the case of rapids, a complex ranking system, involving more than factors like height and volume of water (in falls) was needed. In this special case, a numerical ranking system was developed to include consideration of turbulence, bed width, length gradient and other factors. A report identified the state's 40 most significant rapids out of approximately 189 areas originally judged worthy of investigation.

Like many other states, Maine developed an energy policy in the early 1980's. The effort to prepare this policy was collaborative and involved some of the people working on Maine's Critical Areas Program as well as energy planners. Because they realized that conflicts were bound to occur, the framers of the policy included a recommendation that Maine's rivers and streams be classified by natural value. Because potential hydropower sites were already known, comparison of two lists (potential hydro-sites and high-value river stretches) would allow identification of specific sites where conflicts were likely. Natural areas information was critically important to this evaluation. Resulting recommendations led to Maine's Rivers Study. The study compiled existing information to classify rivers as: a) of statewide significance, b) regional significance, or c) local significance.

The Rivers Study led to an executive order and eventually legislation protecting 1,100 miles of Maine's most outstanding river resources. Of the 70 natural areas known to have statewide significance on Maine's rivers, 39 were afforded protection by Maine's Rivers Policy.

#### **Lakes:**

With the recent boom in second-home development, Maine citizens

have become concerned that they are losing the special natural values of undeveloped lakes. Road access to Maine's lakes has improved rapidly in the last two decades, and now development pressures are substantial. To plan for the best use of these resources, state agencies have undertaken assessments of Maine's lakes. Like the Maine Rivers Study these efforts characterize the overall natural quality of individual Maine lakes. The lake study has now been completed for the northern half of the state, and the regulatory agency that controls this area has adopted policies regarding management of different lake classes. The Critical Areas Program is completing an assessment of lakes in the southern half of the state. Once again, natural areas information has proved critical in developing a meaningful lake classification system.

#### **Public Lands:**

Another area where natural areas information has influenced public policy is in the management of public lands. Maine's Bureau of Public Lands, responsible for managing over 400,000 acres, contracted with the Critical Areas Program to identify significant natural areas on their highest priority holdings. The Critical Areas Program identified some 30 important natural areas on parcels of public lands totaling several thousand acres. As a result of this work, protection of these natural sites is built into the management plans for their respective areas.

### **CONCLUSION**

The development of Maine's Rivers Policy demonstrates how important natural areas information has been in shaping Maine public policy, and Lakes Planning offers an example of how natural areas information is influencing emerging policy issues.

Maine's program has been effective for a variety of reasons, including: clearly defined focus, strategic location in state government, thorough, systematic implementation and other factors. Similar issues to those faced by Maine's Critical Areas Program will undoubtedly arise in other programs. While merely cloning a successful program from one jurisdiction or subject area to another may not always be successful, understanding the underlying reasons for the success of Maine's program should contribute to successes in other efforts.

# Using Continuing Education to Inform the Public About Resource Management

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**Abstract:** Environmental resource managers who worry about the public's understanding of their work have the opportunity to educate people through adult continuing education programs. This paper describes the author's experience of teaching an adult education course on wildlife management at a local community college.

Pages 313 - 314. Ecosystem Management: Rare Species and Significant Habitats. New York State Museum Bulletin 471. 1990.

Wildlife management professionals at times express concern about the public's understanding of their profession and of the scientific principles on which the profession is based. Kellert (1987) notes that appreciation of wildlife among the public is typically narrow in its emotional and intellectual focus, being largely directed at a small component of the animal community. Kellert and Brown (1985) believe that to deal with this situation, environmental and wildlife education will need to move beyond emphasizing factual knowledge and increased concern for animals, in order to provide a broader ecological understanding of species in relation to their habitats.

Some segments of society question the basic principles of resource management, especially as they relate to wildlife. Heavily-populated Long Island, New York, includes many individuals who seem to feel that nature is best served by keeping man's influence out of natural areas. This sentiment is illustrated in a paper submitted by a continuing education student asked to state her beliefs about wildlife: "...I would like to see wildlife left as much to nature as possible. I believe the only way to preserve a habitat is to keep man out." This philosophy stands counter in many instances to what people involved in resource management are trying to accomplish by using management techniques to alter habitats and to restore plant and animal communities. Leaving nature alone may result in further losses of endangered species, particularly if an area is already under a stress or imbalance from some previous man-caused or natural disturbance.

It takes those working in the resource management field years of study to become familiar with the various sciences upon which resource management is based, through formal curricula at colleges and universities. Those outside the field do not share our background. Can we then expect the public to understand and support resource management and habitat manipulation if we do not help them to understand the goals and rationale of the field? Since most of our funding is dependent on public resources, public support of our work is a necessity. We should ask: what educational channels are available to build public understanding of resource management outside of formal curricula that are not readily accessible to most people? Continuing adult education courses are one option. Such courses are often short-term with no prerequisite, and they are often scheduled at convenient times for working people; however, a survey of continuing education course listings on Long Island indicates that the public had very little opportunity to learn about resource management through such an educational channel. Of over 2,500 courses offered, only a handful were environmentally oriented, and most of these centered on bird watching. The breadth of offerings suggests that many adults are indeed interested in lifelong education, both for career advancement and personal enrichment. Teachers are a particularly important group to reach, since they may use the information in their own classrooms. In such cases it

might be appropriate to offer credit, rather than a non-credit course, since teachers' raises are usually tied to credit courses.

To attempt to reach adults with resource management information, especially as it relates to wildlife management, the Eastern Campus of Suffolk County Community College and Cornell Cooperative Extension developed and offered a course entitled "Wildlife In The Long Island Environment." It was offered without any prerequisites and scheduled to meet in the evenings, once a week, for seven weeks. The objective of the course was to help people understand the principles of wildlife management through a series of case studies of species of particular interest in the Long Island area (hardshelled clam, *Mercenaria mercenaria*; least tern, *Sterna antillarum*; Ridley's turtle, *Lepidochelys kempi*; striped bass, *Morone saxatilis*; osprey, *Pandion haliaetus*; bay scallop, *Pecten irradians*). In each case study, the current population situation was described and numerous management options were presented. Discussion and student opinions were encouraged.

There were 21 individuals enrolled, from a very wide range of backgrounds. Some were professionals, working in the environmental field that wanted information about local wildlife, while others were simply people who enjoyed their backyard birdfeeders and wanted to know more about wildlife in general. The varied backgrounds of students did provide a real challenge, but a sincere attempt was made to present information that was comprehensible and useful to most in attendance. Each student participating in the class was given an evaluation form, of which twelve were returned and nine were not. Eleven students strongly agreed with the statement that the topics in the class were explained clearly and in a way they could understand. To the statement "I believe I now have a better understanding of the techniques and tools available to the wildlife manager in managing wild population of animals" six people agreed strongly and six agreed somewhat. Was the course worth the effort? There is no doubt that teaching a course takes time away from one's other professional duties, yet, for many of the people in the class, this was their first exposure to wildlife management concepts and practices. It is hoped that they will now share their new information with others. Future years will allow contact with additional students.

It is useful to think of public education as a marketing tool for an organization and its work, since education can help elicit public support by increasing public awareness and understanding. In the Cornell Cooperative Extension system, marketing of the organization is not left in the hands of a few specialists or public educators, but considered part of everyone's job.

In the resource management arena, Decker and Goff (1987) note: "Natural resource professionals who believe that they can sequester themselves and concentrate on biological or ecological study apart

from social and economic influences, are naive, misled, or both. Social and economic values exert pressure on all aspects of natural resource management today." This paper is presented with the hope that others involved in resource management will consider getting involved in public education. There are many diverse opportunities for professionals to contribute and to improve public understanding of their field. Continuing education programs are constantly seeking new classes that will attract students to their institutions. For us, the professional environmental scientists, not getting involved runs the risk of leaving environmental education in the hands of others. As Gilbert and Dodds have stated, "Our attention has been too heavy on the side of research and management and too light on information, education, and public relations. As managers we have tended to be unable to articulate adequately what we are doing and why."

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New York Botanical Garden Library



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**ISSN 028-3355**  
**ISBN 1-55557-192-1**