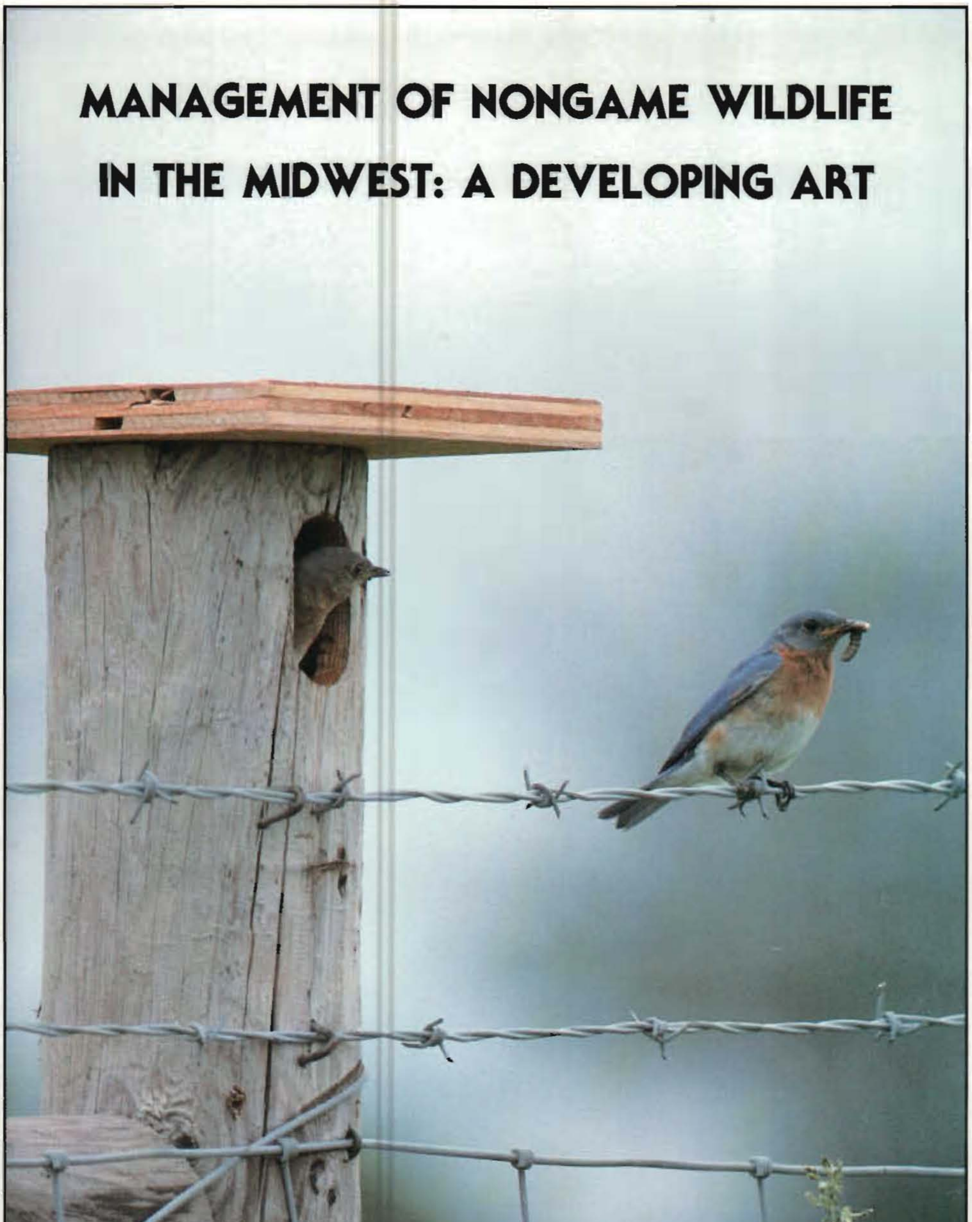
MANAGEMENT OF NONGAME WILDLIFE IN THE MIDWEST: A DEVELOPING ART



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FOREWORD

"Management of Nongame Wildlife in the Midwest: A Developing Art" is the proceedings of a symposium sponsored by the North Central Section of The Wildlife Society. The symposium was held on 17 December 1985 at the 47th Midwest Fish and Wildlife Conference in Grand Rapids, Michigan. Since 1975, The North Central Section has sponsored 5 biennial symposia on a variety of important wildlife species or groups. Proceedings of each have been published for use in management, research and education. This publication on management of nongame wildlife is a fitting addition to the series that includes volumes on grouse, furbearers, waterfowl and deer.

Symposium co-chairmen Richard Clawson, Louis Best and James Karr, and editor James Hale have assembled review papers from many of the principal proponents of nongame management and research in the midwest. Topics addressed include theory, principles, funding, public relations, promotion and research needs, as well as overviews of nongame management in lakes, streams, wetlands, grasslands, forests and agricultural systems.

The North Central Section of The Wildlife Society is pleased to have been a sponsor and the Section appreciates the efforts of the authors, organizers and editors in behalf of progress in the management of nongame wildlife in the midwest.

Donald H Rusch

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President
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Introduction

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In the field of resource management today we are witnessing a revolution—a revolution in our thinking, in the scope of our responsibilities, and in the very foundations on which our profession rests. We have found that we are still young, still in need of growth, still pushing the frontiers of our knowledge and abilities. We have learned a great deal, but we have found that there is a great deal more to learn.

We began by managing and studying game animals for sportsmen, largely using single-species approaches. Then came the realization that some species were in trouble and in need of help; we called them "endangered" and again applied single-species approaches. But close on the heels of the endangered-species concept came concerns for environmental quality and mandates to manage all wildlife. Suddenly the complexity of the tasks confronting us had increased many-fold. New buzz-words were coined, and today we are attempting to apply concepts such as "ecosystem management," "habitat fragmentation," and "landscape ecology" to our management.

The first symposium devoted solely to the management of nongame wildlife was held in 1975 at Tucson, Arizona. Other regional and national conferences since then also have focused on nongame species. Many state and federal laws have been enacted to address the needs and long-term maintenance of nongame wildlife. These events signal the most significant trend in wildlife management in recent times.

Considerable thought and effort have gone into nongame wildlife programs at the state and federal levels. Nongame management techniques are being developed and applied on large scales for the first time. The problems of integrating these new concerns into existing management programs, combined with funding and manpower limitations, create challenges at all levels of resource management. They raise a number of important issues: How do we pay for expanding programs? How do we mesh them with programs that have been in place for years? How do we choose among competing needs for limited financial and personnel resources? How do we integrate nongame management principles and traditional game management practices to create balanced management programs at the local, regional, state, and national levels?

This symposium was planned with the above concerns in mind. Our objectives were to: summarize the current state of the art of nongame wildlife management, discuss the incorporation of nongame considerations into balanced programs of resource management, and encourage the implementation of such programs. These proceedings include 11 papers in 4 broad topic areas: approaches to nongame management, planning considerations, ecosystem management, and the future of nongame wildlife management.

As resource management professionals we have embarked upon a challenging journey. We must apply new principles to old problems, and address new problems in innovative ways. These proceedings will not provide a "cookbook" for nongame wildlife management for all time, or even for today. We hope, however, that they promote principles that all managers can apply to their own unique circumstances, and provide approaches that will accommodate the needs of the entire wildlife resource spectrum. We face both great challenges and great opportunities. How we respond will be the measure of our profession as we conclude a century of stewardship, and look ahead to the challenges of tomorrow.

An Integrated Approach to Nongame Management

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Abstract: A survey of nongame wildlife supervisors in Illinois, Indiana, Iowa, Ohio, Michigan, Minnesota, Missouri, and Wisconsin indicated that all states have some form of nongame program. Annual budgets for nongame wildlife management range from $120,000 to > $800,000. Five nongame supervisors felt that needs of nongame wildlife species are for the most part not being considered in land management decisions, but all believed nongame species would be receiving increased attention in the future. Except in 2 states, wildlife habitat capability and simulation models are not being used to support nongame management. Six states, however, are using electronic data bases. The federal government had legislated nongame management responsibilities as early as 1916 with the signing of the migratory bird treaties. Focus on nongame species increased significantly in the 1970s with passage of the Endangered Species Act (1973) and the National Forest Management Act (1976). Federal agencies have spent much effort in developing data bases, wildlife habitat capability models, and simulation models. These tools are being used in land-management planning and alternative assessment for a variety of wildlife species, including nongame species. Numerous examples of models and computer-based tools exist. The most common in the Midwest are the Natural Heritage, PROCEDURES, TMIS, and WMIS data bases; HSI and PATREC habitat capability models; and FORPLAN, DYNAST, and STEMS simulation models. The type of tool used depends upon the user's objectives and the land base (stand, forest, region) being managed. With these data bases and models, wildlife managers in general and nongame managers specifically will be able to take a more active role in land-management decisions. But these tools must be applied with discretion. In their current state of development they are useful for general management direction, but not for detailed critical analysis.

In the past, wildlife programs were primarily, if not solely, oriented towards game and furbearer species. During the environmental movement of the late 1960s and early 1970s, public concern was focused on threatened and endangered species, and by extension, nongame species. In the last decade, state and federal agencies, professional societies, and private interest groups have publicized the importance of nongame wildlife and have stimulated a growing public interest in managing natural resources for nongame species. As the interest grows, so does the need for more direct and specific management of nongame species.

Nongame biologists are being sought for expertise and guidance in formulating habitat management guidelines, in developing new management techniques, and in evaluating land-management alternatives. As more wildlife species are considered in land management, the potential for conflict grows among alternative objectives. Fortunately, electronic data bases and other computer tools are available to help the resource manager compile, sort, and analyze the information needed to make rational management decisions concerning nongame wildlife.

NONGAME PROGRAMS

State

An informal survey by telephone was conducted with the nongame supervisors of 8 midwestern state natural resource agencies (Illinois, Indiana, Iowa, Ohio, Michigan, Minnesota, Missouri, and Wisconsin) to ascertain the current status of nongame management on state land. Five of these states have clearly defined nongame programs with designated staffs and budgets. In the other 3 states, responsibilities for nongame are spread throughout agency programs.

The nongame budgets within these state agencies are only 1% to 5% of their total wildlife budgets and range from $120,000 to more than $800,000 anually. It was difficult to determine the number of personnel assigned to a nongame program because of the variable degree of autonomy of the programs. In those states with clearly defined nongame programs, professional staffing ranges from 1.5 to 8 full-time equivalents.

When asked if models and other computer-based tools were used to support nongame management, 6 of the 8 nongame supervisors indicated that they use electronic data bases, with the most common one being the Natural Heritage data base. Although the degree of use varies, none of the supervisors felt their data bases are being used at near their full potential. Two of these 6 states are also using wildlife habitat capability models to support nongame habitat assessment, but none are using simulation models in planning. However, this is "an area (they) hope to get into."

The final question in the survey was whether or not nongame really is being considered in land-management decisions within the state. Three nongame supervisors felt that it definitely is and that it has been an increasingly important factor in the past 5 years. The other 5 supervisors felt nongame is being considered in certain instances, particularly in the case of endangered species or where the land manager has a personal interest in nongame. However, for the most part, nongame in these states really is not an important factor in management decisions as there are no formal requirements for its consideration. They added, however, that nongame is becoming an increasingly important element in their programs.

Federal

The federal government has been concerned about nongame management for many years. Authority for management of nongame birds by the United States Department of Interior (USDI) Fish and Wildlife Service was as established through migratory bird treaties, the first of which was signed with Canada in 1916 and the most recent with Russia, ratified in 1978 (Anderson 1979). These treaties provide protection for nongame birds listed in the treaties. A complete list of protected species was published in the Federal Register of 16 November 1977 (Greenwalt 1977).

Focus on nongame management increased significantly in 1973 with the passage of the Endangered Species Act. In 1976, the National Forest Management Act and subsequent regulations established a requirement for the United States Department of Agriculture (USDA) Forest Service to maintain viable populations of all native and desirable non-native vertebrate species on our National Forests (Salwasser and Tappeiner 1981).

The Fish and Wildlife Conservation Act of 1980 established a program to provide federal funding to states for nongame management. Although the act provides direction for the Fish and Wildlife Service through congressional intent, it has not been particularly influential because appropriations never have been authorized. Still, nongame concerns have become increasingly important in management on federal lands. Today, nongame management is a required element in forest plans, receiving much attention in the on-going public-comment phase of national forests planning.

Federal agencies have spent considerable time and effort in developing data bases and models to assist land managers in integrating wildlife needs, including those of nongame species, with other uses of public land. Particularly noteworthy here are the activities of the Fish and Wildlife Service Habitat Evaluation Procedures (HEP) program and the Forest Service Wildlife and Fish Habitat Relationships program. Development of these tools has come a long way, but use and verification are really just beginning.

MODELS AND COMPUTER-BASED TOOLS FOR NONGAME MANAGEMENT

The combination of socio-political and biological concerns in the real world where the resource manager must operate produces a maze of management alternatives. Resource managers such as nongame biologists must have a functional tool that will reduce at least some of the complex biological impacts to a comprehensible level for analysis and presentation if they are to participate effectively in the management (political) process.

Over the past decade, numerous models or computer-based tools have been developed to assist the resource manager. These differ greatly in style and complexity, but as a group they give the manager the ability to fully integrate all resources, including nongame, into land management. These are, however, only tools and not the process.

Electronic Data Bases

Data bases are simply large amounts of detailed information stored electronically in a computer system. They vary widely in scale depending on their purpose. The United Nations Environment Programme has a global resource information data base (GEMS-Global Environment Monitoring System) containing a wide range of environmental data for world-wide environmental monitoring and assessment (Croze 1983).

The Fish and Wildlife Service has developed a standard procedure for collecting fish and wildlife information from published literature and other sources, and storing that information in a data base called PROCEDURES. The USDA Soil Conservation Service has the National Resources Inventory (NRI) information system to assess the status and trends of soil, water, and related resources every 5 years. The Forest Service has detailed forest stand information compiled in its Timber Management Information System (TMIS), and within the Eastern Region has wildlife habitat data stored in a parallel Wildlife Management Information System (WMIS) (USDA Forest Service 1980, 1983).

State agencies and private foundations also have or are developing data bases useful to the nongame manager. Missouri is completing its Fish and Wildlife Information System (MFWIS) which will house data for all the state's 778 vertebrate species and 100 aquatic invertebrate species important to stream management and pollution control programs (G. Kelly, pers. commun.). MFWIS is Missouri's applied version of PROCEDURES. The Natural Heritage data bases, initially established by The Nature Conservancy in cooperation with state agencies, house information on rare and endangered species in a format standardized for easy access.

One type of electronic data base important to wildlife habitat modeling is the GIS (Geobased or Geographical Information Systems). The juxtaposition of vegetation with other land features such as water and topography is an important attribute of wildlife habitat. All natural resource information housed in GIS has land-based coordinates, allowing the user to restructure or merge data files into a variety of multi-resource maps, regional summaries, or specific area details as needed. Systems such as this have been described by Mead et al. (1981), Myers (1983), and Davis and DeLain (In press). Munro (1983) reviewed the selection process for acquiring an appropriate GIS. As in the case with all computer tools, the first and most important step is to develop a specific goal statement.

Habitat Capability Models

Habitat capability models may or may not be computer based. The earliest habitat models, an outgrowth of our data bases, were long matrices that provided listings of wildlife species in various vegetative communities (Thomas 1979). Other models provided a subjective score for key species of wildlife, based on favorable habitat attributes determined from the literature (Flood et al. 1977, Baskett et al. 1980). More recently, however, habitat models have incorporated the use of graphs, mathematical formulae, and probability functions to give the resource manager the ability to assess quantitatively the capability of a tract of land, pond, or stream to support wildlife at a given point in time.

The 2 most common types of habitat capability models are the Habitat Suitability Index (HSI) models developed by the Fish and Wildlife Service's Western Energy Land Use Team (Schamberger et al. 1982) and the Pattern Recognition (PATREC) models originally adopted for wildlife habitat use by Williams et al. (1977). HSI models provide an index of the capability of a given habitat to support a particular wildlife species; PATREC models also estimate the population density of a given wildlife species.

Simulation Models

Other more complex models are available as tools to integrate wildlife, including nongame species needs, into resource management. These simulation models project future forest habitat conditions from current inventories and growth patterns. The Forest Service uses a linear optimization program called FORPLAN (Johnson et al. 1980) to evaluate alternative management strategies. DYNAST, developed by S. G. Boyce at Duke University, is a continuous simulation model that can be used for cumulative effects analysis on timber, wildlife, and other related forest resources (Boyce 1977, 1978, 1980). DYNAST and FORPLAN are multi-stand simulation models based on projections of growth in age or size classes. The North Central Forest Experiment Station and Duke University have completed a cooperative study that expands DYNAST to include a grasslands simulation sector, making it more responsive to the entire terrestrial community.

Individual stand simulation models also exist. PROGNOSIS is a forest stand development model (Stage 1973) that recently has been interfaced with COVER (Moeur 1985) to include projections of canopy structure and understory for wildlife habitat evaluations. STEMS, along with its microcomputer counterpart TWIGS, is a simulation system developed in the north central states, based on individual tree evaluation and growth (Belcher et al. 1982). CLIMACS (Dale and Hemstrom 1984) is another individual-tree simulation model for timber stands in the Pacific Northwest, which was modified from FORET (Shugart and West 1977), a succession model for eastern Tennessee.

Computer tools also are available for planning at other levels. For example, generalized plant-growth models are available for planning on very large scales. FORPLAN can be formulated to simulate generalized conditions for regions or even for the nation as a whole. GROWEST is a plant growth estimator used to elucidate biogeographic patterns and processes at a continental scale in Australia (Nix and Gillison 1985).

Simulation models such as these will permit the resource manager to predict future wildlife habitat conditions that might result from various land-management strategies specific to a manager's objectives and area of responsibility. When coupled with nongame habitat capability models, the nongame biologist has a tool to effectively evaluate management alternatives.

INTEGRATED MANAGEMENT—THE PROCESS

The above is not a complete listing of all computer data bases or models available. It does provide, however, some examples of the types of information-processing tools available to the nongame manager for alternative assessment. The types of tools used depend upon the user's objectives, the land base being managed (stand, district, state, region), and the place in the planning process (goal and objective setting through impact analysis) where the tools are being applied. The specific tools chosen depend as much upon the user's familiarity with and the local availability of certain models as with the advantages and disadvantages associated with each.

Within a general type of tool, most are quite comparable. However, any one of these tools by itself may not be very useful. Their real value comes when several are combined into an analytical package used to support decision-making along the entire planning process.

A detailed example of how these tools can be used in integrating terrestrial wildlife habitat concerns into a multiple-resource forest plan was presented by Kirkman et al. (1986). Forest-wide and district management goals were set using the linear optimization routines of FORPLAN, a multi-resource, multi-stand, forest-simulation model.

To develop meaningful options to meet these goals, wildlife biologists on the Mark Twain National Forest first used electronic data bases (TMIS and WMIS) to summarize present terrestrial vegetation conditions. Current wildlife population densities were estimated by using PATREC habitat capability models. A multi-stand simulation model (DYNAST) was used to describe future forest structure under a variety of forest management options. PATREC models again were used with these predicted habitat conditions to estimate population responses of wildlife species. Economic factors were measured and displayed along with other resource parameters for consideration by administrators. All participants agreed that the process enhanced understanding of options and consequences, and thereby improved the decision-making capability.

In a different scenario, a state wildlife biologist responsible for nongame might be asked to evaluate two tracts of land and recommend which tract should be purchased with nongame checkoff funds. In this case, where no inventory exists in data bases, the tool to use might be a set of HSI capability models. The models could be used in the field to score a sample of representative stands on each area, and a composite index could be developed for each area. The specific models chosen might be dictated by the objectives of the purchase, or a data base might provide a listing of all species likely to occur on the areas or those of special management concern in the county. No simulation models would be needed because the goal is to evaluate current conditions on alternative sites.

Through the use of electronic data bases, forest growth simulation models and habitat capability models, the nongame manager can review a greater base of information and examine a wider variety of management alternatives for a more complete list of species. Armed with this type of analysis, the nongame biologist will be able to interact in more land-management decisions.

In addition, quantitative outputs in the form of tables and graphs will allow the nongame manager to more clearly document and justify a recommendation and thereby more effectively present that recommendation to colleagues, superiors, and the public. With these types of tools, wildlife managers in general and nongame managers specifically will be able to take a more active role in land-management decisions.

Long-term forecasting of wildlife populations using these systems assumes that plant-community simulation and wildlife-habitat relationships embodied in these models are accurate enough to allow managers to reach reasonable conclusions about the viability of future populations. Although most of the underlying wildlife habitat relationships within these models have not been field tested, the models do represent a compilation of professional knowledge on species and their habitats.

Most models to date have been developed through detailed literature summaries and subsequent reviews and refinements by species experts. The models produce results consistent with the whole of professional knowledge on the relationships modeled and in that sense are valid tools for use in alternative analysis.

These tools must be applied with discretion. In their current state of development they are meant to be used for general management direction, not for detailed critical analysis. The user must be aware of the assumptions within models and the limitations on outputs to correctly interpret results. This is particularly true in the case of nongame species models, which often are based on fewer studies and more limited information than game species models.

In addition, not all wildlife species are modeled. Very little is known about the habitat requirements of many rare or particularly secretive species. These species must not be overlooked when evaluating management alternatives. Data bases, habitat capability models, and simulation models should supplement the nongame biologist's knowledge, not replace it.

The foregoing discussion has focused on tools, but in each of the examples presented, the goals or specific objectives dictated which tool or tools should be used. Setting goals and delineating specific objectives are critical first steps in planning. We can't have more of everything; compromise is a must. This is a tough task, requiring an interpretation of federal and state laws in addition to public needs and desires. Consideration of nongame species is not going to be "integrated" without soul searching, conflict, and goal setting.

Data bases and models will vastly help with sorting information, identifying alternatives, evaluating consequences of those alternatives, forecasting, etc. However, they are only tools and they alone will not integrate nongame in the management process. Nongame biologists are still the operative element. Only through a concerted effort by the biologist to integrate nongame into all phases of planning, from goal setting on, will a substantive change be realized.

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Ecological Principles of Wildlife Management

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Abstract: Wildlife management is based on ecological principles that typically are framed with reference to basic ecological units (populations or communities). Objectives of wildlife management also can be divided into categories dealing with the control of population characteristics (population size, density, age structure, sex ratio) or community characteristics (species composition, species richness, patterns of relative abundance). The dichotomy between game and nongame management has no basis in either management objectives or underlying ecological principles. The important principles dealing with the management of populations and communities have not always been followed in developing management objectives, but they must be recognized in planning the direction wildlife management will move in the future.

The field of wildlife management is an important branch of applied ecology, and since gaining self-consciousness through Aldo Leopold's (1933a) seminal text, our field has been based on ecological principles. It can be argued, in fact, that wildlife management actually has few, if any, principles that can be considered uniquely its own; rather, nearly all of the principles that form its basis are borrowed from the basic science of ecology. As the scope of wildlife management has expanded over the years, it has done so by gradually embracing and applying an ever-growing body of ecological principles that may be relatively new to the field but are usually not new to ecology.

At its inception and throughout much of its relatively short history, the field of wildlife management has focused its primary attention on game species—those animals exploited for harvest. There has been, however, a steady trend to include the management of other types of wildlife as well. These other animals have become collectively designated nongame species. The person responsible for the term "nongame" probably has done the field of wildlife management no favor. Actually, the nongame designation encompasses several discrete types of animals with different management problems, including wildlife pests, rare and endangered wildlife, and the remaining members of the wildlife community that have less direct interactions with human beings but are judged to be worth preserving.

Some wildlife managers have made much ado about the field's steady expansion into nongame management (Graul et al. 1976, Brocke 1979, Bolen et al. 1980), and indeed, the distinction between game and nongame management has become firmly codified in many bureaucracies. Possible reasons for the rapid acceptance of this dichotomy within the field include differences in the interests and concerns of the individuals (both in the public and the management field) that espouse these types of animals (Kellert 1976, 1980), differences in the sources of financial support for management of these types of animals, and differences in the types and numbers of animals that fall into the two categories. On many other grounds, however, making distinctions between game management and nongame management may lead more to confusion than clarification of the basic management approaches that are involved (Brocke 1979).

OBJECTIVES OF WILDLIFE MANAGEMENT

Wildlife management is any purposeful attempt to influence the characteristics of a wildlife population or community. Depending on the particular species in question, however, management objectives can vary widely. The objectives of managing game species are remarkably uniform among species.

The common objective of game management is to maintain each game population at a size that allows a sustained yield or harvest with predetermined characteristics. It is important in this type of management to maintain the game population at a particular size, because so many of the characteristics of the harvestable individuals will depend on the population's size (Caughley 1985). The number of harvestable individuals depends on the population's size. Numbers of individuals available for harvest are greatest when the population is at a size that allows maximum productivity. This is the basis of traditional "maximum sustained yield" management.

In addition to affecting the quantity of harvestable individuals, population size also can affect the quality of harvestable individuals. Smaller populations relative to available resources may, for example, produce larger individuals or a greater proportion of trophy animals (Caughley 1977). In any event, whatever the desired characteristics of the harvestable individuals might be, management of population size will play a central role in achieving these characteristics.

In contrast to game management, nongame management does not have a common objective for all species. In fact, it is futile to attempt to propose a uniform objective. Herein lies one of the problems with the game *vs.* nongame dichotomy. Nongame species are such a heterogeneous lot that they cannot be categorized into a single unit. The term nongame is basically an exclusionary term; it describes what the species are not, but conveys nothing about what they are. For this discussion I will subdivide the presently defined nongame species into 4 more meaningful categories: pest species, endangered species, rare species, and other species that do not require specific management. Within these units it is much easier to identify common management objectives and the basic principles that apply to management tactics.

The objectives of managing a wildlife pest species usually are two-fold. Initially, the objective is to reduce the population's size to the point at which interactions between the pest species and human interests become tolerable (Caughley 1981). For a variety of reasons, the tolerable population size varies greatly among pest species, but the basic objective remains similar. After the pest has been "controlled," a secondary management objective often is to prevent the pest from regaining its numbers. This objective usually involves identifying and taking advantage of limiting factors for the pest species.

The objectives of managing endangered species are, in a sense, the reverse of those for pest species. Initially, the objective is to increase the endangered population's size to the point at which the probability of its extinction becomes small (Temple 1978). Once an endangered species has recovered its numbers, a secondary management objective is to prevent the recovered species from declining again.

Objectives of managing rare species of wildlife usually involve managing the environment in such a way that the rare species do not decline and either become endangered or suffer local extirpations.

This leaves the final, catch-all category of "other species." There certainly are far more wildlife species in this category than in the other 3 combined. The objective of collectively managing these species is to maintain or preserve the status of these species as members of the wildlife community of an area. Status is usually evaluated by the relative abundance of the species within the community.

An examination of these objectives of wildlife management reveals that a game *vs.* nongame dichotomy does not describe the system nearly as well as a dichotomy based on population (or single species) orientation *vs.* community (or multi-species) orientation. Management of game species, pest species, and endangered species always involves objectives and activities that are focused rather narrowly on populations of a single species and that are defined in terms of population characteristics such as population size, density, age structure, or sex ratio. Managers usually are intimately familiar with "their species," whether it is a game species, pest species, or endangered species, and their work often is so concentrated on that species that professionals become labelled with titles such as "deer biologist," "eagle biologist," etc.

In contrast, management of rare species and all of the other species that are members of a wildlife community seldom involves objectives or activities that are focused on only one species. Rather, management usually is based on objectives and activities defined in terms of characteristics of wildlife communities or other multi-species assemblages, such as guilds (e.g., Mannan et al. 1984, Verner 1984). These characteristics include species diversity, patterns of relative abundance, and species composition. Managers who are involved with managing these multi-species units usually are not so easily "labelled" and often are merely called "nongame biologists."

The traditional bias of the wildlife management field toward population-oriented rather than community-oriented management often is revealed by attempts to reduce community-level management to the management of an "indicator species." This reduces the responsibility to the more familiar "single-species" scope with which many managers feel more comfortable, but it remains to be demonstrated that management focused narrowly on indicator species will, in fact, satisfy the needs of an entire community (Verner 1984).

At the level of management objectives, then, there appears to be no clear-cut dichotomy between game and nongame management. Game management is almost always oriented toward objectives that deal with populations of single species, but nongame management also involves these types of objectives, particularly when dealing with pest species or endangered species. On the other hand, nongame management often involves objectives that deal with wildlife communities, and these objectives are encountered rarely in traditional game management.

ECOLOGICAL PRINCIPLES OF WILDLIFE MANAGEMENT

It is outside the scope of this presentation to review all of the specific principles that wildlife management has borrowed from ecology and applied to the challenges of attaining management objectives. Some ecological principles, however, stand out as having fundamental and uniform application to particular types of management problems. These central principles are statements of basic ecological relationships that almost invariably describe the way a population or community will respond to some change in its environment. Hence they provide wildlife managers with some confidence that their purposeful manipulations of a relationship will have an anticipated outcome. Ecological principles of this type are the basis of our field; they are the rules by which we play the management game.

Ecological principles, however, are often perceived as less absolute than principles of mathematics or the physical sciences, which often deal with invariable relationships. For most ecological principles, there may appear to be exceptions. It is often the case that ecological principles must be presaged by the phrase "All else being equal," because several environmental factors may affect the response of an organism. The need for this proviso may become clearer as we review some of the central principles that are important to wildlife management.

Principles Applicable to Management of Populations

The ecological principles that form the basis of management of game species, pest species, and endangered species deal primarily with population dynamics. This should follow logically from the basic objective that management of these types of species share—controlling population size.

Although one may subdivide this objective into several more narrowly stated principles, for the sake of simplification, the following principle seems to be at the heart of all management of populations. It is, in fact, probably one of the first ecological principles that our field adopted in its formative years:

> "A population's size can be controlled by varying the rates of mortality, natality, ingress, or egress in the population."

Whether one is managing a game species, a pest species, or an endangered species, the common approach is to manipulate one or more of the fundamental population processes in order to achieve a desired change in population size. This is the principle that was elaborated upon by Aldo Leopold (1933a), and it is a principle that has provided the wildlife management field with many successes.

As wildlife management has expanded its horizons over the years, additional principles dealing with populations have been incorporated into the manager's capabilities, and some previously accepted principles have been modified as a result of new knowledge. For example, managers have long known that manipulating a species' habitat was one of the effective ways to influence the basic population processes. There was, however, a general lack of agreement on the basis for deciding what comprised "optimum habitat" for a population. Attempts to identify "optimum habitat" on the basis of population density seemed in many cases to be related to a basic ecological principle, but too many exceptions to the relationship between high density and habitat suitability emerged (Wiens and Rotenberry 1981, Van Horne 1983). More recently it has been proposed that the following principle is a more accurate statement dealing with habitat quality for a population:

> "The optimum habitat for a population is the one in which the population achieves the highest intrinsic rate of growth."

As the number of endangered species has grown and wildlife managers have assumed responsibilities for helping endangered species recover, many new principles that deal with the problems faced by small populations have become important in the field. Traditionally, wildlife managers dealt with species that were typically present in larger numbers. Most classic game animals, for example, became exploited species because of their natural abundance as well as their value. When management of small populations became an important activity, new principles were derived from ecological or population genetics (Soulé and Wilcox 1980, Schonewald-Cox et al. 1983). Perhaps one of the most fundamental of these new principles deals with minimum critical size of populations (Shaffer 1981):

> "Every population has a minimum critical size below which extinction due to stochastic events in the population's environment, demography, or genetics becomes highly likely."

This principle is the basis for our concern over populations that have dwindled to critically low numbers. It also provides a basis for deciding upon the population size for which management should aim during recovery efforts.

In summary, the principles of population dynamics that are important in the management of populations are well understood by wildlife managers; they are the principles that have shaped game management and that have been easily adapted to the management of certain nongame species such as wildlife pests and endangered species. Some modifications of principles that had been incorrectly applied in the past recently have been made (e.g., density as an indicator of habitat quality), and many newly adopted principles have become important as managers undertook the management of endangered species (e.g., concept of minimum critical size).

Principles Applicable to Management of Communities

Because they form the basis for managing certain types of nongame species, many of the ecological principles that are important in the management of wildlife communities are relatively new to the field. Although wildlife managers always have been aware of many principles of community ecology—indeed, Leopold's (1933b) concept of a land ethic is based on such principles—they did not often put them to direct use in the management of game populations. With the expansion of management responsibilities to include members of the local wildlife community that have not been singled out for population-oriented management, new principles had to be adopted.

For some wildlife managers the initial reaction to the challenge of managing wildlife communities was to take a *laissez faire* attitude. Comments such as: "the best management is no management," "if it's not broken, don't fix it," "let nature take its course," "nature knows best," and other statements of this general theme were heard. The basic conclusion being, if a species doesn't warrant population-oriented management because it is either exploited, a nuisance, or in danger of extinction, it doesn't need management. There is a small element of truth in these attitudes, but they reveal a naive misconception of the deteriorating condition of most wildlife communities. The world has been transformed from a condition in which most of its area was in a primarily natural state (undisturbed by human activities) to one in which disturbed areas are steadily expanding and reducing relatively natural areas to small, disjunct, isolated remnants. The result has been a situation in which many wildlife communities will not be able to maintain themselves in a natural or seminatural state without conscious intervention by wildlife managers to preserve them.

To preserve the wildlife community of an area in a natural or seminatural state, wildlife managers have had to understand the principles of community ecology and how they can help in the planning of management strategies. Clearly, it is beyond the means of the wildlife management field to manage each member of a wildlife community on a population-by-population basis. Rather, actions are needed that will affect many species simultaneously and that will have the tendency to preserve rather than disrupt natural patterns of species composition, species richness, and relative abundance.

Wildlife managers have long known that activities undertaken on behalf of certain managed populations had effects on the entire wildlife community. Perhaps one of the most fundamental principles of community ecology that has been with the wildlife management field for a long time deals with the effects of succession:

> "The species composition of the wildlife community on an area varies with the successional stage of the ecosystem that occurs on the area."

Another principle of community ecology that wildlife managers have long been aware of and have occasionally applied in the management of certain populations states that:

> "Changing the abundance of one or more species in a wildlife community will result in changes in the relative abundance of the other members of the community."

The spin-off effects of early predator control or elimination efforts on the prey and competitors of the managed predator are well known in the profession, as are the consequences of inadvertent human activities that alter natural limiting factors and allow particular species to expand their numbers in a community.

In a similar way, the consequences of introducing a new and exotic species into a natural community have been well known to wildlife managers. The underlying principle is a basic one of community ecology:

> "The successful addition of a new species into a wildlife community will change the pattern of relative abundance and often the species composition of the natural members of the community."

Although these and other principles that have been involved in wildlife management for some time have played a relatively minor part in the management of game populations, they now play a major role in the management of wildlife communities. The introduction of an exotic species into a natural community is viewed as anathema by many wildlife managers (Laycock 1966, Labisky et al. 1975), although a few still naively believe that the benefits of introducing new exotic species somehow override the inevitable consequences for the recipient natural community.

In addition to these and other well-understood principles, a new generation of ecological principles is having a major influence on how wildlife managers approach problems of managing natural communities. Perhaps the source of the most far-reaching principles has been the study of island zoogeography (MacArthur and Wilson 1967). These principles, which apply to the distributional ecology of animals living on islands, have broad implications for species and entire communities that are forced to occupy small, isolated patches of once extensive ecosystems that have been fragmented by human development. Ecologically, many of the consequences of living in an artificially fragmented ecosystem are similar or identical to the consequences of living on islands (Diamond 1975, Harris 1984). This leads to the application of these principles to the problems of managing wildlife communities in a world in which natural ecosystems are increasingly fragmented by the development activities of an expanding human society (Temple and Wilcox 1986).

Perhaps the most fundamental principle that has emerged from the study of island zoogeography is the species-area relationship:

> "The number of species in a wildlife community is affected by the area of the ecosystem in which the community exists."

This has lead to the concept of a minimum critical size for ecosystems and an underlying ecological principle that is important in managing of wildlife communities:

> "Every ecosystem has a minimum critical size (area) below which it

> will not be capable of supporting all of the species that are typical of the ecosystem's natural wildlife community."

Wildlife species that typically exist in low densities because they have large spatial requirements or other ecological constraints are the species most likely to be absent from the wildlife community in an ecosystem below the critical minimum size. Large animals, predators, and ecological specialists are the groups that have proven to be most difficult to maintain in small fragments.

Although the area of an ecosystem has a major effect on the wildlife community, area is not the only characteristic of a fragmented ecosystem that has an effect. Additional aspects of the landscape's geometry have an influence as well. In a fragmented ecosystem, the degree of isolation between fragments and the shape of fragments also affect the species that are typical of the community in an unfragmented ecosystem:

> "The species composition of the wildlife community in a fragmented ecosystem is affected by how isolated the ecosystem fragments are from each other."

Wildlife populations that have poor dispersal abilities and hence have problems moving between ecosystem fragments easily may be isolated from other fragments. Each isolated population becomes closed, with little or no gene flow among the isolated populations. These isolated and now small populations are at great risk of local extirpation which will not be followed by recolonization if distances among patches of habitat are too great for dispersal to occur. Small animals are often especially disadvantaged by isolation.

In addition to area and isolation, the shape of an ecosystem fragment affects the species composition of the wildlife community that exists on the fragment. Fragments that are elongated in shape or have an indented border are less likely to contain as many species typical of the community in an unfragmented ecosystem than a fragment which is compact in shape and has an entire border:

> "The shape of an ecosystem fragment affects the species composition of the wildlife community that exists in the fragment."

The implications of this principle are important for certain members of the wildlife community that are typical of an unfragmented ecosystem. These are the species that are affected negatively by ecological edges—including a component of the wildlife community that lives and thrives in interior habitats, far from edges.

These ecological principles derived from studies of island zoogeography have had their most direct and important implications for managers in the design of nature reserves or wildlife refuges intended to preserve a representative or typical example of the natural wildlife community that exists in an ecosystem (Diamond 1975). However, even when managers are not directly involved in efforts to preserve examples of an intact wildlife community, they should be aware of the consequences of activities that tend to fragment natural ecosystems. The unrelenting fragmentation of the world's ecosystems is likely to have serious consequences for predictable segments of wildlife communities (Temple and Wilcox 1986). An "ounce of prevention" by managing landscapes to reduce the negative impacts of fragmentation certainly will be worth a "pound of cure" in trying to restore populations that have declined and become rare or endangered because they cannot survive in a world of fragmented ecosystems.

CONFLICTING MANAGEMENT ACTIVITIES

As the scope of wildlife management has expanded to include an increasingly diverse assemblage of species, it seems almost inevitable that conflicts in management activities should emerge. During the period in which game management was paramount, managers were dealing with a small number of species, many of which responded positively to the same types of management activities. As the realm of wildlife management has encompassed more and more types of species, the chances grow for conflicts in which management objectives for one type of wildlife are incompatible with those for another.

Because of the traditional emphasis on management of game populations, there is occasionally an assumption among some wildlife managers that these programs should somehow take precedence over programs for other types of wildlife. There also has been some degree of self-delusion in the face of conflicts. The attitude revealed in the philosophy that "what's good for game species is good for all wildlife," is demonstrably naive and incorrect. One of the emerging conflicts of this type centers on the impact of edge habitat on wildlife species (Ambuel and Temple 1983, Brittingham and Temple 1983, Temple 1986).

It is well established that many of our most popular game animals respond positively to ecological edges; it is also clear that many nongame species are affected negatively by edge. Herein lies the root of the conflict. Intentional ecosystem modifications undertaken by game managers, with the specific goal of creating additional ecological edges in an area, are likely to have a negative impact on a segment of the local wildlife community that is already suffering population declines because of the widespread creation of ecological edges as an unintentional consequence of development activities.

Such conflicts will become increasingly common as the natural areas that all wildlife species require steadily become reduced in size and must, therefore, accomodate multiple management objectives. How the wildlife profession responds to these challenges will reveal much about how strong the ecological basis of our field actually is in the face of social, economic, and political pressures that are imposed on management institutions. Will the conflicts be resolved on ecological grounds within the profession or will other arenas outside the profession dictate our actions?

CONCLUSIONS

Wildlife management is a growing field in terms of the ecological scope of its responsibilities, if not in terms of the financial support it receives for its work. A consequence of this growth in the field's scope has been the steady accumulation of new ecological principles that provide the basis for solving newly encountered management challenges. As these new ecological principles are incorporated into the foundations of the field, wildlife managers must keep abreast. They also must try to avoid the very human tendency to resist change.

As the field has expanded to include species other than game animals, there has been an unfortunate precedent of segregating the field into game and nongame camps. There is little basis in either relevant ecological principles or fundamental objectives to support this dichotomy. Rather, a division based on the population (single-species) or community (multi-species) orientation of management seems more logical.

Game management would become one segment of population-oriented management. Community-oriented management will inevitably become more and more important as the pressures on remaining natural areas increase. This should not be viewed as a liability, but as an opportunity for the field to take on a more influential role in preserving the earth's natural diversity. A steadfast adherence to ecological principles that have guided our profession throughout its existence will insure a measure of success that cannot be realized if other factors become dominant in the decision-making processes.

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Fiscal Constraints to Nongame Management Programs

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Abstract. Funds for state nongame management programs declined significantly in recent years. This is somewhat surprising since the total amount of money from state nongame income tax checkoffs nationally increased from $338 thousand in 1978 to $8.6 million in 1984. Despite this increase, there was a decrease in nongame funds of approximately 30% in constant dollars from 1980 to 1984 ($14.3 million to $9.9 million). The decline appeared to be related to a tendency for states to reduce the amount of support from general revenue with the advent of a checkoff, to competition from other income-tax checkoffs, to a reduction in federal grants for endangered species, and to poor economic conditions in the early 1980s. State agencies need to adopt innovative funding approaches for nongame programs and to develop aggressive public education programs about nongame to elicit support in their legislatures. Given the estimated 93.2 million nonconsumptive wildlife users in the United States, additional funding opportunities do exist if proper "user-pay" revenues are developed.

The status of many nongame species has declined in recent years due in part to inadequate funding for programs designed to protect them. However, given that the 1980 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Fish and Wildlife Service and Bureau of the Census 1982) estimated that there were approximately 93 million nonconsumptive wildlife users in the United States who spent over $14 billion pursuing this activity, there should be multiple opportunities to fund nongame programs. Twenty-eight million of the nonconsumptive participants spent over $4 billion while taking trips specifically to observe, feed or photograph wildlife. The 1980 National Survey estimated that there were 17.4 million hunters and 42 million fishermen aged 16 years and older, in contrast to 93.2 million nonconsumptive wildlife users (including 35.1 million dual users who participated in both consumptive and nonconsumptive activities).

Because wildlife management programs have been funded traditionally through fees and taxes imposed an anglers and hunters, game management programs have been emphasized, with negative implications for some nongame species. To compensate for this situation, some states, such as Missouri and

Minnesota, have been leaders in establishing nongame programs and developing innovative funding mechanisms. Some states have adopted income-tax checkoff programs with varying results, whereas other states have attempted other innovative approaches, including special sales taxes, nongame endowment funds, conservation stamps, and sales of merchandise such as posters and T-shirts.

NONGAME FUNDING PROBLEMS

The lack of a single constituency directly related to the use of nongame continues to complicate efforts directed toward funding nongame management. Nongame fish and wildlife make up roughly 90% of the vertebrate species under the jurisdiction of fish and wildlife agencies in the United States—an average of 748 species per state (Nongame Wildlife Committee 1977). However, the remaining 10% of the species (an average of about 125 species per state) receives the most management attention from state agencies because they are harvested by fishermen and hunters. This use pattern has influenced funding mechanisms for the management of wildlife.

The visibility of bagged game or fishing and hunting equipment has resulted in a "user-pay" philosophy for sport fishing and hunting, in contrast to nonconsumptive use of nongame species. Anglers and hunters pay fees for the privilege of pursuing their activities in the form of licenses, permit fees, and excise taxes on fishing and hunting equipment. Federal excise taxes are collected specifically for fish and wildlife restoration under the Federal Aid in Wildlife Restoration Act of 1937 (Pittman-Robertson) and the Federal Aid in Sport Fish Restoration Act of 1950 (Dingell-Johnson). Since their inception, the Pittman-Robertson program has allocated $1.446 billion to states for wildlife restoration (Fiscal Year 1985 - $79.1 million), and the Dingell-Johnson program has allocated $467 million (Fiscal Year 1985 - $35.1 million) for sport-fish restoration. Since nongame species usually are not harvested, it is difficult to assess the degree of actual use for the purpose of assigning appropriate taxing mechanisms, and there continues to be a dearth of funds for nongame management.

Another problem is that the exact amounts needed by the states to meet the priority needs of nongame are not known. The Fish and Wildlife Conservation Act of 1980 (P.L. 96-366) specified a need for comprehensive management plans that will help states to develop priorities, but as of December 1985, the U.S. Congress had not appropriated funds to implement the Act. When such plans are completed, the total nongame management needs that will be identified are expected to approach the amounts currently expended on game species.

An analysis of potential funding sources was made by the Wildlife Management Institute in 1974 in preparation for discussion of the various nongame bills under consideration at that time in the U.S. Congress. This analysis showed that the amount of new funds needed annually to achieve continuing program obligations and to undertake new programs primarily related to nongame and nonconsumptive use of all fish and wildlife would be considered low if nationally they fell between $25 and $125 million, and high if they fell between $250 and $1,250 million (Wildlife Management Institute 1975).

Testimony before the U.S. Senate's Environment and Public Works Committee estimated that the amount necessary for the states to fund nongame programs at an adequate level would be about $18.4 million per year for the period 1978-1983 or $92 million in total (Odom 1981). Current trends seem to indicate that because of inflation, this figure now may be about $30 million per year and may reach $50 million annually by the year 2000. A more definitive study on current state nongame funding needs would certainly facilitate future Congressional deliberations on funding alternatives for nongame management programs.

NONGAME FUNDING TRENDS OVER TIME

In 1974, nongame programs constituted only 2 percent of state and 11 percent of federal wildlife management and research spending (Wildlife Management Institute 1975). A report of the U.S. Fish and Wildlife Service and the International Association of Fish and Wildlife Agencies (1983) summarizing fish and wildlife characteristics of the 50 States indicated that in 1980, the states' collective expenditures for harvested species totalled approximately $443.5 million, whereas expenditures for nongame were only about $14.3 million—roughly 3% of the amount spent on game species. It appeared that nongame funding increased about 1% between 1974 and 1980, but given inflation, the difference actually represented a decrease in funding.

In 1984, the Southeastern Assocation of Fish and Wildlife Agencies initiated a survey of nongame and endangered species management in all state fish and wildlife agencies. Survey results showed that total state funds budgeted for nongame and endangered species programs in 1984 were $13 million (B.C. Thompson, pers. commun.). In spite of the differences in accounting systems and fiscal reporting periods that complicate precise determination of expenditures, this still would seem to represent a further decline in funding for nongame programs. When these amounts are adjusted for inflation, there was an apparent decline in buying power of roughly 30% in funding in the brief period from 1980 to 1984. The adjusted figures showed an increase from $5.6 million in 1974 to $14.3 million in 1980, but a decline to $9.9 million in 1984.

This decline in nongame funding is somewhat difficult to interpret. One might assume that with the growth in tax-checkoff programs, the amount of funding for nongame would have risen. An examination of where funds are derived for nongame programs may provide some insight as to the origin of this problem.

Since the concept of nongame often includes endangered species, some states tend to rely on Endangered Species Act Section 6 grants for restoration of those species. The federal funds for this program were eliminated totally for Fiscal Year (FY) 1982, and reinstituted in FY 1983 and 1984 at reduced levels ($2 million in FY 1983 and FY 1984 *vs.* $3.7 million in FY 1981). Although this reduction was partially reversed, with $3.9 million appropriated for both FY 1985 and 1986, those states that began to rely on federal endangered species grants experienced substantial cutbacks in the 1980s.

Another partial explanation for the downturn in funds for nongame may be a tendency of state governments to cut back the funding for nongame from general tax revenues once an income tax checkoff has been approved for nongame. Thus nongame funding would tend to fluctuate with the state of the economy. The downturn in the economy during the early 1980s probably had a negative impact on generating revenue for nongame programs. Midwestern states heavily dependent upon agriculture, as well as those areas more heavily reliant on industry, were substantially affected by this downward trend in the economy, and this may be reflected in the declines in state nongame program funding.

ALTERNATIVE FUNDING MECHANISMS

Since hunters and fishermen have provided most of the money for wildlife management, game species have been emphasized more than nongame species. Zwank et al. (1980) found that a lack of funds was the major problem facing nongame programs, that 65% of the states would expand their nongame programs if additional funding was provided, and that 22% felt that state programs should seek legislation to assure funding. The need for additional funds for nongame programs was identified clearly as the crux of the problem.

There are several potential sources of funds for state nongame programs. For example, states have used Endangered Species Act Section 6 matching grants (up to 90% federal - 10% state), sales taxes, sales of stamps and other items, sales of personalized auto tags, Pittman-Robertson wildlife restoration funds (for birds and mammals only), general appropriations, voluntary income-tax checkoffs, and taxes on uses, alcohol, tobacco, real-estate transfers, soft drinks, and severances (Nongame Wildlife Association 1981).

The U.S. Fish and Wildlife Service and International Association of Fish and Wildlife Agencies (1983) showed that state nongame programs were supported fairly evenly by federal and state sources in 1980. Fifteen of 29 reporting agencies indicated that most funding for their nongame programs came from state sources and 14 indicated that most was derived from federal sources. When expenditures for all game and nongame programs are added together, most funding came from federal sources for only 15% of the states.

The principal source of nongame funding in 1980 was general tax revenues, but the recent growth in the number of income tax checkoffs may have altered that pattern. Thirty-eight percent of the states reported receiving most of their funding for nongame programs from general tax revenues, 24% from income tax checkoffs, 10% from voluntary purchase of a conservation stamp or other items, and 29% from other sources, such as license plates or nongame endowment funds.

The actual funding pattern differs from the preferred new sources of funding for nongame activities described in the 1980 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (Mangun and Shaw 1984). The means of raising funds most preferred by survey respondents was through the sale of conservation stamps (37%). Tax checkoffs were second (24%), and general tax monies were close behind (20%). This could be interpreted to mean that nonconsumptive wildlife users tend to prefer voluntary ways of supporting their programs as opposed to specific taxes on their activities or the use of general tax funds. In fact, when asked about their level of support for different potential funding mechanisms, 42% of the nonconsumptive enthusiasts specifically opposed a special tax on items purchased for wildlife observation or other uses.

NONGAME INCOME TAX CHECKOFFS

The Nongame Wildlife Association of North America (1985) reported that total revenue for the 31 states with tax-checkoff programs in 1984 was $8.6 million (Table 1). The states with the largest incomes were New York ($1.7 million), Minnesota ($643,500) and California ($511,000). Revenue varied from a minimum of $27,600 (Arkansas) to the maximum in New York, with an average of $298,700 per state per year. The greatest average donations per contributor were made in Delaware ($8.98), New Mexico ($8.32), and North Carolina ($8.06), compared to the national average donation of $5.54. Despite the tremendous difference in populations, revenues in Minnesota exceeded those in California because the former had the highest participation level in the country—11.2% of eligible taxpayers contributed. Participation rates ranged from the high in Minnesota to a low of 0.5% in Arkansas.

The $8.6 million total appears to represent a significant increase when compared with the total of $6.5 million raised nationally in 1983, but with the addition of checkoffs during 1984 in heavily populated states like California, Illinois, and Ohio, as much as $10.4 million had been anticipated from the checkoffs (Nongame Wildlife Association 1983). The discrepancy between expectations and reality is related to the fact that there were declines in total contributions in 11 of the 18 states that had reported revenues in the previous year.

Primary among possible explanations for the decline in revenue from income-tax checkoffs has been the sudden growth of competing checkoffs in several states. Colorado, for example, experienced a major reduction in funding. The nongame checkoff produced $124,000 (22%) less in 1984 than it did the previous year (Table 1). Colorado was the first state to have a nongame checkoff and had remained the number-one revenue generator of funds until New York, a far more populous state, established a checkoff and began receiving funds in 1983. The decline in funding in Colorado was directly related to the addition of competing checkoffs on the state income tax form.

The experience in Colorado has definite implications for future funding by income-tax checkoffs across the country. Success attracts followers. By 1984, 6 of the 11 states that added nongame checkoffs in 1983 already had more than one checkoff (Nongame Wildlife Association 1985). Harpman and Reuler (1985)

Table 1. Revenue from state income-tax checkoffs.

State	1st Tax Year[a]	Revenue in Thousands of $$			
		1984[a]	1983[b]	1982[b]	1981[b]
Alabama	1982	55.8	74.5		
Arizona	1982	245.2	300.1		
Arkansas	1983	27.6			
California	1983	511.0			
Colorado[c]	1977	447.7	571.7	721.0	746.5
Delaware	1983	84.8			
Idaho	1981	88.6	90.3	106.2	
Illinois	1983	260.3			
Indiana	1982	251.3			
Iowa	1982	220.0	238.6		
Kansas	1980	135.9	131.6	137.5	128.8
Kentucky	1980	90.4	85.9	80.5	85.0
Maine	1983	111.0			
Massachusetts	1983	381.3			
Michigan	1983	267.9			
Minnesota	1980	643.5	490.1	533.0	476.6
Montana	1983	35.4			
New Jersey	1981	465.8	466.0	410.0	
New Mexico	1981	255.1	261.5	265.6	
New York	1982	1701.1	1748.4		
North Carolina	1983	228.9			
Ohio	1983	488.2			
Oklahoma	1982	175.6	204.5		
Oregon[d]	1979	234.0	220.1	272.2	360.0
Pennsylvania	1982	395.8	221.6		
South Carolina	1981	158.9	146.7	100.2	
Utah	1980	164.2	172.0	208.0	217.0
Virginia	1981	446.2	394.5	369.8	
West Virginia	1981	98.5	130.0	164.6	
Wisconsin	1983	291.7			

[a]Source: Nongame Wildlife Association of North America (1985). In addition to listed states, a checkoff in Georgia will become effective in the 1989 tax year, and in Mississippi in the 1985 tax year.

[b]Source: Harpman (1984). No revenues cited for Nebraska, where 1984 was the first tax year for a checkoff.

[c]Pre-1981 Colorado revenues were $665,000 in 1980, $511,400 in 1979, and $338,200 in 1978.

[d]Pre-1981 Oregon revenue was $347,000 in 1980.

determined that the presence of one additional checkoff on an income-tax form would result in a 70% decline in the average donation to a nongame checkoff. Based on their analysis of data through 1982, Applegate and Trout (1984) found that the presence of a competing checkoff reduced nongame revenue by about $100,000 annually per state.

Obviously, there are certain dangers associated with relying on nongame checkoffs as a source of revenue for state programs. Although they may currently constitute the best funding opportunity, checkoffs do not necessarily represent a stable form of income in the long run. The success of nongame-checkoff systems is heavily dependent upon the health of the economy. There is potential competition from other deserving causes, and most checkoffs are subject to "sunshine laws" where their continuing worth and importance is periodically reviewed. Furthermore, there are always well-intentioned bureaucrats who are interested in simplifying forms and who may propose the elimination of such checkoffs on income-tax forms. This suggestion has been made at least once by a task force in Minnesota (Nongame Wildlife Association 1982).

Applegate and Trout (1984) determined that several measures are important in explaining variations in checkoff revenues, including the number of people in the state and the extent of involvement of a state's citizens with the out-of-doors. But the most important factor in predicting revenue was the number of refund tax returns filed. A person who was not eligible for a refund was not likely to make a contribution. Other than the amount of money spent on radio promotions, which were positively related to checkoff revenue, Applegate and Trout concluded that most variation in checkoff revenues appeared to be outside the control of state fish and wildlife agencies.

Some states already have initiated means to combat the decline in revenues from nongame checkoffs. To counter the tendency for checkoffs to proliferate, Pennsylvania state legislators, for example, inserted language into their nongame-checkoff law that prohibits the placement of additional checkoffs on their state income-tax forms (Boggis and Hamilton 1984). Boggis and Hamilton also indicated that the key to sustaining nongame checkoff programs may be through active public relations programs that fund high visibility projects, devote considerable time and effort to public education, and provide training for public information specialists in fund-raising skills. Clearly, a good public relations program is essential for effective fund raising. This was demonstrated by the extensive public relations efforts of the Minnesota nongame wildlife staff which was extremely effective in eliciting a high level of support from Minnesota residents.

FUTURE FUNDING POSIBILITIES

To maintain existing nongame programs and to initiate new programs for increasing needs, state agencies will have to find alternative sources of revenue. With effective public education, agencies may be able to elicit more support from state legislators to get more funds from general tax revenues. Lessons could be learned from the efforts of the proponents of the Endangered Species Act at the national level. They expended a tremendous amount of time and energy educating Congressmen and their constituents about the importance of endangered species and their potential linkage to the fate of mankind. Perhaps the declines in habitats for nongame species could be related to growing threats to mankind caused by declining precipitation, shortages of food and firewood, and growing shortages of adequate housing worldwide. The rights of animals are also under considerable discussion. The debates in Congress over the Fish and Wildlife Conservation Act that came into law in 1980 included similar thoughts.

Although no money was appropriated to implement the Act, Congress directed the U.S. Fish and Wildlife Service to conduct a comprehensive study to determine the most equitable and effective mechanism for nongame funding. Also, even though the Act is often referred to as the "Nongame Act," its purpose is to fund development of plans for the conservation of all types of fish and wildlife and not nongame species alone.

A study by the U.S. Fish and Wildlife Service (1984) produced detailed evaluations of the potential of 18 sources of revenue for state nongame programs based primarily, though not exclusively, on excise taxes on equipment used by nonconsumptive wildlife users (Table 2). This approach is similar to the Pittman-Robertson and Dingell-Johnson excise taxes on hunting and fishing equipment. Findings included annual funding potentials through the year 2000, expressed in 1980 dollars and based on varying levels of excise taxes, and, for selected funding sources, an apparent loss in economic efficiency, plus information on the economic ability of taxpayers to pay. Detailed information about specific benefits received from the use of each source are itemized throughout the report.

The State of North Carolina already is moving in the direction of new funding sources by exploring the possibility of a sales tax on sporting equipment to be used for fish and wildlife conservation. Other state legislatures could consider similar taxes or fees in association with several of the potential funding sources evaluated by the U.S. Fish and Wildlife Service. Such taxes would tend to provide more stable sources of income than income-tax checkoffs. Missouri's innovative 0.125% sales tax for the management of both game and nongame species is a stable source of revenue despite its slight regressivity (i.e., that it is not proportional with income). As more states evaluate potential funding sources, more innovative ideas surely will be developed.

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Table 2. Potential funding sources to implement the Fish and Wildlife Conservation Act of 1980 (U.S. Fish and Wildlife Service 1984).

	Funding Potential (Millions of 1980 $$)	
Potential sources	**1980**	**2000**
A. Annual general fund appropriations	variable and uncertain	
B. 5-10% excise tax on wild-bird seed	3.8-7.3	4.5-8.6
C. 5-10% excise tax on wild-bird houses	0.5-0.9	0.5-1.1
D. 5-10% excise tax on wild-bird feeders	1.3-2.5	1.5-3.0
E. 5-10% excise tax on wild-bird waterers, baths, and heaters	0.6-1.2	0.7-1.4
F. 5-10% excise tax on wild-animal furs	11.2-21.4	57.0-108.5
G. 5-10% excise tax on backpacking and camping equipment	14.3-28.1	25.2-42.1
H. 2-5% excise tax on off-road vehicles	76.8-147.3	2,334.0-5,460.0
Snowmobiles	3.5-8.0	Negligible
Off-road motorcycles	5.9-14.6	53.6-128.8
Other all-terrain vehicles	0.8-1.5	0.8-1.8
Four-wheel drive vehicles	66.6-123.2	2,280.0-5,329.0
I. 5-10% excise tax on binoculars, monoculars, and spotting scopes	2.3-4.6	5.4-10.8
J. 5-10% excise tax on wildlife identification books	0.5-1.0	2.0-4.0
K. Fees of $0.50-$2.00 for use of selected Federal lands and waters	103.1	123.6
Fish and Wildlife Service	12.2	14.6
National Park Service	15.0	18.0
Forest Service	50.1	60.0
Army Corps of Engineers	7.5	9.0
Bureau of Reclamation	18.3	22.0
L. Voluntary contribution by federal income tax checkoff	40.0	54.5
M. Sale of semipostal stamps with contribution of 25-50% of postage value	11.3-203.4	13.4-240.3
N. 5-10% excise tax on recreational diving equipment	1.7-3.1	Negligible
O. 1-5% excise tax on certain photographic equipment and film	25.2-124.0	102.8-512.3
P. 1-5% tax on certain locatable minerals extracted from the public domain	0-120.0	0-141.2
$10-$25 claim renewal fee	12.1-30.2	12.1-30.2
Q. 1-5% excise tax on travel trailers and campers	5.3-23.0	19.1-90.5
R. 1-5% excise tax on motorhomes	4.5-19.5	8.6-37.6

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Long-Term Trade-Offs in Wildlife Management

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Abstract: With the growing emphasis on non-game wildlife has come a great increase in both number and types of species that may need to be managed. At some point, if it has not already arrived, wildlife managers may ask the question, "Given limits on time, money, and space, can we afford to save a particular species, and if so, are we doing it at the expense of other species?" Among the arguments that will have to be considered in answering that question is the endangered species approach, including its potential taxonomic biases and asking whether waiting for a species to make an endangered list is not waiting too long. The idea of "triage" as a compromise approach is presented. Here species are put into one of 3 categories depending upon whether or not the species is safe even without management, needs management for existence but can be saved, or is beyond saving by management techniques and should be left to fend for itself. Problems with such an approach vary from public relations to the decision-making process. Among problems associated with the latter is the lack of knowledge on many non-game species. Island biogeographic theory is an example both of the breakthroughs needed to manage some non-game wildlife and the kinds of controversies we face in deciding what to do. Compromises will be inevitable in future wildlife management.

In recent years, at least two major changes have greatly affected the field of wildlife management. The first of these reflects the generally great increase in the number of organisms actively monitored and managed. The addition of so-called nongame species is the most obvious change, although the game-nongame dichotomy has caused controversy of its own (Brocke 1979).

Papers published in the Journal of Wildlife Management reflect this change. Only about 15% of the major papers published there dealt with nongame species from 1960 to 1975, but in the 1980s their number hovered around 30%. The focus of these nongame papers has also shifted from control of pest species to modern papers that really focus on nongame management.

This expansion in the number of species potentially under management has been reflected in governmental organization. In some states, game management agencies have remained as such, with nongame management organized in other state agencies (departments of natural resources, etc.). In other states, the former game management agency has broadened its role to wildlife management, including (at least theoretically) all species of plants and animals. Although in many states it may be true that one agency or another always has had a technical authority to protect and preserve all species, only in recent years have active programs been developed to manage and protect a wide array of non-harvestable species.

With this increase in the number of plants and animals that concern managers has come an increasing sophistication in our understanding of the wide array of factors that may affect the population dynamics of plant and animal species. Many findings of so-called "pure" ecologists have management implications. Among these are:

1. Applied concepts of island biogeography (Diamond 1975; Wilson and Willis 1975) that tell us species-specific requirements in habitat quantity in addition to habitat quality.
2. Studies on plant-pollinator interactions (Howell 1976) that have shown how the demise of a pollinating species can affect reproductive success enough to threaten a plant species.
3. Increasing knowledge of fruit dispersal mutualisms that has shown how the absence of natural fruit and seed dispersers affects reproductive success in plants (McKey 1975).
4. Observations of community structure patterns (Faaborg 1979) that have shown how subtle interactions among species may affect survival independent of habitat parameters.

Many of these studies have shown that the interrelationships among organisms are often subtle, yet a knowledge of these relationships is critical to proper management. We are well past the days when a little habitat manipulation was all that was needed to increase populations of a handful of harvestable species.

Early game management efforts focused on those species that could be managed easily or were somewhat pre-adapted to local agricultural and habitat conditions. Most of these were on the r-selected end of the r and K gradient (MacArthur and Wilson 1967). With the broader mandate of recent times, we must manage for both r-selected and K-selected species.

These changes often make modern management more complex. The addition of many more species and new types of species to management plans makes it inevitable that new ecological factors enter into final management decisions. With a vast array of plants and animals under consideration, it also will become inevitable that management decisions will have to be made that pit one species against another, for one management plan cannot be beneficial for everything in a single location (Faaborg 1980, Noss 1983).

At some point, if it has not already been reached, wildlife managers may have to face the realization that they cannot save it all, at least on a local scale if not on a regional scale. At this point, some hard decisions will have to be made about what can and cannot be saved, and at some point we undoubtedly will have to ask the question, "Given limits on time, money, and space, can we afford to save a particular species, and, if so, are we doing it at the expense of other species?" Some have suggested we are already at this stage with the California condor (*Gymnogyps californianus*) restoration program (Pitelka 1981). Certainly such choices are being presented in tropical situations (Lovejoy 1983) and sooner or later will have to be made most everywhere. Many factors will have to be weighed in making the decisions about what can and cannot be managed to avoid extinction on at least some geographic scale.

THE FOCUS ON ENDANGERED SPECIES

To date, most management practices can be divided into two categories, those that attempt to maximize the production of harvestable species and those that attempt to avoid the extinction of species whose populations are threatened. With the addition of so many new forms to management controls, a vast effort has been made by agencies such as The Nature Conservancy (usually working with state agencies) to ascertain which species have populations that are threatened. I think it is safe to say, at least in the Temperate Zone, that most forms of life are not game species or threatened and endangered species. Most species of plants and animals appear to be able to survive by themselves and have been ignored by managers. Within nongame management, is a focus on threatened and endangered species a good way to proceed?

Among the arguments in favor of a focus on endangered species are those expounding the tragedy of extinction of any form of life. We need not spend time here going through the many reasons, both practical and aesthetic, why an effort should be made to save every species (see Ehrlich and Ehrlich 1981). With such a moral imperative, it seems logical that our first step in managing wildlife species should be to ensure that none become extinct, and many noteworthy efforts to save endangered species exist. To ensure that nothing is missed, conservationists have developed lists of species in danger of extinction. In many cases, the list of threatened and endangered species for an area provides enough material to easily monopolize money available for nongame management, so the result has been the beforementioned focus on this select subgroup. It also has been argued that we know too little about which of the remaining species need to be managed to shift the focus onto less-threatened varieties.

Even if a focus on endangered species is the best approach, one must be aware of potential problems and biases involved in dealing with many diverse species. Have we been, or can we be, consistent in attempting to save all threatened species? Taxonomic biases seem readily apparent among those people concerned with preserving species (Lovejoy 1979) and even appear in our laws (Spencer 1979). Birds and mammals seem to rank higher than "lower" vertebrates, with plants somewhere among the lower vertebrates, but probably higher than invertebrates such as insects.

Within-taxon biases also seem to occur. For example, among birds, raptors and parrots seem to get a tremendous amount of attention, whereas butterflies may be favored among insects. Although some of this reflects ecological characteristics of these forms, much of it is related to the visibility and glamour of these forms compared to smaller "dickey" birds or other "bugs." Although attempts were made to save the dusky seaside sparrow (*Ammodramus maritimus nigrescens*) (Kale 1983), if it had been a hawk or parrot, efforts might have been initiated sooner and more intensively. If we should face a scenario where limited time, money, or space requires that we choose one or the other, how many people will choose a Mead's milkweed (*Asclepias meadii*) over an upland sandpiper (*Bartramia longicauda*), or an endangered moth over a threatened mammal?

Another weakness of a focus on endangered species is simply that such approaches inevitably will fail on occasion, so we must incorporate the acceptance of such failure into the decision-making process. There will be occasions where unlimited amounts of money and effort will not be enough to save a species, and continuing such an effort will dilute other efforts and also will result in negative public relations. The noted ecologist Frank Pitelka (1981) has suggested that the California condor might be such a case where there is nothing we can do to save the species, so we should put the money into programs that will have greater probability of success. While the merits of this case can be debated, a program focusing on endangered species cannot expect 100% success, so contingency plans must be developed that tell the manager when to give up on a species and, sadly, let it disappear.

The realization that we probably cannot save all species by waiting to direct efforts at saving them until they have reached threatened or endangered status suggests that such an approach is wrong. It is possible that waiting until a species has reached threatened or endangered status may be too late to save it in terms of long survival periods. Several modern cases of species preservation have dealt with populations of less than 100, such as with whooping cranes (*Grus americana*) or Puerto Rican parrots (*Amazona vittata*). Even if short-term population increases have occurred, there is no guarantee of long-term success for a variety of reasons. Recent work with minimum population sizes suggests that populations of less than 100 may result in inbreeding depression or other genetic disorders (Shaffer 1981). These may not appear for some time, but waiting until populations reach such a critical level may limit the genetic variability with which managers can work. Waiting also may limit the ability to use native gene pools in management, as occurred with the reintroduction of the peregrine falcon (*Falco peregrinus*) to the eastern United States (Barclay and Cade 1983).

In some cases, waiting to take action until populations reach some lowered level may be self-defeating because high adult survivorship may have masked long-term reduction in production of young, so that by the time adult populations decline, recovery actions will have to cope with unstable age distributions, aging populations, etc. Such long-lived, slow-reproducing creatures as parrots and whales may exhibit this problem. Also, waiting for reduced population levels to initiate activity may greatly reduce management options because of limited distribution of the species in question. Waiting until the Puerto Rican parrot was confined to the Luquillo Mountains, or the black-footed ferret (*Mustela nigripes*) was restricted to one or two white-tailed prairie dog (*Synomys leucurus*) colonies certainly limits management options.

Some of these weaknesses in an endangered-species approach can be overcome, but they require extra money and effort. The final argument against waiting until a species reaches some critical status might simply be an economic one, with the idea being that "an ounce of prevention is worth a pound of cure" and costs less in the long run.

TRIAGE AS A COMPROMISE APPROACH

If we accept the fact that we may not be able to do it all in terms of species preservation, what can we do? One option is based on the "triage" concept of medical care used in battlefield situations (Myers 1981). This system classified wounded soldiers into 3 categories: those too wounded to be helped, those not wounded enough to require immediate care, and those who would be most affected by immediate care. The idea was to save the most people by recognizing that effort would be wasted on some, whereas others could survive for a while without care.

A triage approach to nature preservation also would categorize species into three classes. Hopefully, the largest of these would be those species that did not require effort for preservation because of their compatability with present land-use practices and management schemes. The second group also would include species not actively managed because even the most intensive management program would not be successful. These species would be left to become extinct if they could not adapt to whatever habitat changes were made within their zones of distribution. This leaves the third group as the one on which we would focus our future careers, for this includes those species where active management should result in viable populations for long periods of time.

The obvious initial problem with a triage approach is deciding which species go into each category. Perhaps there are places in the United States where such a system is too severe because all species eventually could be saved. Certainly tropical systems are facing situations where triage will have to be applied today. Although we may have more options in the temperate zone, the increase in the number of species under management consideration and the pressures of development inevitably will lead to some sort of decision making about what can and cannot be done, at least on a local scale.

PROBLEMS WITH A TRIAGE APPROACH

A triage approach undoubtedly would cause many problems for the wildlife manager because of the vast amount of information necessary to make logical categorizations and the long-term implications of these decisions. Perhaps the first and largest problem concerns those species effectively condemned to extinction by this approach. How can one justify letting species go extinct? As managers, we may have made this problem more difficult by our intense campaign to preserve endangered species. After intensive efforts to preserve species like the whooping crane and stop projects like the Tellico Dam to save the snail darter (*Percina tanasi*), how can we convince the public that suddenly we know what is best and that, for example, making the California condor fend for itself is part of the solution?

In this day of animal rights activists, where the harvesting of even stable animal populations is protested, suggesting that extinction is inevitable will not be a popular decision. Conversely, if we suggest that we should spend money on species that seem at the moment not to be in danger of extinction, public relations problems also could result.

Who will decide which species are condemned to extinction and which deserve protection? Given that there are varying viewpoints on how to manage even highy studied species, what sort of panel of experts can we expect to gather to accumulate the available (and probably insufficient) information on all species for applying triage designations? How do we compensate for biases with regards to taxa or other factors, ecological points of view, etc? How do we predict which species are "threatened but saveable" before they actually become too rare to save?

Past history suggests that trophic characteristics (raptors), size (large animals always seem in more trouble), pesticide susceptibility (through egg shell thinning), insularity, and habitat specialization are good factors to consider when looking at a species' susceptibility to extinction (King 1977). Recent work has identified "area sensitive" species that require large nature reserves to maintain viable populations (Robbins 1979), although we understand few of the mechanisms behind such sensitivity.

For some organisms, especially birds, established censuses (e.g., Breeding Bird Surveys and Audubon Counts for birds) give us clues to population shifts as they occur, but such basic data are lacking for most organisms. How do we know a species is declining if we lack such data? Interactions among species seem to be receiving increased emphasis in understanding the population dynamics of some species, but others are arguing that interactions such as competition are of little importance (Simberloff 1978, Wiens 1977). When ecologists cannot agree on the importance of such potentially important factors as these, how can wildlife managers make the proper, intelligent decisions about what should and should not be managed, let alone how a specific species should be managed?

Another problem with a triage approach (although not confined to such an approach) involves the fact that plant and animal species distributions do not match political boundaries. Yet these political boundaries are often critical in delimiting the missions of state agencies. State endangered species lists long have been one of my pet peeves, because I feel they, too, often accentuate the negative status of species on the margins of their ranges. Depending on the criteria used in categorizing species, a triage approach might either enlarge this problem or minimize it.

If species that are really extra-limital are put in the "no-effort" category, a more efficient system might emerge. It will be hard for a state ornithologist or mammalogist to encourage policies that reduce the diversity of a particular taxonomic group, let alone diversity within the state. Cooperation among states in these cases might help alleviate this problem.

Associated with regional biases among managers are the constraints on where they can do work to protect the species involved. A Missouri wildlife manager is expected to work in Missouri although recent findings on many of our migratory birds suggest that we already may be doing all we can for them on the breeding grounds (Terborgh 1980). How will the governor look at a budget that includes several months work on a warbler's wintering grounds in Jamaica, even if all biologists agree it is necessary work? Can state agencies legally put funds into projects in other states or even foreign countries? Private foundations such as The Nature Conservancy and World Wildlife Fund have recognized how essential it is for some species to maintain adequate habitat on the wintering grounds or along migration corridors. Can governmental agencies do their share to help with these activities?

Biogeographic considerations exemplify the sorts of dilemmas wildlife managers will face in the future. First of all, the verity of applied biogeography concepts are being hotly debated among ecologists. Simberloff and Abele (1976) and Whitcomb et al. (1976) started the argument. One group suggests that bigger reserves are better because they protect more species from extinction, whereas a smaller group of scientists suggests that we really have no evidence supporting this idea and that several small reserves are as good as a big reserve and are easier to purchase. Each side includes bright scientists, so it is not a simple problem to deal with, and the potential exists to make some big mistakes in management guidelines.

I strongly support the view that large areas will protect species that small ones will not (Faaborg 1979), although small areas have their advantages, too. I also see advantages in putting as much of the available resource as possible into large reserves simply because I think that is the safest bet. If those of us who think bigger is better are correct, large reserves will have done the job. If we follow the several-small-reserve argument and are wrong, it may be too late to remedy the situation. The World Wildlife Fund is presently doing an elegant experiment on this phenomenon in the rainforest of Brazil with the hope that this dilemma will at least partly be resolved.

Another aspect of applied biogeography that presents problems to wildlife managers is a rather subtle factor, yet one that cannot be ignored. If you are managing an area for both bobwhites (*Colinus virginianus*) which like edge, and ovenbirds (*Seiurus aurocapillus*) which like forest interiors, you might be forced to deal with habitat units of solid forest of 100 ha (250A.) to keep the ovenbirds, even though a single pair of ovenbirds uses only a small fraction of this area. It is not an easy task to convince a quail hunter that the ovenbirds really need such a large area, particularly when ecologists do not understand why that is so. Yet, cutting the management plan into blocks of 40 ha (100A.) may be enough to cause local extinction of ovenbirds. The population dynamics of area-sensitive species are difficult to understand, but managing without recognition of area constraints could lead to local or regional extinctions.

An associated fact is that many island-related phenomena occur over long periods of time. Short-term management with 40 ha-blocks, as described above, might not result in the immediate exclusion of ovenbirds, but over longer periods of time this could be the case. How do we convince the public that trade-offs that will occur in decades or even centuries are worth giving up something now?

Island biogeography presents guidelines that exemplify clearly the sorts of conflicts we will encounter in trying to manage for all types of species. It is difficult to manage for both quail and ovenbirds, but if all game and nongame species found in a typical woodland are included, we can see the vast complexity of the system involved. Although we do not clearly understand the mechanisms involved, we can make some guidelines now and continue work that will allow us to better understand the functioning of the whole system in the future. It seems certain that we will have to make trade-offs in deciding what we can and cannot do with a specific area; here, the concept of landscape diversity management enters in so that maximal species preservation is achieved on a regional basis (Noss 1983).

THE INEVITABILITY OF COMPROMISES

Many managers may be in situations where their management area or district, or perhaps even state, may not have to face the hard decisions I have discussed above. They may have large enough budgets and small enough endangered species lists to accomplish all that they want in today's world. They may be blessed with migratory species that can adapt to changing conditions in the tropics or along their migratory pathways. Sooner or later, however, with increasing human populations and the habitat destruction or modifications that accompany them, most of us or our children will face situations like those outlined above. Perhaps our best hope is that, through meetings like this symposium and continued research and discussion, we will be prepared for such conditions when they occur and will attack them with a long-term strategy for species survival such that we can, in fact, save the species that we have selected. This has to be a better approach than jumping from endangered species to endangered species as circumstances demand.

Although we may sit smugly here in our temperate habitats and feel that these problems are far away, we need only look at the tropical forests of the world or the savannahs of Africa to see that such problems are occurring today. Perhaps we can feel smug because the first wave of temperate extinctions caused by humans occurred so long ago that we no longer consider these species in management plans.

In addition to famous total extinctions like the passenger pigeon (*Ectopistes migratorius*) and ivory-billed woodpecker (*Campephilus principalis*), many species underwent huge reductions in range over 100 years ago. In my home state of Iowa, this includes such species as swallow-tailed kite (*Elanoides forficatus*), common raven (*Corvus corax*), and several large herbivores (Anderson 1907). Restoration of these species is probably not feasible. With increases in human populations and subsequent habitat modification, it may be just a matter of time until we face a potential second wave of extinctions. This time gives us the chance to determine the things we need for optimal wildlife management under future conditions, such that we can choose the best set of trade-offs over the longest term.

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Nongame Perspectives in Aquatic Resource Management

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Abstract: Although nongame aquatic species greatly outnumber game species and share many of the same values to society (e.g., aesthetic, ecological, educational), only recently have they been considered in management programs. Traditionally, freshwater aquatic systems have been managed to maximize sustainable harvest of game and commercial species. However, federal legislation over the last fifteen years has emphasized the maintenance of ecological integrity and the conservation of all biota. Reconciling these perspectives provides a major challenge to aquatic resource managers, who face escalating demands on water resources. For example, common fisheries management practices such as the use of biocides, introductions of non-native species, and impoundment of flowing waters can be highly disruptive to aquatic systems, but impacts of such practices on nongame species and on ecological integrity are seldom assessed adequately. In addition, the ecological integrity of many aquatic resources is threatened by dewatering, pollution, and landscape alterations, which adversely affect both game and nongame species. Solutions to the management problems associated with conflicting demands on water resources must come from comprehensive planning at the landscape level that simultaneously regulates uses of both land and water.

NONGAME VALUES

All animal species have certain aesthetic, ethical, ecological, educational, and scientific values for human society. A small proportion of all wild species also are economically important because of their nutritional, recreational, or commercial values. Collectively, they are called "game species," and historically have been the target of virtually all fish and wildlife management programs. Unfortunately, this approach to management may have allowed, or even enhanced, the demise of various "nongame species" through habitat modification, shifts in energy flow, or other large-scale manipulations.

Any loss of biotic diversity is usually undesirable from aesthetic, ethical, or ecological perspectives, but a compelling argument for maintaining the integrity of ecosystems and their full complement of organisms also may be made on more pragmatic grounds. The basis for this argument is that perceived values of all species are dynamic, and their relative importance may change as society's technology, culture, and standards change. Moreover, potential benefits derived from future uses of species are largely unknown, and species currently considered unimportant may become extremely valuable. For example, in the early 1800's the lake sturgeon (*Acipenser fulvescens*) was considered a nuisance and valued only as fertilizer or livestock feed; today sturgeon flesh and eggs are gourmet foods (Scott and Crossman 1973). Thus prudence in management dictates that a primary goal of fish and wildlife agencies should be to maintain all native biota, thereby preserving as many options as possible for future management needs.

Perhaps the ultimate value of a complete, integrated biota to human society lies in the maintenance of earth's life-support systems. All species play a finite (but typically undefined) role in the processes that maintain biospheric integrity. An important property of the biota with respect to biosphere functions is its redundancy, or the fact that ecosystem processes, such as energy flow and nutrient cycling, do not usually hinge on only one species. This property ensures that ecological processes (and the functions they serve for humans) are not disrupted by fluctuations in the abundance of one species, but maintain some degree of continuity (Bormann 1985).

Our current level of understanding ecosystem processes seldom allows us to predict reliably which links in the system are critical or how many can be lost without causing irrevocable damage. Any loss of a species erodes biotic redundancy and increases the likelihood that an important ecological function will be impaired.

Federal and State Perspectives

Although values of nongame species historically have been largely ignored in management considerations, recent public concern for these species is embodied in two acts of federal legislation, the Endangered Species Act of 1973 and the Fish and Wildlife Conservation Act of 1980. Both acts indicate a much broader interpretation of fish and wildlife resources than management agencies traditionally have adopted.

The Endangered Species Act, which presents an especially broad interpretation of wildlife, currently protects 22 North American mussel species and 79 plant species, as well as many vertebrates (U.S. Fish and Wildlife Service 1984a). This act has been crucial in providing funds to protect fish species not viewed as sport, commercial, or forage species.

Of the 52 rare, threatened, or endangered fishes in North America, only 10 (19%) have or have had significant economic values associated with them (Ono et al. 1983). A similar pattern is apparent in the Ohio River, where only 3 of the

13 (23%) fish species that have disappeared since 1920 were economically valuable (Pearson and Krumholz 1984).

The Fish and Wildlife Conservation Act, commonly called the Nongame Act, encourages comprehensive conservation planning at the state level for all wildlife species, but restricts consideration to wild vertebrate species of ecological, educational, aesthetic, cultural, recreational, economic or scientific value to the public (Bean 1983). Thus, needs of invertebrates and little-known fishes are unlikely to be included in state conservation plans. However, if appropriations ever are approved to implement the Nongame Act, conservation efforts directed at "plan species" still may benefit other species.

In addition to focusing on the conservation of particular aquatic species or faunal groups, federal legislation also regulates human-induced contamination of water resources through the Federal Water Pollution Control Act Amendments (1972) and the Clean Water Act (1977). These acts indirectly benefit many aquatic species by creating allowable standards for a variety of physical, chemical, and radiological factors. Although Congress may have intended a broader interpretation, these laws typically are applied only to the maintenance of water's suitability for human use (including fishing, swimming, etc.).

An explicit objective of the Clean Water Act is to maintain chemical, physical, and biological (i.e., ecological) integrity of water resources. However, biological monitoring of aquatic systems often has been waived under the assumption that assessment of water quality on the basis of physicochemical standards is sufficient for the assessment of biological quality (Karr and Dudley 1981). A recent survey conducted by the United States Environmental Protection Agency (USEPA) indicated that this assumption is invalid. Fish management experts across the United States judged water quality (i.e., physicochemical features) to adversely affect fish communities in 56% of all flowing waters. Most degradation of water quality was attributed to nonpoint sources; indeed, only 10% of all waters are adversely affected by point sources (Judy et al. 1984).

Habitat conditions, which are not assessed during physicochemical monitoring, were judged to limit fish communities in at least 49% of all streams. Most habitat degradation was attributed to siltation and bank instability (Judy et al. 1984), which are indirect results of landscape modification. We infer that maintaining the physical and chemical integrity of water does not ensure the maintenance of biological integrity, and biological expertise therefore should be incorporated into monitoring and decision-making phases of water resource management (Karr 1981).

Aquatic nongame species play an important role in the assessment of biological integrity by providing a comprehensive "bioassay" of the aquatic environment. Because any perturbation is likely to affect distribution and abundance of some species, information on aquatic populations and communities can be used to assess the severity of degradation. Recent applications of this approach have indicated that fish and mussel populations are useful in assessing many types of aquatic degradation (Karr 1981, Fausch et al. 1984, Gunning and Suttkus 1985).

Despite the apparent values of conserving nongame species and the ecosystems in which they live, funds for nongame management are scarce. "Potential" sources of funds for the Nongame Act include numerous excise taxes, user fees on federal lands, and semipostal stamps (U.S. Fish and Wildlife Service 1984b), but no money has yet been appropriated. Federal expenditures for natural resource management still are dominated by more "traditional" activities. For example, over half of the $46.1 million budgeted for the Fisheries Resources Program of the U.S. Fish and Wildlife Service in fiscal 1985 went to hatchery operations (Chandler 1985). To date, income-tax checkoffs in 33 states and a special sales tax in another provide the only significant sources of nongame funds. State fish and wildlife agencies typically are charged with managing their resources for the benefit of all residents, but management goals and activities are dictated by public demand and funding sources. Because these agencies historically have been funded through hunting and fishing license fees and excise taxes on hunting and fishing equipment, the primary management goal has been the enhancement of consumptive uses.

Increasing public awareness of ecological principles and concern for wildlife conservation are apparent, as illustrated by increases in the number of nonconsumptive users of wildlife and in the membership of animal-protection and nature groups. Management philosophies of many fish and wildlife agencies probably lag behind values and expectations of the general public they are supposed to serve (Bromley and Bryan 1980). A recent mail-survey of randomly selected Virginia residents indicated that nonconsumptive values of wildlife were important to 92% of the respondents, but only 50% valued consumptive uses. Moreover, 79% of the respondents agreed that wild species should be managed for their own benefit rather than for human benefit (Moss and Fraser 1984). These patterns of public attitudes suggest that many fish and wildlife management agencies face a major challenge in re-ordering their priorities to match public demands on nongame resources.

Effects of new nongame programs on traditional fisheries management are likely to be complex, conflicting, and highly variable among regions. For convenience, we identify two broad types of potential conflicts: 1) game vs. nongame management programs; and 2) fishery vs. non-fishery uses of water and riparian land. In reality, these conflicts are highly interrelated, and most cases probably involve both categories.

CONFLICTS WITH TRADITIONAL FISHERIES PROGRAMS

The primary goal of freshwater fisheries management historically has been to maximize the sustainable harvest of game or commercial species. The most important ecological implication of this goal is that managers attempt to maximize the energy channeled into game species, usually top carnivores, thereby minimizing energy flow to less desirable species. Of the many management practices used to pursue this strategy, the three types that we judge most likely to irrevocably alter aquatic systems are the use of biocides, the introduction of non-native species, and the impoundment of flowing waters. In reality, most major watersheds of North America already have received liberal doses of all three manipulations. However, actual impacts of these management practices on ecosystem function and resident nongame species remain poorly known, and often can be inferred only from historical accounts by ichthyologists and natural historians.

Use of Biocides

Toxicants have been used widely to manipulate the species composition of fishable waters. Between 1963 and 1972, over 121,000 surface acres (49,000 ha), including 4,200 stream miles (6,800 km), were treated with biocides in the Midwest (Lopinot 1975). Some "rehabilitation" programs attempted to exterminate entire fish assemblages, and then start anew with an artificially stocked assemblage. Other programs were aimed at undesirable species such as common carp (*Cyprinus carpio*) and sea lamprey (*Petromyzon marinus*).

Most biocidal programs were justified on the presumption that they enhance fishing quality, but a few (e.g., Meffe 1983) were conducted to protect nongame endangered species. Although the use of biocides for enhancing fishing quality is probably not as widespread as it was in the 1960s, biocides still are used in efforts to control non-native species and in routine sampling protocols. Any application of biocide carries a potential for long-term damage to aquatic ecosystems and should be conducted cautiously.

Rotenone and antimycin, the most widely used piscicides, are also toxic to a broad array of non-target aquatic organisms. Concentrations of antimycin to which fish are sensitive vary by at least 1500 times among species tested (Marking and Dawson 1972). It takes 13 days for antimycin toxicity to be reduced by 50% at a pH of 6.0, but less time at higher pH. Some species most often targeted for eradiction are highly resistant. For example, bullheads (*Ictalurus* spp.) and goldfish (*Carassius auratus*) require $>$ 1.0 mg/1 rotenone for lethal dosage (Cumming 1975). The necessity of applying high dosages obviously increases the likelihood of also eliminating nontarget species. Additional impacts of toxicant application may occur where efforts are made to neutralize the toxicant, because chemicals used as detoxicants (e.g., chlorine, potassium permanganate) also are highly toxic to aquatic species.

Although massive mortalities of non-target (i.e., nongame) species are undoubtedly associated with many piscicide applications, there is little documentation of impact assessment on nongame species in the literature. Only one of the 10 papers included in the 1973 Midwest Fish and Wildlife Conference symposium on the use of toxicants in fisheries management expressed concern for the "inadvertent" effects of toxicants on aquatic systems. In that paper, Becker (1975) reported the effects of rotenone and antimycin application in the Rock

River basin of Wisconsin. The project's primary objective was to eradicate common carp, but significant ecological impacts were indicated by widespread reductions in species richness and fish numbers in treated streams, even after 5 years. Although minnows and darters apparently suffered the highest mortalities, densities of most invertebrate taxa, including mussels and clams, also were reduced sharply.

Introduction of Non-Native Species

Species introduction is perhaps the most commonly used technique for enhancing recreational fishing. Introductions usually involve piscivorous species, but often include forage fish to sustain predatory game species. Introductions may be conducted by state or federal agencies or unauthorized individuals, or they may be accidental. Rationales for introductions include enhanced fishing, mosquito control, and species protection (Courtenay and Stauffer 1984). Although introduction of brown trout (*Salmo trutta*) into the United States is considered by many to be a success, few other introductions are considered beneficial; some species, such as common carp, are severely damaging to aquatic systems and cost millions of dollars annually to control (Courtenay and Hensley 1980). From an ecological perspective, the worst aspect of a species' introduction is that once a viable population is established, the introduction generally is irreversible, regardless of whether the desired effect is realized.

Unsuccessful attempts to remove introduced species from stream systems are widespread (Moore et al. 1983, Meffe 1983). Indeed, the common carp has withstood numerous eradication programs, and currently is the most widespread fish species in U.S. waters (Judy et al. 1984).

Potential ecological impacts of non-native species on the native biota are diverse, but their extent is poorly documented, especially for nongame species. Most impacts of non-native introductions fall into one of four categories: 1) habitat alteration, 2) changes in predator-prey or foraging dynamics, 3) introduction of parasites or disease, and 4) hybridization between native and non-native genotypes (Welcomme 1984). Some species seem especially adept at disrupting native faunas. For example, cichlid introductions are correlated with declines in a broad array of nongame taxa including speckled dace (*Rhinichthys osculus*), California killifish (*Fundulus parvipinnis*), and gizzard shad (*Dorosoma cepedianum*) (Courtenay and Hensley 1980).

Habitat alteration by non-native species is exemplified by increased turbidity and loss of weed beds induced by common carp and grass carp (*Ctenopharyngodon idella*), respectively. Loss of weed beds may be especially important to small nongame fishes that depend upon the food (invertebrates), cover, and spawning sites provided by vegetation. Ware and Gasaway (1976) attributed the elimination of several forage species from Florida lakes to vegetation removal by grass carp. In addition, loss of vegetation cover may increase the vulnerability of ordinarily non-prey species to predators. Indirect effects of habitat alteration may extend to terrestrial wildlife. Gasaway and Drda (1977) argued that the loss of aquatic vegetation and its associated invertebrates contributed to declines in Florida waterfowl abundance after grass carp introductions.

Introductions of non-native predators may cause dramatic shifts in the species composition and abundance of native fishes, especially small, nongame species. The introduction of peacock bass (*Cichla ocellaris*) into Gatun Lake in Panama and the subsequent precipitous declines in densities of its prey are well documented (Zaret and Paine 1973). This introduction also altered abundances of fish-eating birds (herons, terns, gulls, kingfishers) of Gatun Lake (Karr 1985), again illustrating the interdependence of aquatic and terrestrial components of ecosystems.

Few studies have documented impacts of introduced predators in U.S. waters. Garman and Nielsen (1982) observed declines in torrent sucker (*Moxostoma rhothoecum*) and central stoneroller (*Campostoma anomalum*) in a Virginia stream after large brown trout were stocked. Predation on young by non-native centrarchid species is cited as the most important factor in the decline of razorback sucker (*Xyrauchen texanus*) (Minckley 1983).

Presumably, introducing large, voracious predators such as striped bass (*Morone saxatilis*) and muskellunge (*Esox masquinongy*), or maintaining unnaturally high densities of native predators may profoundly impact species at lower trophic levels. Conversely, reductions in the abundance or size of top predators by angling (Goedde and Coble 1981) may trigger population increases in other species, thereby upsetting natural dynamics of competition and predation.

Impacts of introduced species on native species due to competition for food are more difficult to assess than impacts of predators, but clearly can be severe. Non-native species that are omnivorous (e.g., *Tilapia* spp., common carp) may have especially adverse effects on the food base of native species. One example of food competition between introduced and native fauna comes from the Great Lakes. Increases in the exotic alewife (*Alosa pseudoharengus*) and rainbow smelt (*Osmerus mordax*) in Lake Michigan between 1930 and 1960 were associated closely with declines in 10 native planktivores, including the emerald shiner (*Notropis atherinoides*), a nongame species (Stewart et al. 1981). Impacts on native species apparently resulted from a combination of direct competition for food and predation on eggs and larvae by rainbow smelt and alewife. However, recent maintenance of high predator densities through salmonid stocking may have substantially reduced competitive effects of alewife and rainbow smelt on native planktivores in the Great Lakes.

Introduction of new parasites and diseases, which infect game and nongame species, commonly are associated with non-native species. At least 25 foreign parasites now inhabit North American waters (Shafland 1979). Other imported diseases and parasites include furunculosis, infectious dropsy of cyprinids, and various trematodes (Welcomme 1984). Impacts of these introductions on populations of nongame species essentially are unknown.

Clearly, introductions of non-native species should be performed only after a careful estimate of long-term benefits and consequences. Unfortunately, introductions generally are undertaken without regard for impacts on native fishes and ecosystems (Courtenay and Hensley 1980), and expected effects are based on intuition more often than on quantitative or comprehensive analyses (Li and Moyle 1981).

If preserving the ecological integrity of aquatic systems is a primary mandate, management agencies should develop strict protocols for introducing non-native species. Such protocols should include evaluation of the full range of ecological impacts of all life stages, as well as review and approval by personnel outside the agency seeking the introduction (Courtenay and Hensley 1980). Finally, we concur with Crossman (1984) that rehabilitation of native species and habitats generally is a more responsible solution to degradation of aquatic resources than the "cures" afforded by non-native introductions.

Impoundment of Flowing Waters

Although fisheries managers seldom have the authority to decide whether a stream is to be impounded or remain free-flowing, their assessment of the relative fishery values of river *vs.* reservoir may influence such a decision. For example, the nature and expense of what is judged to adequately mitigate a loss of free-flowing stream may influence the feasibility of a proposed impoundment. Because most popular sport fishes in the Midwest are well adapted for lakes, and because new reservoirs typically provide very productive fisheries, the general trend is to favor impoundments over free-flowing waters, especially in large streams. However, the geologic longevity of rivers far exceeds that of most lakes (millions vs. thousands of years), and most aquatic species are adapted for lotic, not lentic, environments. Over 60% of the rare, threatened, and endangered fish species (Ono et al. 1983), and all of the threatened and endangered mussel species of North America are riverine. These patterns indicate that further impoundment of lotic systems must be undertaken cautiously if permanent damage to the biota is to be avoided.

Impoundments profoundly affect ecological processes and biota above and below the dam. Impounded waters differ from free-flowing waters in their daily and seasonal dynamics of such basic physicochemical factors as temperature and dissolved oxygen. In addition, production pathways shift from benthic to pelagic species, the extent of this shift depending on the relative position of the dam along the stream continuum. Indeed, dams interrupt all biological processes in streams and shift factors such as nutrient concentration, species diversity, and P/R ratios to those that would be expected at up- or down-stream sites in the absence of dams (Ward and Stanford 1983).

Most impoundments cause declines in species diversity, especially for invertebrates (Stanford and Ward 1979). Petts (1984) reported that at least 6 fish species have been eliminated from impounded rivers in the United States. Loss of heterogeneity in current and substrate (especially through siltation), water level fluctuations, and inadequate oxygenation are major factors that cause species losses (Isom 1971). Inevitably, species that require flowing water during some stage of their life cycle (e.g., many minnows, darters, insects, mussels) decline or are extirpated following impoundment. Even species that can tolerate the physicochemical environment of impounded waters may decline because of alterations in the dynamics of predator-prey (Meffe 1984) or competitive interactions. Shifts in biotic interactions and production pathways may be exacerbated by stocking additional predators or forage species, as discussed above. Mussel species, many of which require specific fish hosts for successful reproduction, may be lost if their hosts decline in abundance (Zale and Neves 1982).

Artificial hydrologic regimes imposed by dams can influence the biota for extensive reaches downstream of impoundments. Because many processes that maintain riparian and wetland systems depend on pulses of discharge, regulation of flow often results in losses of backwaters and side channels, as well as the fish and wildlife resources associated with those habitats (Petts 1984). Unfortunately, successional changes in riparian vegetation and equilibration of channel morphology downstream of dams occurs on time scales of decades, and therefore may be overlooked in routine impact assessments (Petts 1984). However, short-term changes in the tailwater environment also may be severe. For example, daily fluctuations in discharge may devastate fishes that spawn in shallow habitats (Petts 1984), and cold water releases from high dams may cause declines in warm-water species (Holden and Stalnaker 1975).

CONFLICTS WITH OTHER LAND AND WATER USES

The greatest threats to nongame species and the integrity of aquatic ecosystems do not originate from fisheries management practices, but from non-fisheries uses of water resources. Some of the most ominous threats to aquatic biota include 1) dewatering through groundwater depletion and diversion for agriculture; 2) pollution by pesticides, industrial and agricultural wastes, and acid precipitation; and 3) landscape and habitat alteration through land clearing and channelization. The management problems associated with these large-scale activities are exceedingly complex because of their socioeconomic and political implications. Although both game and nongame species are affected by general degradation of aquatic systems, game species are more likely to be considered in management programs, and thereby avoid extirpation. Maintaining the ecological integrity of highly diverse systems (e.g., most large rivers of the Midwest) may be especially difficult because they seem to be more dramatically altered by stress than less diverse systems (Long 1974, Goodman 1975).

In the Midwest (where intensive agriculture dominates the landscape), siltation, nutrient enrichment, and stream-channel modification usually are the major causes of degradation of water resources. Significant improvements in the quality of water and aquatic biota could be achieved with minimal reductions in agricultural production through selective use of land-management practices in and along streams (Karr and Schlosser 1978). Such practices include 1) maintenance of near-stream vegetation to inhibit entry of light, nutrients and sediment into water courses, and 2) maintenance of natural-channel morphology to reduce bank erosion and concentrations of suspended solids, and to promote species diversity and stable production (Karr and Schlosser 1978, Schlosser 1982, Angermeier and Karr 1984). Although widespread implementation of such practices at land-water interfaces clearly would enhance the long-term health and productivity of aquatic resources, comprehensive state or federal programs to integrate land and water management at the watershed level are glaringly absent. Yet (as we mentioned earlier), landscape modification is probably the major cause of water degradation in the U.S. (Judy et al. 1984). Until policy-makers develop broad-scale programs for landscape management, water-resource managers will remain relatively powerless in their attempts to maintain or improve the quality of aquatic systems. Before effective water-resource management programs can be implemented, some changes in socioeconomic ideology may be necessary. In particular, our society's view of land "ownership" should shift toward one of land "stewardship," and the practice of discounting future resource values in economic analyses should be discontinued.

INTEGRATIVE MANAGEMENT

An integrative approach to identifying various types of degradation to aquatic systems is needed to replace the traditional chemical-standards approach of assessing water quality (Karr and Dudley 1981; Karr, Toth and Dudley 1985) An integrative approach should take into account the primary variables (and interactions) affecting aquatic community organization. For streams there are at least 5 such variables:

1) *Flow regime*—temporal distribution of water availability, including annual and seasonal fluctuations.
2) *Water quality*—regimes of temperature, pH, dissolved oxygen, suspended and dissolved materials, and other physicochemical factors.
3) *Energy source*—relative availability of allochthonous *vs.* autochthonous organic material, including particle size distribution and spatio-temporal dynamics.
4) *Habitat structure*—physical features of the environment, including depth, current, substrate, and cover attributes; also juxtaposition and frequency of various habitat types (e.g., pools and riffles).
5) *Biotic interactions*—competition, predation, disease, parasitism, and mutualism.

Because aquatic biota may be limited by alterations in any combination of these 5 primary variables, successful maintenance of biotic integrity in aquatic ecosystems will require managers to simultaneously monitor and assess all of them (Karr, Toth and Dudley 1985). Such a program obviously extends the concerns of water resource management beyond the boundaries of shorelines and stream channels to the entire watershed.

This approach also requires the development and use of monitoring tools that directly assess biological integrity. The development of integrative biological monitoring has lagged behind that of other monitoring protocols that focus only on physicochemical features of water. However, a recently developed protocol for assessing the biological integrity of streams on the basis of fish community attributes has been shown to be useful for a variety of degradation types and in several geographical regions (Karr 1981; Fausch et al. 1984; Karr, Heidinger and Helmer 1985; Angermeier and Karr 1986). This protocol warrants further testing, especially to determine its applicability for other taxa and types of systems.

Guidelines

Although we cannot specifically address all problems that managers of nongame aquatic species encounter, we can offer some general guidelines for approaching the most likely problems.

1. Adopt a landscape perspective. The quality of an aquatic system reflects the sum of all natural and human-induced impacts in that system's watershed. Identify impacts of land use, and exercise available authority to correct or minimize adverse effects. Anticipate impacts of changing land use.
2. Avoid practices that involve system-level perturbation. Large-scale changes in species composition or habitat features are unlikely to improve a nongame species' environment relative to that in which it evolved. Species introductions (except re-introductions) and impoundments are unlikely to benefit the system as a whole, especially in the long run.
3. Be familiar with the biota. Find out what is known about life-history patterns. Determine why rare species are rare. Ideally, a manager should be able to rank all species (and life stages) with respect to tolerances to various perturbations. This information would allow monitoring and mitigation efforts to be most effectively apportioned.
4. Adopt an integrative approach. Aquatic species and ecosystems are extremely complex, and their functions may be altered by a vast array of perturbations. Management programs must be comprehensive, yet flexible, thereby enabling management efforts to shift among potential limiting factors when appropriate. Avoid over-managing one factor (e.g., water quality) at the expense of neglecting another (e.g., habitat structure). Attempt to assess responses of the biota to perturbations by examining patterns and processes at individual, population, and community levels.

In conclusion, maintaining the quality of aquatic resources will require shifts in philosophy by both state and federal management agencies. These agencies need to work together in a more concerted fashion to implement comprehensive programs that protect all aquatic biota, thereby maintaining the integrity of the systems on which human society depends.

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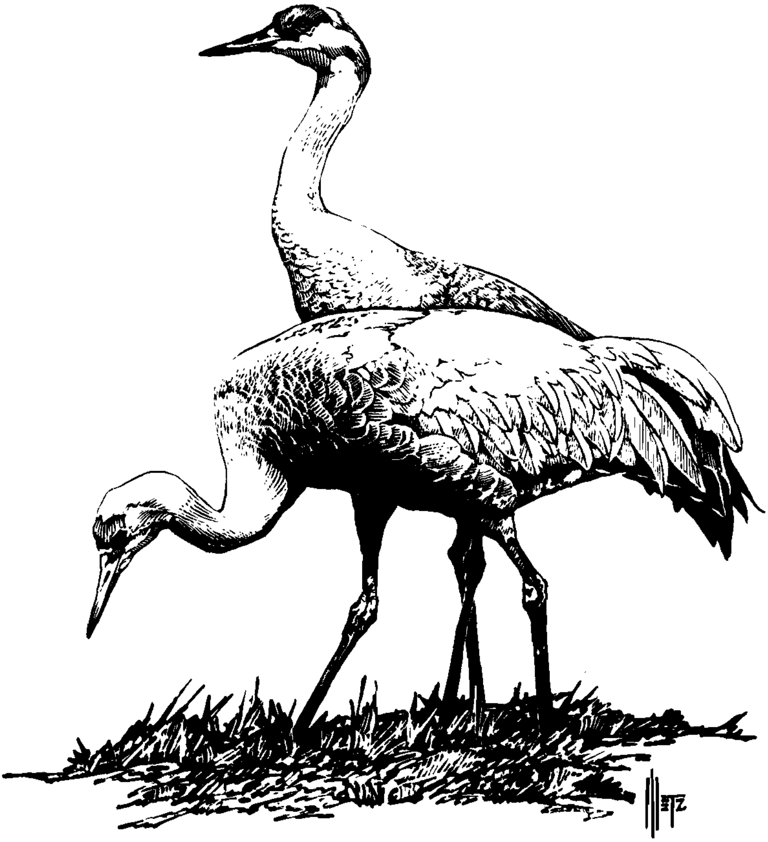

Wetland and Riparian Habitats: A Nongame Management Overview

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Abstract: Wetland and riparian habitats have been severely disrupted by man's activities, especially agriculture. Over one-half of these lands, transitional between terrestrial and aquatic systems, have been destroyed and few of the remaining habitats in the 48 coterminous states have not been impacted adversely. Birds are obvious members of wetland and riparian communities and are the focus of this paper. Effective management for wildlife requires an understanding of biotic and abiotic factors influencing these habitats. Hydrology, a key element among these factors, determines the composition and productivity of plants and corresponding animal associations. Wetland complexes are required to meet the broad needs of diverse bird faunas and the constantly changing requirements of organisms in the annual cycle. Wetlands with naturally occurring hydrological regimes should be protected, and managed habitats should be manipulated as complexes with dynamic water regimes. Drawdown management is a common procedure in wetland and riparian habitats that promotes high productivity and, in turn, attracts a diverse bird fauna. Discing, fire, and reflooding are techniques commonly used to enhance the effectiveness of drawdowns to meet the needs of a variety of species. A holistic management approach can meet the needs of game and nongame alike while providing plant structure and food requirements for wetland wildlife.

Some of the earliest efforts associated with wetland protection in the United States were for nongame wildlife (Salyer and Gillett 1964). Populations of plumed birds decreased precipitously at the beginning of the 20th century as harvest for the millinery industry increased. In response, Pelican Island Refuge, Florida was established by Executive Order of President Theodore Roosevelt in 1903 to protect brown pelicans (*Pelecanus occidentalis*) and other colonial nesting birds.

Beginning in the drought years of the 1930s, wetland acquisition priorities for wildlife have been primarily for waterfowl. Although federal wetland acquisition has been influenced primarily by habitat needs for waterfowl, other needs, including flood control, water quality, and endangered species considerations, also determine priorities of current acquisitions. Whatever the acquisition goals, nongame wildlife have benefitted tremendously from state and federal programs directed toward protection and acquisition of wetlands.

Wetlands are transitional lands between terrestrial and aquatic systems where the water is at or near the soil surface, or the land is covered by shallow water (Cowardin et al. 1979). Riparian habitats are also transitional between terrestrial and aquatic systems, often have woody vegetation, and usually are restricted to the 100-year floodplain (Swift 1984). Although riparian habitats are often classed as wetlands because they have many similar characteristics, some differences warrant separate attention (Brinson et al. 1980). Riparian lands have a very high edge-to-area ratio because they are linear or dendritic and ecotonal in nature. Riparian areas have large energy, nutrient and biotic interchanges between aquatic and terrestrial systems (Odum 1978).

Wetland and riparian habitats have been impacted severely by man's activities. Society continues to exploit these habitats extensively because they provide sites for cities or ports, are easily converted to agriculture or are considered a nuisance. Since the late 1800s, expansion of agriculture and incessant efforts to control floods and promote navigation on waterways has led to a catastrophic reduction in area and quality of wetland and riparian habitats throughout the United States (Tiner 1984, Swift 1984, Reinecke et al. 1986). Currently, only about 40 percent of our original wetlands remain in the coterminous states, and some states such as Iowa have lost nearly 100 percent of their wetlands (Bishop 1981, Office of Technology Assessment 1984).

Nongame wildlife include a vast array of cold- and warm-blooded vertebrate and invertebrate species that are not harvested for fur, for sport, or by commercial food industries. In many cases, these animals have economic values because of recreational uses, but their value often is intangible. In this paper, we will emphasize the management of waterbirds. (See Table 4 for scientific names that follow American Ornithologists' Union [1983]. Scientific names for species not included in Table 4 are in the text.) Nevertheless, goals for effective management of wetlands and riparian areas should be to maintain productive and diverse systems rather than to focus on specific target organisms such as birds. Attempts to perpetuate a stable vegetative structure or to provide required foods over extended time periods for a single species often result in unrealistic management of dynamic wetland systems. Thus the concepts developed in this paper are appropriate for all wetland species regardless of whether game or nongame.

Research and management of nongame wetland wildlife have been limited when compared to efforts and expenditures for migratory waterfowl. Only recently have federal and state wildlife agencies expanded staffs and budgets to address nongame issues and to increase efforts to meet the needs of nongame (Applegate and Trout 1984, Boggis and Hamilton 1984).

Some public and private groups have developed management programs designed for nongame species. For instance, the National Audubon Society manages wood storks (*Mycteria americana*) in Corkscrew Swamp, Florida. Similarly, the U.S. Fish and Wildlife Service (FWS) devotes a considerable effort to protect and increase the continental population of endangered whooping cranes (*Grus americana*). Research at the FWS Patuxent Wildlife Research Center and management at Gray's Lake, Idaho and Aransas National Wildlife Refuge, Texas are indicative of the extent of research and mangement efforts for species with critical needs (Archibald and Mirande 1985). At the state level, the Missouri Department of Conservation has purchased and managed wetlands for the protection of endangered herpetofauna and bats. For example, the Illinois mud turtle (*Kinosternon flavescens spooneri*) has been protected by the purchase of key wetland habitat, and riparian-zone caves have been protected or purchased for the endangered Indiana bat (*Myotis sodalis*) and gray bat (*M. grisescens*).

A diverse group of government agencies has responsibilities that will determine the fate and productivity of the remaining riparian and wetland habitats. The U.S. Army Corps of Engineers and the Environmental Protection Agency have regulatory responsibilities for wetlands through Section 404 of the Clean Water Act. The U.S. Fish and Wildlife Service is responsible for the migratory bird resource and its management.

In general, agencies lack the specific information to determine nongame needs or to understand the complexities of wetland processes for effective management. Some information is available on presence or absence of species, chronology of annual events or numbers of organisms, but nutritional, energetic and habitat needs of most species are poorly understood. Much of the information on nongame wildlife associated with wetland and riparian habitat has accumulated because of specific research interests in a species or a taxonomic group, or to test hypotheses relating to ecological questions. Such published information on nongame species often has high value for management, but the syntheses needed by land managers generally are lacking.

Another important deficiency for wetland managers and administrators is the lack of information that identifies wetland management values for nongame wildlife that are not obligate wetland species. For example, bats and swallows feed on insects emerging from wetlands, raptors prey on wetland wildlife, and many seed-eating passerines forage on foods produced in wetlands. None of these vertebrates are traditionally recognized as wetland species, yet wetlands play an important role in providing nutritional needs in their annual cycle.

CURRENT STATUS AND TRENDS OF WETLAND AND RIPARIAN HABITATS

Classification and Inventory

Effective management requires identification of specific environments and some measure of the distribution and status of the remaining habitats. Wetland classification systems were developed for national (Martin et al. 1953) and regional use (e.g., prairie, Stewart and Kantrud 1971; arctic coastal, Bergman et al. 1977;

and northeastern U.S., Golet and Larson 1974). Although destruction of riparian habitats and wetlands has been ongoing since the 1800s, the severe destruction following World War II caused much concern within the FWS. These losses stimulated the first national wetland inventory, often referred to as "Circular 39, Wetlands of the United States" (Shaw and Fredine 1956).

Severe habitat losses continued in the 1950s and 1960s (Frayer et al. 1983), but as society began to recognize the importance of wetlands for flood control and water quality in the 1960s, there was increasing public support for wetland protection. The need for a more comprehensive classification document became obvious by the 1970s. A hierarchical system of wetland classification was developed that used criteria based on soils, flooding, and vegetation (Cowardin et al. 1979). Concurrently, the need for information on the status and trends of wetlands and their distribution increased. In 1974, Congress directed the FWS to conduct another national wetlands inventory. The first published status report became available in 1984 (Tiner 1984), and wetland maps at scales of 1/24,000 and 1/62,500 are now available from FWS Regional Offices for use as overlays on United States Geologic Survey maps of similar scale. Congressional interest in wetlands is also apparent from the important document produced by the Office of Technology Assessment (1984).

Wetlands

The original wetland area in the coterminous United States was estimated to be between 77 and 90 million ha (190-220 million A.) (Office of Technology Assessment 1984, Tiner 1984). Alaska has another 84 million ha (208 million A.) of wetlands (Tiner 1984). Today, wetland area is estimated at around 42 million ha (104 million A.) or about half of the original total in the coterminous United States. Conversion of palustrine wetlands to agricultural purposes has been of primary concern in the prairie pothole region of the north central United States, the Central Valley of California, and the Mississippi Alluvial Valley (MAV).

Within the rich prairie pothole region, 99% of wetland habitat has been lost in Iowa (Bishop 1981), and total losses in North and South Dakota have been about 60% (Tiner 1984). The Drift Prairies of eastern North Dakota and western Minnesota have experienced losses approaching those in Iowa.

Destruction of southern forested wetlands in the MAV are more severe than losses on the prairies. Southeastern Missouri has lost 97.5% of the original forested wetlands (Korte and Fredrickson 1974) and only 19% or 1.9 million ha (4.7 million A.) of the original 10 million ha (25 million A.) remain in the MAV (MacDonald et al. 1979).

Originally, the southern forests represented about 13% as much area as the extensive prairie breeding habitats. By 1985, the widespread destruction of southern forested wetlands had reduced this forested habitat to only 5% in comparison with the remaining prairie wetlands. Coastal plain wetlands of Louisiana currently disappear at the astounding rate of 102 km^2 (39 mi^2) per year, largely because of gas and oil exploitation, and flood control projects in conjunction with natural processes of subsidence and uplift (Gagliano et al. 1981).

Wetland and riparian areas in the Central Valley of California historically served as habitat for myriads of wetland birds. By 1982, wetlands in pristine condition were nonexistent in the Central Valley and the remaining highly modified wetlands account for only 8% of the original wetland habitat (Gilmer et al. 1982). The extensive modifications of wetland and riparian habitats throughout the United States have affected every wetland type and process as well as every wetland species.

Riparian Habitats

Four primary woody riparian habitats are recognized: elm-ash (*Ulmus-Fraxinus* spp.), northern floodplain, southern floodplain, and mesquite (*Prosopis* spp.)-bosque (Swift 1984). Documentation of riparian destruction has been less thorough than for wetland losses. Some riparian habitats also are recognized as wetlands, so the area of loss is not additive for riparian and wetland habitats. Originally, the coterminous United States had an estimated 30.3 to 40.5 million ha (75-100 million A.) of riparian habitats (Swift 1984). Currently, between 10 and 14 million ha (25-35 million A.) remain in the 48 coterminous states. Only 11% of the original 364,000 ha (900,000 A.) of riparian habitats remain in California's Central Valley. Riparian areas are extremely important to wildlife in the arid west because of their proximity to water.

Because of their location in floodplains, destruction of riparian habitats is largely associated with man's activities, especially urbanization, stream-channel modification, water impoundments, and clearing for agriculture (Swift 1984). Narrower riparian areas are more easily altered and potentially degraded. These inordinately high losses adversely affected fish and wildlife, recreation, water quality, wood production and other values (Lowrance et al. 1984). Specific perturbations that impact riparian habitats include road construction, because riparian zones often follow the gradual elevational changes of a watershed; recreational developments that destroy natural plant diversity and structure, lead to soil compaction and erosion, and disturb wildlife; improper grazing practices; and reservoir development that impounds lotic floodplains (Thomas et al. 1979).

The disappearance of fire from the prairie ecosystem resulted in major changes in prairie riparian habitats. As woody vegetation became widely established, habitat conditions for grassland avifauna deteriorated, and birds of woodlots and savannahs increased in abundance (Krapu et al. 1982, Knopf pers. commun.).

CHARACTERISTICS OF WETLAND AND RIPARIAN HABITATS

Wetland and riparian habitats resemble sieves because the biotic and abiotic components that determine habitat characteristics readily pass through the system (Fredrickson 1982). Abiotic components of wetlands include soil, climate, fire, hydroperiod (period when soils are saturated), and hydrological regime, as well as water quality, quantity and chemistry. Hydrology is a key component within the system and deserves special attention by managers. Naturally productive wetland systems have dynamic water regimes. Plant composition, habitat structure and productivity are determined by the timing, duration, and extent of flooding. Thus hydrology has an overriding influence on the system.

Understanding the effects of short- and long-term fluctuations on the system is one of the greatest challenges facing managers because natural hydrology is continually modified by man's activities. Hydrological regimes vary daily, seasonally and over longer periods, and wetland and riparian productivity is largely determined by these fluxes. Modifications to the natural dynamic regimes can lead to extended extremes of drought or flooding with a resultant drastic decline in productivity (Belt 1974).

Climatic factors control the seasonal availability of habitats for wildlife, with considerable variation among the 48 coterminous states and Alaska. North Dakota has about 120 frost-free days per year (Cowardin et al. 1985), whereas southeastern Missouri normally has over 200 days (Fredrickson 1979). Arctic habitats may be ice-free for less than 3 months annually, but food resources are plentiful and growth rates are rapid during the period of continuous daylight. Adaptations to cold environments allow high-latitude macroinvertebrate forms to have high production in spite of lower annual mean temperatures (Longsdale and Levington 1985).

The seasonal pattern of rainfall determines when foods are produced or available in wetland and riparian habitats. In southeastern Missouri, peak precipitation accumulations occur in spring, whereas accumulations are minimal in September. Annual precipitation averages 122 cm (45 in.), ranging from 64 to 190 cm (25-75 in.) among years (Fredrickson 1979). Unusually heavy rainfall can occur during any month in the MAV. In contrast, rainfall on the North Dakota prairies averages 44 cm (18 in.) per year, with a peak occurring in early summer. Rainfall on the prairies fluctuates considerably among years and droughts occur regularly. Accumulations of snow, and the timing of the thaw in relation to when soils are frost-free determine runoff and surface water accumulations in wetland basins each spring. These water conditions impact the timing and extent of nesting by birds. Rainfall in the early part of the growing season partially regulates the number of wetland basins that are flooded by mid summer and available for use by young and post-breeding waterfowl assemblages.

Nutrients, energy, and biota are constantly moving between the terrestrial and aquatic systems (Likens and Bormann 1974). Wetlands have complex interchanges of energy and nutrients at the local, regional, continental or even global level, because of the characteristics of some abiotic components, as well as the migratory behavior of some biota. Soils impact the fertility of the adjoining wetland water (Moyle 1956, Hoyer and Reid 1982). The heart of the best agricultural area in the Midwest once had the most productive prairie potholes. Water quality in wetland waters is closely associated with substrates within and adjacent to the wetland boundary. Nutrients from rich soils move into the water column and determine the potential for aquatic production.

The major biotic components within the "sieve" include litter, macrophytes, invertebrates, and vertebrates. The seasonal and long-term fluxes in water levels influence the production of macrophytes within wetland basins. The entire floor of ephemeral pools or temporary wetlands may be vegetated by late summer each year. More permanent wetlands have longer trends of vegetational changes, but during the drier years, entire basins or major portions of basins may be heavily vegetated.

Litter is a key element in the productivity of wetlands and eventually determines the value of a site for animal life (de la Cruz 1979, Nelson and Kadlec 1984, Batema et al. 1985, White 1985, Wylie 1985). In seasonal environments, the entire above-ground biomass becomes litter following senescence. Structurally intact plants, living and dead, are colonized by periphyton that provide foods for grazing macroinvertebrates. Other macroinvertebrates function as shredders and change coarse particulate organic matter to fine particulate organic matter. This material then can be transported to adjacent areas in the water column where gatherers, collectors and filterers meet their nutritional needs with these smaller particles.

Macroinvertebrates are key food resources for wetland birds. Thus successful management requires a basic understanding of the relationships among hydrology, water quality, water chemistry, plant production and structure, decomposition, invertebrate ecology and subsequent availability of invertebrates for higher life forms.

Waterbirds have an important influence on wetland systems. Their mobility results in transfer of energy and nutrients among adjacent habitats (resident and migratory species) or among widely separated habitats (migratory species). Migratory movements occur within regions, continents, and sometimes between continents. Some of the most spectacular movements are the annual migrations of shorebirds between hemispheres. Managers have special obligations with migratory species because natality may be profoundly influenced by factors in a distant habitat. A greater percentage of a bird's total energy may be channeled into reproduction if habitats used by migrants have provided adequate resources for physiological needs and for increasing endogenous reserves. For example, the reproductive performance of the purple heron (*Ardea purpensis*) in the Netherlands is influenced by water conditions on the wintering grounds in the Senegal and Niger river floodplains of Africa (den Held 1981). Other species may well have such requirements; hence, managers on wintering areas may play a key role in assuring that breeding on some distant wetland is successful.

Activities of birds within riparian and wetland habitats may promote conditions conducive to plant germination or growth. For example, intensive foraging may trample vegetation and reduce vertical structure or disturb the substrate in such a way that early-successional plants are likely to be present in the subsequent growing season. Parasites, pathogens, and predators also influence wetlands and riparian environments and have important implications for successful management.

HABITAT COMPLEXES

Biologists and managers have gradually identified a wide array of habitat types resulting from variations in the abiotic and biotic components within a locale (Tables 1 and 2). These groupings of different wetland habitats arranged in close juxtaposition are called wetland complexes. For example, prairie wetland complexes consist of groupings of wetlands classed as Type I through IV, V, VI or VII. Once these different habitat types were described for various wetland systems, the role of each habitat type in the life cycle of animals was identified (Derksen et al. 1981, Ryan et al. 1984, Fredrickson and Heitmeyer 1986). In general, a single habitat type fails to provide the food and cover required for all stages in the life cycle of a species. Note that of the 10 habitat types identified for southern floodplain forests (Table 1), all have different flooding regimes, 3 are often flooded continuously, and 4 others normally are dry for at least 6 months each year. Likewise, of the 7 types of glacial wetlands, seasonal flooding is highly variable (Table 2).

The importance of diverse habitat complexes cannot be overemphasized. The role of different wetland types has been documented for mallards on wintering habitats (Heitmeyer 1985, Fredrickson and Heitmeyer 1986, Reinecke et al. 1986). Unlike prairie wetlands that have distinct wetland basins, southern hardwoods have wetland habitats distributed along flooding gradients. These palustrine habitats can be typed as open water, moist-soil, scrub/shrub, cypress-tupelo (*Taxodium-Nyssa* spp.), overcup oak (*Quercus lyrata*), and pin oak (*Q. palustris*), among others. Flooding of these habitats varies seasonally, as well as in depth and duration (Table 1).

Mallard use of these habitats varies with pairing status and flooding. For example, unpaired birds use open habitats, pairing occurs in scrub/shrub habitats and paired birds use flooded forests, especially the pin oak and overcup oak habitat types. Thus, mallards are distributed among the habitats on the basis of their individual status in the annual life cycle rather than solely on seasonal flooding regimes.

Prairie marshes have distinct basins that are flooded for varying periods each season (Table 2). Less permanent potholes provide foods early in the season, but as drying occurs, aquatic foods are available only in the larger, deeper marshes. Habitat used by marbled godwits (*Limosa fedoa*) is a good example of the importance of diverse prairie wetland complexes. Godwits prefer shallow ephemeral and aklali wetlands during the breeding season, but a mosaic of these types, including semi-permanent and temporary wetlands and associated grasslands, are necessary to satisfy all the habitat needs of this species (Ryan et al. 1984).

These recent findings suggest that we have much to learn about habitat requirements of nongame wildlife. Effective management will require more definitive life-history information on each species and the importance of habitats used during each period in the annual cycle. Undoubtedly, nongame wildlife have constantly changing needs in relation to their annual cycle, but seasonal use of habitats in relation to stage in the life cycle is poorly known. Major differences in habitat needs are apparent for some nongame species, but an understanding of the more subtle requirements is lacking.

CONCEPTUAL APPROACHES TO MANAGEMENT

All too often, management is viewed only as active manipulation of habitats. Errington (1963) pointed out the pricelessness of untampered nature, and his comments are particularly appropriate at a time when so many natural habitats have been disrupted severely. The desire to improve upon nature is a common perspective held by society and some professional managers as well. When large undisturbed wetlands are present, the dynamic processes of natural systems cannot be improved upon by man. In the few places where functional natural systems remain, effective and responsible management requires protection of the habitat rather than manipulations. In contrast, manipulations often are essential where hydrology has been modified or habitats have been degraded.

Where management objectives require manipulations, careful assessment of management potential is essential because of the tremendous cost of development, operation, and maintenance of wetland areas. Manipulations may be physically impossible, counter to other desired responses, or beyond the means of realistic budgets. Successful development requires recognition of important site characteristics, including climate, soils, water supply, presence of natural wetland complexes, plant composition, animal populations, and wetland or riparian location, particularly as it relates to disturbance (Table 3). If a site has the potential for development, then development considerations must be matched with the potential to provide the needs of target species (Table 3). This aspect of management is continuous and information on biological activities and habitat requirements of a species should be incorporated into the ongoing operation when such information becomes available. This decision-making process is particularly important for nongame wildlife because many species have specific habitat requirements that are presently unknown.

Once areas are developed, complex factors that influence plant establishment, growth and structure are the key elements that determine management actions. Important interactions among the seedbank (including seeds, tubers, rhizomes and other propagules), topography, season, time of drawdown, type of drawdown, type of disturbance, time since disturbance, and time since continuous flooding determine the composition and structure of the plant community. This complexity of interactions is further complicated by long-term precipitation cycles. Vegetative responses during the drier part of the cycle are different than responses during the wet portion of the cycle, even though similar management actions are implemented. These variations require management experience within a system to

Table 1. Habitat types of southern floodplain forests.

Wetland Type	Flood duration	Plant foods	Animal foods	Cover	Wildlife Users[a]
Open-water	Usually 12 mos.	None	Fish	None	Turtles Grebes Fish Pelicans Diving ducks
Aquatic bed Submergent	11-12 months	Seeds Browse	Fish Insects Snails	None	Turtles Fish Grebes Pelicans Diving ducks
Aquatic bed Water shield	10-12 months	Seeds Browse	Fish Insects Snails	None	Turtles Frogs Fish Gallinules Diving ducks Wood ducks
Emergent bed American lotus	8-12 months			Roosting	Snakes Frogs Wood ducks
Emergent wetland Moist-soil	Variable 1-12 months	Seeds Tubers Browse	Crayfish Insects Snails	Nesting Roosting Feeding	Waders Rails Frogs Snakes Swallows Marsh wrens Dabbling ducks Blackbirds

Table 1 (cont.)

Wetland Type	Flood duration	Plant foods	Animal foods	Cover	Wildlife Users[a]
Scrub/shrub	8-12 months	Seeds	Crayfish Insects Snails	Nesting Roosting	Snakes Frogs Waders Dabbling ducks Wood ducks
Forested					
Cypress-tupelo	6-8 months (Dec-July)	Tupelo drupes	Crayfish	Nesting Roosting	Fish Frogs
Overcup oak	4-6 months (Dec-May)	Large acorns Berries Fruits Samaras	Crayfish Small crustaceans Fingernail clams	Nesting Roosting	Frogs Snakes Woodpeckers Prothonotary warbler Tanagers Dabbling ducks Wood ducks
Pin/nuttall oak	1-6 months (Dec-May)	Small acorns Berries Fruits Samaras	Crayfish Spiders Small crustaceans	Nesting Roosting	Frogs Woodpeckers Red-shouldered hawk Dabbling ducks Wood ducks
Cherry bark/ willow oak	1-3 months (Jan-Mar)	Small acorns Berries Fruits Samaras	Spiders Small crustaceans	Nesting Roosting	Woodpeckers Red-shouldered hawk Dabbling ducks Wood ducks

[a] Only major groups are designated except for a few specific examples

Table 2. Habitat types of glacial wetlands.

Wetland Type	Flood duration	Plant foods	Animal foods	Cover	Wildlife Users
Type I — ephemeral ponds	< 1 month	Seeds Browse	Insects Fairy shrimp	None	Shorebirds Dabbling ducks
Type II — temporary ponds	< 1 month	Seeds Browse	Insects Fairy shrimp	None	Shorebirds Waders Dabbling ducks
Type III — seasonal ponds and lakes	2-4 months	Seeds Browse Tubers	Insects Snails	Nesting Roosting	Frogs Shorebirds Waders Rails Sedge wrens Dabbling ducks Diving ducks
Type IV — semi-permanent ponds and lakes	12 months	Seeds Browse	Insects Snails Crustaceans Chironomids	Nesting Roosting	Turtles Frogs Waders Grebes Rails Gallinules Pelicans Blackbirds Terns Dabbling ducks Diving ducks

Table 2 (cont.)

Wetland Type	Flood duration	Plant foods	Animal foods	Cover	Wildlife Users
Type V — permanent ponds and lakes	12 months	Seeds Browse	Insects Snails Chironomids	Nesting Roosting	Turtles Fish Diving ducks
Type VI — alkali ponds and lakes	6-12 months	Submergent seeds, Browse	Small crustaceans	None	Shorebirds Dabbling ducks Diving ducks
Type VII — fen ponds	12 months	Seeds	Snails	Nesting Roosting	Blackbirds

Table 3. A checklist of variables important in the development of management plans for wetland and riparian habitats.

Management Considerations	
Site Characteristics	**Biological Aspects of Target Species**
Climate	Chronology
Precipitation cycle	Migration
Temperature ranges	Breeding
Length of growing season	Molt
Soils	Nutritional requirements
Structure	Population size
Fertility	Migration
Topography	Breeding
Residual pesticides	Molt
Water control potential	Social behavior
Water supply/source	Foraging modes
Levees	Breeding strategies
Control structures	Significance of location
Pumps	Local
Impoundments in complex	Regional
Number	Continental
Size	Status
Juxtaposition	Endangered — rare
Plants	Recreational value
Composition	Multispecies benefits
Structure and maturity	
Seedbank	
Disturbance	
Man-induced perturbations	
Public Use	
Research and management activities	

assure continuing success in meeting management goals. Because some managers may have traditional agricultural training, they attempt to manage for monocultures of natural plants as replacements for agricultural crops. This biologically unsound approach may lead to a decline in animal diversity as well as the total number of birds using manipulated wetlands.

On breeding areas, management goals traditionally have focused on plant structure rather than on foods (Weller 1981). In contrast, wetland habitats used by migratory and wintering wildlife have been managed more for foods than for structure. Although seed production has been a traditional management goal, recent evidence has shown the additional importance of other plant foods such as tubers, browse, and rootlets. Likewise, habitat structure for animal prey and cover for the target organisms are important.

Invertebrates are of obvious importance on breeding areas, but their value at more southern locations has only recently been recognized. Management of invertebrates requires the integration of efforts to develop plant structure and litter with attempts to emulate natural water regimes (Reid 1985, Batema et al. 1985). The large array of aquatic and aquatic-related invertebrates have such a broad range of requirements that no single wetland nor single management approach will provide the diversity of invertebrates required to meet the needs of a rich avifauna. Common invertebrates may not be present within all wetlands of the same complex because water regimes or vegetative structure differ (Wiggins et al. 1980, Reid 1985). For example, organisms that respond to summer drawdowns are not present on ephemeral wetlands with spring flooding. Likewise, the composition and abundance of invertebrates in forested wetlands is dependent on time and duration of flooding as well as the type of litter available (Batema et al. 1985). These diverse invertebrate responses further emphasize the need for wetland complexes.

Once food sources have been developed, management must attempt to make foods available. Few studies have addressed the difficult problem of food availability, but experience in Missouri suggests that general patterns of avian use are based on vegetative structure and water depth (Fredrickson and Taylor 1982). Constantly changing water levels are essential to maintain good bird usage. Reduction of avian use is commonly associated with areas that are flooded too deeply, whereas few problems have been associated with water that is too shallow. The normal decrease of spring water levels in seasonal ponds results in changing avian-use patterns. Species with deep water requirements (grebes) appear early in the season. As water levels decline, great blue herons, great egrets and night herons appear. By the time mudflats are present, green-backed herons, shorebirds and grackles are in abundance. Finally, scavengers like crows and vultures forage on the foods or carrion remaining in small pools or in the last bit of drying mud. Rapid reduction in water levels on deeply flooded sites may result in fish kills. Masses of dying fish often lead to a poor public image and reduced public support for wetland managers.

SPECIFIC MANAGEMENT PRACTICES

Marsh Management

Drawdowns commonly are used to produce desired habitat conditions on prairie wetlands where there is potential for water control. Because prairie wetland complexes consist of distinct basins of variable size and permanency, management of water regimes is restricted to the larger, more permanent basins located on drainage systems. Thus, the prairie wetland manager is at the mercy of environmental conditions on the majority of areas. Uplands surrounding wetlands provide sites for nesting and foraging. Manipulations of uplands are easier and more economical than the manipulation of wetland basins to enhance habitats.

Productivity in marshes is tied to the water cycle, so the hydrologic regime determines habitat structure and productivity. Traditionally, managers have tried to control habitat structure in the larger semi-permanent marshes. The ideal cover-water interspersion is about 50-50, the "hemimarsh" of Weller and Spatcher (1965). Drier periods produce extensive litter that provide structure or nutrients for macroinvertebrate populations (Nelson and Kadlec 1984). Peak abundance of nesting pied-billed grebes, least bitterns, redheads, ruddy ducks, common moorhens, American coots, Forster's terns, black terns (*Chlidonias niger*), yellow-headed blackbirds (*Xanthocephalus xanthocephalus*), and red-winged blackbirds occurred when open water covered 50 to 70 percent of the surface area in an Iowa marsh (Weller and Fredrickson 1974). Although foods were not measured in the study, other studies have attempted to relate the abundance of food to the cover-water rations (Kaminski and Prince 1981, Murkin et al. 1982). However, no clearcut relationships were apparent between duck use and food abundance.

In uplands, fire, grazing, mowing or developing dense cover may be necessary for attracting target species. Upland sandpipers responded well to native grasslands that were grazed, burned or hayed every 2 to 3 years, but annual manipulations were detrimental (Kirsch and Higgins 1976, Kaiser 1979).

Moist-Soil Management

In the midwest, manipulated sites that produce natural foods for wetland wildlife are called moist-soil impoundments. The practice was originally described for the Illinois River Valley and has matured in recent years (Fredrickson and Taylor 1982, Havera and Bellrose 1985). The management practice is more of an art than a science. Each region, locale, impoundment, or part of an impoundment has characteristics that determine the success of management attempts. Further, uniform responses among years or among sites on the same management area are unlikely because of variations in successional stage, seasonal temperatures and rainfall, soil structure and type, drawdown duration, and topography. Once these characteristics are recognized, the potential to meet management objectives increases; consequently, the technique has wide acceptance and has been used effectively to attract a variety of wildlife (Fredrickson and Taylor 1982), not only in Missouri, but in sites as diverse as South Carolina tidal marshes, California floodplain wetlands, Hawaiian coastal wetlands, and Swedish lacustrine wetlands (Hertzman 1980). Effective use of the technique requires a basic understanding of a conceptual framework that is tempered by a manager's experience in a region or at a specific location.

Although seeds were originally the objective in moist-soil management, recent evidence suggests that tubers, rootlets, browse, invertebrates, and herpetofauna provide a wide variety of foods for birds. These diverse resources attracted over 150 avian species from 14 orders to the Ted Shanks Wildlife Area and Mingo National Wildlife Refuge, Missouri (Table 4).

Table 4. Conditions associated with avian utilization of moist-soil impoundments in Missouri.[a]

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
PODICIPEDIFORMES														
Pied-billed Grebe (*Podilymbus podiceps*)	B,M	CB	AQ				25+	X			X		X	X
CICONIIFORMES														
American Bittern (*Botaurus lentiginosus*)	B,M	CB	AQ,T				0-10			X		X	X	
Least Bittern (*Ixobrychus exilis*)	B,M	CB	AQ				0-50	X		X		X		
Great Blue Heron (*Ardea herodias*)	B,M	CB	AQ				0-60	X			X		X	X
Great Egret (*Casmerodius albus*)	B,M	CB,WB	AQ				0-50	X			X		X	X
Snowy Egret (*Egretta thula*)	B,M	CB	AQ				0-20	X			X		X	X
Little Blue Heron (*Egretta caerulea*)	B,M	CB	AQ				0-20	X			X		X	X
Cattle Egret (*Bubulcus ibis*)	B,M	CB,WB	AQ,T				0	X			X		X	X
Green-backed Heron (*Butorides striatus*)	B,M	CB	AQ				0-20	X	X		X		X	X
Black-crowned Night Heron (*Nycticorax nycticorax*)	B,M	CB	AQ				0-40	X			X		X	X
Yellow-crowned Night Heron (*Nycticorax violaceus*)	B,M	CB	AQ				0-30	X			X		X	X
White Ibis (*Eudocimus albus*)	M	CB	AQ				0-30	X		X	X		X	X
White-faced Ibis (*Plegadis chihi*)	B,M	CB	AQ				0-30	X		X	X		X	
ANSERIFORMES														
Tundra Swan (*Cygnus columbianus*)	M,W			X	X	X	25+	X			X		X	
Trumpeter Swan (*Cygnus buccinator*)	M,W			X	X	X	25+	X			X		X	
Greater White-fronted Goose (*Anser albifrons*)	M,W				X	X	0-10	X	X		X		X	
Snow Goose (*Chen caerulescens*)	M,W				X	X	0-10	X	X		X		X	
Canada Goose (*Branta canadensis*)	M,W			X	X	X	0-10	X	X		X		X	
Wood Duck (*Aix sponsa*)	B,M		AQ	X			5-25	X		X	X		X	
Green-winged Teal (*Anas crecca*)	M		AQ	X	X		5-25	X	X	X	X		X	
American Black Duck (*Anas rubripes*)	M,W		AQ,T	X	X		5-25	X	X	X	X		X	
Mallard (*Anas platyrhynchos*)	B,M,W		AQ,T	X	X		5-15	X	X		X		X	
Northern Pintail (*Anas acuta*)	M,W		AQ,T	X	X		5-25	X	X	X	X		X	
Blue-winged Teal (*Anas discors*)	B,M		AQ	X	X		5-25	X	X	X	X		X	

Table 4 (cont.)

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
Cinnamon Teal (*Anas cyanoptera*)	M		AQ	X			5-25	X			X		X	
Northern Shoveler (*Anas clypeata*)	M		AQ	X		X	5-25	X			X		X	
Gadwall (*Anas strepera*)	M,W		AQ	X		X	5-25	X			X		X	
American Wigeon (*Anas americana*)	M		AQ	X	X	X	25+	X		X	X		X	X
Canvasback (*Aythya valisineria*)	M		AQ	X			25+	X		X	X		X	X
Redhead (*Aythya americana*)	M		AQ	X	X		25+	X		X	X		X	
Ring-necked Duck (*Aythya collaris*)	M,W		AQ	X	X		25+			X	X		X	X
Lesser Scaup (*Aythya affinis*)	M,W		AQ				25+	X			X		X	X
Oldsquaw (*Clangula hyemalis*)	M		AQ				25+	X			X		X	X
Black Scoter (*Melanitta nigra*)	M		AQ				25+	X			X		X	X
White-winged Scoter (*Melanitta fusca*)	M	CB	AQ				25+	X			X		X	X
Common Goldeneye (*Bucephala clangula*)	M	CB	AQ				25+	X					X	X
Bufflehead (*Bucephala albeola*)	M	CB	AQ				5-25+	X			X		X	X
Hooded Merganser (*Lophodytes cucullatus*)	B,M,W	CB	AQ				25+	X			X		X	X
Common Merganser (*Mergus merganser*)	M	CB	AQ				25+	X		X	X		X	X
Red-breasted Merganser (*Mergus serrator*)	M		AQ				25+	X		X	X		X	X
Ruddy Duck (*Oxyura jamaicensis*)	M						25+	X			X		X	X
FALCONIFORMES														
Black Vulture (*Coragyps atratus*)	B,M						NA		X		X		X	X
Turkey Vulture (*Cathartes aura*)	B,M						NA		X		X		X	X
Osprey (*Pandion haliaetus*)	M	CB					NA	X			X		X	X
Mississippi Kite (*Ictinia mississippiensis*)	B,M		AE,T				NA			X	X	X	X	
Bald Eagle (*Haliaeetus leucocephalus*)	M,W	CB,WB					NA	X	X		X		X	
Northern Harrier (*Circus cyaneus*)	M,W	CB,WB					NA	X	X	X	X	X	X	
Northern Goshawk (*Accipiter gentilis*)	M,W	WB					NA	X	X	X	X	X	X	
Red-Shouldered Hawk (*Buteo lineatus*)	B,M	CB,WB	AQ				NA	X	X	X	X	X	X	
Red-tailed Hawk (*Buteo jamaicensis*)	B,M,W	CB,WB					NA			X	X	X	X	

Table 4 (cont.)

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
Rough-legged Hawk (*Buteo lagopus*)	M,W	WB					NA			X	X	X	X	
Golden Eagle (*Aquila chrysaetos*)	M,W	WB					NA			X	X	X	X	
American Kestrel (*Falco sparverius*)	B,M,W	WB					NA			X	X	X	X	
Peregrine Falcon (*Falco peregrinus*)	M,W	WB					NA			X	X	X	X	
GALLIFORMES														
Wild Turkey (*Meleagris gallopavo*)	B, W		T	X	X	X	0			X	X	X	X	
Northern Bobwhite (*Colinus virginianus*)	B, W		T	X		X	0			X	X	X	X	
GRUIFORMES														
Yellow Rail (*Coturnicops noveboracensis*)	M		AQ,T	X			0- 5			X		X		
King Rail (*Rallus elegans*)	B,M		AQ,T				0-10	X	X	X		X		
Virginia Rail (*Rallus limicola*)	M		AQ,T				0-10	X		X		X		
Sora (*Porzana carolina*)	M		AQ,T	X			0-25	X		X		X		
Purple Gallinule (*Porphyrula martinica*)	M		AQ,T			X	0-25+	X		X	X	X	X	
Common Moorhen (*Gallinula chloropus*)	B,M		AQ,T			X	20+	X		X	X	X	X	
American Coot (*Fulica americana*)	B,M		AQ			X	20+	X		X	X	X	X	
Sandhill Crane (*Grus canadensis*)	M		AQ,T	X	X		0	X	X		X		X	X
CHARADRIIFORMES														
Black-bellied Plover (*Pluvialis squatarola*)	M		AQ				0		X		X		X	X
Lesser Golden Plover (*Pluvialis dominica*)	M		AQ,T				0		X		X		X	X
Killdeer (*Charadrius vociferus*)	B,M		AQ,T				0		X		X		X	X
Semipalmated Plover (*Charadrius semipalmatus*)	M		AQ,T				0		X		X		X	X
Greater Yellowlegs (*Tringa melanoleuca*)	M		AQ				0- 5	X			X		X	X
Lesser Yellowlegs (*Tringa flavipes*)	M		AQ				0- 5	X			X		X	X
Solitary Sandpiper (*Tringa solitaria*)	M		AQ				0- 5	X	X		X		X	
Willet (*Catoptrophorus semipalmatus*)	M		AQ				0- 5	X	X		X		X	X
Spotted Sandpiper (*Aetitus macularia*)	B,M		AQ				0		X		X		X	X

Table 4 (cont.)

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
Upland Sandpiper (*Bartramia longicauda*)	M		AQ,T				0		X		X		X	
Sanderling (*Calidris alba*)	M		AQ,T				0		X	X			X	X
Semipalmated Sandpiper (*Calidris pusilla*)	M		AQ,T				0		X		X		X	X
Western Sandpiper (*Calidris mauri*)	M						0				X		X	X
Least Sandpiper (*Calidris minutilla*)	M		AQ				0		X		X		X	X
White-rumped Sandpiper (*Calidris fuscicollis*)	M		AQ				0	X	X		X		X	
Baird's Sandpiper (*Calidris bairdii*)	M		AQ				0		X		X		X	
Pectoral Sandpiper (*Calidris melanotos*)	M		AQ				0- 2	X	X		X		X	X
Dunlin (*Calidris alpina*)	M		AQ				0	X			X		X	
Stilt Sandpiper (*Calidris himantopus*)	M		AQ				0- 2	X			X		X	
Short-billed Dowitcher (*Limnodromus griseus*)	M		AQ				2-10	X			X		X	X
Long-billed Dowitcher (*Limnodromus scolopaceus*)	M		AQ				2-10	X			X		X	X
Common Snipe (*Gallinago gallinago*)	M		AQ				0- 2	X	X	X	X	X	X	X
American Woodcock (*Scolopax minor*)	B,M		AQ				0		X	X	X	X		
Wilson's Phalarope (*Phalaropus tricolor*)	M		AQ				5-25+	X			X		X	X
Forster's Tern (*Sterna forsteri*)	M	CB						X						X
COLUMBIFORMES														
Mourning Dove (*Zenaida macroura*)	B,M,W						NA		X				X	X
CUCULIFORMES														
Yellow-billed Cuckoo (*Coccyzus americanus*)	B		T				NA			X	X	X	X	
STRIGIFORMES														
Common Barn-Owl (*Tyto alba*)	B, W	CB,WB					NA			X	X	X	X	
Great Horned Owl (*Bubo virginianus*)	B, W	CB,WB					NA			X	X	X	X	
Barred Owl (*Strix varia*)	B, W	WB	AQ,T				NA			X	X	X	X	
Short-eared Owl (*Asio flammeus*)	W	WB					NA			X	X	X	X	

Table 4 (cont.)

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
CAPRIMULGIFORMES														
Common Nighthawk (*Chordeiles minor*)	B,M		AE				NA			X	X	X	X	
APODIFORMES														
Chimney Swift (*Chaetura pelagica*)	B	CB	AE				NA			X	X	X	X	
Ruby-throated Hummingbird (*Archilochus colubris*)	B,M						NA			X	X	X	X	
CORACIIFORMES														
Belted Kingfisher (*Ceryle alcyon*)	B, W	CB					NA	X			X		X	
PICIFORMES														
Northern Flicker (*Colaptes auratus*)	M		T								X		X	X
PASSERIFORMES														
Eastern Phoebe (*Sayornis phoebe*)	B		AE,T				NA			X	X	X	X	
Great Crested Flycatcher (*Myiarchus crinitus*)	B		AE,T				NA			X	X	X	X	
Eastern Kingbird (*Tyrannus tyrannus*)	B		AE,T				NA			X	X	X	X	
Purple Martin (*Progne subis*)	B		AE				NA			X	X	X	X	
Tree Swallow (*Tachycineta bicolor*)	B		AE				NA			X	X	X	X	
Northern Rough-winged Swallow (*Stelgidopteryx serripennis*)	B		AE				NA			X	X	X	X	
Bank Swallow (*Riparia riparia*)	B		AE				NA			X	X	X	X	
Cliff Swallow (*Hirundo pyrrhonota*)	B		AE				NA			X	X	X	X	
Barn Swallow (*Hirundo rustica*)	B		AE				NA			X	X	X	X	
Blue Jay (*Cyanocitta cristata*)	B,M,W		T				NA			X	X	X	X	
American Crow (*Corvus brachyrhynchos*)	B,M,W	CB	AQ,T	X			NA		X		X		X	X
Carolina Chickadee (*Parus carolinensis*)	B,M,W		T				NA			X	X	X	X	
Tufted Titmouse (*Parus bicolor*)	B,M,W		T				NA			X	X	X	X	
Carolina Wren (*Thryothorus ludovicianus*)	B, W		T				NA			X	X	X	X	
Winter Wren (*Troglodytes troglodytes*)	W		T				NA			X		X		

Table 4 (cont.)

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
Sedge Wren (*Cistothorus platensis*)	B,M		AQ,T				0			X		X		
Marsh Wren (*Cistothorus palustris*)	B,M		AQ,T				0			X		X		
Loggerhead Shrike (*Lanius ludovicianus*)	B, W	WB,CB	T				NA			X	X	X	X	
Eastern Bluebird (*Sialia sialis*)	B,M,W		T				NA			X	X	X	X	
Gray Catbird (*Dumetella carolinensis*)	B		T				NA			X	X	X	X	
Northern Mockingbird (*Mimus polyglottos*)	B,M,W		T				NA			X	X	X	X	
Water Pipit (*Anthus spinoletta*)	M		AQ,T				NA			X	X	X	X	
European Starling (*Sturnus vulgaris*)	W		T	X			NA			X	X	X	X	X
White-eyed Vireo (*Vireo griseus*)	B		T				NA			X	X	X	X	
Bell's Vireo (*Vireo bellii*)	B		T				NA			X	X	X	X	
Warbling Vireo (*Vireo gilvus*)	B		T				NA			X	X	X	X	X
Blue-winged Warbler (*Vermivora pinus*)	B		T				NA			X		X		
Yellow-rumped Warbler (*Dendroica coronata*)	B,M		T				NA			X		X		
Common Yellowthroat (*Geothlypis trichas*)	B		AQ,T				NA			X	X	X	X	
Yellow-breasted Chat (*Icteria virens*)	B		T				NA			X		X		
Indigo Bunting (*Passerina cyanea*)	B		T				NA			X		X		
Dickcissel (*Spiza americana*)	B		T				NA			X		X		
Rufous-sided Towhee (*Pipilo erythrophthalmus*)	W		T				NA			X	X	X	X	
American Tree Sparrow (*Spizella arborea*)	W		T	X			NA			X	X	X	X	
Savannah Sparrow (*Passerculus sandwichensis*)	W		T	X			NA			X	X	X	X	
Grasshopper Sparrow (*Ammodramus savannarum*)	B, W		T	X			NA			X	X	X	X	
LeConte's Sparrow (*Ammodramus leconteii*)	B, W		T	X			NA			X	X	X	X	
Field Sparrow (*Spizella pusilla*)	B		T	X			NA			X		X		
Lark Sparrow (*Chondestes grammacus*)	B,M		T	X			NA			X	X	X	X	
Fox Sparrow (*Passerella iliaca*)	M		T	X			NA			X	X	X	X	
Song Sparrow (*Melospiza melodia*)	B,M		T	X			NA			X	X	X	X	
Swamp Sparrow (*Melospiza georgiana*)	B,M		T	X			NA			X		X		

Table 4 (cont.)

Orders and Species	Season	Foods					Water Depth (cm)	Openings		Vegetative Cover				
		Verts	Inverts	Seeds	Tubers	Browse		Water	Mud	Rank	Short	Dense	Sparse	None
White-throated Sparrow (*Zonotrichia albicollis*)	W		T	X			NA			X		X		
White-crowned Sparrow (*Zonotrichia leucophrys*)	W		T	X			NA			X		X		
Harris' Sparrow (*Zonotrichia querula*)	M		T	X			NA			X	X	X	X	
Dark-eyed Junco (*Junco hyemalis*)	W		T	X			NA			X	X	X	X	
Red-winged Blackbird (*Agelaius phoeniceus*)	B,M,W		T	X			0-25+			X	X	X	X	
Eastern Meadowlark (*Sturnella magna*)	B, W		T	X			0			X	X		X	
Common Grackle (*Quiscalus quiscala*)	B,M,W		AQ,T	X			0				X		X	X
Brown-headed Cowbird (*Molothrus ater*)	B, W		T	X			0				X		X	X
Orchard Oriole (*Icterus spurius*)	B		T				NA			X	X	X	X	
Purple Finch (*Carpodacus purpureus*)	W		T	X			NA			X	X	X	X	
American Goldfinch (*Carduelis tristis*)	B,M,W		T	X			NA			X	X	X	X	

[a] Symbols

B = during breeding season, includes nesting and foraging.
M = migrants, may be present for several months or only a few days.
W = winter, present in December and January.
AE = aerial foraging on insects.
AQ = aquatic and aquatic-associated invertebrates.
T = terrestrial invertebrates.
WB = warmblooded.
CB = coldblooded.
NA = not applicable.

Moist-soil management attracts such diverse bird populations because manipulations to control and enhance vegetation result in diverse habitat conditions that are attractive to wetland and non-wetland birds alike. Of the 153 species listed in Table 4, 14 are residents, 22 are present exclusively as breeders, 9 are present in winter, 47 are migrants, 32 are both migrants and breeders, 16 are migrants and winter residents, and 13 are breeders, migrants, and winter as well. A general seasonal pattern of moist-soil use by orders of birds is apparent. Passeriforms are the most numerous birds present exclusively in winter or as breeders, whereas birds that are exclusively migrants are primarily charadriiforms. The ciconiiforms are predominantly migrants and breeders.

The value of diverse foods that range from vertebrates to green browse in attracting wildlife is obvious. Thirty-seven species feed on vertebrates. Cold-blooded vertebrates are particularly important for 30 species. Warm-blooded vertebrates are important prey for 14 species and, in some cases, the aggregations of birds become the prey for raptors. Interestingly, only 42 species are attracted by seeds, yet management objectives in the recent past were primarily for seeds or other vegetative parts. Fifteen species use tubers and 13 use browse. The recent recognition of the importance of invertebrates is supported by the number of birds that consume invertebrates from moist-soil impoundments in Missouri where invertebrates are important foods for 129 species.

Although aquatic invertebrates are important foods, the value of terrestrial forms that become available during flooding or on vegetation above the water surface cannot be dismissed. Management that enhances the production and availability of invertebrates offers unique opportunities to provide resources to species not normally identified with wetlands. Bats and birds such as nighthawks, swifts and swallows that forage on flying insects often respond to emerging midges or other insects that occur in great abundance in wetlands.

Managers should be particularly cognizant of flooding conditions. Only 25 of the species in Table 4 can effectively use water depths greater than 25 cm (10 in.), but 10 of these regularly use depths less than 25 cm. We reiterate that shallow waters provide the best foraging conditions for most species, and costs for flooding and development can be reduced considerably when less water is required. A mix of habitat conditions ranging from open water and mudflats to dense, rank vegetation attracts the greatest diversity of organisms. In general, certain bird orders have rather uniform habitat requirements—rails need dense, robust cover, whereas shorebirds require open habitats with sparse cover.

One of the concerns in meeting management objectives relates to how management for a target species affects use by another group of birds. Obviously, some conditions attractive to one group are not compatible with the needs of another (e.g., rails vs. shorebirds). Nevertheless, some management scenarios are compatible. Even when the target groups were either shorebirds or rails in Missouri, waterfowl numbers remained constant (Rundle and Fredrickson 1981). However, the composition of waterfowl using the area managed for shorebirds and rails shifted toward species such as pintails and blue-winged teal that are attracted to shallowly flooded sites. Thus, moist-soil management has high potential to meet the needs of nongame wildlife on areas where game animals are primary target species.

The impacts of rodents such as beaver (*Castor canadensis*) and muskrats (*Ondatra zibethicus*) on water control structures, levees, and vegetation must be considered. Certain configurations of structures and levees reduce the impacts of these burrowing mammals and result in major savings in management costs (Fredrickson and Taylor 1982, Buech 1985).

Riparian Habitat Management

Estimating how alterations will affect riparian habitats is essential for successful protection and management of nongame wildlife. In Iowa, the most common alterations were timber removal, grazing and stream-channel realignment. These practices converted woodlands to open communities dominated by herbaceous vegetation (Grier and Best 1980). Loss of forested habitat and changes in composition of bird communities were documented in southeastern Missouri when rivers were channelized (Fredrickson 1979). Because each species has a preferred habitat, the type of perturbation determines whether the habitat changes will be detrimental. Species such as house sparrows (*Passer domesticus*), American robins (*Turdus migratorius*) and European starlings that are adapted to urban situations were affected least by alterations of natural riparian forests (Stauffer and Best 1980). Specialists that require mature forests are most susceptible to alterations.

Control of activities that cause habitat alterations and create more open habitats have high importance for agencies, but implementation of protective action has met with varying degrees of success. For example, in California protection of riparian corridors had limited success in the Sacramento River Valley even though the value of the corridors was recognized (Burns 1978). These responses identified the value of public ownership to assure protection, thus emphasizing acquisition or presevation of habitats rather than actual manipulation.

Riparian management may include silvicultural practices, but special restrictions should be applied. For example, in Missouri Ozark forests, special considerations include prohibition of grazing, even-aged timber management, broad-scale application of insecticides and herbicides, and skidding practices that run parallel to or in the streambed. These restrictions apply to corridors along permanent rivers and streams, spring branches, and intermittent streams with permanent pools. Potential timber practices should not jeopardize the natural riparian community. Uneven-aged and single tree removal methods may be applied if timber is harvested (R. Kirkman, pers. comm.).

Other attempts at manipulations have been in progress for years. Planting trees is one of the oldest programs to enhance riparian zones and has been used by the Tennessee Valley Authority (TVA) since 1935 (Bates et al. 1978). Initially, plantings were for mosquito control. After 32 years, anopheline mosquitoes were reduced five-fold in comparison to numbers found in adjacent open herbaceous habitats (Bates et al. 1978). Nongame wildlife benefited from the planting because habitat diversity increased around TVA reservoirs.

In Arizona, cuttings from several riparian trees were started in greenhouses and then planted after the invading salt cedar (*Tamarix chinensis*) was removed (Anderson et al. 1978). Cuttings that were watered through the growing and dormant seasons had better survival and grew more than unwatered trees. Insect populations also were greater on watered sites and provided foods for insectivores. Seed-eating birds increased dramatically and were most abundant in the revegetated sites where seed-producing annuals grew in dense patches. Although bird response as measured by numbers was good immediately after the manipulations, the long-term reestablishment of riparian forests is a slow process with many pitfalls (Anderson et al. 1978).

CASE HISTORIES

Shorebirds

Because few shorebirds breed in Missouri, shorebird management is primarily oriented toward providing habitats during migration. Most shorebirds use open habitats or those with only sparse cover. These requirements dictate that sites with management potential must either have basins with more prolonged flooding that may be drained to form mudflats, or vegetational control such as fire or mechanical means before shallow flooding is initiated. A variety of migrant shorebirds responded to manipulations from March to November. Many northern nesting charadriiforms begin the southward passage through Missouri by midsummer, just a few weeks after the major northward movement terminates. In most cases, shorebirds consistently responded to the creation of ideal habitat conditions in a matter of hours (Rundle 1980). Drawdowns on sites with prolonged flooding attracted species that normally forage on mudflats. Drawdowns must be timed to meet the migratory chronology of each species. When waters receded gradually, new mudflats were exposed continually and provided suitable habitats for extended periods.

Dewatering of moist-soil sites for late summer and early fall migrants in Missouri is often difficult because few permanent water bodies are present, vegetation develops rapidly in mudflats, and permanent water may be choked with aquatic vegetation. Discing and reflooding proved to be an effective alternative for shorebird management. Shallow discing is best because crushing and partial burial of vegetation resembles the natural process of senescence. When the resulting litter is flooded, invertebrate response is immediate and foraging birds appear in numbers within a few days. Because of Missouri's long growing season, sites disced early in the season were revegetated rapidly and soon became unattractive to most shorebirds.

Attracting specific shorebird species requires an understanding of their habitat needs. Time-budget studies of five shorebirds and vegetation samples from foraging sites on Mingo Refuge, Missouri (Rundle 1980) identified specific needs for each species (Fig. 1). Lesser yellowlegs use visual cues for feeding while moving constantly in shallow water (Fig. 1). In contrast, common snipe are tactile feeders that have low mobility and use the interface between mud and water.

These data can be placed in a model (Fig. 2) to enable a manager to visualize the niches utilized by each species as well as the overlap of habitats needed to attract all five shorebirds. By understanding habitat structure and the chronology of migration, managers can attract shorebirds to sites where the viewing public can enjoy large numbers or several species at the same time (Reid et al. 1983). For example, a manipulation (drawdown or flooding) might be timed so that an Audubon Society group could enjoy seeing a variety of shorebirds on a predetermined weekend. Further, ideal placement of viewing areas is facilitated by information on microhabitat needs of target species.

Rails

Rails require a very different management scheme than shorebirds, because they select dense, rank vegetation. Like shorebirds, most rails are migrants in Missouri, except for the king rail and common moorhen which nest where suitable

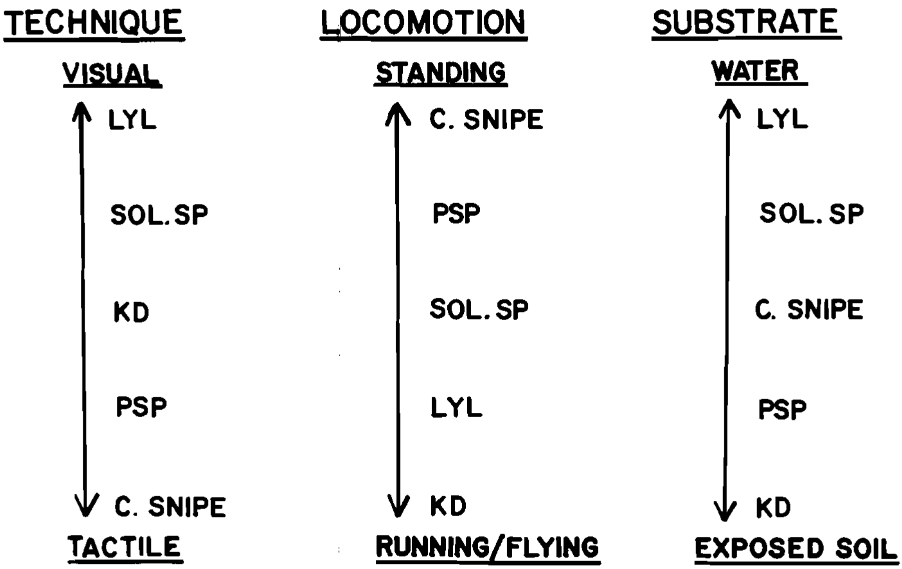

Fig. 1. Differences in foraging behavior of 5 species of shorebirds (LYL—lesser yellowlegs, PSP—pectoral sandpiper, KD—killdeer, Sol.SP—solitary sandpiper, C. snipe—common snipe). After Rundle (1980).

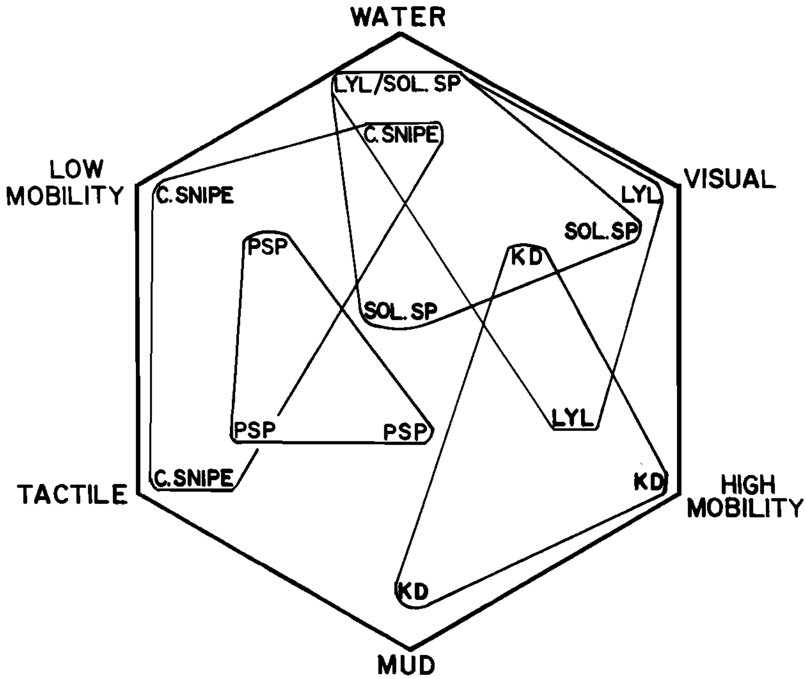

Fig. 2. Foraging niches of 5 species of shorebirds (LYL—lesser yellowlegs, PSP—pectoral sandpiper, KD—killdeer, Sol.SP—solitary sandpiper, C. snipe—common snipe) based on feeding substrates, feeding method, and mobility (Rundle 1980).

habitat conditions are present. Migrant soras mainly consume seeds and few invertebrates, whereas Virginia rails and king rails consume more animal than plant foods (Meanley 1955, Horak 1970, Rundle and Sayre 1983, Sayre and Rundle 1984). Soras, Virginia rails, and king rails are easier to attract with habitat management in fall than spring because rank vegetation is readily available immediately following the growing season. Decomposition, snow, and heavy use by waterfowl usually destroy the vertical structure of annual plants before spring.

Inland-breeding Rallidae display a continuum of preferred water depths at nest sites (Fig. 3). This suggests that stabilized water regimes may be catastrophic to a diverse breeding-rail community. King rails and black rails (*Laterallus jamaicensis*) nest in the driest sites, and these species probably are most affected by drainage and conversion of habitats to agriculture. Natural swales within manipulated sites are important to maintain, as they are predictable places of prey concentrations for king and Virginia rails during drying of natural basins or drawdowns of managed sites.

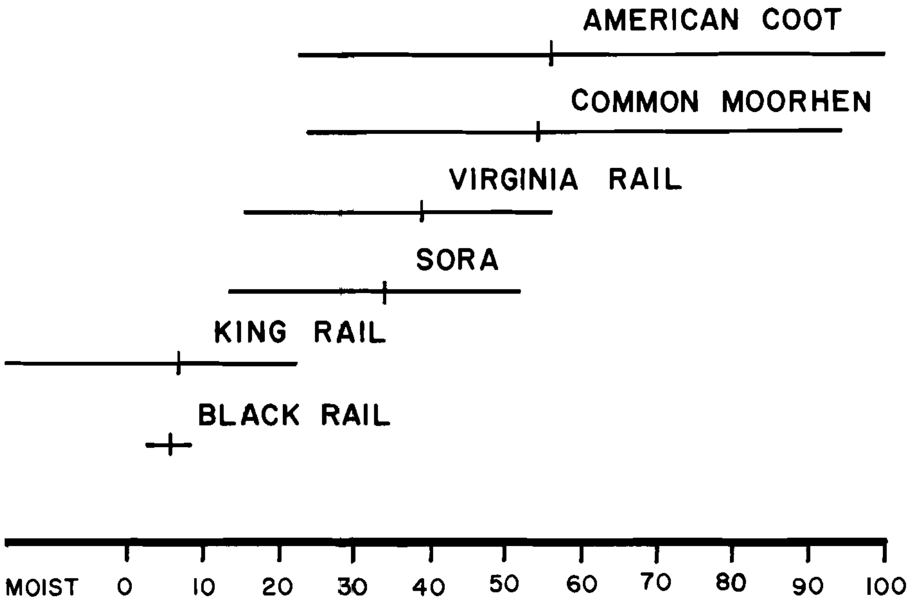

Fig. 3. Water depth in cm at nest sites of inland-breeding Rallidae. Vertical line indicates mean; horizontal line indicates range of depths.

Wading Birds

Wading birds such as little blue herons, cattle egrets and snowy egrets nest in Missouri's southeastern lowlands in mixed-species concentrations reaching well over a thousand nests. Riparian areas in southeastern Missouri, as well as along streams statewide, provide excellent nesting habitats. Wetlands provide foraging sites to meet breeding-season food requirements, when most wading birds consume cold-blooded vertebrate or macroinvertebrate prey. Slow drawdowns concentrate prey within the heron species' foraging range, whereas rapid drawdowns fail to emulate the natural recession of waters which makes fish or crayfish more susceptable to predation. Rapid oxygen depletion also may result from rapid drawdowns when temperatures reach 30°C or more. Protection of nesting habitat and the continuing availability of wetland foods throughout the breeding and post-breeding season is essential to assure breeding success of waders within riparian and wetland complexes.

INTEGRATED MANAGEMENT

Although some organisms may have a high priority for management, their long-term needs likely will be met more effectively if the system is managed as a habitat mosaic to maintain productivity. Discing to reduce cover followed by shallow flooding for shorebirds in summer can reduce woody vegetation. The

Table 5. Suggested management of a theoretical wetland complex of 5 wetland basins in Missouri, including manipulations for the target species in each basin and expected habitat conditions resulting from management and successional changes.[a]

Management Unit	Spring I	Summer I	Fall I	Winter I	Spring II	Summer II	Fall II
I							
Target species	B, M—waders Late M—ducks	B—waders duck (broods)	M—ducks	W—ducks, geese passerines, raptors	Early M—ducks, shorebirds	B—turkeys, passerines	M—rails, ducks
Manipulation	Late PD	CD	Late SF	MF	Early CD	None	Early SF
Plant condition	10-0W,10-U,80-MT	GE	90-A,10-P	5-0W,45-0,50-MT	10-0W,15-U,75-MT	GE	60-A,40-P
II							
Target species	B,M—waders, rails	M—shorebirds	M—ducks,	W—ducks geese	M—ducks, waders	B—waders	M—ducks, rails
Manipulation	Mid CD	disc deep, I	Late RF	MF	B—waders Mid PD	Early CD	Mid SF
Plant condition	10-0W,60-U,30-MT	GE	100-A	20-U,80-MT	5-U,95-MT	GE	90-A,10-P
III							
Target species	M—shorebirds, ducks	B-turkeys, waders	M—ducks, rails	W—ducks, raptors geese, passerines	M—grebes, ducks, waders, shorebirds	B—waders, duck (broods)	M—ducks
Manipulation	Early PD	Early CD	Early SF	MF	Early PD	Late CD	Late SF
Plant condition	90-0W,10-MT	GE	100-A	10-0W,10-U,80-MT	40-0W,5-U,55-MT	GE	95-A,5-P
IV							
Target species	M—rails ducks	B—turkey, rails passerines,	M—ducks, rails	W—duck, geese raptors, passerines	B—ducks waders, rails	B—turkey passerines	M—ducks, shorebirds
Manipulation	Early CD	None	Early SF	MF	Late CD	SD,RF	RF
Plant condition	5-0W,75-U,20-MT	GE,	40-A,50-P,10-W	50-U,50-MT	40-U,60-MT	GE	50-W,40-A,10-P
V							
Target species	M—ducks shorebirds	B—turkeys, passerines	M—ducks, shorebirds	W—ducks, geese	M—shorebirds, B—waders	B—turkey, passerines	M—ducks, rails
Manipulation	Late CD	None	SD,SF,MF		Mid CD	None	Mid SF
Plant condition	30-0W,5-U,65-MT	GE	70-0W,20-A,10-P	70-0W,30-MT	90-0W, 10-MT	GE	80-A, 20-P

Table 5 (cont.)

[a]Acronyms for:

Bird chronology
- B = breeding
- M = migration
- W = wintering

Water Regime or technique
- PD = partial drawdown
- CD = complete drawdown
- MF = maintain flooding
- SF = shallow reflood
- I = irrigation (soils brought to saturation and then drained)
- DD = deep disc
- SD = shallow disc

Plants
- A = annuals
- P = perennials
- W = woody

Plant structure or process
- GE = germination and establishment
- MT = matted
- U = upright
- OW = open water

Numbers indicate percentages of specified plant structures.

effects of discing stimulate the growth of robust annuals with good seed production in the next growing season. Both seeds and cover are required by rails, whereas the dense cover is not attractive to shorebirds. Continued management for robust vegetation will increase the presence of perennials that maintain their upright structure much better than annual plants. Rails are attracted to shallow fall flooding of robust plants with good upright structure. If flooding of perennials occurs before senescence, plant vigor is reduced. As the perennials decline, seed production from annual plants increases in the next growing season. Drawdowns scheduled for waders throughout the breeding and post-breeding season also provide foraging sites for shorebirds as mudflats appear.

Suggested management practices on a manipulated complex with five impoundments provide options to meet such diverse goals (Table 5). Although we prefer seven or more units to optimize management options in a man-made complex, for purpose of demonstration only five units are included in Table 5. Our approach identifies the target species, manipulation and potential plant condition resulting from the manipulations for the target species. For management purposes, the biological aspects identified in Table 3 are used to determine the timing and chronology of manipulations. Note that in many cases, target species can include game and nongame species (Table 5) within the same season. This multi-species aspect is one of the great advantages of moist-soil management. Suggested manipulations (Table 5) influence water levels, change vegetative structure or modify the soil surface. The plant condition (Table 5) identifies the types of vegetation expected (annuals, perennials, or woody) and the cover condition (open water, matted, or upright).

Table 5 provides guidelines for obtaining desired responses to meet the demands of target species. For example, Unit 2 has breeding and migrant rails and waders as a target group in Spring I. Because these species are not early migrants, a mid-season complete drawdown is scheduled to provide habitats and associated foods. The plant condition indicates that 60 percent of the area is in upright vegetation, a condition desirable for rails. This abundance of perennials or woody cover indicates that some control measures are probably desirable.

The mid-season complete drawdown sets the stage to allow adequate drying so that the deep discing necessary to control woody and perennial vegetation can be scheduled. The manipulation (discing) should be timed to provide optimum benefits for target species. In Missouri, soils dry so rapidly after summer discing that germination and establishment of desired plants are often limited. An irrigation will assure that vegetative response will be acceptable. The deep discing stimulates annuals and reduces perennials and some tuber-producing plants, hence the plant response in Fall I is predominantly annuals that are good seed-producers. Seeds not only meet the needs of the target species (ducks) but many other seed-eaters as well.

Annual vegetation breaks down readily, so by winter, 80% of the cover already has become matted. This condition precludes rails and most shorebirds as target species. However, migrant ducks and breeding and migrant waders find these conditions acceptable in Spring II. A mid-season partial drawdown concentrates invertebrate prey for target ducks and waders. An early complete drawdown in Summer II corresponds with the breeding season for waders and concentrates foods in puddles. In Fall II, the timing of reflooding is scheduled to coincide with the rail migration and for mid-fall duck movements. The abundance of upright annual vegetation in fall at the time of flooding provides suitable rail habitat throughout the period of their peak use.

Within any season (columns, Table 5) every management area should have a variety of target species because of different vegetative conditions resulting from site characteristics, succession and manipulations. The timing of manipulations in man-made complexes allows the manager to emulate cover and food conditions in large natural wetland complexes. Such an approach leads to continuing wetland productivity while optimizing benefits from scarce management dollars.

CONCLUSIONS

Managers are faced with many challenges to meet the diverse needs of species whether game or nongame. In some cases, even species that may not be recognized as wetland forms receive benefits from wetland managment. An appreciation of the current status of wetlands and the importance of the managed habitat at the local, regional and continental level provides guidance for acquisitions, development requirements and manipulations. Remnant wetland habitats and their natural hydrology should be protected from destruction and modification. A conceptual understanding of wetland function, particularly the importance of dynamic hydrological regimes, allows a manager to function more effectively on an area or a specific impoundment. Development of man-made complexes has the potential to provide some measure of replacement for naturally occurring wetlands that have been lost or modified. Timing of manipulations to match plant life-history strategies, and managing to meet cover and food requirements of target animal species result in maximum benefits from management investments. Additional information on the ecology of plants and nongame species will enable managers to meet the needs of wildlife as wetland areas continue to be lost or degraded.

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A Framework for Nongame Management in Midwestern Forests

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Abstract: We present a framework to be used in developing a management plan that incorporates nongame habitat considerations into the forest management process. Depending on the desires of the landowner, one of four management approaches can be used in designing a wildlife management plan. These include management for a featured species, for indicator species, for whole guilds, or for maximum diversity. The planned management may be as simple or complex as needed to meet the goals of the landowner, but it always should be followed by a monitoring process to evaluate the success or failure of the management plan. Monitoring should measure changes in both wildlife populations and vegetation (including special habitat features) of the forest stands. The development of a forest management plan that includes consideration of nongame as well as the more traditional game species can be a complex undertaking. As we move into the age of forest and wildlife management by computer, new tools will become available for both the agency and consulting wildlife manager to aid in the consideration of all wildlife species in the management of forest resources.

Many articles are available concerning options and strategies for managing wildlife in midwestern forests. The emphasis of these publications has shifted from more traditional game management to the broader area of wildlife management. A decade ago, interest focused on white-tailed deer (*Odocoileus virginianus*), wild turkey (*Meleagris gallopavo*), or gray squirrel (*Sciurus carolinensis*) management in these forests. Today, with recent legislation and increasing public interest in nonconsumptive uses of wildlife, many forest managers are faced with the challenge of maintaining populations of all wildlife species. Although most wildlife biologists are well-versed in the field of game management, the complex issue of managing all kinds of forest wildlife means that they must become experts at nongame management as well.

In recent years, publications on nongame have provided much information on natural diversity, community structure, species distribution, and habitat suitability. Even though we are beginning to understand more about the needs of a wide variety of species, the profusion of publications also has resulted in some confusion. There appear to be conflicts between the goals of different nongame management strategies or between nongame management and management for other forest resources. The following are examples of conflicts that have arisen:

1. The positive aspects od edge *vs.* the negative aspects of type conversion that may be necessary to create edge.

2. The positive aspects of habitat diversity (especially microhabitats within a habitat) *vs.* the negative aspects of forest fragmentation.

3. The positive aspects of snags and other dead and dying trees *vs.* the introduction of insects or diseases and the impacts on fuelwood or salvage wood production.

4. The positive aspects of hardwoods and mast production *vs.* the higher productivity and better markets for softwoods.

The wide variety of forest types in the Midwest and the various age and condition classes that can be found in all these types make the task of developing specific management guidelines for nongame forest wildlife a complex undertaking beyond the scope of this paper. However, some general concepts and management opportunities for nongame wildlife in forest ecosystems are worthy of discussion. Our main objective is to propose a systematic approach to forest wildlife management that will assist midwestern forest managers and landowners in developing management plans. This approach to nongame management in forested habitats allows flexibility in goal setting and plan implementation, and aids the enhancement of habitats for many nongame species.

FORESTS OF THE MIDWEST

The variety of habitats found in midwestern forests is great. They encompass bottomland hardwoods, upland hardwoods and conifers, and heterogeneity is increased by land-use patterns. These patterns create some areas of very fragmented forests, while other areas have low human populations and large tracts of contiguous forests.

Total land and water area in Minnesota, Iowa, Missouri, Wisconsin, Illinois, Michigan, Indiana, and Ohio is 330 million acres (133.6 million ha), of which 78 million acres (31.6 million ha), or 24 percent, is classed as forestland. Most of the forestland, 68 million acres (27.5 million ha), or 87 percent, is in non-federal ownership. The nonforested acreage consists of 170 million acres (68.8 million ha) in cropland, pastureland, and rangeland; 43 million acres (17.4 million ha) in lakes, ponds, and waterways; and 39 million acres (15.8 million ha) in other classes (U.S. Forest Service 1984a).

Hardwoods dominate the commercial forests in this eight-state region, with 23 billion cubic feet (bcf) (644 million m^3) in sawtimber, 26 bcf (728 million m^3) in poletimber, and 8 bcf (224 million m^3) in rough, rotten, and dead timber. In contrast, coniferous forest volumes for the region include only 6 bcf (168 million m^3) in sawtimber, 6 bcf (168 million m^3) in poletimber, and one bcf (28 million m^3) in rough, rotten, and dead timber.

The midwestern forests include many types, subtypes, and age classes. The major forest types in the area are shortleaf pine, jack pine, red pine, black spruce, aspen, oak-hickory, northern hardwoods, and lowland hardwoods. The following brief descriptions of each type are based on information from U.S. Forest Service (1973), Ohmann (1979), and Eyre (1980).

Shortleaf Pine (*Pinus echinata*)

Although shortleaf pine is much more common in the south where it occurs in stands with loblolly pine (*Pinus taeda*), shortleaf pine extends its range into the southern portion of the midwest. Here, shortleaf pine occurs as an early successional species on thin, rocky, and droughty mountain soils. This pine is a light-demanding species and is replaced by a broad spectrum of hardwoods if succession is unchecked. Even-aged management usually is used with clearcutting and some form of hardwood control to reduce competition. Prescribed burning often is used to prepare the seedbed.

Jack Pine (*Pinus banksiana*)

Jack pine is best adapted to sandy soils and is found on more than 2 million acres (0.8 million ha) in the Lake States. Jack pine is a relatively short-lived species that often establishes itself on burned areas. Jack pine usually is managed by even-aged techniques. Rotation age is usually 50-70 years. If a seed source is available, natural regeneration will occur when mineral soil is exposed. Jack pine is a temporary type on more productive soils and will be replaced by shade-tolerant species.

Red Pine (*Pinus resinosa*)

Before the extensive logging around the turn of the 20th century and widespread fires after logging, red pine was an important component of the Lake States pine forests. Since that time, however, natural regeneration on sites previously supporting red pine has led to the development of aspen (*Populus* spp.) and jack pine stands. Current red pine stands, which cover 1.3 million acres (0.5 million ha), are the result of intensive even-aged management, with clearcutting, machine site preparation, and seeding or planting for regeneration.

Black Spruce (*Picea mariana*)

Black spruce occurs mainly in pure stands on organic soils of old lake beds, but will occur in mixed stands or in stands with a shrub understory on mineral soils. Black spruce is a slow-growing, shallow-rooted species and, in the absence of fire, is succeeded by northern white-cedar (*Thuja occidentalis*) and balsam fir (*Abies balsamea*). Black spruce is well suited to even-aged management systems for pulpwood. Uneven-aged management for Christmas tree production is applicable on poor sites.

Aspen (*Populus* spp.)

Aspen occurs on about 13 million acres (5.3 million ha) in the Lake States. Aspen grows on a wide variety of soils and sites. Growth rate, vertical diversity of the stand, and many other stand characteristics vary with soil fertility, available moisture, and past land-use patterns. Aspen is an early successional, light-demanding species that reproduces poorly under its own shade. Aspen is replaced by more shade-tolerant species where ecological succession is not interrupted by fire, cutting, or wind. Aspen stands are best reproduced by clearcutting on a 35-50 year rotation. Aspen is susceptible to several diseases, and longer rotations provide ample opportunities for cavity-nesting wildlife species.

Oak-hickory (*Quercus-Carya* spp.)

The oak-hickory forest type is very extensive, covering approximately 120 million acres (48.6 million ha). Oak-hickory forests occur over a wide range of soil moisture, geographic, and climatic conditions. This variety of conditions is expressed in a wide variation in stand composition, growth rate, and understory characteristics.

Although even-aged management with clearcutting techniques generally are suitable for oak-hickory management, the wide variety of stand conditions and composition make management considerations too complex to discuss adequately here. On more moist sites, the oak-hickory type can be successional, giving way to more shade tolerant species such as maples (*Acer* spp.). On drier sites, the oak-hickory type generally is considered a climax type.

Northern Hardwoods

The forests classified as northern hardwoods are quite varied. The stands are usually dominated by sugar maple (*Acer saccharum*), American beech (*Fagus grandifolia*), and yellow birch (*Betula allegheniensis*), but younger stands often contain paper birch (*Betula papyrifera*), white ash (*Fraxinus americana*), red maple (*Acer rubrum*), and other hardwoods. Inclusions of conifers such as eastern hemlock (*Tsuga canadensis*), balsam fir, and red spruce (*Picea rubens*), sometimes occur.

Suitable silvicultural systems vary with species composition, age distribution, site productivity, deer browsing pressure, and management objectives. Northern hardwoods can be managed as an uneven-aged forest if the shade tolerant species are desired.

Lowland Hardwoods

The lowland-hardwood type is a mixture of riparian forests and also is called bottomland hardwoods, flood-plain hardwoods, or elm-ash (*Ulmus-Fraxinus* spp) type. The primary tree species are American elm (*Ulmus americana*), black willow (*Salix nigra*), green ash (*Fraxinus pennsylvanica*), red maple, silver maple, box elder (*Acer negundo*), and several species of oak. This is a wildlife-rich type and, in most cases, should be managed by single-tree selection to protect water quality, leaving some large trees for production of dens and cavities. Even-aged management is appropriate when used on small blocks with long rotations to benefit wildlife species needing large trees or snags.

MANAGEMENT BACKGROUND

Forest management activities play a major role in shaping the habitats of forest wildlife species. Any change in the type, age, stocking, or structure of the forest carries with it both advantages and disadvantages for the various wildlife species that occupy the site or move into the area as a result of habitat change. In the development of a nongame management plan, it is especially important that wildlife managers recognize the costs and benefits of forest management activities for all wildlife species and evaluate the projected responses in terms of the goals for a particular forest stand.

Ownership patterns in the midwest play a dominant role in developing nongame management plans. The acreage involved and the goals of owners may limit the kinds of management that can or should be considered. Sixty-five percent of the forest land is owned by farmers and small private landowners; over 40% of this land is in parcels of less than 50 acres (20 ha) (U.S. Forest Service 1982). The primary reason for ownership may be for a summer home, hunting camp, Christmas tree farm, or other purposes. In a recent survey of small non-industrial forest landowners in Wisconsin, 74% of the respondents expressed a desire to manage their land for wildlife—all kinds of wildlife (Roberts et al. 1985). To best meet this goal of managing a small parcel of forest land for a wide variety of wildlife, a landowner must consider the land-use patterns that exist on surrounding lands. Often, nongame management could be enhanced if groups of adjacent landowners banded together to form a neighborhood association for which one management plan could be developed.

Larger blocks of forestland in single ownership often are managed for several primary goals. These may include wood pulp or lumber production on industrial forestland, multiple-use management on public forestland, watershed protection on municipal land, preservation on National Park land, and special-use areas such as state wildlife management lands. Many of these landowners may include nongame wildlife management as a secondary management goal to improve public relations or to meet the legal requirements of maintaining several resource benefits.

When wildlife professionals began delving into the area of nongame management, new concepts were needed and old philosophies needed change. With the more traditional approaches to management, biologists developed expertise that centered on one species. Questions and problems facing the single-species biologist were more focused and direct. For example, a ''deer biologist'' could discuss what was optimum deer habitat, or a ''grouse biologist'' could recommend programs to increase ruffed grouse (*Bonasa umbellus*) populations. For the nongame biologist, the questions become more complex. The answer to the question, "What is good bird habitat?" is much more elusive and depends on whether we are thinking of chestnut-sided warblers (*Dendroica pensylvanica*), pileated woodpeckers (*Dryocopus pileatus*), or any of the other approximately 400 bird species inhabiting midwestern forests.

Several ecological concepts have been elucidated that are important in the development of nongame management plans for midwestern forests. These concepts have not solved completely the problems of dealing with complex, multiple-species systems, but they have provided insights and information. The most useful concepts in forest nongame management include diversity, edge, forest fragmentation, guilds, and the importance of a dead and dying tree component in the stands.

Diversity

The diversity concept is most useful when applied to management programs that preserve a mosaic of habitats supporting a broad spectrum of native species. In these instances, indices of biotic diversity can be used to measure ecosystem quality. The abstract nature of these indices and the development of several different ways to measure diversity led to much discussion of the concept by applied ecologists. Samson and Knopf (1982) provided an excellent discussion of the diversity concept and its application to wildlife management.

In any discussion of diversity, it is important to keep in mind the scale at which diversity will be measured and managed (Noss 1983). Landowners or managers of fairly small forested tracts most likely will be interested in "alpha" diversity; i.e., the diversity of species within a given habitat or community. When managing a larger property that includes forest stands of different types and ages, and perhaps nonforested areas as well, "beta" diversity becomes more important. Beta diversity measures the diversity among adjacent habitats or along an ecological gradient. Some managers also need to consider the role of "gamma" diversity, the overall diversity of species in all habitats located within a fairly large geographic area, for this is critically important in planning efforts to maintain the natural diversity of a region. National Parks, National Forests, states and a few other large landowners have the opportunity to consider gamma diversity in their management plans.

By selecting "diversity" as a management goal, many judgments on the value of individual species are avoided. Lack of consideration of the value of species involved has led to criticism of diversity-enhancing management plans, especially those emphasizing alpha diversity (Samson and Knopf 1982, Noss 1983). Although species richness may be increased by such management plans, species requiring large blocks of contiguous forest or species with other specialized needs often are adversely affected. The popularization of management plans based on diversity, more specifically alpha and beta diversity, has led to several studies relating the decline in some wildlife populations to forest fragmentation. The results of these studies will be discussed later.

Instead of managing for diversity on a local scale, Samson and Knopf (1982) proposed that future wildlife management practices identify important resources within an area, determine the extent and ecological value of each resource, and incorporate a resource-based diversity measurement into regional and local planning procedures. Important or highly vulnerable wildlife resources are identified as those that are threatened or endangered, have a low tolerance to habitat variability, have one or more specialized needs for survival and production, are subject to competition pressures from the common or nuisance species, have a limited range, or exhibit a low resilience to changes that already have occurred throughout a large portion of their range.

Edge

One of the long-accepted rules of wildlife management is that the variety and density of wildlife is greatest at the edge of a particular habitat type (Leopold 1933). Edges attract wildlife species that inhabit the communities on either side of the ecotone and that have relatively low mobility. This combination of species from very different habitats creates a unique animal community. The creation of edges can increase local populations of species that prefer these areas. However, forest management plans that call for the creation of as much edge as possible (as suggested by Yoakum and Dasmann 1971) may be detrimental to other species, particularly the forest interior wildlife species that often disappear in the face of increased disturbance and competition associated with edge. Often there is a much greater need to manage or protect forest interior species than there is for the more opportunistic edge species. Current land-use patterns in the Midwest have created an abundance of edge. Many edge species are more abundant now then they were at any time in the last 200 years.

In midwestern forests, edge usually refers to a forest-field, forest-pasture, or forest-clearcut edge, although it may refer to the ecotone between two forest types. The impact and longevity of these different types of edge vary in terms of changes in animal communities. For example, as one moves from a forest into a field, there is a dramatic shift in the niches available, but as one moves between one forest type and another, the changes are much more subtle. Bird species' preferences for specific tree species as foraging and nesting sites may cause a shift in the bird communities, but the shift between forest types probably is less pronounced than at the forest-field edge. The longevity of the forest-field and forest-pasture edge generally is much greater than for a forest-clearcut edge. Plants and animals tend to maintain the distinctness of the edge in the former cases, but the process of forest regeneration soon eliminates the sharpness of the forest-clearcut edge. Our casual observations in oak-hickory forests of the Missouri Ozarks indicated that the "edge effect" disappeared between 10 and 12 years after clearcutting.

The edge effect is an integral part of efforts to maximize local diversity. Efforts to increase species diversity often call for an increase in horizontal habitat diversity or "patchiness." This increase in the interspersion of different habitats also leads to an increase in edge. Thus, the conflict between the positive aspects of increased wildlife diversity along edges and the negative aspects of forest fragmentation when edges are created within a large forest block actually is the same conflict that exists between managing for wildlife diversity on a local scale and on a regional scale.

Forest Fragmentation, Corridors, and Patch Size

Closely related to the edge effect, but sometimes in sharp contrast to it, is the idea that increased fragmentation of our forests is associated with decreased species richness. With the advent of agriculture and urban developments in the Midwest, the originally vast forests have been broken up into patches of forests, some still extensive in area, others quite small (Curtis 1956). This fragmentation of forests created edges that may have added species, thus increasing species richness in the ecotone between the forest and adjacent habitat, but usually reduced species richness within forest patches by eliminating species that require large forest blocks.

In recent years, researchers have applied the principles of island-biogeography theory to forest fragments, treating these fragments as "habitat islands" surrounded by a "sea" of unsuitable or inhospitable habitat. The principles of island-biogeography theory have been useful in providing insights into the importance of patch size, isolation, and juxtaposition of forest habitats. Some of the most important findings of these studies include:

1. Larger forests tend to support more diverse faunas. Not only do larger patches of habitat have the potential to meet the varied microhabitat requirements of a wider variety of species, but large forests also attract both edge species and those forest interior species that can not tolerate forest fragmentation (Moore and Hooper 1975, Forman et al. 1976, Galli et al. 1976, Matthiae and Stearns 1981, Whitcomb et al. 1981).

2. Forest patches or woodlots located far from the nearest extensive forest usually have a less diverse species list. The isolation of such remnant forest areas limits the flow of new individuals and new species into these areas (MacClintock et al. 1977, Butcher et al. 1981, Whitcomb et al. 1981, Opdam et al. 1984). The impact of isolation probably is greatest on species with less mobility (e.g., reptiles, amphibians, small mammals) or those that find the surrounding habitat inhospitable.

3. The mosaic or pattern of distribution of forest islands can be extremely important in maintaining regional faunal diversity (Harris 1984). In those portions of the Midwest where agricultural or urban land uses dominate, corridors such as hedgerows or linear forests between adjacent woodlots may lessen the impact of forest fragmentation and increase species richness (MacClintock et al. 1977, Butcher et al. 1981, Whitcomb et al. 1981). A "stepping stone" arrangement of forest islands between two large forests also may increase species rich-

ness. In areas of predominantly forest cover, close attention to the pattern of distribution of forest stands of a particular age class may help to maintain viable populations of certain species by allowing movement among suitable habitats (Harris 1984).

4. Some species, especially forest interior species, are sensitive to forest fragmentation and are not found in forests smaller than a certain critical size (Robbins 1979, Whitcomb et al. 1981). Special consideration needs to be given to meet the size as well as habitat requirements of these species when trying to maintain the natural diversity of the region. These species should be considered as candidates for indicator species to monitor the impact of forest management practices on wildlife diversity.

The Guild Concept

According to Root (1967), a guild is "a group of species that exploit the same class of environmental resources in a similar way." Thus, species that feed in the same layer of the forest canopy on similar types of food or that nest or den in similar places could be considered to belong the same guild. In the past two decades, the guild concept has been useful in ecological studies of competition and resource partitioning among coexisting species. The way in which species have been assigned to guilds has varied. Thus the same species may be assigned to different guilds in different studies. In an effort to standardize guild assignments, DeGraaf et al. (1985) classified 674 species of North American birds into foraging guilds, but their assignments for species with broad ranges should be evaluated for a given area since variation in feeding habits may occur among different geographic locations.

Severinghaus (1981) suggested that because all members of a guild use the same resource, they also should respond similarly to changes in their environment. Thus, one member of a guild could be selected as an indicator species to represent the well-being of the entire guild and hence reduce the costs of environmental assessment and monitoring. This idea has been rejected by researchers trying to develop techniques that will be useful in monitoring the response of forest-bird communities to habitat alterations. Mannan et al. (1984) showed that not all members of selected guilds responded in the same way to environmental changes that did not completely destroy their habitat. Slight changes in forest canopy or understory may result in increases in some bird populations and decreases in others. Verner (1981) suggested an alternative to using guilds to select indicator species. He found that by monitoring the populations of all members of a particular guild and combining these population estimates into a single number, the sampling intensity required was far less. The amount of time involved in counts of whole guilds was the same as for counts of guild indicator species, since both could be tallied on 5-minute counts, but fewer counts were necessary to detect environmental impacts with the whole-guild approach than for guild-indicator species alone.

The Dead and Dying Tree Component

Until recently, not much attention had been given to managing dead and dying trees for wildlife on forested stands. Approximately one-fifth of the fauna of a forest depend on fallen timber or slightly decayed trees for food, cover, or reproductive sites (Elton 1966). Although some attention was given to the importance of den trees for certain game species, the impetus for gathering information about snags and down wood generally was lacking under the single species management system. With increased interest in nongame management, it became evident that many species relied on snags, hollow trees, and rotting logs to provide certain life requirements.

Most research concerning the dead and dying tree component of forest stands has centered on snag management for cavity-nesting wildlife. Davis et al. (1983) provided information on the species and diameter of snags preferred by several cavity-dwelling wildlife species, and reviewed snag-management alternatives for different forest types and ownerships. Although there is still much to learn about specific snag requirements of wildlife species and the most beneficial and cost-effective ways to accomplish these goals, forest management professionals are beginning to use the available information to develop guidelines for the number, size, location, and preferred species of snags to be retained. For National Forests, the impetus for snag management stems from legislation (National Forest Management Act, 1976) that requires the maintenance of viable populations of all native wildlife species, whereas several industrial forests have developed snag management policies for public relations reasons.

Research is lacking on the number of logs or amount of slash to leave for wildlife habitat improvement after a timber sale, although information on the role of fallen trees in terrestrial and aquatic habitats is available (Maser and Trappe 1984). Some specific management recommendations for down wood have been suggested, but these are not based on research (Pierovich et al. 1975, Evans and Conner 1979). Very few multiple-use forest plans include guidelines for management of woody debris on forest floors for wildlife. In certain parts of the country where fire hazards are higher than they are in the midwest, logs and slash are considered potentially dangerous fuels. Intensive management efforts often are prescribed, including manual or mechanical treatment of the debris to reduce the hazard of wildfire. Rarely is wildlife habitat management included in prescriptions for the treatment of down wood.

In forests managed intensively for timber production, the amount of dead and dying material available for food, cover, and wildlife reproduction sites is less than optimum. We recommend that modifications of standard timber practices be made such that at least minimum levels of dead and dying trees are retained. However, further research is necessary to determine exactly what amounts of these materials are required by which species of wildlife. Modifications in need of study include:

1. Extending rotations in some forests stands under even-aged management.

2. Setting structure goals to leave larger diameter trees in forests under all-age management.

3. Retaining specific minimum numbers of snags, dying trees, and culls during intermediate cuttings.

4. Retaining clumps of uncut trees (0.1 ha per 2 ha) in even-age reproduction openings.

5. Maintaining a certain proportion of large forest ownerships in mature and over-mature stands to ensure the continued existence of large dead and dying trees within the forest.

6. Discouraging the removal of dead, dying, and decayed trees in stands where cavity sites are limited.

7. Creating snags or cavities through girdling, routing, or inoculation with a decay organism in stands where snags are lacking and limiting certain wildlife populations.

8. Providing artificial cavity sites such as nest boxes or leaving tree species that are prone to heartrot. In certain areas, artificial nesting platforms for ospreys (*Pandion haliaetus*) or double-crested cormorants (*Phalacrocorax auritus*) may be called for if traditional nesting sites are no longer available.

9. Leaving permanent uncut buffer strips 30m to 60m wide along streams and waterways. Not only will this protect the water quality of the streams, but dead or dying trees are particularly valuable as wildlife habitat in these areas.

Appropriate combinations of treatments will depend upon the forest type in question, the wildlife species to be favored, and the relative importance of timber and wildlife resources to the landowner.

MANAGEMENT APPROACHES

One definite advantage to a nongame management plan is the nearly unlimited opportunity to provide habitat for a sizable number of species. However, when the management goal for a particular forest stand is to increase or sustain a healthy population of one species or group of species, the breadth of management opportunities is limited. For example, many management options exist if the goal is to provide habitat for all nongame birds, but the number of options decreases if the goal is to provide snags for cavity-nesting species, and are limited even further if a pileated woodpecker population is desired.

There are four major approaches in developing a wildlife management plan to meet the landowner's goals: management for a featured species (also called emphasis species), indicator species, whole guild, or diversity (which includes species richness).

Featured Species

When managing for a featured species, the landowner's goals might be to improve hunting opportunities (e.g., deer or grouse); to protect rare, threatened, or endangered species (e.g. Kirtland's warbler, *Dendroica kirtlandii*); or to high-light a particular species with a quality that is special to the landowner, such as the song of an ovenbird (*Seiurus aurocapillus*), wood thrush (*Hylocichla mustelina*), or spring peeper (*Hyla crucifer*), or the color of a scarlet tanager (*Piranga olivacea*).

Indicator Species

Both management and ecological indicators may be used in planning and monitoring forest management activities. Management indicators generally are highly valued wildlife species that represent a significant issue in the maintenance of forest diversity or productivity. Land-management plans currently being developed for National Forests use management-indicator species to monitor the impacts of forest management on wildlife. The key to success of the management-indicator approach is in the selection of suitable indicator species. Forest Service regulations require that indicators include any federally endangered or threatened species, those species commonly hunted or fished, species with special habitat needs, and species that are particularly sensitive to management activities or environmental quality (Szaro and Balda 1982). Such species should be a "significant resource in their own right, or . . . represent important aspects of forest diversity" (USDA Forest Service 1984b).

In other cases, ecological-indicator species have been selected to monitor the impacts of forest management on other species with similar habits, habitats, or needs. The guild indicators proposed by Severinghaus (1981) are an example of ecological-indicator species, but when forest management does not involve a drastic change in forest structure (e.g., thinning *vs.* clearcut), indicator species may not adequately represent other species with similar habitat requirements. Small variations in the species' strategies for utilizing a layer of vegetation or a food type may mean that individual species respond differently to slight or even moderate habitat alterations (Mannan et al. 1984).

Whole Guilds

Verner's (1984) whole-guild approach to nongame management can be used to monitor the impacts of forest management on bird communities. The whole-guild approach may not be as cost-effective for mammals, reptiles, or amphibians, depending on whether or not all guild members can be monitored at the same time and with the same technique.

Diversity

The diversity approach to nongame management may be used by a variety of landowners. Small nature centers or environmental-education demonstration areas may develop a management plan that is based on maximizing alpha diversity. In these instances, small patches of a variety of habitats should be juxtaposed to provided habitat for the maximum number of species. The plan might include a small pond with some woods on one side and perhaps a field on the other. Hedgerows, conifer patches, or streams might also play an important role in maintaining within-habitat diversity.

Landholders of larger forests, such as state forests, state wildlife areas, industrial forests, National Forests, and National Parks may have a nongame management goal to maximize beta diversity. With this goal, land-use patterns of forests and nonforest lands, forest types, and age-class distribution become important.

Resource agencies with a national or regional perspective are the only managers likely to choose gamma diversity as their goal for nongame management. Planners at the national level should try to coordinate management efforts among forests and parks to ensure regional maintenance of large blocks of unfragmented forests.

Because chances to see wildlife in natural habitats are a high priority with the public, projects that improve wildlife observation opportunitues may be desirable. These might include building impoundments or observation towers, creating small clearings, planting agricultural crops or fruit-bearing shrubs to attract birds, and maintaining winter feeding stations. In the case of rare or endangered species, tours may be scheduled to avoid disturbance of these animals at critical periods in their life cycles (e.g., Kirtland's warbler tours in Michigan).

THE MANAGEMENT PLAN

The planning process is complicated by the number of wildlife species that need to be considered and the variety of treatments that can be implemented. Lipscomb et al. (1983) outlined a 4-step approach to develop a management plan that can be complete and complex for large land ownerships or can be simplified where the area is small or if only a few species are to be considered. These steps are:

1. Inventory—what do I have?
2. Goals and objectives—what do I want?
3. Species needs—what do they need?
4. The plan—how do I meet the needs?

We have combined these 4 steps with a monitoring process to provide an overall system of inventory, planning, management, and evaluation for nongame wildlife management (Fig. 1)

The first step is inventory. An inventory needs to be complete enough to be meaningful, but care must be taken not to do more than is actually necessary. The basic requirements include delineation of the area, identification of forest types and condition classes, identification of areas that potentially can be influenced by management actions, and an inventory of wildlife species.

Management goals are broad statements of purpose for a management effort. These goals are based on the objectives of the landholder and, especially in the case of publicly owned lands, should be accompanied by a justification statement as to why the goals were chosen. Once goals have been defined, the manager should select one of the management approaches described earlier—management for a featured species, indicator species, whole guilds, or diversity—or some combination of these approaches.

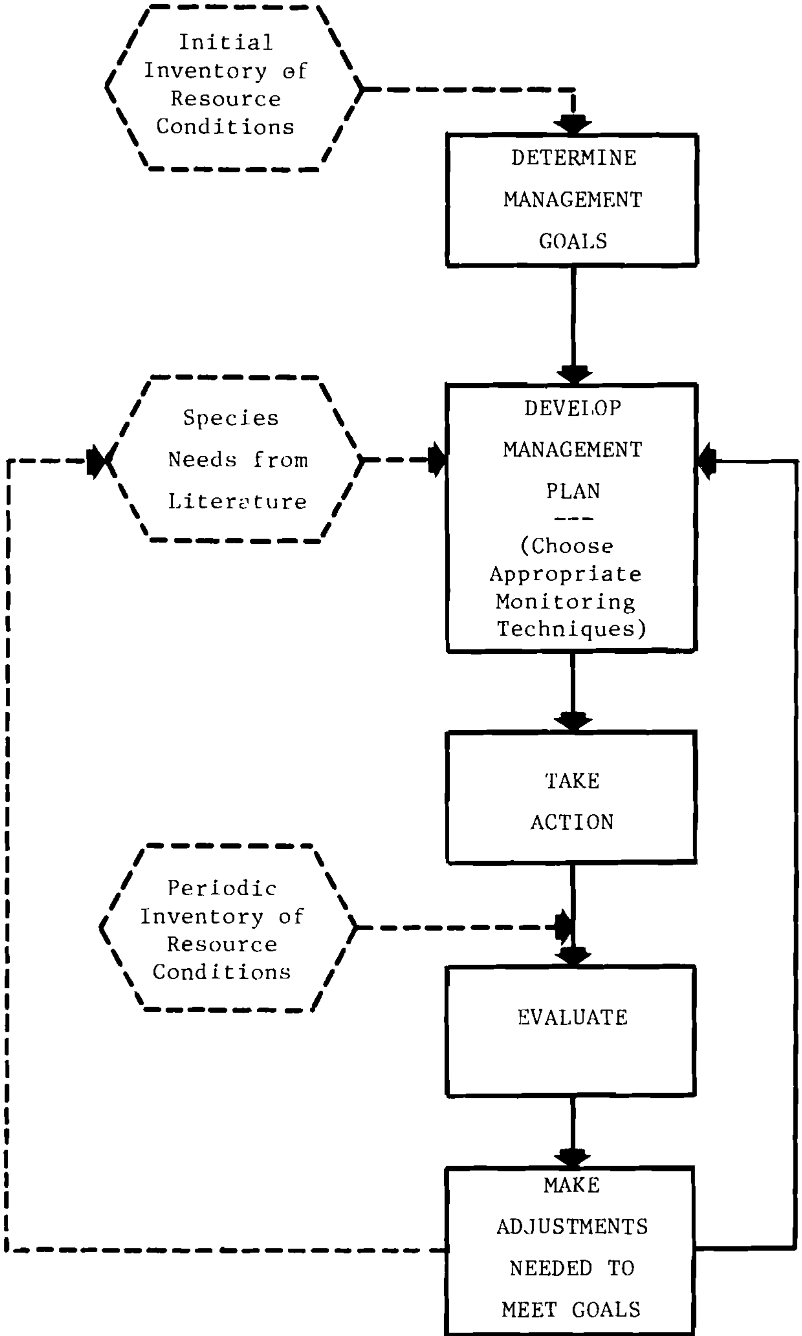

Fig. 1. The planning and monitoring process.

Information on species needs usually can be obtained from available literature or computer data bases. Compilations by Thomas et al. (1979) of information on the habitat requirements of forest wildlife species for reproduction and feeding sites provided the theoretical basis for the development of numerous wildlife-habitat relationships publications and data bases in recent years. If this information is not automated, the task of collecting information can become more difficult if many species are selected, or if very little information is available on the selected species.

If the diversity approach is selected, it is possible that information on the habitat characteristics of individual species will not be needed. The diversity approach can be totally habitat-oriented. For example, the plan could call for a mix of early and late forest successional stages, distributed to provide a mosaic of stand conditions and special habitat features such as riparian zones, snags, edges, travel corridors, and food sources.

The actual nongame management plan should outline the prescribed treatments needed to reach a particular nongame wildlife goal, a schedule and layout of the stands to be treated, showing the juxtaposition of treatments, size and shape of stands, methods for enhancing habitat features, and other special criteria. For most forest managers, this planning still has to be done manually. For small forest ownerships with fairly straightforward goals, a nongame wildlife management plan can be developed by a manager familiar with the forest type, the species desired, and the techniques involved.

With larger ownerships and more complex goals (especially when goals conflict with other forest resource goals), computer programs are needed to sort through large data bases, to keep track of wildlife habitats currently available, to predict future habitats that are the result of natural succession or planned forest management, and to prescribe additional treatments needed to produce the desired kind and quantity of nongame wildlife habitats. At present, such computer programs are in the early developmental stages, with many refinements and broader applications becoming available on a regular basis. In the near future, it should be possible to enter a slightly expanded set of forest-cruise data from a small forest ownership into a microcomputer and have a forest management plan printed out, including prescriptions designed to produce optimum habitat for selected wildlife species, including nongame.

MONITORING AND EVALUATION

Monitoring may be defined as the acquisition of information that allows an evaluation of accomplishments. Monitoring can take on many forms, from simple to complex and from qualitative to quantitative. It almost always is desirable to follow the implementation of any action plan with some form of monitoring. Landowners investing their own money and time might be satisfied with an "it looks good" evaluation. However, when others invest in a wildlife improvement program, they generally require a more complex monitoring program to clearly define the progress that has been made towards an expected attainment. This progress may be in terms of achieving a desired population goal for certain wildlife (e.g., featured species, indicator species, or whole guilds), in terms of the number of species present (diversity), or in terms of wildlife habitats provided (habitat suitability).

Habitat-suitability indices represent a recent technique for assessing the potential detrimental impacts or improvements associated with a particular forest management action. These indices are based on characteristics of the vegetation and special habitat features present in the forest stand. Thus, this approach measures how well the species habitat requirements are met and does not deal with actual population response. Regardless of which management goals have been selected, periodic inventories of both the vegetation and wildlife of the forest are necessary to evaluate and identify adjustments needed in the management plan (Fig. 1).

For most public forests, periodic inventories of woody vegetation already are being made. For example, information on the timber resources of the National Forests in the midwest is readily available in the Timber Management Information System (TMIS) data base. Additional information on the understory vegetation of shrubs and herbaceous plants often is necessary to complete the vegetation inventory, along with an inventory of special habitat features such as cliffs, rocks, caves, snags, streams, or lakes. A few midwestern National Forests have adopted the Wildlife Management Information System (WMIS) data base for storage of wildlife habitat information collected for each stand in the forest. Other data bases have been developed by individual National Forests to store similar information on word-processing machines. But many National Forests, state forests, and industrial forests have not begun to collect this kind of information on forest structure and wildlife habitat features. In many instances, the information is stored in the minds of field personnel, but is not easily available for use in developing and evaluating a forest management plan to produce habitats for nongame wildlife.

The other half of the monitoring system is to gather data on wildlife populations for the same forest stands. The number of species to be monitored depends upon the management approach selected and the goals of the management plan. The techniques that may be used to sample these populations are many (Kendeigh 1944, Robbins 1978, Ralph and Scott 1981, Vogt and Hine 1982), but the selection of suitable techniques is critical in designing a monitoring system. Such a system must provide the necessary population information in the most efficient and productive manner possible. Methods for the initial inventory and subsequent monitoring of wildlife populations should be detailed before implementation of the plan.

An inventory of forest resource condition (vegetation and wildlife) is necessary both for the preparation of the forest management plan and to provide feedback in the system. Feedback may be either direct or indirect. If a particular wildlife population (or diversity goal) is not responding to the prescribed treatments, then adjustments can be made directly in the forest management plan to provide habitat more suitable to the wildlife population goals. The periodic inventory of wildlife populations by habitat condition also will provide information to refine species-habitat relationships originally gleaned from the literature. Incorporating these new relationships into the model will provide a more predictable assessment of species' responses to habitat changes. The role that field personnel will play in providing information to correct wildlife-habitat relationship models is paramount to the success of the system.

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Nongame Management in Grassland and Agricultural Ecosystems

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Abstract: The pristine prairie was a mosaic of habitat types that comprised a continuum of vegetative conditions. Cover types varied from short sparse habitats to tall, dense growth and from grass-forb dominated sites through shrub-grass communities to oak savannahs. Prairie wildlife species have adapted to all components of this continuum. Conservation of prairie wildlife requires the management of grassland and agricultural ecosystems so that all aspects of the prairie continuum are provided. Management decisions must be made on a state or region wide basis. Size of habitat units will influence management options. Fire and grazing are the primary tools for managing grassland wildlife habitats. Other management options include mowing, use of herbicides, reestablishment of prairie vegetation, and reintroduction of grassland fauna. Public and private agricultural habitats can contribute to the conservation of grassland wildlife. Pastures and hayfields may provide habitat conditions that meet the requirements of species adapted to grazing and associated shorter, sparser cover. Croplands are seemingly of lesser value in meeting grassland wildlife conservation goals. Conservation tillage seems to offer significant advantages over conventional tillage methods in providing wildlife habitat. Minimum tillage technology may result in the conversion of other land-use types to agriculture, thus effecting net losses in wildlife numbers.

Native prairie habitats once occupied approximately 3.6 million km^2 (1.4 million sq. mi.) in North America (Kuchler 1964, Rowe 1972, Constable 1985). For the midwest, loosely defined as from southern Manitoba south to Oklahoma, east to Ohio and north to Ontario, grasslands dominated an estimated 40% of the landscape. Tall-grass or true prairie, with characteristic flora such as big bluestem (*Andropogon gerardii*), Indian grass (*Sorgastrum nutans*), and switchgrass (*Panicum virgatum*), dominated the eastern two-thirds of the Midwest. The western third was occupied by mixed-grass prairie with its variety of grass species: needlegrasses (*Stipa* spp.), gramas (*Bouteloua* spp.), wheatgrass (*Agropyron* spp.), etc.

Despite the legacy of this extensive habitat base and its unique faunal components, the art and science of wildlife management in grassland systems is poorly developed relative to that in other habitats. The widespread destruction of prairie habitats, minimal public ownership of grasslands, especially in the midwest, low faunal species richness in prairie communities, and low aesthetic appreciation of grasslands by the public undoubtedly have contributed to the lagging development of integrated management approaches for grassland resources.

For the Great Plains, Klopatek et al. (1979) estimated that 85% of the bluestem prairie, 65% of the bluestem-grama prairie, and 45% of the grama-buffalo grass (*Buchloe dactyloides*) prairie have been destroyed. In Iowa, Illinois, and Missouri well over 90% of the original tall-grass prairie and savannah habitats have been converted to other land-use types. In the U.S., only about 1.7 million ha (4.2 million A.) of prairie are preserved as National Grasslands, mostly in the western states. State and private conservation organizations (notably The Nature Conservancy) own smaller grassland holdings throughout the Midwest. This low level of public ownership seemingly has made it difficult for management agencies to justify grassland wildlife management as a priority. Compared with forest ecosystems, grasslands support relatively few animal species. Wiens and Dyer (1975) reported mean number of nongame breeding bird species for tall-grass prairie as 4.1; mixed-grass prairie, 4.7; and short-grass prairie, 4.3. In contrast, the deciduous forests of eastern North America commonly support 10-30 breeding bird species (Kendeigh 1946, Back 1979, Probst 1979). This relative paucity of species seemingly has contributed to a lack of interest in prairie nongame wildlife ecology and management.

The grassland resource base undoubtedly will continue to shrink in the face of intensified agricultural development, urban sprawl, energy resource exploition, etc. In recent years, the conservation of grassland resources has benefitted from an increased awareness of and sympathy toward environmental problems. This increased concern demands the development of integrated approaches to the management of grassland ecosystems. If we are to conserve the genetic diversity of our native prairie flora and fauna, the acquisition, reclamation, and proper management of grassland resources must become a priority. It is essential that we better understand grassland ecosystems, examine the role of agricultural ecosystems in prairie wildlife conservation, and develop management strategies and procedures that will satisfy the requirements of all prairie wildlife species.

PRISTINE PRAIRIE: A DYNAMIC MOSAIC

The key to managing wildlife in grasslands, as in any environment, is understanding the basic attributes of the ecosystem and the way wildlife species have adapted to them. The management of prairie wildlife often suffers from misconceived views of grassland ecosystems. Early prairie immigrants, fresh from the forests of eastern North America, were awestruck by the "vast sea of grass." They saw the prairie as an arid, inhospitable "desert," stretching monotonously to the horizon. This early view of the prairie as a desolate and homogenous environment continues to plague us. Modern views also may be distorted by the remaining grassland tracts that are too small to show the characteristic variability of the natural prairie ecosystem. Although subtle in comparison with forests or coastal environs, grasslands evidence substantial habitat diversity, primarily in the horizontal plane.

In the pristine prairie, soil moisture, influenced by soil type and topography, combined with native grazers and wildfire to create a mosaic of habitat conditions. This mosaic can be viewed as a two-dimensional continuum (Fig. 1). Along one axis, vegetation structure varies from short and sparse to tall and dense. Soil moisture and grazing intensity are the primary controlling factors along this axis. The other axis is a gradient varying from grass- or grass-forb-dominated sites through increasing shrub densities to savannah-like conditions. Fire frequency and intensity, and soil moisture are the controlling variables along this axis.

Wildlife species have evolved adaptations to exploit different regions across this continuum. Mountain plovers (*Charadrius montanus*) or vesper sparrows (*Pooecetes gramineus*) prefer short-grass habitats maintained by heavy grazing (Knowles et al. 1982, Skinner et al. 1974). Sedge wrens (*Cistothorus platensis*) and bobolinks (*Dolichonyx oryzivorus*) are examples of species adapted to tall, rank cover on moist sites that can be eliminated easily by heavy grazing (Kantrud and Koligiski 1982, 1983; Skinner et al. 1984). Many prairie vertebrates, such as deer mice (*Peromyscus maniculatus*), respond positively to periodic fires that eliminate litter and eradicate shrub-tree invasion (Kaufman et al. 1983). But others, e.g., clay-colored sparrows (*Spizella pallida*) and Henslow's sparrows (*Ammodramus henslowii*), may be extirpated by frequent fires that destroy required shrub nest sites or preferred song perches. This view of the grassland ecosystem as a mosaic of habitat types from a vegetational continuum sets the stage for the development of wildlife management goals and strategies.

MANAGEMENT APPROACH

The goal of nongame grassland management should be the conservation of wildlife species native to the prairie habitats of a particular region. As discussed earlier, habitat diversity in natural grassland ecosystems is achieved in the horizontal plane. Therefore, diversity increases most readily with area. However, the opportunity to manage prairie tracts of sufficient size to include all aspects of the prairie vegetation continuum is rare in the midwest. The alternative is to manage smaller tracts as components of the overall prairie mosaic. Problems of minimum area requirements of wildlife species are discussed later.

The implementation of a component management approach necessitates the coordination of management actions at the state or region-wide level. If management decisions are made only at the local level, there is a risk that some components of the prairie vegetation mosaic will be underdeveloped and the associated fauna not conserved. If species-richness goals are established at the local level, managers may be justified in creating and maintaining habitat conditions that provide for only a portion of the native fauna. Managers on

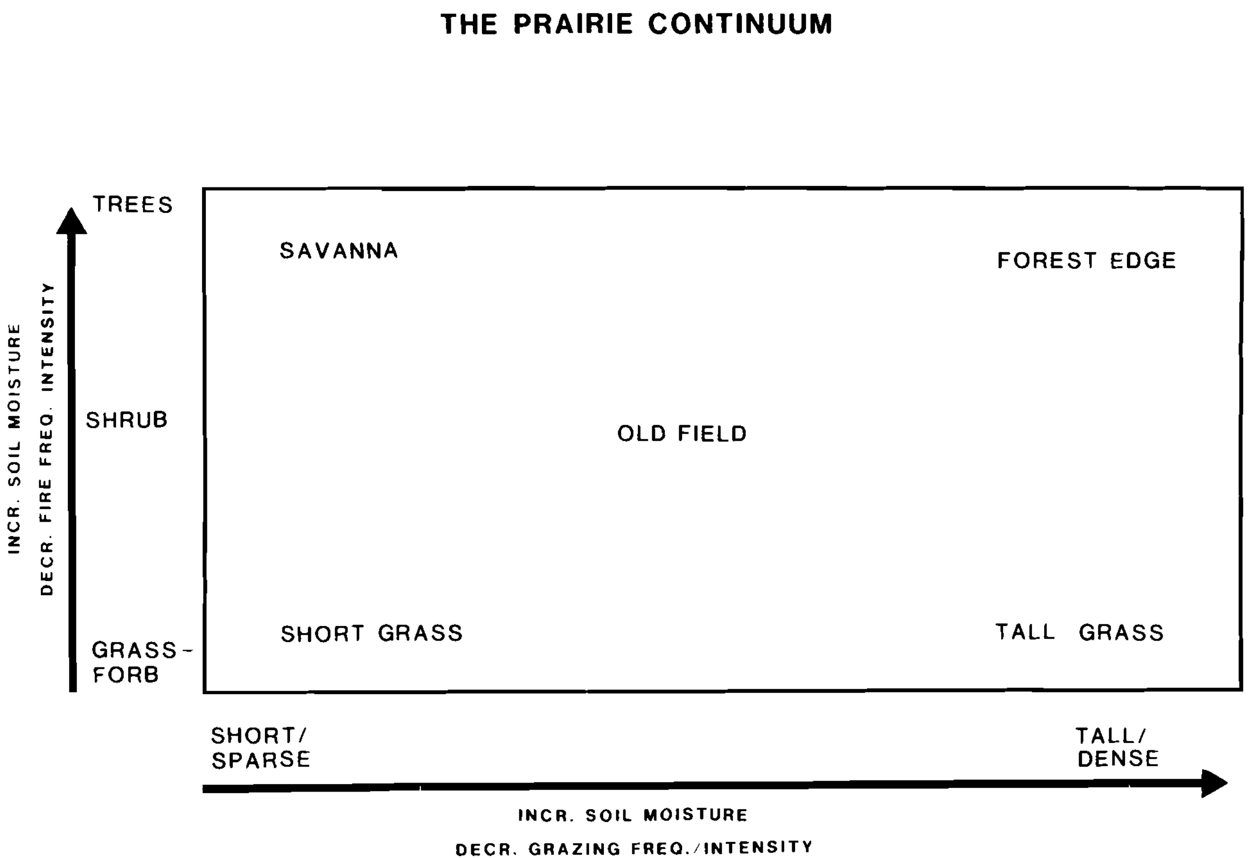

Fig. 1. A continuum of habitat conditions in the prairie ecosystem.

different areas may meet their local goals using similar habitat conditions. Whereas each unit may have high species richness, the statewide species richness may be lowered. An example will illustrate this problem.

Renken (1983) studied nongame bird communities on managed grasslands in North Dakota. Three different site manipulations produced 3 distinct habitat conditions. Grazed sites had short, sparse cover, dominated by grasses and forbs; idled areas (no fire or grazing) had moderately tall vegetation with high shrub densities; and planted grass-legume treatments produced tall, dense cover.

Shrub-dominated plots had the highest bird-species richness, but 11 of the 23 species observed in shrub-dominated sites were forest-edge species. The 12 grassland species accounted for only 57% of the 21 potential grassland, nongame birds native to the area. Each of the 3 treatments supported species unique to that habitat type: grass-legume 3, grazed 2, idled 2. Combined, the three cover conditions accounted for 18 (86%) of the 21 expected grassland species. The three missing species were characteristic of moderately to heavily grazed grasslands. Obviously, species richness at the regional level could be maintained only by a component management scheme using different manipulations to create a range of habitat conditions optimal for the entire grassland bird fauna.

In summary, species richness goals must be developed at the state or regional level and implemented through integrated management actions at the local unit level. I suggest the following steps to develop an ecosystem-wide management program for grassland fauna: 1. Identify species native to grassland habitats of the management unit (preferably statewide). 2. Identify current distribution of extant grassland species and list extirpated species. 3. Identify habitat requirements of individual grassland species, including possible minimum area requirements. 4. Evaluate the potential of existing management areas as components of the prairie habitat continuum and for maintenance of statewide grassland species richness. 5. Review need for acquisition of additional habitat components. 6. Develop integrated management plans for specific management areas such that all aspects of the prairie continuum are represented on a continuing basis.

MINIMUM SIZE OF GRASSLAND MANAGEMENT AREAS

The possible effect of habitat tract size and isolation on maintenance of species richness and genetic variability is a controversial issue. The argument derives from the application of the theory of island biogeography (MacArthur and Wilson 1967). In theory, species richness is a function of relatively constant immigration and extinction rates determined by island size and distance to mainland. Supporting data are available for oceanic islands (Simberloff 1976), but application of the theory to terrestrial habitat "islands" is less clear-cut. Most ecologists would agree that, for similar habitats, a large block will contain and conserve more species than a small unit. The controversy centers around the question of whether a large habitat tract will maintain more species than several small tracts that, combined, equal the large one.

Several studies presented convincing evidence that some species will not be maintained in reserve designs using several smaller tracts (mammals, Brown 1971; birds, Galli et al. 1976; Robbins 1979; reptiles, Heatwole 1975). The causal factors are far from clear, but large home-range requirements, seasonal variation in space of required resources, large minimum population needs, psychological factors, and competition with 'edge' species have been suggested (Diamond 1976). Samson (1980) provided the only data (albeit sketchy) for North American grasslands, focusing on bird species richness only. For prairie relicts in Missouri, he concluded that some species—greater prairie chicken (*Tympanuchus cupido*), upland sandpiper (*Bartramia longicauda*), and Henslow's sparrow—could not be maintained on small tracts. Samson recommended grassland reserves of at least 300 ha to preserve the most area-sensitive species, the greater prairie chicken. However, viable populations of prairie chickens are being maintained on smaller, closely juxtaposed grassland tracts in Illinois, Minnesota, Missouri, and Wisconsin (Hamerstrom et al. 1957, Westemeier 1980, Cannon and Christisen 1984). Samson (1980) also concluded that such reserves invariably would provide appropriate habitat for all the other prairie bird species.

Simberloff and his colleagues (Simberloff and Abele 1976a,b, 1982; Simberloff and Gotelli 1982) have criticized strongly the "bigger-is-better" concept. They cite data, from a variety of ecosystems, that demonstrate that a complex of small reserves will conserve as many or more species than a large single unit.

Both sides of the controversy frequently have over-simplified their arguments. Some proponents of large reserves assume tract size to be the overriding factor determining species richness and imply that within-and between-habitat diversity is of minimal importance (e.g., Diamond 1975, 1976), whereas the presence of some species (notably forest-interior birds) may be affected primarily by habitat-unit size, other species may be habitat specialists. Additionally, an implicit assumption that large areas invariably will have greater habitat diversity seems to be made in the "bigger-is-better" arguments. Although the probability of greater habitat diversity increases with block size, the relationship is not one of cause and effect. The relationship cannot be used as a decision rule in grassland reserve design or management. A large grassland tract managed as a homogenous habitat unit certainly will have less value than several smaller units, each of which is a distinct component of the prairie mosaic.

Proponents of the small, multiple tract approach have been criticized for weighing all species equally in their evaluation of reserve configurations (Blake and Karr 1984). Two small areas with large numbers of common species may be less valuable than a large area supporting fewer species, of which some are at high risk of extirpation or extinction.

The lack of clear consensus on species-area relationships and paucity of data for prairie systems makes size-related recommendations for grassland management difficult. But with the grassland resource base ever-dwindling, preservation actions cannot wait. The limited availability of funds requires the establishment of criteria for prioritizing grassland areas for management action.

The highest priority for acquisition, management, or reclamation should be prairie sites that, *regardless of size*, contain rare wildlife species or habitats that could support them. Secondary priority should be given to large prairie tracts containing a variety of habitat conditions or that can be managed easily to create a mosaic of habitat types. Large, vegetationally complex tracts offer not only substantial benefits to prairie wildlife, but preserve the spiritual feeling of the wide-open prairie that is an integral aspect of the human experience. I recommend third-order priority be assigned to clusters of small tracts that provide, or can be managed for, a variety of components of the prairie mosaic. The reestablishment of small native prairie sites will be most useful if the sites are developed near existing grassland habitats. Fourth- and fifth-level priority should be given to large, homogenous blocks of grassland and to small, highly isolated units, respectively.

In summary, the priority given to a grassland area for acquisition, management, or reclamation cannot be based on size alone. Species composition, habitat diversity, management potential, human appreciation, and cost all must be considered. There is no shortcut to valueing grassland habitat for wildlife conservation. Decisions must be based on sound knowledge of ecological attributes and long-term potential for species conservation.

GRASSLAND MANAGEMENT OPTIONS

Left undisturbed, prairie habitats are dynamic and may change rapidly in structure and species composition; litter will build-up, woody vegetation may invade, and shifts in grass and forb dominance may occur. Periodic manipulations will be required to maintain a diversity of grassland cover types and assure the integrity of the prairie community.

An important consideration in managing public grassland habitats is the availability and condition of privately owned grasslands. For example, if grazed pasturelands or hayed grasslands are common, there is little need to manage for the short-grass components of the prairie continuum.

Many management tools are available. Those that simulate natural processes are likely to be the most effective and aesthetically appealing, but many techniques may be needed to create and maintain the multi-habitat solution to prairie wildlife conservation.

Fire

Although climate likely was the dominant factor controlling the distribution of North American grasslands, fire played an important role in limiting shrub and tree development, especially in the eastern tall-grass prairie (Wright and Bailey 1982). Any grassland management program will involve fire as a tool. The extent and frequency of prescribed burns will be determined by local conditions and management goals. In the midwest, the major benefits of prescribed burning will be to control invading shrubs and trees, remove litter, increase grass and forb production, and control introduced cool-season grasses (Wright 1974, Wright and Bailey 1982). An advantage of using fire as a management tool is the simultaneous achievement of several objectives (Wright and Bailey 1982).

The effect of fire on wildlife will be highly variable. Wildlife responses to fire will be mediated by intensity and uniformity of the burn, the season in which the burn occurs, presence of nearby refuge habitat, as well as habitat preferences of individual species.

Small mammals, reptiles, and amphibians are the taxa most likely to be killed directly by fires (Erwin and Stasiak 1979). Burrow-dwelling species usually will be well insulated from the heat of burning, but slow-moving species or those far from burrows may be killed (Howard et al. 1959, Wright and Bailey 1982).

Small-mammal populations will be affected most by short-term losses of cover and food, exposure of surface runways and burrow openings, and increased predation (Cook 1959, Crowner and Barrett 1979). Cool fires that leave patchy cover and food will have less dramatic impacts than hot, extensive fires. I found no data on herptile population responses after fire, but patterns seemingly would be similar to small mammals. Most studies have demonstrated a short-term reduction in total density of small mammals and, often, a decline in species richness (e.g., Vacanti and Geluso 1985). The presence of alternative refuge habitats may be important for repopulation of burned sites. If rare species are present, burning small areas while leaving refuges may be essential.

Small mammal species, such as deer mice, that are adapted to open grassland habitats with minimal litter and high grass species dominance will be favored by more frequent fires (Tester 1965, Schramm 1970, Kaufman et al. 1983). Small mammals favoring dense grassland vegetation (e.g. *Microtus* spp., *Reithrodontomys megalotis*) or shrubby vegetation (e.g., *Peromyscus leucopus*) will respond more favorably to less frequent fires (Birney et al. 1976, Kaufman et al. 1983).

Large, mobile mammals and birds rarely are killed by fires, but nests and young may be destroyed and refuges for renesting or escape cover may be needed. The major effect of fire on wildlife populations will be through altered habitat conditions. Reduced litter conditions will result in differential response by avian species. Birds requiring a deep litter layer for nest placement will be affected negatively by frequent burns, but reduced litter may improve brood habitat for avian species with precocial young, allowing greater mobility and access to food resources. Increasing forb and grass production will favor species using those resources for food and cover, whereas shrub-dependent species will decline.

Frequency of prescriptions will vary substantially with site characteristics and management goals. Annual burns may be necessary to eradicate high densities of woody vegetation. Maintenance intervals will vary from 2-5 years in moist tall-grass sites to 10 years in drier, mixed-grass prairies. An excellent review of prescribed burning techniques is given by Wright and Bailey (1982).

Mechanical Treatments

Mowing can be an effective management tool in creating open grassland habitat. The advantages include relatively low cost, low labor intensity, and a high degree of control over height reduction and size of area treated. Costs may be reduced further by selling the hay or trading it for the mowing.

Hay removal may result in loss of nutrients from the system that can affect grass and forb production negatively and change species composition. If cuttings are left, the increased litter may counteract the effectiveness of mowing to produce short, sparse cover conditions. Mowing must be timed carefully to reduce nest loss and exposure of immobile young to predation. Typically, mowing later in summer results in fewer nests being destroyed. Late mowing may, however, reduce sale potential of hay as nutritive quality declines.

Shrubs or brush also may be eradicated by mechanical means as varied as chainsaws and bulldozers. These methods have been criticized because brush removal usually resulted in reductions in density of game species (e.g., Darr and Klebenow 1975). The use of chainsaws for limited shrub and tree elimination may be useful where fire is impractical for safety reasons or because of lack of equipment and manpower. In my view, chaining, root plowing, or roller-chopping are unlikely to be needed in the midwest, except to reclaim sites well along in succession. I do not recommend the use of these techniques on areas with native grass cover because damage to native vegetation may be greater than any gain afforded by brush removal.

Herbicides

Herbicides are used in grassland management to reduce shrub cover or to eliminate non-native vegetation prior to reseeding native grasses and forbs. The most commonly applied herbicides reduce broad-leaved plants but do not affect grasses (Robinson and Bolen 1984). These herbicides may be effective in eliminating woody species but also are detrimental to forbs that may be important as wildlife foods or song perches (Beasom and Scifres 1977, Skinner et al. 1984). The herbicide Tebuthiuron, however, has been found to reduce shrub cover while increasing grass production and maintaining forb availability (Doerr and Guthery 1983). Cannon and Knopf (1981) reported increased lesser prairie chicken (*Tympanuchus pallidicinctus*) numbers after herbicide applications to reduce shinnery oak (*Quercus havardii*) cover.

Grazing

Grazing and browsing by ungulates, hares, rodents, and insects were integral aspects of the native grassland ecosystem. Bison (*Bison bison*) are thought to have been a primary force in maintaining the short grass component of the pristine prairie (Larson 1940) to which many wildlife species are adapted (Kantrud and Koligiski 1982, Ryan et al. 1984, Skinner et al. 1984). The literature on grazing effects on wildlife reported conclusions ranging from completely detrimental to absolutely essential. Clearly, for species requiring tall, rank cover, grazing is an inappropriate management tool. But for those species adapted to exploit short, sparse grassland habitats, grazing may be the most effective management tool available, especially in the mid- and tall-grass prairies.

Bison created a mosaic of vegetation over a huge area (Allen 1876, England and DeVos 1969). Domestic livestock pastured on small grassland units will not simulate native grazing patterns, but can be effective in creating short-grass compartments in a larger management unit. There is often considerable opposition to grazing livestock on grassland wildlife management areas. Native prairie enthusiasts commonly are offended by the unaesthetic Holstein or Hereford in a prairie remanant, although the presence of longhorn cattle in Theodore Roosevelt National Park, North Dakota, seems acceptable to the public as part of the historical theme of the area. In the midwest, at least, public education may be needed before establishing a grazing program for grassland management. Care also must be taken to assure that domestic grazers will not outcompete native species (rodents, hares, etc.) for forage resources.

An important consideration in assessing the need for grazing public grasslands is the availability and adequacy of privately owned pasturage. If publicly owned grasslands managed for mid- to tall-grass cover, shrubby areas, etc. are juxtaposed with private grazing lands providing short-grass habitat, the spectrum of wildlife needs can be met most efficiently. But in areas where private grazing is inadequate or public access to all prairie wildlife is a goal, grazing may need to be incorporated in management plans.

Initiating grazing may be costly if fencing must be erected and permanent water sources developed. However, personnel needs to monitor grazing are relatively small, and the leasing of grazing rights may offset costs in the long term.

Water

Water can be a limiting factor for wildlife in grassland habitats. The value of small prairie tracts, especially, may be reduced severely by lack of available water for drinking, bathing, or as egg-laying sites for amphibians. The need to develop ponds, etc. on grassland tracts should be evaluated carefully.

Re-establishing Grassland Habitats

In areas where grasslands have been eliminated, reseeding sites with grasses and forbs may be desirable. Reestablishment of grassland vegetation may require substantial effort, but can be effective in attracting nongame species. Higgins et al. (1984) reported colonization of a reseeded native grassland plot by nongame birds one year after planting.

On mesic sites throughout the former tall-grass prairie, big bluestem, Indian grass, and switchgrass can be established readily. For more xeric sites, mixed-grass species such as western wheatgrass (*Agropyron smithii*), sideoats grama (*Bouteloua curtipendula*), needle-and-thread (*Stipa comata*), and little bluestem

(*Andropogron scoparius*) can be planted with success (Duebbert et al. 1981). A major factor in the success of reseeded grassland cover is the use of site-adapted seed. For example, if varieties of grass seed from southern sites are planted in northern regions, success may be very low due to poor winter hardiness. Northern-adapted seeds planted at southern locations mature earlier produce less herbage, grow to shorter heights, and are less disease-resistant than southern varieties (Duebbert et al. 1981). Other factors affecting establishment (e.g., site and seedbed preparation, planting equipment and methods, and dates of seeding) were reviewed by Duebbert et al. (1981). This excellent publication, although intended for the northern Great Plains, can be used throughout the grassland region. George et al. (1979) and Missouri Department of Conservation (1980) also gave management recommendations for establishing tall-grass species.

Reintroducing Prairie Wildlife

Almost no data are available on nongame-species reintroductions in grassland ecosystems. Kansas has begun reintroductions of mountain plovers and American swallow-tailed kites (*Elanoïdes forficatus*) (J. Schaefer, pers. comm.), and Iowa has attempted a greater prairie chicken reintroduction, but the results are not yet known or are unreported. Information on reintroductions is critically needed, and any agencies making such attempts are urged to report their procedures and results.

AGRICULTURAL ECOSYSTEMS

Agricultural ecosystems occur over much of the area formerly occupied by natural grasslands in North America. Most wildlife species using pastures, hayfields, or croplands evolved in native prairie habitats. Agricultural ecosystems, therefore, may provide alternative habitats to meet grassland wildlife management goals. Species that can exploit agricultural resources for all of their needs will require little or no management effort on public lands. With few exceptions, agricultural habitats support lower numbers of grassland species and lower overall densities than managed grasslands (Graber and Graber 1963, Owens and Myres 1973, Higgins 1975). But because of the huge area involved, any changes in agricultural practices that benefit wildlife, even in a small way, can have a substantial effect (Castrale 1985).

For private landowners to implement changes in agricultural practices for the benefit of wildlife, incentives of some type will be needed. The widely held view has been that cash payments were likely to be the most effective inducement (Shellberg 1981). Kirby et al. (1981), however, reported that Missouri farmers preferred free seeds or plants, technical advice, or tax considerations over cash payments. These results suggest that opportunity exists for substantial gains in wildlife conservation with minimal public expenditure.

Pastures

The role of grazing in the management of public grasslands for wildlife already has been discussed. On private lands, domestic livestock production functions similarly by creating open cover types. However, the wide variety of range and livestock management options used on private grasslands can produce vastly differing effects on wildlife. Livestock stocking rates, timing and distribution of grazing, and even the livestock species used affect plant-species composition and associated structure which in turn affect wildlife use of pasture habitats.

Most wildlife species select cover based on structural characteristics of the vegetation. The vegetation structure of pasture communities is determined by extrinsic factors (e.g., soil moisture, grazing intensity) and by the physiognomic characteristics of the plant species themselves. For example, on similar sites, introduced tall fescue (*Festuca arundinacea*) produces low, dense cover, whereas native switchgrass or Indian grass create less ground cover and more cover at taller heights (Skinner et al. 1984). Even if fescue pastures are grazed moderately, the shorter, sparse cover required by, for example, upland sandpipers, will not be produced (Skinner et al. 1984). Native tall-grass species can offer greater cover-type variability. If lightly grazed, taller dense cover is maintained for animal species such as voles (Eadie 1953, Birney et al. 1976), Henslow's sparrows or sedge wrens (Skinner et al. 1984), and if more heavily grazed, is preferred by sparse-cover specialists (Skinner et al. 1984). Forbs may be important food items as well as cover for wildlife species. Skinner et al. (1984) reported lower forb coverage in fescue pastures than in native grass pastures. In northern regions, introduced cool-season grasses, such as smooth brome (*Bromus inermus*), do not stand up well under snow, resulting in dense mats of vegetation and poor residual cover in spring (Higgins and Barker 1982). Native warm-season grasses stand up well over winter, and in general, are preferable for wildlife in pastures.

Recently, several publications (e.g., George et al. 1979) have discussed the economic advantages of incorporating warm-season pastures into livestock production operations. Native warm-season grasses provide excellent forage for livestock in mid- to late-summer. Typically, livestock are moved to warm-season pastures after many nongame birds have completed at least one nesting cycle. Combinations of cool- and warm-season pastures can improve livestock weight gains (D. Kirby, pers. comm.) and increase wildlife production.

Different grazing systems affect the timing and degree of use of range resources by livestock, and thereby, the wildlife use of pastures. At low stocking rates, continuous or season-long grazing results in uneven use of forage. This patchiness can be highly attractive to a variety of wildlife species, but low stocking rates usually are economically unattractive. At high rates, continuous grazing can result in long-term damage to the range, as well as a high degree of disturbance to wildlife.

Pasture management such as rest-rotation, twice-over rotation, short-duration systems, etc. (Stoddart et al. 1975) generally result in more complete utilization of forage and greater weight gains by livestock. Increased grass utilization reduces patchiness within units of the system, but rested units provide undisturbed habitats. In some systems, pasture cells are not grazed until well after nongame species have reproduced. The data on wildlife responses to different pasture management systems primarily addresses game species. Although differential responses have been documented, no clear pattern emerges to recommend one system over another across the entire grassland region (Bryant et al. 1982). These systems hold promise for increased wildlife production coupled with sound economic incentives, but more data, especially on nongame responses, are needed.

Hayfields

Annually mowed hayfields provide a limited array of habitat conditions along the prairie continuum. Early-nesting birds that prefer short cover, such as longspurs, (*Calcarius* spp.), horned larks (*Eremophila alpestris*), or killdeers (*Charadrius vociferus*), will benefit the most, but lack of water near hayfields may reduce the fields' value even to these species. Migrating birds (gulls, upland shorebirds, raptors) and mammalian predators may use cut-over sites to feed on readily available invertebrates (especially grasshoppers) or small mammals.

Other species may be attracted to hayfields by good cover conditions early in the summer, but fail to reproduce because nests or young are lost during mowing operations. Under these conditions, hayfields may act as population drains. For example, some small mammals, such as voles may be reduced or eliminated by mowing and unable to recolonize sites cut each year (Lemen and Clausen 1984). In evaluating the contribution of hayfield habitats to grassland faunal conservation, managers must carefully distinguish those species that can exploit mowed sites successfully from species in residence but negatively affected.

Planted, monotypic hayfields are less likely to be of value to nongame wildlife than multi-species native fields. Monotypic tame-hay sites are very low in structural diversity and high in disturbances (planting, fertilizing, and harvesting operations). Legume-dominated sites may be highly attractive to a few species, such as dickcissels (*Spiza americana*), savannah sparrows (*Passerculus sandwichensis*), or bobolinks (Stewart 1975), but early cutting may negate any benefits.

Hayfields composed of native species can provide a variety of structural conditions and potential food plants. Native grasslands cut for hay typically have greater structural diversity than tame hay fields. Even if mowed annually, native hay sites can provide habitat for species adapted to short, open cover types, such as Richardson's ground squirrel, (*Spermophilus richardsonii*), thirteen-lined ground squirrel (*S. tridecemlineatus*), marbled godwit (*Limosa fedoa*), chestnut-collared longspur (*Calcarius ornatus*), and upland sandpiper (Stewart 1975, Jones et al. 1983, Ryan et al. 1984).

Croplands

Most nongame wildlife using croplands are species with grassland habitat affinities. Agricultural fields typically support low numbers of wildlife species and at lower densities than natural grass habitats. However, over large areas, croplands may be the only habitat available to support grassland wildlife (Best 1985). Wildlife use of cropfields is influenced by crop type, field size, adjacent habitats, tillage operations, and pesticide applications.

Data on the relative value of crop types to wildlife are scant. Higgins (1975) presented data on shorebird nest distributions among growing grain, stubble, and fallow fields. Several studies of corn and soybean systems also are available (e.g., Young 1984, Castrale 1985, Basore et al. 1986), but direct comparisons of vertebrate communities in various crop types remain to be done. Small-grain fields, with their greater similarity to natural grasslands, might be expected to support higher wildlife diversity than row crops such as corn or soybeans.

In general, increasing field size results in overall reductions in wildlife density because of minimized interspersion of cover types (Robinson and Bolen 1984). Many wildlife species that use intensively tilled lands also need adjacent habitats to meet food, cover, and song perch requirements. Rodenhouse and Best (1983) found that vesper sparrow territories in corn and soybeans were restricted to within 80 m (260 ft.) of field edge, possibly because of limited song perch availability early in the growing season. Small mammals may occur throughout crop field environments (Warburton and Klimstra 1984), but Young (1984) reported greater species diversity at field margins.

Grass waterways within crop fields, and fencerows and roadside ditches adjacent to crop fields typically support higher numbers of species and greater densities than tilled areas. Best (1983) reported 62 bird species (30 during the breeding season) using fencerows in Iowa, although most were forest-edge birds. Basore et al. (1986) recorded 14 avian species nesting in strip cover (waterways, terraces, fencerows, and roadside ditches) and reported nesting densities there up to 70 times greater than in agricultural fields. Rodenhouse and Best (1983) suggested that eliminating fencerow habitat would drastically reduce vesper sparrow use of tilled areas.

Detailed information on the effects of tillage operations on nongame wildlife is just beginning to emerge (Young 1984, Castrale 1985, Basore et al. 1986). Best (1985) and Wooley et al. (1985) provided excellent reviews, and the following summary borrows heavily from those sources.

Tillage operations range from conventional plowing, in which surface soil and crop residues are turned over in the fall and followed by spring discing, through varying degrees of "conservation" tillage to no-till practices that leave surface soil and crop residue undisturbed (Hayes 1982).

Tillage systems affect wildlife use in varying ways through differences in amount of residual cover, surface availability of grain, amount of disturbance by farm machinery, and pesticide applications (Best 1985). No-till operations result in substantially greater cover available in winter for wildlife than do conventional practices (Castrale 1985). Winter food abundance also is increased by conservation tillage that does not bury waste grain left after harvesting. Best (1985) noted, however, that the greater cover on no-till fields traps more snow and thus may reduce availability of food during critical periods. Differences in waste-grain availability among tillage practices continue through spring and summer, although the importance of this food source to wildlife is reduced at these seasons. Some studies (Blumberg and Crossley 1983, House and Skinner 1983, Warburton and Klimstra 1984) suggested that higher crop residues, associated with conservation tillage, support greater insect abundance, but see Basore et al. (1986) for contrary findings.

Increased cover availability in conservation-tilled fields may be beneficial as nesting or escape cover for nongame birds. Basore et al. (1986) and Castrale (1985) reported higher species richness and density values for breeding birds in no-till habitats compared with conventionally tilled fields. Primary nesting species in Iowa no-till fields were vesper sparrow, grasshopper sparrow (*Ammodrammus savannarum*), western meadowlark (*Sternella neglecta*), killdeer, and mourning dove (*Zenoida macroura*) (Basore et al. 1986). Young (1984) recorded greater small mammal diversity in no-till sites than conventionally farmed areas but densities did not differ. Deer mice and thirteen-lined ground squirrels dominated no-till sites. Castrale (1985) reported no difference in small mammal diversity and density between no-till and conventionally tilled fields.

In conventionally tilled fields, Rodenhouse and Best (1983) found that vesper sparrows were not producing enough young to offset adult mortality. Conservation tillage systems require fewer passes over fields with farm machinery and thus may reduce direct nest destruction or disturbance leading to abandonment. Species that place nests between crop rows will be particularly benefitted (Best 1985).

No-till operations involve greater reliance on chemical, as opposed to mechanical, control of weeds (Best 1985). Herbicides vary widely in their toxicity to wildlife; eggs and young are most likely to be affected. Paraquat, used to "burndown" early weeds and retard sod growth, may be especially detrimental to wildlife (Hoffman and Eastin 1982, Best 1985). Best also noted that incorporated herbicides are likely to be less harmful than surface applied chemicals.

In general, conservation tillage methods seem to offer greater benefits to wildlife than conventional practices. Wooley et al. (1985), however, cautioned wildlife conservationists that it is too early to reach any conclusions. They point out that the availability of no-till technology may result in formerly marginal lands being brought into production. These lands are likely to be important wildlife producing habitats. Gains accrued in wildlife conservation by changing from conventional to conservation tillage may be offset by other land-use conversions.

As a final note, the most significant gains for wildlife conservation in agricultural habitats may be realized not through the application of any new techniques, but through changes in agricultural policies set by state and federal governments. The 1985 Federal Farm Bill, with "sodbuster" and conservation-reserve provisions, could have far-reaching and long-term benefits for grassland wildlife.

The "sodbuster" program would withhold subsidies to farmers who plow lands classified as highly erodible. The conservation reserve program is intended to retire erodible land from crop production. Payments would be made to farmers that transform former cropland to pasture, trees, or permanent grass and legume cover. The program is intended to retire up to 25 million acres for periods of not less than 10 years. Efforts by individuals and conservation groups to assure the passage of these or similar provisions will result in substantial benefits to the conservation of nongame wildlife.

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Interfacing With The Public

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Abstract: "Interfacing" refers to the process by which an agency interacts with the public to achieve program goals and to generate public support. This is a multi-step process: Define program goals and objectives, identify publics, understand the learning process necessary to achieve desired action, identify key messages, select appropriate techniques to reach selected publics, communicate with those publics by using the multiplier effect, encourage public input to the program design, develop public involvement and volunteer participation in the program, and provide recognition and feedback to donors, supporters, and selected publics. Implementation of this process will help insure success in achieving the organization's goals.

In 1976, an income-tax checkoff allowing taxpayers to designate part of their tax refunds for nongame wildlife conservation was passed by the Colorado state legislature. This "Nongame Wildlife Checkoff" was an immediate success and the concept has been adopted in 33 states. Although this technique of raising money for nongame wildlife has its limitations, it is now a major source of funds for state nongame wildlife programs in the United States. In the tax year 1984, $8.96 million were donated nationally to this cause (McCance 1985).

An integral part of each checkoff program is the publicity associated with the tax-preparation season. The intensity and ingenuity necessary for successful checkoff promotion has generated some important peripheral benefits that can help natural resource agencies in general and wildlife agencies in particular. The primary benefit is a significant improvement in the ability to interface with the public. Unfortunately, there usually has not been enough attention given to interfacing with the traditional publics associated with fish and wildlife resources (Todd 1980, Butler 1983). This can lead to a lack of public support for agency programs.

Interfacing with the public can help resolve such problems and facilitate reaching an agency's goals. In the context of this paper, interfacing with the public shall refer to the following sequence of interactions between a natural-resource agency and the public: 1) Define program goals and objectives, 2) identify publics, 3) understand the learning process necessary to achieve desired action, 4) identify key messages, 5) select the appropriate techniques to reach those publics, 6) communicate with those publics by using the multiplier effect, 7) encourage public input to the program, 8) develop public involvement and volunteer participation in the program, and 9) provide recognition and feedback to donors, supporters, and various publics.

This 9-part sequence of interactions may appear complicated, but it was essential to the development and implementation of Minnesota's nongame wildlife program, and can be adapted readily to similar programs in other states or to broaden other wildlife programs.

DEFINE GOALS AND OBJECTIVES

The first step is to state succinct program goals and objectives so that both agency and publics have a clear idea of where the agency is going, why the public should be concerned, and how program success will be measured. An example might be a goal "To preserve the diversity and abundance of a state's nongame wildlife resource." An objective for that goal might be to raise $500,000 per year to fund a state nongame wildlife program. Such goals and objectives are an integral part of the comprehensive planning process that should be the initial phase of program development.

IDENTIFY PUBLICS

The second step is to identify your publics. Remember, there is no such thing as the "general public." There are many specific publics, each with their own special needs and interests. Different publics may need to be reached by different techniques. Examples of publics for a state nongame program include garden clubs, bird clubs, naturalists, industrial-arts teachers, science teachers, urban home owners, college biology and wildlife professors, librarians, state revenue department personnel, hunters and trappers. Others include youth groups; community education program coordinators; county, state and federal foresters; county extension agents; sportsmen's clubs; civic groups and media persons. Each public offers special opportunities for program involvement and interaction, and each public has its own information network.

UNDERSTAND THE LEARNING PROCESS NECESSARY TO ACHIEVE DESIRED ACTION

My approach to understanding the conservation-education process is called the "chocolate chip" model. People generally pass through 6 developmental steps in relation to natural resource problems: 1) No awareness or concern, 2) awareness of a program or problem, 3) appreciation, 4) understanding, 5) concern, and 6) action. The best way to visualize these stages in a population is with a chocolate-chip-shaped diagram (Fig. 1). Most of the people are in the wide basal portion, representing those unaware of a program's existence. The top of the chocolate chip represents those persons who are fully informed on an issue and who are actively concerned and involved with the program.

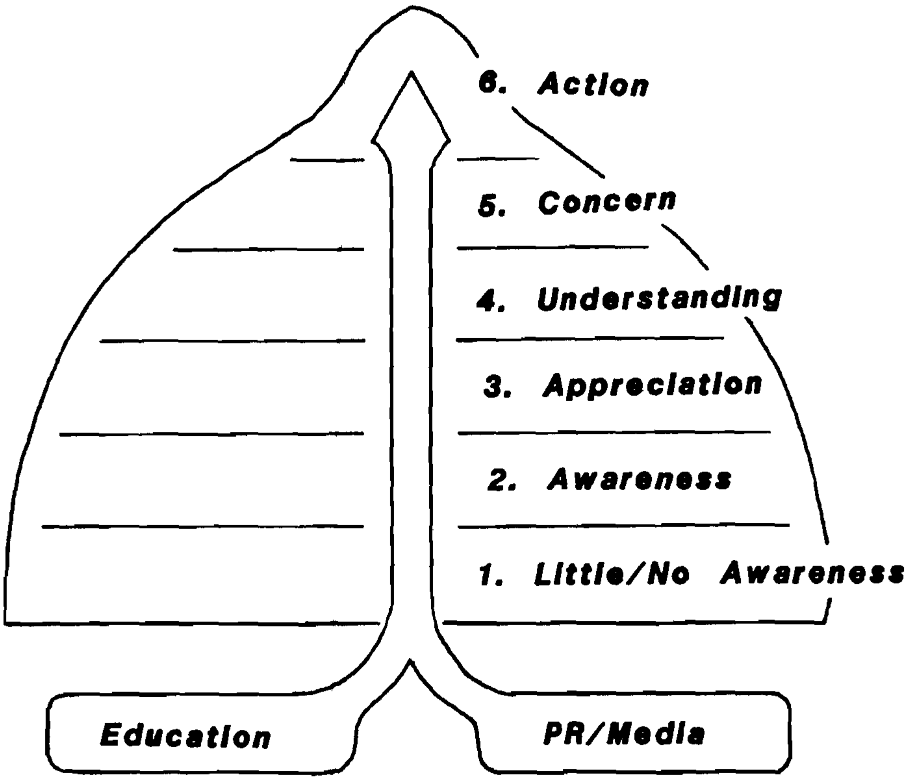

Fig. 1. The Chocolate Chip model of the conservation education process.

Interfacing with the public is a key element of that process, and it can be done in 2 ways. One is to carry out an active education program that is presenting your message to targeted publics. Examples might be sponsorship of "Project WILD" workshops to teach wildlife ecology and conservation to teachers, workshops to teach people how to help bluebirds (*Sialia sialis*), or acquiring films, slide-tapes, and video tapes about wildlife for public use.

The second and probably the best way to reach the most people is to work with radio and television personnel to achieve continuing exposure to your program. Strategies and techniques for doing this were described by Henderson (1984).

IDENTIFY KEY MESSAGES

Next, it is necessary to develop brief and catchy messages to convey to the public. Think in terms of a message that can be conveyed in a 30-second public-service announcement or a 7-word billboard message. For example, "Help wildlife.

Don't mow roadsides until August 1." Or, "Look for the loon on your state tax forms." Longer articles can be published in state conservation magazines.

SELECT PROPER TECHNIQUES TO REACH THE PUBLIC

Given a limited budget, select publicity techniques necessary to reach the public. Examples include radio, television, newspaper, magazines, billboards, posters, and other public-service techniques. An example of selecting the proper technique for a specific public is printing posters for tax preparers to give to clients who make donations to the nongame wildlife checkoff. This step initially involves the development of a communication plan.

Television

The most important form of television advertising in Minnesota has been 30-second public-service advertisements (PSA) featuring well-known species that are benefiting from the checkoff, such as bald eagle (*Haliaetus leucocephalus*), peregrine falcon (*Falco peregrinus*), trumpeter swan (*Olor buccinator*), common loon (*Gavia immer*), and western grebe (*Aechomorphorus occidentalis*).

Television news coverage can be another important source of publicity. There are 2 ways to obtain TV news coverage. First, call the news assignment desk whenever a special event or project is to take place. Second, special "media days" can be held to publicize major nongame projects such as peregrine falcon, burrowing owl (*Athene cunicularia*), or trumpeter swan releases. New-release packages should be provided to participating reporters to help insure that project details, names, and titles are reported accurately.

Radio

There are 2 important types of radio publicity: 30-second public-service advertisements and interview programs. For example, 5 30-second PSA may be sent to all radio stations in the state, each featuring a single wildlife species and its call. Weekly programs on wildlife ecology and phenology might be featured on some stations. Local radio stations also should be invited to media days. An outline of key questions about the nongame program and other informational material should be prepared and sent to a radio-station interviewer prior to a live interview.

Newspapers, Magazines, Newsletters

Several techniques can be utilized to publicize a checkoff in newspapers, magazines and newsletters: Post-Christmas news releases to encourage taxpayers, camera-ready public service ads, photo news releases with attached cutlines (scripts), and feature stories about winter bird feeding, building bird houses, or general program accomplishments. Excellent coverage in major newspapers may be derived from media days. Always arrange for a live, captive specimen to be present as a "backup" in case the wild animals do not cooperate with the photographers.

Posters

Posters are a surprisingly important type of publicity. In Minnesota, for example, a 17″ x 22″ (43 cm x 56 cm) wildlife poster was printed each year to publicize the state's tax checkoff. Posters were distributed free of charge to tax preparers, libraries, banks, county courthouses, newspapers, radio stations, and to a mailing list of friends of the nongame program. The cost of poster production ranged from $0.08 each to $0.20 each.

Corporate Support

Free corporate support for publicity may be relatively easy to obtain because the checkoff is so popular as a non-controversial, public-service cause.

COMMUNICATE WITH PUBLICS BY USING THE MULTIPLIER EFFECT

As a communications plan is implemented, rely heavily on the "multiplier effect." With the same amount of effort, 10, 1,000, 10,000 or 100,000 people can be reached. It all depends on how well you are able to reach teachers and the media, primarily radio and television, and have them relay your message. The traditional approach of printing pamphlets or giving talks to people 25 at a time will not generate the level of support needed to maintain a statewide nongame wildlife program.

ENCOURAGE PUBLIC INPUT

Effective communication is a 2-way process. It is not enough to deliver messages to the public. You also must encourage initial and ongoing public input to your program. It is important to conduct public meetings to allow members of the public to express their opinions on program priorities and direction. This can be done with one central meeting to kick off the process, followed by regional meetings across the state. Thereafter, annual or biennial meetings may be held to allow continuing public input.

DEVELOP CITIZEN VOLUNTEER PARTICIPATION IN SELECTED PROGRAM ACTIVITIES

There are many wildlife research and conservation activities that must be carried out by trained professionals. However, the support that exists for a state nongame program may be increased significantly if opportunities are provided for personal involvement by volunteer citizens. The key is to select volunteer activities that can be coordinated easily by mail or telephone from a central office without requiring supervision of volunteers in the field. Observation-card surveys for easily identified species work well for sandhill cranes (*Grus canadensis*), common loons, bald eagles and winter birds. Bluebird-trail projects also work well. In this way, the nongame program for volunteers becomes "their" program and they will fight for it and lobby for it when the legislative chips are down.

PROVIDE RECOGNITION AND FEEDBACK TO CITIZENS

To bring the whole process of citizen information and involvement full cycle, it is necessary to provide recognition of volunteers and to provide information on program accomplishments and budgets to the public. This is perhaps one of the most critical elements in the whole process, and helps to insure long-term support and involvement.

A loon jacket-patch has been used as reward for volunteers in the Minnesota nongame wildlife program. An annual report and a project map may also be useful to inform people of how their donations are being spent. There are 2 useful publicity periods, one during the tax-collecting season to solicit donations, and one during the summer field season to show people what projects are underway and how their money is being spent.

CONCLUSIONS

Interfacing with the public is a daily way of doing business through a determined effort to establish a warm, continuing, open relationship with the public. Let them know that their opinions and their support is important, and let them know what you are doing. By using the general guidelines explained in this paper, Minnesota's nongame wildlife program grew 22-fold between its inception in 1977 and 1985. The key to that growth was "interfacing with the public."

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The Future: Agency Perspective

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Abstract: Nongame programs will continue to grow in size and influence as the public broadens its appreciation and use of fish and wildlife species that traditionally have not been harvested or hunted. State conservation programs evolve through 5 stages: Protective, regulatory, biological, ecological, and sociological. Consideration of nongame matters is an integral component of the latter stages. For nongame considerations to be incorporated into future agency activities, the functional organization of a nongame program should consist of 8 features: 1) Citizen support, 2) funding base management, 3) planning, 4) data acquisition and management, 5) coordination, 6) information and education, 7) habitat management, and 8) species management.

Nongame wildlife conservation programs still are in their first decade of existence in most states, yet they have progressed rapidly during those few years. In spite of the progress that has been made, much remains to be accomplished. Herein I will discuss the role that a nongame wildlife program can play within a state wildlife agency, and make some predictions about opportunities for future development of nongame conservation.

Butler (1983) described how wildlife agencies have broadened the scope of their activities in recent years. Changing environmental values, continuing loss of habitats, and increased concern for endangered species all have contributed to this change. Butler explained that the management of wildlife rsources is undergoing an evolution that has 5 distinct phases:

First is the "Protective Phase," in which habitats are designated or acquired and other actions are taken to protect species that are undergoing significant declines. This phase often is typified by the closure of harvest seasons for declining game species and the establishment of refuges.

Second is the "Regulatory Phase," wherein game laws attempt to allocate harvestable resources among hunters, fishermen and trappers.

Third is the "Biological Phase," in which research is used primarily to fine-tune our ability to manage game species. Population modeling is an integral part of this process. This phase is one in which biologists are primarily "game managers" or "game biologists." There still is a lack of concern for or knowledge of nongame species. Research is focused almost exclusively on animals themselves and not on sociological aspects of wildlife management.

Fourth is the "Ecological Phase." This stage is marked by expanded attention to a full range of wildlife species and the use of wider ecological input in management decisions. This includes development of state nongame and natural heritage programs. New emphasis is placed on nongame vertebrates, invertebrates, wildflowers, urban wildlife, preservation of unique habitats, and preservation of endangered species. Objectives of this phase favor retaining and maintaining representative, intact wildlife associations. This holistic approach is further marked by expanded planning for wildlife. During this phase, sportsmen no longer are perceived as the primary audience, and support from society as a whole results in a surge of interest in habitat and species protection.

The emergence of this fourth management phase generally corresponds with the hiring of ecologically minded biologists. Managers and biologists draw upon each other's experience to further develop their own projects. Greater emphasis is placed on public education, with programs developed from a broader perspective of environmental education. Terms such as "dickey-birds" or "rough fishes" are dropped and personnel speak and act knowledgeably about the ecological requirements of species such as piping plovers (*Charadrius melodus*), five-lined skinks (*Eumeces fasciatus*), and wood turtles (*Clemmys insculpta*). Additional critical wildlife habitats, such as heronries and shorebird nesting areas, are acquired and protected during this phase.

Fifth is the "Sociological Phase," in which wildlifers seek a new level of rapport and understanding with their publics. This includes scientific research and opinion surveys about the users of the wildlife resource—their attitudes, perceptions, motivations, and values. More effort is placed on obtaining public input during the planning process. Greater emphasis is placed on keeping the public informed through improved public relations and media contacts. A well-informed public will use wildlife resources more wisely, and concurrently will give greater support to the wildlife resource agency. This phase may occur simultaneously with the fourth phase.

So what does all this mean in terms of where we are and where we are going? The goal of a state wildlife agency should be an holistic program for the protection and enhancement of wildlife habitats and populations, with no distinction between game and nongame. To keep the nongame wildlife conservation movement advancing, it is necessary to incorporate it into a wildlife agency in a compatible and comprehensive manner. This will help the agency to be more effective in reaching its objectives and will provide a logical approach through comprehensive planning for all wildlife activities.

Comprehensive and strategic planning will be invaluable in guiding the direction of nongame programs for the coming years. Budgeting limited nongame funds becomes much easier when priorities have been established through a planning process. Planning must identify the key issues that affect the nongame resource and the program functions that address those issues.

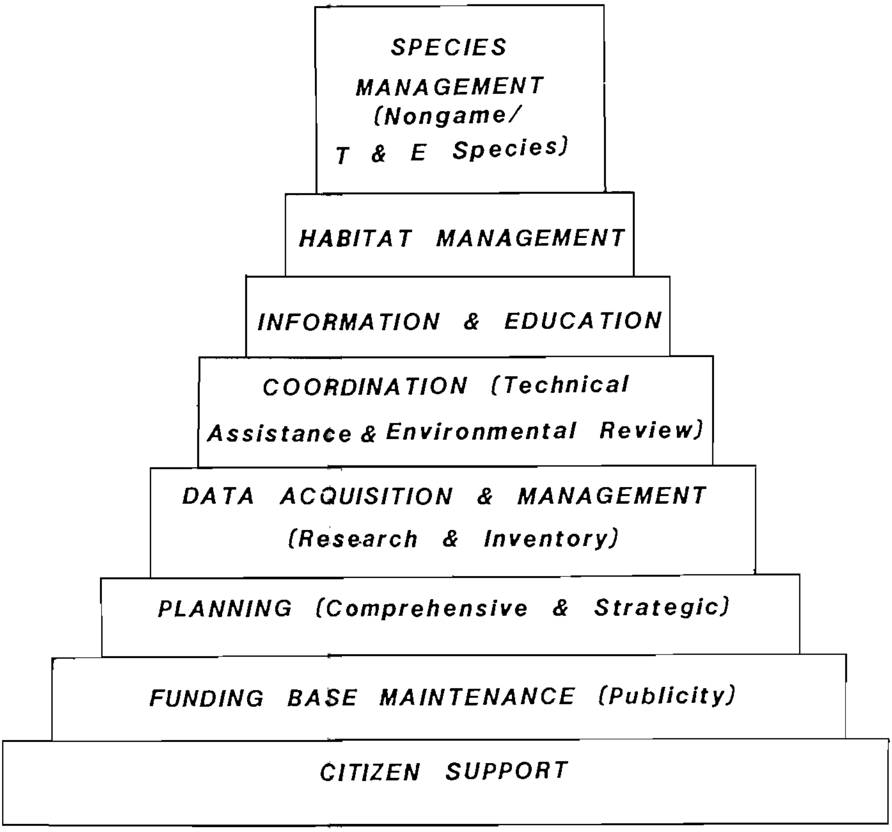

Fig. 1. Suggested organization of functions for a nongame wildlife program.

Figure 1 shows the functional organization that exists for the Minnesota Nongame Wildlife Program. Its 8 levels provide a pyramid of program functions.

CITIZEN SUPPORT

The success, extent and effectiveness of a nongame wildlife program depends on the amount of citizen support. As citizen support increases or decreases, the ability of the program to meet its responsibilities will be increased or decreased accordingly. Citizen support is the foundation of the entire program.

FUNDING BASE MAINTENANCE

Maintaining a funding base is next in the organizational pyramid. Citizen support must be converted into funding support if program benefits are to occur. The funding must be adequate and stable in the long term, and specific monetary goals should be set. If a nongame wildlife tax checkoff is the primary source of program funding, checkoff publicity will be an integral part of this function.

PLANNING

The third level is planning, both comprehensive and strategic. Given adequate citizen support and a funding base, planning provides the guidance and priorities essential to achieve the goals that have been identified. Planning, therefore, becomes an ongoing function within the program.

DATA ACQUISITION AND MANAGEMENT

Data acquisition and data management are the biological backbone of the program. Data is acquired through research and inventory of nongame species and their habitats. Emphasis is placed on nongame priorities, problems and opportunities that were identified in the planning process. Data is managed manually and with computer techniques so that information is both accessible and useful to a variety of potential users.

COORDINATION

Nongame wildlife information is in demand by many publics, including private individuals and organizations, and county, state and federal agencies. Technical assistance is provided to these groups to enhance their impacts of nongame species and habitats, or to minimize adverse environmental impacts. Environmental review services also may be provided to these agencies. The impact of this coordination function should not be underestimated because many thousands of acres of wildlife habitat owned by private individuals and public agencies may be affected in this manner.

INFORMATION AND EDUCATION

With the benefits of comprehensive planning goals and information from data acquisition and management, it becomes possible to design an information and education effort that addresses special needs or problem areas, such as promoting the ecological values of snags, snakes and bats, or informing the public on how to help bluebirds (*Sialia sialis*). This function may help reduce problems of illegal or needless killing of nongame species, or the unnecessary destruction of nongame habitats. It also will promote a better understanding of the principles of wildlife ecology and management.

HABITAT MANAGEMENT

Habitat management in the context used herein refers to activities carried out directly by program personnel, such as management of islands for shorebirds, prescribed burning of prairies, or acquisition of important habitats. These activities usually are designed to benefit a variety of species. The selection of projets is based on a review of other programs functions, including planning goals and priorities, budgets, and biological data. This helps ensure that the chosen projects will significantly benefit wildlife.

SPECIES MANAGEMENT

The final function to be carried out by a nongame wildlife program is species management. It consists of projects or activities designed to benefit single nongame species such as bluebirds, trumpeter swans (*Olor buccinator*), great blue herons (*Ardea herodius*), or big brown bats (*Eptesicus fuscus*). It also may include state or federally threatened or endangered species such as the wood turtle, common tern (*Sterna hirundo*), bald eagle (*Haliaeetus leucocephalus*), or piping plover. Projects may include restoration of species, management of selected sites to enhance the survival of a species, or preparation of a management plan for a species. Successful species management will depend upon data and input from virtually all of the preceding nongame functions.

CONCLUSIONS

Figure 1 portrays how each function depends on those functions below it in the pyramid. Keeping these functions in balance will contribute to the successful functioning of the entire program. Whereas species-management projects often generate the most interest and attention in a nongame program, many other less visible functions are equally important and essential. High visibility projects such as the reintroduction of trumpeter swans and peregrine falcons (*Falco peregrinus*) provide a positive feedback to "Citizen Support" by increasing support for the entire nongame program. Planning helps ensure that program actions will occur in concert with other wildlife conservation activities.

It is obvious that nongame management is not a passing fad. By understanding that nongame management is an integral part of the ecological phase of wildlife management, we can broaden our directives to encompass responsibilities for natural heritage programs, scientific and natural area preservation, native plant conservation, and endangered species preservation. We also can evaluate how well we are incorporating sociological aspects into wildlife management work.

If we are to meet the challenges of these broader responsibilities, however, it will be necessary to go beyond the levels of funding that currently are available. Nongame checkoff funding is inadequate for state programs, and more money must be generated at the state and federal levels to realize the full potential of these programs and responsibilities. The best opportunity to generate federal funding is, in my opinion, either a semi-postal (surcharged) postage stamp or an excise tax on birdseed, bird feeders and other equipment associated with the enjoyment of nongame wildlife. Such revenues would fund the federal Fish and Wildlife Conservation Act of 1980, which has yet to be funded by the U.S.

Congress. We must present a strong and united front to our legislators in Washington to convince them of the need to fund this Act. We also must explore and generate new funding mechanisms at the state legislative levels. Nongame wildlife programs will continue to grow in state wildlife agencies, and the funding levels we can establish will determine the rate of growth.

Be opportunists. Don't ask "When are they going to give us more money?" Go out and get it. Get involved with the big issues and look for ways to enhance not only nongame, but the total environment. Look for the chances to promote and become involved in developing big programs such as forest planning, watershed management, and agricultural programs such as the 1985 federal farm bill. The proposed "Reinvest in Minnesota" measure to provide state-level funding for soil and water conservation, and fish and wildlife management, as well as Missouri's sales taxes for soil conservation, state parks, and wildlife conservation, are examples of innovative solutions to meet broad program needs that should be backed.

Finally, we must think in holistic terms. We need an ecological approach to all natural resource management. Rather than emphasizing management of mallards (*Anas platyrhynchos*) or common loons (*Gavia immer*), we should direct our efforts toward broader issues such as wetland protection, prairie preservation, and the maintenance of ecosystem integrity. Only by maintaining a diversity of quality habitats will we preserve their component species.

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Future Trends in Management of Nongame Wildlife: A Researcher's Viewpoint

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Abstract: The challenge for wildlife managers has been multiplied significantly by recent emphases on nongame species. Two aspects of the response by agencies to cope with this added complexity are long-term forecasting of trends in animal numbers, and monitoring actual trends in abundance. Models using information about habitat requirements of various wildlife species are used to assess environmental impacts and to forecast trends in abundance, given knowledge of trends in habitat conditions. The accuracy of many such models is questionable, and even the basic concepts are limited. For example, many models assume that some measure of animal abundance is an adequate index to habitat suitability. We know this is not true in all cases. In other cases, populations are regulated by factors other than habitat carrying capacity (e.g., predation, disease, weather), weakening the manager's ability to predict population trends from knowledge only of habitat changes. These and other complications require much additional research.

Our ability to monitor trends in the abundance of animals is similarly short of the technical development needed to do the job properly and efficiently. Existing inventory techniques are generally too expensive and they require more skilled personnel than are available. Suggested approaches using indicator species, guilds, and habitat relationships all have potentially serious shortcomings. To date, no comprehensive system for monitoring wildlife resources on a major land-management unit has been developed and tested, although elements of such systems can be suggested. For example, a distinction between high-risk and low-risk species allows us to deal with these two groups differently, with considerable potential for cost savings and a minimum likelihood of threatening any population. Other cost savings can be realized from appropriate sampling designs and from monitoring only to detect significant declines in the abundance of target species. In any case, the monitoring challenge will not be met satisfactorily without much additional research and testing.

Long-term maintenance of viable populations of animals is the pivotal wildlife element in the current generation of natural-resource management plans. Without this objective, the future of wildlife is uncertain. With it, planning necessarily includes other elements (e.g., dispersion of habitat patches, monitoring populations) that will dictate many future trends in planning, research, and management. In this paper, I evaluate two major trends: 1) The use of models to forecast long-term trends in animal numbers, and 2) the development and use of monitoring methods and systems to assure that we meet the goals prescribed in management plans.

FORECASTING TRENDS IN ANIMAL NUMBERS

Ecologists and wildlife biologists have been forecasting trends in animal numbers in response to management for decades. The primary difference between that effort and what wildlife biologists are attempting now is one of scale. Previous efforts dealt with one or a few species. They usually were based on detailed ecological studies of those species, and forecasts of population trends were typically short-term. Today the managers of state and federal lands typically are being asked to assure that their activities will not result in the endangerment or extinction of any species. This means that all species must somehow be considered in relation to a host of environmental changes, that very long-term forecasts be made, and that this be done without the luxury of detailed field studies of the species involved.

Major Agency Approaches

Two rather different but complementary approaches have been used to aid this process. First, the USDA Forest Service, following the lead of Thomas (1979), is developing its Wildlife Habitat Relationships (WHR) Program. This program rates all habitat types and successional stages according to their suitability for breeding, feeding, and resting by all terrestrial vertebrates of a defined area. The rating scale is usually a coarse three-rank system, generally in terms of optimum, suitable, and marginal (Fig. 1). This program also uses fairly coarse-grained classifications of habitats and successional stages, the sort used in so-called forest models to predict habitat changes in forested environments. Nonetheless, these WHR models at least permit the wildlife manager to make some judgments about general trends in the abundance of any given species, if he knows how habitats will change when management plans are implemented.

Second, the U.S. Fish and Wildlife Service is developing Habitat Evaluation Procedures (HEP) (U.S. Fish and Wildlife Service 1980a, 1980b), in which a Habitat Suitability Index (HSI) is prepared for selected species (U.S. Fish and Wildlife Service 1981). An HSI model is not unlike the habitat-rating scale for a given species in the Forest Service's WHR system, but it is more detailed and relates habitat suitability to measured values of specific habitat parameters, such as percent canopy cover or distance to free-standing water. This system allows managers to rate habitats for those species modeled, with the hope that they can be used as indicators of habitat suitability for other species known to have similar habitat requirements. Finally, if the manager knows how specific habitat parameters will change as a result of management activities, HSI models can be used

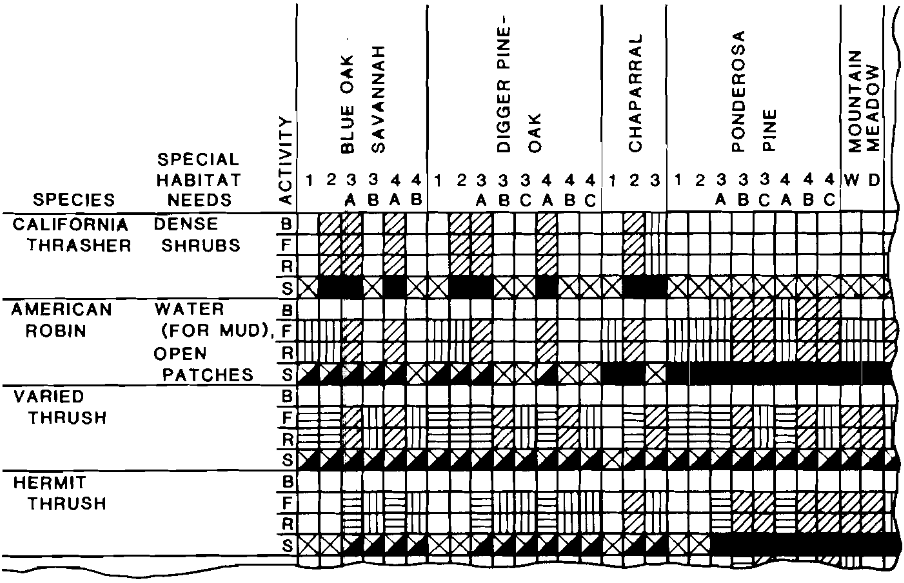

Fig. 1. Portion of a wildlife habitat relationships (WHR) matrix for birds in the western Sierra Nevada (based on Verner and Boss 1980). Species and their special habitat needs are identified in 2 columns at the left. In the Activity column, B = breed, F = feed, R = rest, and S = season of occurrence. Habitat types are identified across the top. Numeric codes at the top identify successional stages: For Woodlands and Forest, 1 = grasses/forbs, 2 = shrubs/seedlings/saplings, 3 = poles/medium trees, 4 = large trees; for Chaparral, 1 = grasses/forbs, 2 = 50% shrub cover, 3 = 50% + shrub cover. Alpha codes at the top identify canopy-closure classes: A = <40% closure, B = 40%-69% closure, C = 70% + closure. For Mountain Meadows, W = wet meadow, D = dry meadow. Habitat suitability ratings for the various activities (B,F,R) in the different habitat conditions are designated in the appropriate cells by vertical bars (optimum habitat), diagonal bars (suitable habitat), horizontal bars (marginal habitat), or no bars (unsuitable habitat). Seasons of Occurrence (S) are designated by blackened quarters (cells are divided into quarters by diagonal lines between corners), as follows: Left quarter = spring, top quarter = summer, right quarter = fall, bottom quarter = winter. Scientific names: California thrasher (*Toxostoma redivivum*), American robin (*Turdus migratorius*), varied thrush (*Ixoreus naevius*), hermit thrust (*Catharus guttatus*), blue oak (*Quercus gouglasii*), digger pine (*Pinus sabiniana*) ponderosa pine (*P. ponderosa*).

for long-term forecasting of population trends in the modeled species.

WHR and HSI models can be used in a complementary way. For example, the finer scale of habitat evaluation in HSI models can be used to improve the ranking of habitat suitability shown for species in a WHR matrix. Fry et al. (1986, in press) developed a method for using a WHR data file to objectively select species to be used in the HEP process. They concluded that "stratified random selection of evaluation species by taxonomic class is generally preferable to selecting species according to guild membership, or by simple random selection. Finally, the number of species chosen to represent a habitat type is inversely related to the maximum percent of error in the sample mean Habitat Suitability Index (HSI). Hence, one should maximize the number of evaluation species within the constraints of a project's budget."

Assumptions of the WHR and HSI Models

Three major assumptions are made when WHR and HSI models are used to evaluate the impacts of habitat changes on wildlife, especially for long-term forecasting of trends. First, we assume that ratings of habitat suitability for the various wildlife species are reasonable. Second, we assume reasonably accurate models of plant succession. Third, when applying the models to major land units, as opposed to a single patch of habitat, we assume that periodic changes in the total habitat profile are periodically updated in our record of the total area of each habitat type. Failure of the first or second assumption will lead to erroneous predictions about trends in animal numbers. Failure of the third assumption foils any effort to maintain an accurate update of those predictions.

Accuracy of HSI and WHR Models

The HSI models used with HEP analyses are typically developed from existing literature, together with expert opinion. Few careful studies have tested the accuracy of any of the many models now available, but the few that have been tested have given mixed results. For example, Lancia et al. (1982) used an HSI model to predict habitat use by bobcats (*Felis rufus*) in North Carolina and found close agreement with real use as measured by radio-tracking the animals. On the other hand, Cole and Smith (1983) tested HSI models for four species of birds and three species of small mammals in West Virginia and found uniformly poor agreement between predicted and measured abundance. These models were then modified empirically to fit their abundance measures well, but the models were not tested in other areas or years to evaluate their general applicability.

The importance of this testing is shown by a study of martens (*Martes americana*) in California (R. H. Barrett, pers. commun.). The general HSI model for martens, developed by the Fish and Wildlife Service (Allen 1982), was used to predict martens' visitation rates to feeding stations in the Sagehen Creek Basin, Nevada County, California. Refinement of the model was needed to make it fit the data better. This "Sagehen" version was then tested at locations in Shasta and Modoc counties, California, where results required additional refinements of the model to make it more generally applicable.

Habitat-suitability rankings in the Forest Service's WHR system are subject to the same criticisms that apply to the HSI models. As with tests of the HSI models described above, tests of predictions using information in the WHR files for national forests gave mixed results. For example, for mixed-evergreen forest in northwestern California, Raphael and Marcot (1986, in press) found that "species richness among stages differed significantly from model predictions for birds but not for amphibians, reptiles and mammals. Overall, 11% of the species predicted to occur were not observed, whereas 14% were observed but not predicted. The model predicted changes in relative abundance between stages less successfully. Of 650 between-stage predictions, 43% were in error but only 10% were serious (i.e., involving unexpected declines in abundance). Error rates were constant across seral stages but varied among taxa." Similarly, for birds of mixed-conifer and black oak (*Quercus kelloggii*) sites in the western Sierra Nevada of California, Dedon et al. (1986, in press) found the predictive ability of WHR models to be poor for some species.

Finally, in a study of bird communities in oak-hickory (*Quercus-carya*) forests of the southeastern United States, Hamel et al. (1986, in press) showed that predictions of WHR models matched study results progressively less well with distance of the study area from the locality for which the WHR information was specifically developed. All of these studies suggest that the models should be applied only with the cautious interpretation of a trained wildlife biologist.

Accuracy of Plant Succession Models

Munro (1974) and Shugart (1984) distinguish "tree models" and "forest models" among the many approaches used today to simulate the dynamics of forests. Tree models use individual trees as the basic unit, projecting the development of forest stands by simulating the growth of individual trees, interactions among trees in a stand, germination and mortality rates, etc. Forest models generally focus on large-scale changes in the composition and structure of forest stands, more or less identifiable as successional stages. Modeling forest changes at the tree level is roughly comparable to using HSI models for predicting animal responses to forest succession, and modeling at the forest level is roughly comparable to using WHR models.

As with the wildlife models, predictions of few models of plant succession have been tested empirically. This is especially true for predictions extending several decades into the future, because we lack long-term sampling in most habitats. One exception is that of the Prognosis Model, used to predict stand development in forests of the northern Rocky Mountains (Stage 1973). It has been used extensively by foresters in that area to predict patterns of stand growth under different management options. The model was tested recently with data from several permanent plots in highly productive coniferous forest stands in northern

Idaho (Stage, in press). The duration of sampling at these plots averaged 35 years (maximum 70 years). Measured volumes of wood production during that period averaged 2898 cubic feet/acre ($SD = 1588$); the original prognosis model overestimated production by 39 percent (4038 cubic feet/acre, $SD = 2000$). The overestimate resulted from inaccurate projections of tree mortality. A corrected version of the model now generates acceptably accurate predictions of timber production. This experience, however, demonstrates the need to test all such models with known data sets covering the longest time period possible. Meanwhile, we must use the models we have and interpret the results with due caution.

Periodic Updating of Information Files

Periodic appraisal of the current status of wildlife populations and the long-term outlook for their continued maintenance in a given land-management unit is an essential aspect of wildlife management. To accomplish this, we must have periodic updates or inventories of the structure and composition of vegetation throughout the management unit. Efforts to do this for national forests in this country have been marginally successful at best. I believe this is in part because suitable computerized data files have not been put into place, and in part because it is prohibitively costly. I agree with Mayer (1986, in press), however, that these constraints can be overcome in the near future by wedding technologies such as remote sensing (including satellite imagery) to identify habitat types and successional stages with Geographic Information Systems (GIS) (e.g., Mead et al. 1981). Indeed, with a good GIS in place and a conscientious land manager, an ongoing record of changes in all habitat polygons (i.e., patches in a habitat mosaic) could be maintained in the computer file, thus eliminating, or at least markedly reducing, the need for periodic inventories, such as by remote sensing.

The direct cost of these systems to wildlife budgets may be minimal, at least for some land management agencies, because the information they produce is vital to major revenue-producing resources, such as timber for the Forest Service. Managers of wildlife resources need only develop models that use the same (or easily translatable) information.

Limitations of the Habitat Approach to Impact Assessments and Trend Predictions

Abundance as a Measure of Habitat Suitability

The many HSI and WHR models now available were developed on the pivotal assumption that some measure of a species' abundance can be used as an index to habitat suitability. This is true whether the model was developed from actual field estimates of density, relative abundance, or habitat use, or from judgments accumulated over time by experts most familiar with a given species in the field. Van Horne (1983) correctly challenged this assumption, pointing out that for various reasons, a species' abundance in a habitat may be a poor index to the suitability of that habitat. Socially subordinate animals are commonly forced to disperse into unsuitable habitats where breeding is seldom successful. But because the availability of optimum habitat may be limited, the density of a species may be greater in less suitable habitats. Unless the observer understands this, measures of abundance could lead to a reversal in the ranking of habitats on a suitability scale.

Wiens and Rotenberry (1981) used the terms "source population" and "sink population" in their discussion of this phenomenon. Source populations occupy habitats suitable for breeding, and have reproductive output exceeding the capacity of the local habitat. Sink populations occupy inferior habitat, where their reproduction is inadequate to maintain their numbers. Sink populations must be maintained by periodic emigration from source populations. I had the same concept in mind in suggesting that the three habitat suitability classes (optimum, suitable, and marginal) used in the Forest Service's WHR system might ultimately be based on knowledge of a population's reproductive success. As data permit, abundance measures as indices of suitability might be replaced by more objective measures, "such as optimum in habitats where reproduction generally results in surplus individuals (r is positive), suitable where reproduction generally results in population maintenance (r is zero), and marginal where reproduction generally is insufficient to maintain the population (r is negative)" (Verner 1980:207). Over the long term, source populations occupy optimum habitat and sink populations occupy marginal habitat.

Indices based on measures of demographic parameters, such as clutch sizes of birds, litter sizes of mammals, or growth rates of young (e.g., Best and Stauffer 1980, Rogers 1985 and references therein), probably will prove to be more reliable than those based on measures of abundance. Because the demographic information is normally so expensive and time consuming to obtain, research in this area should give first priority to the species judged most likely to be threatened by management activities. Until such improved indices are available, however, I believe the current pace of habitat alteration forces us to use measures of abundance to index habitat suitability.

Population Regulation by Factors Other Than Habitat Carrying Capacity

Both HSI and WHR models are based on the assumption that wildlife populations increase and decrease in proportion to the availability of suitable habitat. This is not always the case, because animal numbers may be held well below carrying capacity by factors that act independently, or nearly so, from habitats (Andrewartha and Birch 1954). In a recent review, Carey (1984) identified several categories of these factors: 1) Other organisms may have effects through mechanisms of predation, parasitism, intra- and inter-specific competition, or facilitation (e.g., primary cavity nesters excavate cavities later used by secondary cavity nesters). 2) The local abundance of migrant species may reflect conditions in transit or on the wintering grounds (see Fretwell 1972). 3) Weather extremes can reduce local populations well below carrying capacity (e.g., Hejl and Beedy 1986, in press). 4) Stochastic events, which may be genetic, demographic, or environmental (e.g., catastrophes such as volcanic irruptions and tidal waves), influence populations largely or totally independent from habitats (see Rotenberry 1986, in press). Even fine-scale changes in the physical environment that have little or no visible effect on habitat structure are known to result in marked changes in the abundance of certain bird species (Karr and Freemark 1983).

We know very little of the extent to which such factors negate inferences from habitat relationships about the status of wildlife populations. The general topic represents a high priority area for research, and the need for answers is urgent. Whatever the research may reveal, we can be assured that habitats will never be taken completely out of the picture, because without sufficient suitable habitat, a population of animals cannot maintain itself, regardless of other factors that keep its numbers below carrying capacity.

Landscape Ecology—Effects of Edges, Habitat Mosaics, and Fragmentation

A set of related influences on animal populations is the complex of edges between adjacent habitats of different types, the mosaic of habitat patches, and the existence of isolated "islands" of habitat widely separated from other areas of similar habitat. These factors compose the meat of a relatively new field of study called landscape ecology (e.g., Tjallingii and de Veer 1981, Risser et al. 1984, Pickett and White 1985). The importance of habitat edges to wildlife has been widely recognized and studied for decades, and much productive research has been done recently on the effects of habitat fragmentation on wildlife populations (e.g., Whitcomb et al. 1981, Ambuel and Temple 1983, Lynch and Wigham 1984, Simberloff and Abele 1984). But little quantitative information is available on the effects of habitat patchiness on animals. Certainly the sizes of habitat patches in a mosaic, the types of habitats that adjoin one another, the distances between patches of the same type, and the kinds of habitats intervening between patches of the same type affect whether or not a given species will be able to maintain long-term population viability throughout the mosaic.

Although we know little about the effects of patchiness on wildlife populations, some promising modeling approaches have been developed by H. H. Shugart and his colleagues (e.g., Urban and Shugart 1986). Suffice it to note that quantitative information on landscape ecology, in relation to the abundance and distribution of animals, must ultimately be integrated into any realistic model that attempts to predict the effects of habitat management on wildlife.

Components of a Complete Model for Evaluating Environmental Impacts and Predicting Long-term Trends in Animal Numbers

Assessing environmental impacts on wildlife and predicting long-term trends in animal numbers depend on models that share the same elements (Fig. 2). HSI

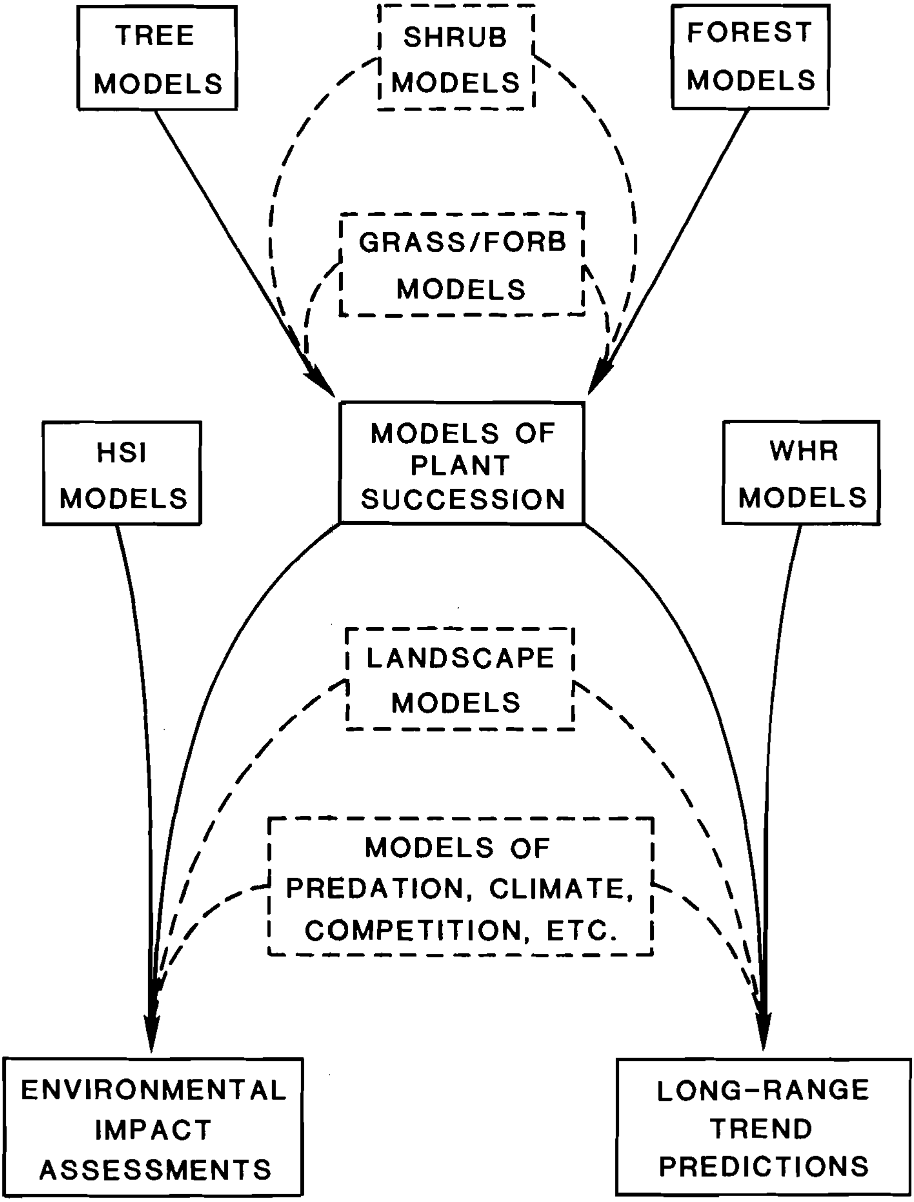

Fig. 2. Components of an integrated model for predicting long-term trends in animal populations. Solid rectangles enclose sub-models already in place, and solid lines depict their relationships to other sub-models. Dashed rectangles enclose sub-models still needed, and dashed lines depict their relationships to other sub-models. This does not imply that sub-models now in place are necessarily accurate.

models like those developed for the HEP program are intended to produce reasonably fine-grained evaluations of habitats for the species modeled. Predicted changes in the habitat can be used to predict short- or long-term trends in the abundance of those species. The kind of habitat information needed for this procedure is best provided for forested habitats by so-called tree models, but because detailed HSI models are not available for most species, we cannot make comprehensive evaluations for whole animal communities. WHR models, on the other hand, provide at least a crude ability to forecast short- or long-term trends in the abundance of all species in whole communities.

Figure 2 shows a set of submodels needed to assure an effective operational model to forecast changes in animal populations in response to habitat change. Most current applications of HSI and WHR models by managers have relied on an incomplete set of these model components. In fact, they have used only those models enclosed by solid lines in Figure 2 (e.g., Benson and Laudenslayer 1986, in press; Kirkman et al. 1986, in press). Only time will tell how accurately these efforts predict real changes in animal numbers as a result of habitat changes.

Obviously, we have only begun to explore the potential for using models in wildlife management. But if we do not make predictions based on existing knowledge, managers will make decisions without our predictions. They will use whatever information is available, with or without the benefit of a wildlife biologist's best judgment. Consequently, I believe that we must be bold and make the best predictions we can, even though we make some mistakes. As Karr (in press) pointed out, economists have not been timid about making predictions, and we all know how poorly they do at times. Certainly we can do better than economists, even with the limited knowledge we now possess about animal ecology.

MONITORING TRENDS IN ANIMAL NUMBERS

Forecasting trends in animal numbers is not enough, as forecasts are only hypotheses. Monitoring—the periodic sampling of populations to determine how closely predictions match actual trends—is a process for testing those hypotheses. Using feedback into the system from the results of monitoring, the methods of forecasting someday may be corrected empirically to a degree that the accuracy of predictions is no longer in question. Indeed that should be an ultimate objective of a well-designed monitoring system. The more immediate objective, however, is to provide sufficient information about the abundance of animals targeted for monitoring to assure that current management practices are not threatening long-term viability of their populations.

Constraints on Monitoring

Cost

We should not waste money on a monitoring system that fails to give the level of confidence needed by wildlife biologists to deduce the most likely effects of management activities on wildlife resources. Monitoring systems that do not meet this criterion will only tend to mislead us into thinking that we are managing our resources wisely when, in fact, they could be declining beyond the point of no return. The key question is, "Can we afford reliable monitoring systems?"

A "worst case" example uses the pileated woodpecker (*Dryocopus pileatus*). This species was chosen because it has been selected as a management indicator species by many national forests. It is also an appropriate choice because it is uncommon, which is typical of the rare, threatened, and endangered species that management agencies are mandated to preserve. (Uncommon animals will be more costly to monitor than common ones.) Although the cost analysis suggests a hopeless situation, this is not really the case. I itemize some cost-saving alternatives in later sections, and further research will undoubtedly reveal others. The messages I hope to get across in this example are that we must use some less-than-ideal designs to monitor population trends of many species for which we have concern, and that persons charged with developing monitoring plans must undertake a serious analysis of the costs before finalizing those plans. I have read several draft plans for monitoring wildlife that showed a total lack of appreciation for the realities of those plans when put into practice.

The cost of monitoring yearly changes in abundance of pileated woodpeckers in the western Sierra Nevada mountains are estimated here on the basis of data obtained in the Sierra National Forest (Verner 1983), together with information on the number of bird counts needed to detect differences in the abundance of birds between years or between areas (Dawson 1981). The assumptions are the presence of an ideally randomized selection of sample sites in the zone of mixed-conifer forests, and a sample size large enough to detect, with 95% confidence, a 10% change in numbers between years in 80% of the cases (statistical power = 0.8). Although one would fail to detect a significant change of 10 percent in abundance in one of five cases, continued sampling in subsequent years would make this a negligible concern. For pileated woodpeckers, "the number of counts needed annually to detect a 10 percent change in the population is on the order of 300,000. If one observer could complete 20 counts per day, the number of counts would require 15,000 observer-days. Assuming a salary of an entry-level biological technician (GS-5, Step 1), this comes to an estimated $825,000 per year" (Verner 1983:356-357). Add to that an estimated $180,000 for per diem and $33,000 for vehicles and mileage, and the total cost exceeds $1,000,000 per year.

Skilled Personnel

The potential magnitude of certain kinds of monitoring systems would quickly exceed the available pool of persons skilled enough to put them into operation. The best time to estimate the numbers of pileated woodpeckers is during the breeding season, when the birds are more conspicuous and their habitats are more accessible. Only about 40 working days are available during the breeding season, so 375 observers would be needed to cover the required 15,000 observer-days of work. Anyone who has attempted to recruit skilled birders for temporary appointments will recognize this as a serious limitation, especially if other management units are competing for the same pool of skilled people to meet the needs of their monitoring programs.

Scale

Any monitoring system must be scaled to the size of the unit in which monitoring is done. Methods applicable to one scale can be totally inappropriate to another. For example, I recommend a timed point-counting system for monitoring trends in numbers of birds (observer stands at a designated station for a fixed period, say 5 min, and records all birds detected) (Verner 1983). I have a test area to study this method in oak-pine woodlands in the western foothills of the Sierra Nevada, Madera County, California. Previous data on the mean number of birds counted per station there suggest 200 stations as a reasonable number to detect declining trends (see later discussion), at least among the most abundant species. The test design uses seven lines of 30 stations each, for a total of 210 counting stations. Allowing a minimum spacing of 200 m (650 feet) between stations, and accommodating vagaries of the terrain and habitat distributions, this design requires about 1,200 ha (3,000 acres). Such a design would obviously not work to monitor trends in relation to a single timber sale or the installation of a small park, because they typically involve land areas that are too small.

The Use of Surrogates in Monitoring Systems

Various systems have been suggested to accomplish the goals of monitoring without actually sampling populations of all species. One such surrogate was identified in a federal statute, the National Forest Management Act of 1976, which specifies that land-management plans of the various national forests will identify ecological indicator species. These are to be monitored to give information on trends in habitat capability and to indicate population trends among other species with similar habitat requirements. Other possible surrogates include groups of species, and specific habitats and their successional stages (as described in the previous section).

Indicator Species

Two fundamental assumptions are made when using population trends of an indicator species to make inferences about population trends among ecologically related species. First, one assumes that the ecological niches of the various species are sufficiently alike that even the specializations of the species for microenvironmental features of their habitat are the same. In other words, the various species have such similar niches that all are affected in the same way by any change in habitat. This assumption was challenged by Landres (1983), Mannan et al. (1984), and Verner (1984), because ecologically similar species are nonetheless different in one or more of their microhabitat needs, or in the ways in which they use their habitats, or in both. Furthermore, because species with very similar ecologies are more likely to compete for needed resources than are species with different ecologies, the presence of one species in a habitat may preclude the presence of other species with similar ecologies (e.g., see Martin 1981). In other words, the presence of an indicator species may be a better predictor of the *absence,* not the presence, of those other species most likely to respond similarly to habitat changes.

Second, one assumes that the populations of all species for which another is selected as an indicator are regulated by the same factors. This may be true in some instances, but it is not likely to be the case for many species. Among birds, for example, migrants may be subjected to sufficient mortality in transit or on their wintering grounds to hold their breeding populations well below carrying capacity. Populations of other species may periodically suffer heavy mortality from climatic extremes, predation, or infectious disease. Obviously, many factors or combinations of factors may set limits on the populations of species that compose a particular assemblage of animals, whether amphibians, reptiles, birds, or mammals. To assume that all species with similar ecologies are similarly regulated is probably to err more often than not.

Recent empirical evidence supports the arguments developed here that one cannot readily find an indicator species whose population trends parallel those of a group of closely related species. Mannan et al. (1984) grouped birds into guilds, sensu Root (1967), to determine whether the members of each guild exhibited similar population changes in response to forest management practices in coniferous forests of Oregon. Concurrence in population responses was poor among the species in each guild, especially those including four or more species. R. C. Szaro (pers. commun.) found similar results among bird communities in pine forests of Arizona. Species in smaller guilds sometimes showed similar population responses, suggesting that smaller guilds may be better. But, as Mannan et al. (1984) and Verner (1984) correctly point out, the advantages of grouping species into guilds are lost if the number of species per guild is reduced to one or two.

Guilds

I proposed the use of "management guilds" in place of indicator species to monitor trends in the capability of habitats to support certain classes of wildlife (Verner 1984:3), and defined a management guild as "a group of species that respond in a similar way to a variety of changes likely to affect their environment. . . . For example, one could group in a management guild all bird species that depend upon tree canopies for their food supply, whether fruit, buds, leaves, or insects taken from foliage or twigs." A management guild approach, at least with birds, has several advantages over that of using indicator species to evaluate the capability of a habitat for supporting various species of wildlife:

1. It is less expensive because fewer counts are needed to obtain a statistically

reliable sample. The sample size needed to measure trends in animal numbers depends on total count, and the count of a guild is always equal to or greater than that of the most abundant species in the guild. Note also that it takes no longer to record all birds detected during a specified period than it does to record all individuals of just one or a few indicator species.

2. Monitoring trends in management guilds requires counting the total assemblage of birds to obtain a complete listing of species at each monitoring interval. The list can at least be studied for possible evidence of problems. Even though significant trends in numbers of most species would not be detectable, they would be detectable for the more common ones (assuming a monitoring design using 5-min point counts at 200 counting stations). But one could at least look for species additions and losses, such as starlings (*Sturnus vulgaris*) replacing bluebirds (*Sialia sialis*). Such options are lost if only one or a few indicator species are monitored.

3. Inferences about habitat capability are likely to be more complete and more accurate because they are based on the differing ecologies of a variety of species rather than on those of a single indicator.

4. When all species are counted, species may be grouped in various ways to form hypotheses about the cause(s) of observed trends. For example, if counts show significant declining trends in most assemblages grouped by zones of the habitat used for feeding (tree canopies, tree trunks and large limbs, shrubs, ground, etc.), one could regroup according to diet (insectivorous, granivorous, omnivorous, etc.) in search of further patterns. Or one could look at trends among migrant breeders as a group, compared to resident breeders. If migrant breeders showed a significant decline, but resident breeders did not, one might conclude that the problem lies somewhere along the migration route or on the winter range. If, on the other hand, resident breeders showed the decline and migrant breeders did not, a reasonable working hypothesis is that severe winter weather resulted in excessive mortality among the residents. Such options are not possible if one has estimated the abundance of only a few indicator species.

Mannan et al. (1984:429) correctly pointed out that "a potential danger of examining only the summed response of all guild members is that if intraguild responses are inconsistent, a large increase in 1 or 2 species could mask the decline or absence of others." This, too, argues for use of a guild approach, because the scenario they suggest is more likely to be detected with an estimate of the abundance of all species in a bird community than with an estimate of just one or a few. As suggested in the second advantage above, the user has a complete list of species and an opportunity—indeed an obligation—to use it wisely.

The guild concept should be considered as an option primarily in connection with "high-risk" habitats. Patterns exist in our uses of land and other natural resources that tend to make certain kinds of habitats more likely to be altered. Riparian ecosystems, for example, are especially vulnerable to a large number of common human uses, such as agriculture (e.g., grazing, tilling), development (e.g., location of industrial facilities, urbanization), and channel "improvements" (e.g., diversions, rip-rapping). Native prairies and hardwood forests of the midwestern United States have been in this category for more than a century. Forests dominated by favored timber species, and wetlands in general, are other examples. It is important to recognize that the objective of monitoring with this approach is to use assemblages of animal species to assess trends in the capability of a habitat to continue supporting those species or others with similar ecologies. Information about trends in abundance of any individual species is only a byproduct—albeit interesting and perhaps even important.

Monitoring habitats Rather Than Animal Populations

As I suggested earlier, the major modeling approaches to forecasting trends in animal numbers by both the Fish and Wildlife Service and the Forest Service assume that knowledge of wildlife-habitat relationships can be used to infer trends in animal numbers from predicted trends in habitats. I will not elaborate further on this assumption here. We all need to recognize, however, that this approach will be the least expensive in the long run, and most of the cost will be shared with other resources such as timber and range. That alone will drive management agencies toward total or near-total reliance on predicted trends in habitat conditions to infer trends in animal populations. The important take-away message is that a tremendous effort in research and development is needed, on a national scale, to assure that we have the most accurate information possible before relying totally on WHR models to predict long-term trends in animal populations. The ultimate system must be dynamic to the extent that it incorporates feedback loops for correcting existing data files as new information is gained from application of the system itself (Verner 1983).

Other Cost-Cutting Strategies

Using surrogates such as models to predict changes in habitat types instead of directly monitoring the population status of all terrestrial vertebrates can markedly reduce the costs. And I believe that the accuracy of these models can eventually be empirically improved to the point that their use will not result in significant, unpredicted losses of wildlife as a result of management actions. Meanwhile, other strategies must be used to reduce the cost of monitoring when direct measures of population change are attempted for any species. Four such strategies are suggested here: 1) Distinguish between "high-risk" and "low-risk" species, and use labor-intensive methods only for the high-risk species. 2) Monitor for high-risk species only in "high-probability" sites. 3) Monitor only to detect significant declining trends. 4) Monitor during only one season.

"High-risk" vs. "Low-risk" Species

High-risk species are those that can be identified as highly vulnerable to existing threats to their populations directly, or indirectly through their environments (change in structure or composition of their habitat, pollutants, exotic competitors, parasites, diseases, etc.). Many species in this group have already been placed on federal or state lists of endangered, threatened, rare, or sensitive species. All harvested species and other species having special public interest should be added to the list. Although they may not be high-risk species in the sense of the first grouping, still we need more direct knowledge of their population status because they have high public visibility. Finally, wildlife specialists responsible for each land-management unit have a responsibility to study the list of all species known to occur on their unit and to determine whether other species need to be added to their high-risk list, following the guidelines given by Salwasser et al. (1984).

Low-risk species are those for which maintenance of long-term population viability is likely assured, given our best understanding of future trends in land and resource management. These are species that exhibit one or more of the following attributes: high reproductive capacity, extensive geographic distribution, successful reproduction in several habitats and successional stages, optimum habitat in intermediate successional stages, and relatively abundant (with a corollary of small territory or home range).

I believe that all low-risk species can be safely monitored, at least for the next several decades, by relying totally on WHR models of the sort being developed by the Forest Service. This is not to say that those models are fully accurate at this time. They are not, but the likelihood that low-risk species will be threatened in the near future is low enough that we should concentrate our meager resources for monitoring on the high-risk group. In the long term, our real goal is to maintain the ecological integrity of natural systems, not just viable populations of target species. Therefore, we must not ignore for long the fact that WHR models for the low-risk species need verification.

Sample at High-probability Sites

Because high-risk species are commonly restricted in their distribution and often occur in only one or a few types of habitats, methods designed to monitor changes in their population status should specify sampling only in those sites most likely to be used by the species. This will require an inventory of potential sites before final selection of sites for later monitoring. If the species is abundant enough, one may have the luxury of randomly selecting monitoring sites from a larger sample of sites found to be used by the species. For rarer species, however, all sites known to have the species may need to be included in the monitoring system (in effect, a total inventory).

Declining Trends Only

Resource managers are normally under pressure only when animal populations decline, and this is especially true for high-risk species. The manager faces a type of question not normally addressed in statistical analyses of biological studies (Toft and Shea 1983, Verner 1983). The question here is, "Has the population declined significantly from one sample period to the next?" Whatever the answer, the manager needs to know how often a real decline would *go undetected* by the sampling method used. In the jargon of statisticians, this is a Type II error—a real change is missed. The measure of our ability to avoid Type II errors is called "power." When the power of a statistical test is 0.5, a real change will be missed 50 percent of the time; one may as well flip a coin. To increase the power of the test, one needs to enlarge the sample size, which means more effort and more cost.

One may wish to know whether any change—either growth or decline—has occurred in a species' population. But if the answer to only half of that question will suffice, i.e., whether only a decline occurred or only growth occurred, the cost of an adequate statistical sample is reduced by more than half. "Consider a species that averages one bird per count. To detect a 25 percent decline in its population, the number of counts needed per year ranges from 93 for a power of 0.5 to 250 for a power of 0.9. Corresponding sample sizes if one wishes to detect *any* change of at least 25 percent, i.e., either an increase or a decrease, are 362 and 2707, respectively . . . Savings using year-to-year declines range from 74.3 percent to 90.8 percent" (Verner 1983:358).

Single-season Monitoring

Ideally, wildlife populations should be monitored during all seasons each year. This is especially true for birds, because the composition of bird communities changes from season to season. However, because the animals that use a habitat in different seasons generally tend to use the same components of that habitat (e.g., tree canopies, bark, shrubs, ground) in similar ways, it may be reasonable to assume that the needs of most species in all seasons will be provided if we assure their maintenance during a single season.

The breeding season is the best time for monitoring population changes for most bird species, because they are more easily detected then, they tend to remain in the same general area for extended periods of time, and weather conditions are more conducive to efficient sampling. Exceptions exist, however. Monitoring of bald eagle (*Haliaeetus leucocephalus*) populations in the United States, for example, has been done during the winter when the birds are concentrated in a few major roost areas, so estimates of their total numbers can be more efficiently obtained.

The assumption that long-term provision of adequate habitat for breeding birds will assure adequate habitat for migrants that use the area in transit, or for winter residents, needs testing. The question has not been studied directly, although limited supporting evidence was given for birds in oak-pine woodlands in the western foothills of the Sierra Nevada: "Transients and winter residents foraged in the same zones . . . as breeders, although not in the same proportions. . . . Tree canopies were used more extensively by the transients and winter residents than by the breeders. Winter residents fed on the ground more than either of the other groups (this reflects a significant addition of granivores to the bird community during the winter). And aerial searching for insects or ground prey was much more common among the breeders. These were differences only in percentages of use of the various zones of the habitat already identified as important to breeders. Consideration of the needs of transients or winter residents did not introduce any new zone or pattern of habitat use. Adequate maintenance of these zones for the existing breeding species should, therefore, assure continued provision for the transients and winter residents that use the same area'' (Verner 1984:12).

Recent studies by Morrison et al. (in press) in Sierran mixed-conifer forests of El Dorado County, California, call for a healthy measure of caution when relying solely on single-season monitoring. Their results show that birds seldom foraged on incense-cedar (*Calocedrus decurrens*) during the breeding season, but they foraged extensively on the bark of this species during the winter. Further study of this relationship showed that the birds were feeding primarily on one species of scale insect, *Xylococculus macrocarpae,* that was plentiful on the bark of the cedars. Because incense-cedars have not been found to be important in the breeding ecologies of wildlife in the Sierra Nevada, and because they have less value as timber than other species that thrive in the same areas, little attention has been given to retaining significant numbers of the trees in managed stands of the Sierra Nevada (Airola and Barrett 1985). The question remains whether or not the scale insects on the cedar bark are an essential food source for wintering birds, but it is clearly premature to judge that the species is of little consequence to wildlife based on its use during the summer.

CONCLUSIONS AND RECOMMENDATIONS

The future in wildlife management—game and nongame alike—includes the development and use of a variety of models, from simple to highly complex. I see no way to accomplish the ambitious goals we have for long-term predictions of plant and animal communities without such models. And I see no way to assure long-term maintenance of viable populations of all wildlife species without major use of long-term predictions. Something akin to HSI models will probably be used for making critical decisions about high-risk species, and more accurate versions of WHR models will be used for low-risk species. Greater sophistication and more reliable predictions will come as the accuracy of existing tree models is improved, and as shrub models, landscape models, and models to predict the effects of predation, competition, disease, weather extremes, etc., are finally integrated into a comprehensive system of predictive models, and as these models are interfaced with a Geographic Information System (GIS).

I believe we will soon see the emergence of acceptably accurate and cost-effective remote sensing technology to classify habitats as defined for HSI and

WHR models. Undoubtedly, GIS will be incorporated into this picture. Without these technologies in place, prospects are shakey for periodic updating of our information on how the plant communities represented by habitat polygons have changed over time. Yet that updating is essential to the success of the kinds of long-term predictions envisioned for future wildlife management.

The implementation of effective monitoring systems is essential for the success of any effort to integrate a number of diverse models to predict long-term effects of habitat management on wildlife. Monitoring will provide two vital, related requirements. First, it will allow us to determine whether model predictions are correct. This will be possible only on a short-term basis initially, but eventually even some long-term predictions can be evaluated. Second, the information gained from monitoring will be our primary source for correcting errors that lead to incorrect predictions. Monitoring, then, has the potential to become *the primary feedback loop into the basic models that determine our predictions.* As such, monitoring is an essential part of the overall system.

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