Conservation Techniques

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She would also like to extend a heartfelt thanks to Glen Kandia for his painstaking and cheerful editing of drafts. His comments were always useful and he consistently made time on short notice to read chapters. His feedback surely helped to improve the quality of the wording in this textbook. She would also like to thank Kathy Mills and Kristi Arend for their comments on early drafts of some of the chapters in this book. Marcia Meixler's department at Rutgers University was very supportive of her decision to write this textbook and she is grateful to them as well.

Marcia Meixler also wishes to thank her immediate and extended family who have been a source of incredible support during this process. Finally, she would like to thank her friends for tolerating her attempts to work conservation-related facts from this textbook into conversations during the writing of this book, and for providing support when the project seemed immense.

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Foreword

Knowing that the resources of our planet are limited, competing interests make managing and conserving these resources an interesting challenge. This textbook "Conservation Techniques," by Drs. Marcia Meixler and the late Mark Bain, provides valuable lessons not only in the scientific principles behind conservation and management efforts but also in how that information is applied in several real-world examples.

The seeds for this textbook were planted in 2009 when Mark began teaching a course titled "Earth Care: Applying Knowledge to Conservation," and after a time decided to use much of the course material to write a textbook. Early in the process, though, he became ill and soon thereafter passed away. I eventually met with Marcia and a number of Mark's colleagues to decide what should be done with the Earth Care materials, but the timing did not work out for completing the book. Fortunately, Marcia used both the earlier course material and information from the drafts of book chapters to teach a course titled "Conservation Techniques." Those pieces, along with updated course material, provide some of the core components of this textbook.

I've known both Marcia and Mark for most of my adult life: not only do I share a similar interest in ecology but regarding Mark, as the famous line from Jane Eyre goes: "Reader, I married him." Over several years I witnessed Marcia and Mark working together as a highly productive team on many projects ranging from research on stream biodiversity, to hydroecology and conservation mapping as a tool for community planning, and analysis of impairments in tributary watersheds. Together they published several scientific papers covering topics such as stream habitat restoration needs, river restoration, and predicting barrier passage and habitat suitability for migratory fish species. Mark served as Marcia's advisor for her Masters and PhD programs, and though both had pursued rigorous academic backgrounds, they also excelled at applying their knowledge to solving pressing environmental problems by working closely with stakeholders from various viewpoints with differing agendas. Both Marcia and Mark demonstrated a keen ability to communicate effectively with groups of people such that conflicting viewpoints would not necessarily hinder what they set out to accomplish. It's been often remarked that Mark's warm laughter and sense of rare good humor drew others to him, enhancing his ability to get people to cooperate on challenging projects. Even the director of the Hudson River Foundation would refer to Mark as his "coordinating genius."

This textbook includes details of several case studies that illuminate the uniqueness of each conservation challenge. One of the most prominent cases is known as the "Peace Treaty on the Hudson River," for example. Several books have been written on this case that cover the dispute over a proposed pumped-storage hydroelectric power plant at Storm King Mountain; a controversy that lasted well over a decade and resulted in several new laws and the creation of regional and national environmental organizations.

A collective effort is required when handling decisions around the conservation of ecosystems. The authors of this textbook recognized that not only are expertise and enthusiasm required, but skilled leadership, a trust in the value of collaboration, and a practical understanding of the issues, conflicts, interests, values, and personalities to advance those management efforts and decisions.

Mark would have been particularly pleased that this opportunity for providing an open-access textbook was recognized by Marcia as a meaningful way to continue spreading their breadth of knowledge and experience with others interested in mastering the scientific principles, making sound decisions, and taking effective actions regarding environmental management and conservation.

Jane L. Bain

Introduction

THE PURPOSE OF THIS BOOK

This book fosters the recognition of options for making progress toward increased environmental conservation through an understanding of the underlying science and practice of a variety of conservation techniques. Today, there are expected benefits from integrated science and practice, and many people are promoting this as the way forward to improve our environment. Over time, trends emerge regarding the best way to conserve the environment, but so far an outstanding solution has not emerged. Each conservation technique has its foundational concepts, limitations, and implementation issues. Reviewing a collection of techniques provides a basis for considering which approach will be best for any specific environmental challenge. This book should advance the recognition of the challenges managing the environment, techniques that can be used to address the challenges, and the ways they might help foster the integration of science and the practice of ecological conservation.

This book is intended for students and management professionals who might benefit from a vision and guiding path that leads toward achieving ecological conservation for the long-term. This book is unique in the Open Educational Resources (OER) space as it seeks to present a range of conservation techniques with differing science concepts and applications.

LOGICAL ORGANIZATION

Our book covers 13 distinct techniques for ecological conservation. Some techniques are old and specified in laws (e.g., standards and criteria, the Endangered Species Act, National Environmental Policy Act), while others are newer and are just beginning to be put into practice more frequently (e.g., ecosystem services, rewilding, and sustainability). Some techniques were proposed by scientists long ago and have more recently become commonly used (e.g., adaptive management, restoration, ecosystem-based management, and ecological engineering). Thus, this book covers a wide range of both old and new ideas about ecological conservation.

Each of the techniques has its own chapter and each chapter begins with a small section introducing the topic and explaining what will be covered. Chapters vary in their content depending on the topic, but common sections include the technique’s basis in science, and a review of the technique in practice with some background on its procedures, implementation issues, controversies, and impediments. Each chapter also includes a case study to illustrate the application of the technique. Finally, the chapter ends with a short summary of the important aspects of the technique.

The 13 techniques fall into four main categories: fundamental techniques, biologically-focused techniques, habitat-focused techniques, and holistic techniques.

Fundamental techniques

The topics in the fundamental techniques group consist of: 1) science and practice; 2) standards and criteria; and 3) National Environmental Policy Act (NEPA). *Science and practice* are distinctly different endeavors that are not easily integrated together to improve ecological conservation. Starting the book with a chapter on the distinctions between science and practice sets the stage for reviewing the techniques in the remaining chapters.

Standards and criteria was the first technique implemented to regulate environmental pollution, and remains a prominent part of current environmental conservation. Science that supports standards and criteria is reviewed, as well as the principles used for assigning these standards. This chapter also reviews how agencies charged with setting standards develop the regulations they enforce. Precise numerical regulations demand clear justification using scientific information and these ideas are discussed. This chapter examines the meanings of standards and criteria, provides lessons on their practical implementation, and explores some examples.

The *National Environmental Policy Act* (NEPA) requires an open process for public consideration when potentially major impacts will significantly affect the environment. Passed in 1969, NEPA is considered the *Magna Carta* of United States environmental laws. There is a long history of NEPA's use, since almost any work that affects the environment triggers NEPA and results in the generation of impact statements. Hundreds of NEPA impact statements are issued each year so it is easy to feel that this is routine government work. However, the NEPA process was established to improve ecological conservation, and that process will continue. Although old and not considered as a hopeful way to improve the environment today, this technique merits attention because it is an active and central part of ecological conservation. This chapter presents some background on NEPA, a review of the process for implementing it, and an exploration of how it has performed.

Biologically-focused techniques

The topics in the biologically-focused techniques group consist of: 1) rewilding; 2) endangered species protection and recovery; and 3) biomonitoring. The concept of rewilding is grounded in the notion that to have truly natural ecosystems, the ecological processes have to be reestablished, then dynamic natural processes will yield a natural ecosystem. Large carnivores and herbivores are seen as necessary to shape the flora and physical characteristics of a natural environment. Rewilding can be considered a proactive conservation strategy that attempts to restore natural environments by reinvigoration. *Rewilding* can be risky, often with unanticipated and catastrophic effects on native flora and fauna, habitats, and ecosystems. Many conservationists and managers view the rewilding approaches as controversial and not commonly advocated. The main issues are the use of non-native species, high risk of unintended consequences, and potentially high public attention. This chapter covers some background on rewilding, the theoretical basis for its use, and examples of implementation.

The United States Endangered Species Act defines the approach to *endangered species protection and recovery*, and this is a prominent part of ecological conservation in the United States. The provisions of the Act are complicated and extensive, but are important for guiding conservation practices under the law. There has been extensive scientific research on endangered species listings and recoveries, which provides a frame of reference for improving the operation of the Act. In addition, there is an extensive record of actions using the Endangered Species Act to save and recover species. This chapter presents the history and effectiveness of the Endangered Species Act, the process of listing (or delisting) species, and the criteria for determining endangerment and recovery.

The use of biological standards and criteria to perform *biomonitoring*, which helps us assess biological integrity, emerged from the water regulation arena in the 1970s. Extensive research and scientific principles support biomonitoring, and also foster a better understanding of the quality of environments. There are a wide variety of guides, manuals, and cases using biological properties to assess ecological quality. One distinctive feature of this technique is its reliance on natural reference conditions to set standards. Given natural reference conditions, indices can be used to estimate numerical quality ratings. This technique relies on the biological community to indicate problems and needs, and is well developed for implementation in conservation. This chapter covers a review of the background and reasons for implementing biomonitoring and how it works in practice.

Habitat-focused techniques

The topics in the habitat-focused techniques group consist of: 1) habitat assessment; 2) restoration; and 3) Ecological Engineering. Development often results in a loss of habitat. Habitat is easily defined, inventoried, and mitigated for losses. In ecology there are principles of habitat analyses, and in applied science there is a rich record of research on *habitat assessments*. Agencies charged with maintaining species and habitats have well-developed methods for identifying habitat losses and mitigation strategies. In practice these methods are routinely employed. In recent years, many ecosystem-scale ecological management plans have been based on habitat analyses for exploring different future scenarios. Landscape scale habitat modeling is fairly new in science, and is applied in practice when considering complex options for ecosystem management. This chapter presents principles of habitat analysis and landscape scale modeling methods, and how these ideas are used in practice to mitigate habitat losses and evaluate tradeoffs.

The traditional definition of *restoration* is returning an ecosystem to its former, undisturbed state with the original functions and structure. The science on restoration increased greatly in the 1990s and is still growing. Restoration science is diverse in its scope, and addresses measures of success, ecosystem properties, and means for reversing environmental damage. In practice, the scope of restoration is broad and includes public interests, partnerships, and education. Most applications target habitat restoration for specific benefits, which may not seek to return the habitat to its original and natural conditions. This chapter explores the background of restoration, its track record, and details on why this has become a very active management technique.

Humans are creating new ecosystems that have novel properties, possess new biological communities, and support people. *Ecological engineering* is focused on designing and reconstructing environments consistent with ecological principles and integrating human society with its natural environment. The science aimed at these ecosystems tends to focus on ecosystem stress, defining ecosystem quality, resistance to change, self-organization capacity, and diverse biological structures. Practitioners design strategies for rehabilitating or renewing ecosystems to make better environments that were irreversibly damaged, abandoned, or permanently altered. Creative practices are especially needed in highly stressed ecosystems that are not expected to return to a near natural state. Common goals in establishing new ecosystems are to support greater biodiversity, integrate human activity, and provide sustainability through internal system processes. This chapter explores the background of ecological engineering within a framework of ecological stress and health, and delves into the ideas that set ecological engineering apart from restoration.

Holistic techniques

The topics in the holistic techniques group consist of: 1) ecosystem-based management; 2) adaptive management; 3) ecosystem services; and 4) sustainability. Holistic environmental management was proposed decades ago, and has only more recently seen widespread implementation, especially in marine ecosystems. Agencies like the United States National Oceanic and Atmospheric Administration and others have developed frameworks for *ecosystem-based management*. Therefore, an extensive background of scientific research and applications are available for use in discussing this technique. This chapter presents the background and justification for ecosystem-based management, and possible avenues for implementation.

Adaptive management is a technique that fits situations which are important to address, but where the information necessary to make confident decisions is lacking. The central basis for adaptive management is to learn from management outcomes. This approach includes iterative adjustments in plans over time using knowledge gained during the process. Adaptive management was introduced by scientists that saw the need to treat management as an experiment for learning. Adaptive management seems most appropriate as agencies and managers shift to ecosystem-scale challenges. Exploring management alternatives, predicting outcomes, monitoring results, and updating management plans is, in short, adaptive management in practice. The track record on this technique suggests that when governments and agencies invest in the approach, it can succeed over time and truly improve ecological conservation. This chapter covers details of the process of adaptive management, and its benefits and limitations.

Natural ecosystems provide humans with many diverse benefits and products. These benefits and products are called e*cosystem services*, and the recognition of these services are one way for increasing investments in conservation. Much recent research has focused on exploring patterns of response of ecosystem services to change, distribution of service flows in space and time, conditions that promote the stability of services, tradeoffs and synergies among services, and resilience of ecosystems when managed for particular services. Valuation of the services an ecosystem provides is challenging, but this process has seen some success when ecologists and economists have collaborated. The priority has been to identify a broad range of ecosystem services and practical measures of service benefits. Payments for providing ecosystem services have been implemented to promote conservation, and provide direct benefits to local people who maintain the ecosystems. There is a good deal of optimism on how this ecological management technique can advance conservation as a mainstream societal need. This chapter presents the background and justification for emphasizing ecosystem services as an approach to conservation, and attempts to economically value those services.

Sustainability as a technique for ecological conservation is not new because it was the principle for exploitation of natural resources decades ago. However, this conservation technique has been redefined over the years and is now a popular concept for current management. The definition of sustainability often includes concepts involving the maintenance of resources for future generations, interactions between humans and the environment, and interdisciplinary collaboration to solve problems. The science focused on sustainability includes both the strategies and mechanics of reshaping the effect people have on the environment with a long-term perspective. In practice, there are varied goals and objectives, measurements of performance, and accounting systems for determining progress made through implementation of sustainable actions. This chapter defines sustainability, provides examples of sustainable actions, covers information on recent developments in the field, and presents illustrations of successful applications of sustainable principles.

THE AUTHORS

Marcia S. Meixler is a professor of spatial aquatic ecology in the Department of Ecology, Evolution, and Natural Resources at Rutgers University. She has worked closely with organizations like The Nature Conservancy, the New Jersey Department of Environmental Protection, the New York Department of Environmental Conservation, Great Lakes Protection Fund, Wildlife Conservation Society, Hudson River Foundation, Charles River Watershed Association, National Park Service, Niassa National Reserve in Mozambique, and more to further the conservation goals of each organization. She uses spatial analyses of mapped data to explore issues in freshwater and coastal ecological areas. The topics of her projects have ranged to include things like: protection of biodiversity, the sustainability of northeast native fisheries, ecosystem planning, wetland degradation, animal migration, renewable energy, transportation, land use planning, and more. Her teaching covers the content of this book and courses on sustainability, freshwater ecology, landscape ecology, and Geographic Information Systems (GIS).

Mark B. Bain was Professor of Systems Ecology in the Department of Natural Resources at Cornell University. He was a quantitative aquatic biologist and ecosystem scientist conducting both basic research and studies driven by current management issues. His research focused on coastal ecosystem restoration and conservation, ecosystem analyses and assessment, recovery of endangered species, and monitoring of invasive pathogens in Great Lakes waters. Mark Bain's environmental policy experience included ecosystem management, endangered species protection, energy - environment conflicts, watershed conservation, and water management on the Great Lakes. His teaching covered the content of this book and a course on environmental systems.

Photo: M. Todd Walter

Fundamental Techniques

Chapter 1 - Science and Practice

The first chapter in the fundamental techniques group is on science and practice. Science and practice are unique and yet linked together in the goal of improving ecological conservation. This chapter on science and practice sets the stage for reviewing the conservation techniques covered in the remaining chapters of this book. We will define the meanings of science and practice, explore parallels, differences, and challenges, and end with a case study exploring the roles of science and practice in a long legal battle over whether to build a pump storage plant in the face of Storm King Mountain on the Hudson River.

SCIENCE AND PRACTICE DEFINED

Science has a tradition of showing and promoting progress. Science is the use of observation, study, and experimentation to derive knowledge about the nature or principles of what is being studied. Ecological science is aimed at understanding nature, and forming principles that explain the processes and properties of environmental systems. The basic scientific method includes (Figure 1.1): 1) Reporting an observation or finding; 2) Developing a hypothesis to explain the observation or finding; 3) Testing the hypothesis using data and analyzing the results to determine if the data are significantly different than expected values under the null hypothesis; 4) Interpreting the results and comparing those with the results of other studies; and 5) Forming conclusions in the context of theory or principles. This scientific method shapes research and scientific activity, and also serves as the basis for making progress. Science contributes to ecological conservation by generating facts, improving understanding, and providing the ability to explain natural phenomena.

Figure 1.1: Scientists taking a soil core as part of the scientific method. Source: National Science Foundation 2021

Conservation practices are aimed at protecting and managing the environment (Figure 1.2). Conservation practitioners must assess the consequences of their decisions through objective analysis of current conditions, plans for possible actions, and subjective evaluation of the significance of any anticipated changes. Practitioners must anticipate changes in order to formulate a best estimate of the expected outcome, without having full knowledge of, or confidence in, what will actually happen. The basic management method identifies goals and objectives, establishes baseline conditions, reviews options for action, anticipates changes under each option by collecting, analyzing and interpreting data to determine if these changes will be significant, documents expected benefits, and chooses a plan for action.

Though this sequence is often performed with uncertainty and knowledge gaps, it is necessary for deciding what to do. Conservation practice contributes to the resolution of environmental problems by making decisions, planning improvements, and anticipating future outcomes.

DIFFERING PERSPECTIVES OF SCIENCE AND PRACTICE

There are ways that science and practice fundamentally differ. Science is traditionally reductionist in that it aims to deal with hypotheses and specific questions. Scientists try to eliminate other factors that can confound responses to treatments. Recent trends in science, however, emphasize interdisciplinary collaboration and a more system-wide orientation. Practitioners are generally holistic in their approach, and consider the big picture so that all perspectives and facts are taken into account. Practitioners often work at a broad scale seeking to assemble, analyze and evaluate all the information associated with actions or change. Thus, science and practice may be converging with current trends, but traditionally diverge in scope.

One polarizing notion is the significance of an outcome. Significance in science is quantitatively estimated as the probability that an observation deviates from a hypothesis, but science tolerates only very small probabilities. More specifically, science uses the probability level associated with rejecting a true null hypothesis. This is described as the probability of having a type I error ($P \leq 0.05$). These data that form a type I error deviate from predictions under the null hypothesis, enough to be considered consistent with the alternative hypothesis. Significance in practice is more subjective and is often based on agency policies, public concerns, legal standards and responsibilities, personal preferences, and past case histories. Significance is not commonly estimated in practice. Instead, over time, a judgment is made as to whether practices are working.

Finally, achievement and progress are viewed differently in science and practice. In science the published scientific literature shows progress through changing principles and advancements in understanding. Major results are frequently challenged, tested and revised, and the theories, principles, and paradigms developed by the synthesis of these findings provide a progression of ideas and accepted truths which accumulate in the scientific literature. These widely accepted principles of science are presented in books and taught in schools. That is why scientists are expected to publish articles and present lectures. For practitioners, the basis for decisions are often poorly documented in a systematic manner, and few true management journals, books and courses exist. Little time is afforded to communicate

Figure 1.2: The practice of conservation. Source: United States Department of Agriculture 2021

methods or results of management studies. Thus, study results and predictions are rarely checked or verified. Rather, institutional experience is derived from successes, and people with institutional experience pass on those lessons inside an organization. Laws, policies, and practices often change for reasons other than management success or failure.

SHARED PERSPECTIVES OF SCIENCE AND PRACTICE

Scientists and practitioners have much common ground as well. Many have similar reasons for directing their careers to include the environment early in their lives. They are often science-oriented, with a technical education focused on a single discipline. They make use of studies to accomplish their work, with a heavy reliance on data and objective analyses. Many have a high regard for precise, quantitative results underpinned with biological knowledge. They often find that written material is the primary measure of productivity and that those writings are almost always subjected to independent reviews. And, most eventually encounter frustrations due to restrictions on topics and priorities often because of funding availability.

CHALLENGES WITH MERGING SCIENCE AND PRACTICE

Ecological conservation is informed by science. Science provides the facts, guidance, and methods to protect and improve our environment. Conservation is the act of putting science into practice. However, science and practice are not necessarily close in perspective, values, methods, and considerations. The relationship between science and practice varies by approach, though a close coordination between science and practice is seen as a key to success in ecological conservation. The differences in thinking between science and practice seem to be a product of the workplace, on-the-job demands, and the local culture (Pouyat 1999). Acknowledging the differences between science and practice across conservation techniques can provide a broader and potentially more effective approach to ecological conservation.

Both scientists and practitioners want to make a better world and solve environmental problems. Both see the need for a merger of science and practice to develop effective decisions, plans, policies, and make progress on ecological challenges. Science offers facts, analytic methods, and understanding of complex ecological systems. Practitioners have the capacity for taking action, identifying feasible options, and making implementation work. Ultimately, ecological conservation should not be treated as a singular perspective. Instead, we should foster the recognition of different strengths and options for making progress by exploring science and practice within many different conservation techniques.

Figure 1.3: Merging science and practice as part of a community conservation program. Source: Warren County Soil and Water Conservation District 2021

Merging science and practice for each ecological conservation technique is challenging because of the inherent differences across applications (Figure 1.3). However, there are expected benefits from integrating science and practice, and many are promoting this as the way forward to improve our environment. In theory, combining these two distinct pursuits of science and practice will promote improvements in ecological conservation, and over time trends will emerge regarding the best way to conserve the environment.

WHAT EACH SIDE NEEDS FROM THE OTHER

Generally, what practitioners need from scientists is a simple, widely applicable technique that will detect the health of the environment. Ideally, this technique would involve steps that are clear and fully described so they can be followed by people with little or no training (Cairns 1985). If a device is involved, it should be inexpensive, readily available, and portable (Cairns 1985). The technique should provide results that can be obtained immediately and are easily understood (Cairns 1985). Results from methods and models should also be reliable, credible, provide guidance on meaning, and take into account key factors, processes, mechanisms, and structural properties that represent the essential characteristics and functions of systems and species. Practitioners often need this information to produce general "rules-of-thumb" and track important variable levels or system conditions that define limits of health (e.g., thresholds or tipping points).

Generally, what scientists need from practitioners is guidance regarding the balance between research that is intensive and extensive. Intensive research experimentally investigates one or a few factors and extrapolates findings to a broader context. This is the process of induction, in which we use specific findings to make general conclusions (e.g., Keeling Curve). Extensive research is broad-based and looks for patterns that suggest processes of importance. This is the process of deduction, in which we use general principles to explain specific observations. Scientists also need guidance regarding the divergence between research aimed at understanding the complexity of nature and research aimed at developing indicators or simple measures of environmental health. Additionally, scientists need guidance on appropriate scale of investigation which can be thought of as the geographic scope of interest (e.g., microhabitat to landscape ecology). Finally, scientists need more guidance on what will be important 5-10 years from now, as well as crises that exist today.

EXAMPLES OF SCIENCE IN USE BY CONSERVATION ORGANIZATIONS

Many examples exist of science successfully integrated into practice at conservation organizations. For example, the Nature Conservancy (2021a; 2021b) says on their website: "Science matters, especially at this critical turning point for nature. Our work is grounded in science." The mission of The Nature Conservancy is to "preserve the plants, animals, and natural communities that represent the diversity of life on earth by protecting the lands and waters they need to survive," and they have currently preserved land on almost all continents around the world.

Another example is Conservation International (2021a; 2021b) whose website says: "Science has always guided our work, and we rely on science and evidence as the foundation of conservation." The mission of Conservation International is "building upon a strong foundation of science, partnership and field demonstration, Conservation International empowers societies to responsibly and sustainably care for nature, our global biodiversity, for the well-being of humanity."

These are just two of many such organizations that use science as the basis for their conservation actions.

EXAMPLES OF SCIENCE IN USE BY GOVERNMENTAL ORGANIZATIONS

There are also many examples of science successfully integrated into practice within governmental organizations (Figure 1.4). For example, the former director of the United States Fish and Wildlife Service (USFWS), H. Dale Hall, was quoted (2008) to have said, “Science underpins everything we do as an agency.”

Figure 1.4: Example of using science in decision-making. Source: United States Fish and Wildlife Service 2021

The National Oceanic and Atmospheric Organization (NOAA) put science directly in its mission statement by proclaiming science as one of its three goals: Science, service and stewardship (2021).

These are just two of many such governmental organizations that use science as the basis for their decision-making and actions.

CASE STUDY: PEACE TREATY ON THE HUDSON RIVER

Figure 1.5: Consolidated Edison Company’s proposed pump storage plant project in the face of Storm King Mountain, Hudson River. Source: Marist College Cannavino Library 2021

In 1963, the Consolidated Edison Company (Con Ed) proposed to embed the world’s largest pump storage plant into the face of Storm King Mountain (Figure 1.5) on the Hudson River (Marist College Cannavino Library 2021). Pumped storage hydropower (PSH) is a type of hydroelectric energy storage system configured with two water reservoirs at different elevations, that can generate power as water moves down from one reservoir to the other (discharge), passing through a turbine (United States Department of Energy 2021). PSH acts similarly to a giant battery because it can store power and then release it when needed (United States Department of Energy 2021). This proposal by Con Ed generated the longest, most expensive, most litigated, and most important environmental controversy in United States history up to that point (Barnthouse et al. 1984). It also spurred the start of the United States environmental movement with the development of many environmental organizations, such as Scenic Hudson (Marist College Cannavino Library 2021), Riverkeeper (2002), Clearwater, Natural Resources Defense Council, the Hudson River Foundation, and the environmental consulting profession (Lubchenco 1998).

Opposition to Con Ed's plan was mounted by several groups. People who valued the aesthetics of the Hudson River, such as the Hudson River School artists (Ferber 2009), formed one group (Figure 1.6). Those opposed to the potential environmental effects formed another group. Those concerned about the impact to striped bass populations formed a third group.

There were already a number of existing power generating facilities on the Hudson River (Figure 1.7).

Figure 1.6: Storm King Mountain and the Hudson River. Source: New York State Office of General Services 2021

Science played an important role in the decision-making process about whether to let Con Ed's plan move forward. The core issue was the effect of electric generation on striped bass populations. There was no doubt that electric generation plants killed striped bass through impingement and entrainment. Impingement is the physical contact of a fish with a barrier structure (e.g., screen) due to intake velocities which are too high to allow the fish to escape (West Virginia Department of Environmental Protection 2021). Entrainment is the unwanted passage of fish through a water intake, generally caused by an absent or inadequate screen surrounding the water intake (West Virginia Department of Environmental Protection 2021). The question under investigation was whether losses through impingement or entrainment would have a significant impact on the striped bass population. Scientists used mathematical population models as environmental assessment tools.

Two models were developed. The first model was developed by a consulting company hired by Con Ed. It compared average daily water use versus daily tidal flow and assumed that fish were uniformly distributed in the water. The results of the model indicated there would likely be negligible impact of the pump storage plant on fish populations. The second model was developed by Oak Ridge National Laboratory. It predicted that small fish would accumulate at the salt front and circulate on an "endless belt" near the power plant. The results of this model predicted that 30-50% of the population would be killed (Barnthouse et al. 1984).

Both models were attacked as being oversimplified and unrealistic (Barnthouse et al. 1984). It was argued by a witness testifying on behalf of Con Ed that, if tidal circulation and cooling water withdrawal were modeled more realistically, impact estimates would be reduced by 75%. So, a more elaborate hydrodynamic model was developed with non-uniform spatial distribution of organisms and 'migration factors' (used to model the movement of juveniles). These migration factors became the next focus of debate.

In its initial licensing decision, the Atomic Safety and Licensing Board (ASLB) decided that the Indian Point plant was the cause of a substantial reduction in the striped bass population, and ordered that cooling towers be built. This decision was appealed by Con Ed, who won using the hydrodynamic model.

New models were developed by both sides Models succeeded models but the results stayed the same. The models developed by the utility companies showed negligible impact, while the models developed by regulatory agencies predicted serious fish population depletion. Neither side could be defended conclusively.

There were two main failings of the models. The first was that hydrodynamics did not actually predict the distribution of fish. The models were forced to match known distributions, but then did not simulate hydrodynamics properly. The second was that population compensation, the idea of allowing entrainment mortality to be offset by a decrease in the natural mortality rate of the non-entrained striped bass, was not well understood. Without accurate estimates of compensation, any of the models could be forced to predict either a large or a small impact with virtually equal plausibility (Swartzman et al. 1977).

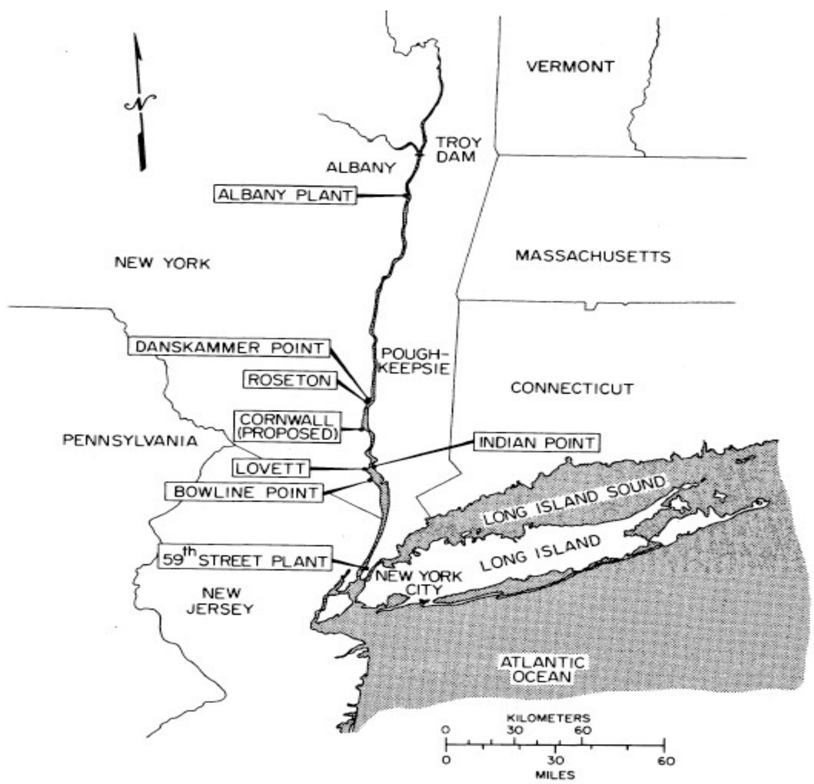

Figure 1.7: Locations of power generating facilities on the Hudson River. Source: Barnthouse et al. 1984

Since the models were not proving helpful, direct impact assessment was tried next. Empirical transport accounting utilized extensive field data and estimated losses by river regions. Estimated reductions in the size of the 1974 and 1975 year classes ranged from 12% to 14% according to utility scientists, and from 12% to 22% according to regulatory agency scientists (Barnthouse et al. 1984). The data also showed that, for most fish, impingement was less of an issue than entrainment. A focus on reductions in abundance of individual year classes eliminated the need to make and defend long-term impact predictions. It also eliminated any remaining hope of resolving the issue of whether cooling towers were needed to ensure the long-term viability of Hudson River fish populations (Barnthouse et al.

1984). Thus, after tens of millions of dollars and a decade of study, it was still not possible to know the effects of the proposed pump storage plant on striped bass populations with confidence. In the end, as part of the “Hudson River Settlement Agreement” in 1980, the utility companies agreed to implement flow reductions and scheduled shutdowns as an alternative to cooling towers to reduce entrainment of fish species. They also funded ongoing environmental monitoring, the creation of the Hudson River Foundation, and agreed to operate a striped bass hatchery on the river. And, the idea of a pump storage plant at Storm King Mountain was abandoned. In return, the environmental organizations and United States Environmental Protection Agency (USEPA) dropped the requirement of closed system cooling, which would have necessitated the building of six costly cooling towers. Additionally, the USEPA would allow the plants to continue operating as they have in the past. This agreement stands as a model for balancing economic and environmental needs.

Lessons learned: 1) Conclusive results were not possible even with a simple case looking at one species (Striped bass) in one river (Hudson River) with known loss mechanisms; 2) In this case, simple empirical analyses were more useful than simulation models; 3) Decisions need to be made despite uncertainties; 4) The ultimate question, "what will be the long-term effects of once-through cooling on Hudson River fish populations?" was unanswerable; 5) An alternative question, "what are the available methods of reducing the impact of once-through cooling, and how can they be most effectively deployed?" enabled scientists to make a positive contribution.

SUMMARY

Science and practice are unique and yet work together toward the goal of improving ecosystem health through environmental conservation. Science provides the knowledge used to create decisions that lead to action. Practice is the act of managing and conserving the environment. These two fields often have different perspectives. When they work together successfully, their collaboration can prove effective at improving ecosystem quality.

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Fundamental Techniques

Chapter 2 - Standards and Criteria

The second chapter in the fundamental techniques group is on standards and criteria. Standards and criteria were first designed to regulate environmental pollution and are an important part of current environmental management plans. Standards are regulations that include designated uses (e.g., water for consumption, swimming) and criteria (e.g., chlorine should not exceed 19 ug/L) that should be applied to protect those uses (United States Environmental Protection Agency 2021). Criteria are used to evaluate or test the quality of something and decide if it passes or if action should be taken. In this chapter, we explain the meanings of standards and criteria, provide lessons on implementation, explore some examples, and end with a case study on New Jersey water quality management.

USE OF STANDARDS

Standards are pervasive in society. They are used by governments to protect the public and the environment. For example, in the United States, food safety standards help to reduce the number of pathogen-related outbreaks. Road safety standards keep us safer while driving. Environmental standards help to protect the environment and are the first and fundamental technique for doing so.

PROPERTIES OF STANDARDS

Standards are meant to be set and then applied repeatedly (Fischhoff 1984). Generally the use of a standard is considered an administrative act unlike decision-making which is a political act (Fischhoff 1984). Standards determine whether some actions, all actions or no actions are acceptable. Assessments based on standards are irreversible unless the standard is changed later (Fischhoff 1984).

Table 2.1: Conditions that favor standard setting. Source: Fischhoff 1984

Feature	Standards	Decisions	Conditions Favoring Standards
Number of options chosen	None at all	One	(1) No choice possible (2) No selection needed
Task facing options	Satisfy standard	Beat opposition	(3) Predictability important (4) Future options need shaping
Range of application	Category	Single case	(5) Competing technologies in same jurisdiction (6) Category members homogeneous
Expression	Rule	Choice	(7) Explicit policy attractive (8) Value issues sensitive
Application	Technical	Political	(9) Political resources limited (10) Process unimportant
Flexibility	Little	Great	(11) Awkward applications avoidable

Conditions that favor standard setting are listed in Table 2.1. In particular, standards are appropriate in the following situations (Fischhoff 1984):

1) When no choice among options is possible.
2) When no choice among options is required.
3) When predictability is important.
4) When regulators hope to shape future options.
5) When competing technologies fall in the same jurisdiction.
6) When category members are homogeneous.
7) When an explicit policy statement is attractive.
8) When value issues are sensitive.
9) When political resources are limited.
10) When process is unimportant.
11) When awkward applications can be avoided.

Deciding to rely on a standard sets into motion many small decisions brought about by translating the standard into operational terms (Fischhoff 1984). There are four generic approaches to setting standards (Table 2.2). The approaches differ in the perspectives that they consider and the methods used for implementing them. The choice among approaches in a particular case is based in part on an empirical question about the potential advantages and disadvantages, as well as a political question about the importance of these advantages and disadvantages (Fischhoff 1984). Hybrid approaches are also possible.

Table 2.2: Methods for setting standards. Source: Fischhoff 1984

Approach	Locus of Wisdom	Description	Potential Advantages	Potential Disadvantages
Formal analysis	Formalized intellectual processes	Choose standard offering highest utility (or best cost-benefit tradeoff)	Systematic explicit sophisticated techniques	Impractical oversold centralizes power
Professional judgment	Intuitive intellectual processes	Let technical experts identify best standard	Realistic implementable creative compromises	Vested interests incomplete perspectives inscrutable
Political processes	Body politic	Have lay groups set standards, informed by technical advice	Broad perspective legitimacy open to criticism	uninformed unrealistic unstable
Revealed preferences	Past social processes	Adopt standard implicitly emerging in actual decisions	Reflects deeds shaped through experience influenced by whole society	Inefficient unfair insensitive to risk

IMPLEMENTING STANDARDS

In the early 1960s, the United States congress passed the first environmental laws utilizing standards (Houck 2003) based on analysis done by scientists. These standards were meant to prevent environmental harm rather than compensate for it and required enforcement to encourage compliance.

This first wave of environmental laws were science-based environmental policy in action. One of the first laws enacted was the Water Quality Act of 1965 which sought to improve conditions based on water quality criteria. It was soon followed by the National Environmental Policy Act of 1969 and the analysis of environmental impact. Then came the Clean Air Act of 1970, which focused on the attainment of national ambient air quality standards (Houck 2003). These Acts were followed by many others, all with the same premise that scientists would draw the lines in preserving and improving environmental health: the Resource Conservation and Recovery Act (waste disposal), the Comprehensive Environmental Response, Compensation, and Liability Act (abandoned waste sites), the Toxic Substances Control Act (chemicals), the Federal Insecticide, Fungicide, and Rodenticide Act (pesticides), and the Safe Drinking Water Act.

A CLOSER LOOK AT THE STANDARDS USED FOR CLEAN WATER

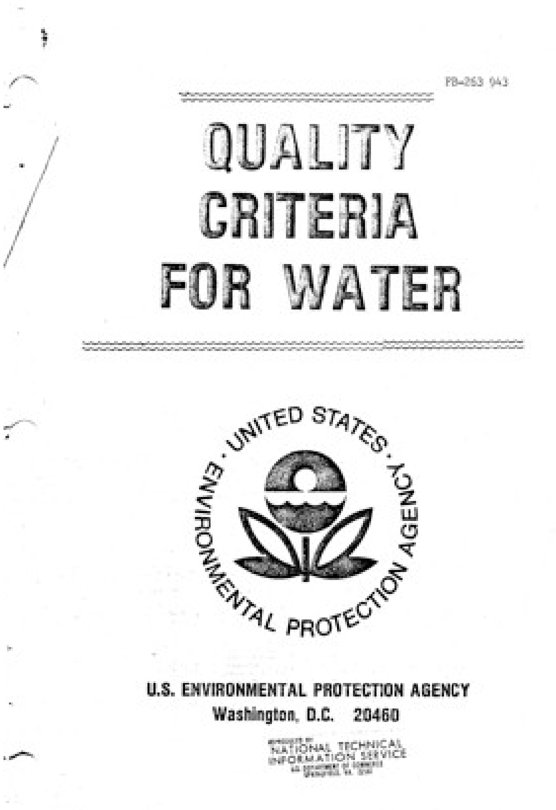
Water quality standards and criteria are the regulatory and scientific foundation for programs established under the Clean Water Act (CWA) to protect the Nation's waters. Early water quality legislation was for the protection of public health. Over time, this purpose was supplemented to include aesthetic and recreational purposes (fishable and swimmable waters) and then with the goal of restoring and maintaining the "chemical, physical, and biological integrity of the Nation's waters" (Hershman and Feldmann 1979) through administration by the United States Environmental Protection Agency (USEPA). The USEPA's strategy is built upon a long-term vision for the future:

> "All waters of the United States will have water quality standards that include the highest attainable uses, combined with water quality criteria that reflect the current and evolving body of scientific information to protect those uses. Further, standards will have well-defined means for implementation through Clean Water Act programs" (Source: United States Environmental Protection Agency 2003).

In practice, each of these purposes must be restated in operational and measurable terms as ambient water quality standards.

The USEPA designates water quality standards and criteria to protect the uses of water and set antidegradation policies (National Research Council 2001). It straddles the dual roles of establishing goals and providing the regulatory basis to enforce strategies. In addition, they provide policy guidance and scientific information to states and tribes; they also review state standards, approve or disapprove them, and can issue federal standards to replace or correct state policy deficiencies where necessary.

Many of the standards issued in the mid-1970s to support the CWA have not changed much since then. For example, the "red book" (Figures 2.1 and 2.2) contains quality criteria for United States waters (United States Environmental Protection Agency 1976). Similarly, the "gold book," issued ten years later, contains additional quality criteria for water (United States Environmental Protection Agency 1986).



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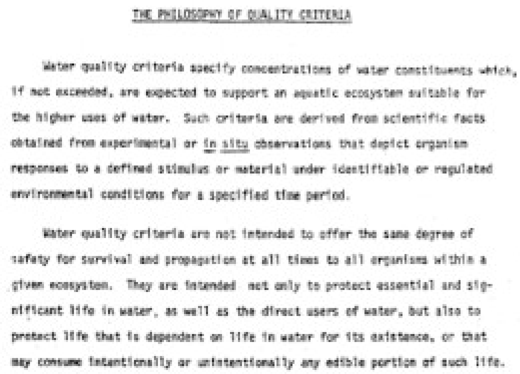
QUALITY CRITERIA FOR WATER

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

U.S. ENVIRONMENTAL PROTECTION AGENCY
Washington, D.C. 20460

NATIONAL TECHNICAL INFORMATION SERVICE

Figure 2.1: Quality criteria for water "red book." Source: United States Environmental Protection Agency 1976



THE PHILOSOPHY OF QUALITY CRITERIA

Water quality criteria specify concentrations of water constituents which, if not exceeded, are expected to support an aquatic ecosystem suitable for the higher uses of water. Such criteria are derived from scientific facts obtained from experimental or in situ observations that depict organism responses to a defined stimulus or material under identifiable or regulated environmental conditions for a specified time period.

Water quality criteria are not intended to offer the same degree of safety for survival and propagation at all times to all organisms within a given ecosystem. They are intended not only to protect essential and significant life in water, as well as the direct users of water, but also to protect life that is dependent on life in water for its existence, or that may consume intentionally or unintentionally any edible portion of such life.

Figure 2.2: Philosophy of quality criteria from the "red book." Source: United States Environmental Protection Agency 1976

Starting in the 1970s, states began developing their own water quality standards and criteria. The intent was to identify specific sources of pollution in violation of these standards. Once these sources were carefully identified, controls on polluting activities were put in place. However, multiple sources of pollutants made it difficult to unambiguously determine which were responsible for violating the standard. Neither the available monitoring data nor the analytical methods in use allowed the states to defensibly mandate differential load-reduction requirements. State level standards and criteria were rarely fine-tuned and proved inadequate when dealing with complex issues like sedimentation, flow, pathogens, feasibility for all sites, or when evaluating the cumulative effects from combinations of pollutants or stressors.

The amendments incorporated into the 1972 CWA recognized this dilemma and shifted the focus of water quality management away from ambient standards. Instead, all emitters of pollutants were expected to limit their discharges by meeting nationally established effluent standards. Effluent standards are specified in National Pollution Discharge Elimination System (NPDES) permits, issued by the states. These standards were set at a national level based on available technologies for wastewater treatment appropriate to different industry groups. The shift to effluent standards eliminated the need to link required reductions at particular sources with the ambient condition of a waterbody.

Instead, each regulated source was simply required to meet the effluent standard for its particular wastewater discharge.

Thus, a shift in thinking occurred: water quality accomplishments could be described in terms of compliance rather than on the condition of the waters themselves. However, it should be noted that effluent standards only applied to point sources of pollution (e.g., pollutants from a pipe or known location). Pollutants from nonpoint sources (agricultural, silvicultural, and construction activities) escaped oversight.

Present-day implementation requires states to identify waters not meeting effluent standards, define the pollutants and the responsible sources, and establish Total Maximum Daily Loads (TMDLs) to achieve these standards (Cooter 2004).

LESSONS ON STANDARDS APPLICATIONS

Unfortunately, the first generation of environmental laws with quality standards didn't work. Administrators began to realize that science is rarely definitive and conclusive. In the world of environmental policy this spells disaster. The reason is political: environmental policy faces a degree of resistance unique in public law. Few who have to comply with environmental law like it and many detest it outright. The reasons for this are many. Environmental laws are intrusive, involve people, state bureaucrats, the general public, the media, and environmental policies are often seen as threatening personal choice.

Resistance to environmental policy brings at least two consequences. The first is: that which is not nailed down by law is not likely to happen. The second is: even for requirements that are nailed down, compliance rates are about 50% percent (Houck 2003). A good rule of thumb is that no environmental law, no matter how stringently written, achieves more than half of what it set out to do (Houck 2003).

Second generation laws worked differently. Congress changed the rules of the clean water game and adopted a new standard: best available technology (BAT). The theory of BAT was very simple: If emissions could be reduced, just do it. It did not matter what the impacts were. It did not matter where a plant was discharging. It didn't matter what scientists said the harm was or where it came from. The theory was, just reduce it. Within 5 years, industrial discharges of conventional pollutants were down by 80% (Houck 2003). Receiving water quality improved by an average of 35% across the board (Houck 2003). For all BAT-controlled sources, the amendments were a stunning success.

Some lessons came out of this process. First, beware the lure of "scientific management." The technology standards initially implemented were criticized as too arbitrary, one size fits all, inflexible, treatment for treatment's sake, and outmoded. They often spurred iterative, impact-based, localized management techniques focused on the scientifically determined needs of a river, airshed, or community. Though good in theory, decades of implementation failed. The biggest losers under the federal air and water quality acts were the science-based TMDL and state implementation plan programs. These were very costly and featured shameless manipulation of the data, crippling political pressure, and little abatement of the water quality issues.

The second lesson involves another caution: beware the lure of "good science." The theory of good science goes like this: good science is the science that supports your case. All other science is bad. If opposed to something, science is never good enough. Such a perspective can give rise to many tactics for delay. In the name of good science, peer review of all science-based decisions may be requested. Decisions can be stalled by lack of consensus among independent reviewers. More studies may be commissioned. Years will pass. Administrations will change. Nothing will get done and the opponents win.

CASE STUDY: NEW JERSEY WATER QUALITY MANAGEMENT

Figure 2.3: Goals of the New Jersey Water Quality management program. Source: New Jersey Department of Environmental Protection 2021a

Let's take a look at New Jersey State water regulations. New Jersey's waters are overseen by the New Jersey Department of Environmental Protection (NJDEP), Division of Water Monitoring and Standards (Figure 2.3) (NJDEP 2021a).

New Jersey's Surface Water Quality Standards were developed and are administered in conformance with requirements of the CWA, the Federal regulatory program established by the USEPA, and the New Jersey Water Quality Planning Act (NJDEP 2021a). The State uses Surface Water Quality Standards (SWQS) to assure that both current decision-making and future planning adequately take into account protection of water quality and quantity. The SWQS include the policies, surface water classifications, and surface water quality criteria necessary to protect the quality of New Jersey's surface waters (NJDEP 2021a).

The SWQS protect the health of New Jersey waters by establishing designated uses, classifying streams based on uses, designating antidegradation categories, and developing water quality criteria to protect the streams and their uses (NJDEP 2021a). In addition, the standards specify general, technical, and interstate policies, as well as policies pertaining to the establishment of water quality-based effluent limitations (NJDEP 2021a).

The SWQS ensure that New Jersey waters are suitable for all existing and designated uses, including drinking water supply, fish consumption, recreation, flood protection, shellfish resources, propagation of fish and wildlife, agricultural, and industrial water supplies (NJDEP 2021a). The SWQS also protect the health of New Jersey citizens and visitors by ensuring that the drinking waters are suitable for consumption, that the bathing waters are safe for swimming, and that the fish and shellfish harvested from our waters are safe to eat. The SWQS also protect waters for other uses such as trout production and maintenance, agricultural and industrial use (NJDEP 2021a).

Surface waters are categorized into stream classifications based on designated uses. New Jersey has both fresh and saline waters. Freshwaters are classified as FW1 (not subject to any man-made wastewater discharges) and FW2 waters (all other freshwaters). Freshwaters are further classified based on trout status; trout production (FW2-TP), trout maintenance (FW2-TM), and non-trout (FW2-NT). Waters within Pinelands Protection and Preservation areas are classified as pinelands waters (PL). Saline waters are classified as saline estuarine (SE) and saline coastal (SC). SE waters are further classified into SE1, SE2, and SE3 based on the designated uses (NJDEP 2021a).

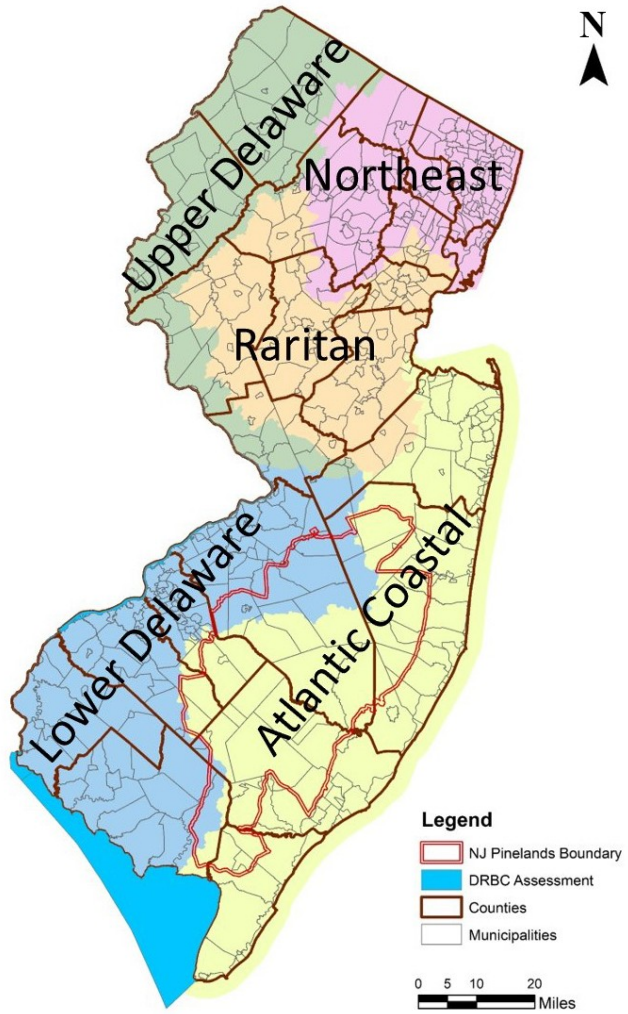


Figure 2.4: Regions of New Jersey. Source: New Jersey Department of Environmental Protection 2020

There are three levels of antidegradation designations: Outstanding National Resource Waters (ONRW), Category One (C1) waters, and Category Two (C2) waters. All waters of the State are classified and assigned one of the three antidegradation designations. Each stream in New Jersey is designated with a classification and an antidegradation designation (NJDEP 2021a).

Finally, Water Quality Criteria were developed for individual pollutants to protect aquatic life (plants and animals that live and reproduce in water) and human health in both fresh and saline waters. Criteria were developed to protect water quality for designated uses including the survival, growth, and reproduction of aquatic life, and drinking water and fish consumption for human health protection. Different criteria may be applicable to different stream classifications. For example, the criterion for dissolved oxygen is different for trout production, trout maintenance, non-trout, SE, and SC waters (NJDEP 2021a).

New Jersey also implements several water quality improvements or restrictions. One method is through the New Jersey Pollutant Discharge Elimination System (NJPDES) which issues permits based on a calculation of water quality based effluent limitations for point source discharges. The calculation is based on the size of the receiving stream, the volume of wastewater, current levels of pollutants in the receiving stream, and effluent characteristics (NJDEP 2021a). Another method is through Flood

Hazard Area Control Act Rules in which a 300 foot riparian zone is imposed on flood hazard areas with certain designations.

How is New Jersey doing with this system? New Jersey is the fifth smallest, yet most densely populated state in the Nation but is also one of the most geologically and hydrogeologically diverse states (NJDEP 2020). The surface waterbody types in New Jersey range from intermittent streams to large potentially-tidal river systems, lakes, ponds, and reservoirs, wetlands (freshwater and saltwater), estuarine and coastal (ocean) waters (NJDEP 2020) and abundant groundwater resources (Table 2.3).

Table 2.3: Areal extent of each resource category. Source: New Jersey Department of Environmental Protection 2020 ([1] state population values from 2019 census)

RESOURCES	EXTENT
State Population (2019) [1]	8,882,190
State Total Area (square miles)	8,772
State Total Land Area (square miles)	7,254
Rivers and Streams:	
Miles of Nontidal Rivers and Streams	13,695
Miles of Tidal Rivers and Streams	5,730
Miles of Rivers and Streams (total)	19,425
Border Miles Shared Rivers	197
Lakes, Ponds and Reservoirs:	
Total Acres of Lakes and Ponds and Reservoirs	47,620
Number of Reservoirs	43
Acres of Reservoirs	14,970
Estuaries and Ocean:	
Square Miles of Estuaries	650
Miles of Ocean Coast (linear miles)	127
Square Miles of Ocean (jurisdictional waters)	470
Wetlands:	
Acres of Freshwater Wetlands	739,160
Acres of Tidal Wetlands	209,269
Total Acres of Wetlands	948,429

The majority of fully supporting assessment units (green) were found in the less densely populated areas of the state in the Northwest Highlands (Upper Delaware Region) and the Southern Pinelands (Atlantic Coastal Region) (Figures 2.4 and 2.5). These areas were characterized as having large intact forested and wetland areas, intact riparian buffers, and limited dense urban development (NJDEP 2021b). Biological impairments were identified as the most frequent reason waterbodies were not able to support aquatic life (Figure 2.6) (NJDEP 2021b). Additional reasons for water quality impairment included high nutrients, total phosphorus, and impairments associated with nutrient over-enrichment (NJDEP 2021b). The NJDEP is developing strategies that will address these impairments utilizing the most effective restoration actions.

Figure 2.5: Geographic distribution of aquatic life support (brown = does not support aquatic life; green = fully supports aquatic life; blue = insufficient data). Source: New Jersey Department of Environmental Protection 2021b

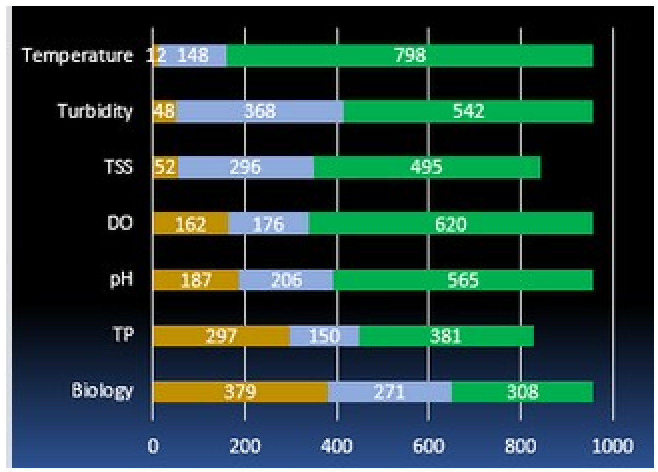


Figure 2.6: Aquatic life major parameter results. Source: New Jersey Department of Environmental Protection 2021b

SUMMARY

Standards were established by the government as rules used to protect the public and the environment. Criteria are used as the principles for evaluating or testing whether something meets a standard. Standards and criteria have successfully been used to protect against high levels of pollution, waste disposal, toxic chemicals, pesticides and more. Standards and criteria have been in use for many decades and have served to provide us with cleaner air, water and land.

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Chapter 3 - National Environmental Policy Act

The third and final chapter in the fundamental techniques group is on the National Environmental Policy Act of 1969 (NEPA). NEPA was the first major environmental law enacted in the United States and is considered the "Magna Carta" of environmental laws (United States Department of Energy 2021). In this chapter, we will provide some background on NEPA, review the process for implementing it, explore how it has performed, and end with a case study on a standoff between groups who wanted a highway vs. groups who wanted to save wetlands in West Eugene, Oregon.

BACKGROUND ON NEPA

NEPA was designed to create a national policy which encourages productive and enjoyable harmony between humans and their environment; to promote efforts to prevent or eliminate damage to the environment and biosphere, and thus stimulate the health and welfare of humans; to enrich the understanding of ecological systems and natural resources important to the Nation; and to establish a Council on Environmental Quality (CEQ) to ensure that Federal agencies meet their obligations under NEPA (Council on Environmental Quality 2007; United States Department of Energy 2021). In short, NEPA was established to make agencies think about the environmental effects that their proposed actions will have prior to making decisions (Baldwin 2012).

Figure 3.1: Overview of the NEPA process. Source: United States Department of the Interior 2021

Congress recognized the fundamental impact of human activity on the interrelations of all components of the natural environment, particularly the profound influences of population growth, high-density urbanization, industrial expansion, resource exploitation, and new and expanding technological advances. Congress further recognized the critical importance of restoring and maintaining environmental quality for the overall welfare and development of human populations. Thus, congress declared that it is the continuing policy of the Federal Government, in cooperation with State and local governments and other concerned public and private organizations, to use all practicable means and measures, including financial and technical assistance, in a manner calculated to foster and promote the general welfare, to create and maintain conditions under which humans and nature can exist in productive harmony, and fulfill the social, economic, and other requirements of present and future generations of Americans (United States Department of Energy 2021).

Federal agencies must comply with NEPA before undertaking any federal actions that could impact the environment. NEPA applies to a very wide range of federal actions, including federal construction projects, plans to manage and develop federally owned lands, and federal approvals of non-federal activities such as grants, licenses, and permits. The Federal Government takes hundreds of actions every day that are, in some way, covered by NEPA.

Private individuals or companies become involved in the NEPA process when they need a permit issued by a Federal agency. The agency that is being asked to issue the permit must evaluate the environmental effects of the permit decision, which triggers the NEPA process. The primary responsibility for NEPA is vested in the CEQ which sits within the Executive Office of the President.

NEPA has two main purposes: 1) To inform decisions and 2) To enable citizen involvement. The point of NEPA is to identify the environmental and social costs and impacts of a project, solicit public comments, and factor this information into a decision (called a Record of Decision (ROD)). NEPA also requires a discussion of alternatives and why an alternative was dismissed, not preferred, or not feasible. Sometimes there really is no alternative. At a minimum, a discussion of the "No Action Alternative" (doing nothing and keeping the status quo) is required.

THE NEPA PROCESS

The NEPA process can best be described in general terms by Figure 3.1 and in detail in the flowchart in Figure 3.2 (New Mexico State University 2021). There are sets of processes and yes/no questions that direct users from identification of a need for action to a decision.

Requirements of NEPA are generally met through the production of an environmental document that analyzes the proposed action (National Oceanic and Atmospheric Administration Program Planning and Integration 2005; Baldwin 2012). There are three levels of NEPA analysis and documentation: Categorical Exclusions (CATEX), Environmental Assessments (EA) for smaller projects, and Environmental Impact Statements (EIS) for larger projects.

A CATEX is a category of actions that the agency has determined does not individually or cumulatively significantly affect the quality of the human environment. These are proposed actions or projects that are too small or insignificant to warrant the preparation of an EA or EIS. Examples include issuing administrative personnel procedures, making minor facility renovations (e.g., installing energy efficient lighting), and reconstruction of hiking trails on public lands (Baldwin 2012). Agencies develop a list of CATEXs specific to their operations when they create or revise their NEPA implementation procedures in accordance with CEQ's NEPA regulations (Baldwin 2012).

An EA is used to determine the significance of the environmental effects of an action or project, and to look at alternative means to achieve the agency's objectives. An EA is intended to be a concise document that 1) Briefly provides sufficient evidence and analysis for determining whether to prepare an EIS; 2) Aids an agency's compliance with NEPA when no environmental impact statement is necessary; and 3) Facilitates preparation of an Environmental Impact Statement when one is necessary.

An EIS must be prepared by a Federal agency if it is proposing a major federal action significantly affecting the quality of the human environment. The regulatory requirements for an EIS are more detailed than the requirements for an EA or a CATEX. An EIS must include (beyond the cover sheet, summary, table of contents, list of preparers, distribution list, index, and appendices if applicable) a statement of the purpose and need, description of the proposed action and alternatives, affected environment, environmental consequences, and mitigation measures (if applicable).

Typical EISs will include sections assessing impacts for the following, although in many cases a project will not impact all of these things:

- Air quality
- Climate
- Soils
- Geology
- Surface water
- Wetlands/floodplains
- Groundwater
- Biological resources
- Cultural resources
- Land use
- Aesthetics
- Solid & hazardous waste
- Safety & health
- Transportation
- Noise
- Utilities
- Community services
- Socioeconomic
- Environmental justice (EJ)

NEPA IMPLEMENTATION ISSUES

NEPA has worked well in some instances but not in others. Poorly planned projects have been justifiably killed as a result of the NEPA process, when previously unforeseen impacts, costs, and inefficiencies were brought to light during the preparation of an EIS. Also, quite commonly, the NEPA process has resulted in mitigation measures being required to offset the impacts identified as a condition of the project moving forward. This requires diligence on the side of fed-

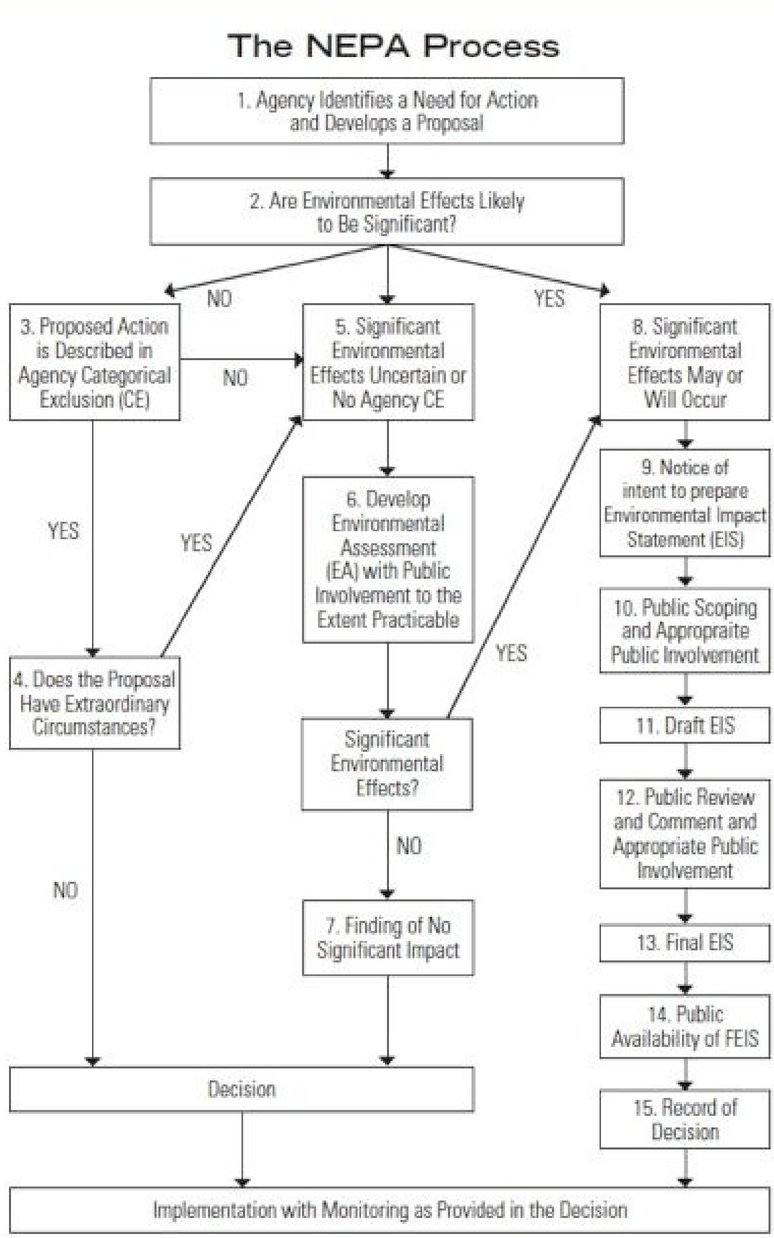

Figure 3.2: Detailed view of the NEPA process. Source: New Mexico State University 2021

eral regulatory agencies to ensure that these measures were actually followed. NEPA has proven effective in the aforementioned instances. Conversely, empty promises are sometimes made in a NEPA document (e.g., the contractor states that they will use top-of-the-line Best Management Practices, but fail to do so) in order to sell a project to the public, but the public doesn't always know what will actually happen in reality. In addition, once the NEPA process has been completed, a project may change in ways that the public might not be aware of and yet the process may or may not be revisited.

The EIS process requires an analysis of alternatives. The process of reviewing alternatives should be rigorous and objective but in reality selection is often subjective and arbitrary. Alternatives often reflect narrow project objectives, agency agendas and biases, and timing too early for public input (Steinemann 2001). More environmentally sound alternatives can be overlooked or eliminated before the formal analyses in EIA. Plus, earlier decisions that guided development of the project idea may not have been subject to EIA. Consequently, inadequate alternatives can undermine the goals of EIA which is to encourage more environmentally sound and publicly acceptable actions (Steinemann 2001).

The most fundamental issue facing those trying to comply with NEPA rests is their attempt to answer the question "What is a significant effect?" The CEQ -requires consideration of (a) Context – the action must be analyzed in several contexts such as society as a whole (e.g., human, national), the affected region, the affected interests, and the locality; and (b) Intensity – the severity of impact. Haug et al. (1984) stated that an environmental issue is significant if there is a high probability that one or more impacts connected with the issue will exceed a threshold of the top priorities which are viewed as:

1) Legal threshold: limits and effects regulated by law;
2) Functional thresholds: impacts that disrupt ecosystem function in an irreversible and irretrievable way;
3) Normative thresholds: impacts on resources at a level of concern relevant to social norms.

Each federal agency has their own idea of what NEPA should dictate, and it is not always consistent from one agency to another or even within the same agency. For example, some federal agencies prepare elaborate EAs for putting solar panels on top of a building, while another agency may CATEX the new construction of an entire building altogether. Most decision-makers at federal facilities are not very educated on NEPA and thus don't always know when to use the process or how it works.

Sometimes agencies enter the NEPA process solely for the appearance of being "green," or simply out of fear of how they may be perceived if they don't use it, instead of carefully considering the costs and whether NEPA is truly warranted. To spend thousands of dollars on an EA to conduct a relatively insignificant project can be considered a waste of tax-payer money if the sole reason is just to maintain appearances.

Sometimes the NEPA process appears to be simply an act of "going through the motions," where a project has basically been pre-determined to go through, regardless of the outcome of NEPA. While many small EAs may receive zero comments from the public, some of the larger more controversial projects requiring an EIS (such as a large coal-fired power plant) may literally receive 25,000 or more comments (mostly from environmental groups like The Sierra Club) and contain a whole host of impacts (acres of lost wetland, degraded threatened or endangered habitat, traffic issues, etc.) but will still end up as a Finding of No Significant Impact (FONSI) and receive approval. In many cases, these may be projects that are crucial for national security or energy. In cases where there may be significant environmental impact, the EIS may require mitigation measures to be included in the project such as wetland restoration, new roads, or sound barriers, which is what NEPA was designed to do. At the same time, many months and funds may be spent sorting through public comments which may ultimately have little to no bearing on the project. These comments will get summarized in a report, but will otherwise be dismissed and the project will move forward regardless.

PERFORMANCE OF NEPA

How much is NEPA used? Between 1973 and 2012, EISs dropped from 2,036 per year (in 1973) to 397 (in 2012) (United States Department of Energy 2018). A total of 2,656 EISs were submitted for the whole period from 2013-2020 (United States Environmental Protection Agency 2021).

Evaluating the effectiveness of NEPA is difficult since it depends on how much of a difference it is making (Jay et al. 2007). There are many challenges that affect the judgment of NEPA's performance. First, NEPA is not an environment management tool since it includes no environmental standards or criteria and has no measurable endpoints. Second, it is not possible to compare environmental conditions with and without a NEPA assessment. That would involve a hypothetical comparison of outcomes that is virtually impossible to measure. Third, as alluded to earlier, it is difficult to ascertain the extent to which decision-makers act in accordance with the environmental information provided. It is very challenging to evaluate whether environmental considerations have been taken into account the decision-making process. There is no requirement that decision-makers give any specific weight to the environmental information provided. Finally, it is hard to quantify the contribution to project design and the fine tuning of developments that may result from stakeholder involvement.

Thus, the possible benefits of NEPA are not easily quantified. However, NEPA can enhance environmental awareness and learning among participants, can bring about change in the values and priorities in planning decisions, and can make a difference through design modifications, institutional learning, and stakeholder involvement. Decisions can be improved through project modifications and mitigation measures, and environmentally damaging proposals that might previously have been approved can be denied.


Figure 3.3: West Eugene Wetlands. Source: Bureau of Land Management 2021

CASE STUDY: A HIGHWAY, A WETLAND AND A DIVIDED COMMUNITY

For twenty years officials in West Eugene, Oregon were locked in a standoff. A transportation agency wanted to build a highway with federal funds through a substantial wetland area to relieve traffic on major surface streets in and out of West Eugene. Meanwhile, a land management agency wanted to establish, expand, and protect the West Eugene Wetlands (Figure 3.3). Each objective was pursued by a different agency with different sources of funding and with non-overlapping planning processes (Environmental Law Institute 2010). NEPA provided a way to resolve this mismatch of agendas.

The NEPA process started in 1985 when the Oregon Department of Transportation (ODOT) and Federal Highway Administration published a draft Environmental Impact Statement (EIS). In 1997, a supplemental draft EIS was published, recommending construction of the West Eugene Parkway (WEP), a four-lane bypass that would cross through a significant wetland area (Environmental Law Institute 2010). Despite the passage of time and intervening recognition of the value of wetlands, the statement of purpose in the EISs was narrowly drawn and did not consider a non-wetlands-crossing alternative to improve transportation in and out of West Eugene (Environmental Law Institute 2010).

In an attempt to respond to both highway and wetlands project advocates, the local county and city governments prepared a transportation plan in 2001 that included one of the four segments of the full WEP, a portion that did not cross wetlands, as a priority transportation project (Environmental Law Institute 2010). Later that year, WEP advocates on the West Eugene City Council initiated a ballot referendum on whether the full highway with all four segments, including the portion crossing the wetlands, should be built. It passed by a vote of 51% to 49% (Environmental Law Institute 2010). This empowered the ODOT to recommend that all four segments of the WEP be built, and the proposal was approved by votes in four local jurisdictions (Environmental Law Institute 2010).

In 2004, however, several events occurred that changed the tone of this ongoing debate. That year, both the United States Army Corps of Engineers and the Bureau of Land Management indicated they had not been given adequate information during the NEPA analysis. Before they would issue permits under the Clean Water Act to fill wetlands and construct a highway across the federally-protected West Eugene Wetlands, both agencies determined that they needed to analyze information on potential non-wetland-crossing alternatives (Environmental Law Institute 2010). That same year, the residents of West Eugene elected a mayor who had campaigned in part on opposition to the WEP citing that alternatives had not been considered during the NEPA analysis (Environmental Law Institute 2010).

By early 2007, pro-highway business people and pro-wetlands community members began jointly discussing options for transportation in and through West Eugene. These discussions led to the formation of the professionally-facilitated West Eugene Collaborative, which included equal numbers of business, neighborhood, environmental, and government representatives (Environmental Law Institute 2010). The Collaborative's purpose during meetings (Figure 3.4) over the next two years explicitly involved consideration of alternatives to the WEP and encouraged development of an integrated transportation and land use solution that would be broadly supported by stakeholders (Environmental Law Institute 2010). The Collaborative strived for a solution that would receive broad community support; be economically feasible; facilitate both the movement of people and commerce through the region; minimize greenhouse gas emissions; avoid wetlands loss; support sus-

Figure 3.4: West Eugene Collaborative meeting. Source: West Eugene Collaborative 2021

tainable business; and enhance both the community and the environment (Environmental Law Institute 2010).

Later in 2007, the Federal Highway Administration issued a "no-build" final EIS decision which disallowed the WEP from being built through the wetlands. After two years of meetings and hundreds of hours of volunteer time, in March 2009, the West Eugene Collaborative published its final report titled "A New Vision for West Eugene" which included a set of recommendations for short, medium, and long-term actions that would simultaneously address environmental, transportation, and community concerns and needs (Environmental Law Institute 2010). The final report was welcomed by the community, and the West Eugene City Council voted unanimously to convene work sessions to discuss next steps for implementing The Collaborative's recommendations (West Eugene Collaborative 2021). One West Eugene Collaborative participant stated: "This is a vision of a place which will not only be nicer to live in, healthier, safer, and more pedestrian friendly, but a place where compact urban development can occur" (West Eugene Collaborative 2021).

NEPA enabled the community of West Eugene, Oregon to organize and collaboratively pursue NEPA's goals: a public process with clear needs and a positive purpose; consideration of the social and environmental impacts of a range of alternatives to address the stated needs and purpose; and an informed community and decision-makers (Environmental Law Institute 2010). Further NEPA analyses may be required in the process of implementing future recommendations of the West Eugene Collaborative (Environmental Law Institute 2010).

SUMMARY

NEPA was established to encourage agencies to think about the environmental effects that their proposed actions might have prior to making final decisions. Submissions to NEPA have declined in recent decades. The possible benefits of NEPA are not easily measured. However, NEPA can act to increase environmental awareness and learning among participants, can bring about change in the values and priorities in planning decisions, and can make a difference through design modifications and stakeholder involvement. Decisions can be improved through modifications and mitigation, and environmentally damaging proposals can be denied that might previously have been approved.

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Biologically-Focused Techniques

Chapter 4 - Rewilding

The first chapter in the biologically-focused techniques group centers on the idea of rewilding. Rewilding has at its core the goal of reestablishing species and interactions in an effort to restore natural ecosystems. This chapter will cover some background on rewilding, the theoretical basis for its use, examples of implementation, and will end with a case study on gray wolf (*Canis lupus*) reintroductions in Yellowstone National Park.

BACKGROUND ON REWILDING

The concept of rewilding is grounded in the notion that to have truly natural ecosystems, ecological processes must be reestablished, allowing nature to create or rebuild these ecosystems. Dynamic interactions among species are regarded as the fundamental bases of natural processes that support natural ecosystems. Large carnivores and herbivores are seen as necessary to shape the flora and physical characteristics of natural environments (Asner et al. 2009). Many large species of carnivores and herbivores were extirpated by humans beginning 150,000 years ago, resulting in altered ecosystem functions and attributes (Martin 2005). Estimating an ecosystem's original functions and processes becomes a near impossible task due to the impact of thousands of years of human activity. Intensive management of parks and landscapes are unlikely to yield a fully natural setting as ecosystems have undergone extensive, long-term alterations (Sutherland 2002).

Rewilding sets out to recreate a full assemblage of interacting species within a specific ecosystem. In essence, the rewilding concept aims to assemble the "biological pieces" and then let nature rebuild populations to reestablish the interrelationships which support and maintain the assemblage and its ecosystem. Top predators constrain herbivore diversity and abundance. The composition of herbivores shape plant cover and the physical attributes of their habitats. Competition among both flora and fauna also determines community composition and its effect on the broader landscape. A core principle is that biotic processes will result in an ecosystem that is more natural than we can predict and create ourselves. Consequently, the rewilding concept may seem unrealistic and risky to implement. However, rewilding efforts have been implemented and some have demonstrated very profound results in reestablishing biotic and abiotic ecosystem characteristics. Some of the principles of ecology justify this approach and some applications of this concept have demonstrated success for conservation and restoration purposes.

A basic and distinctive attribute of the rewilding concept is reestablishing large predators and herbivores that were targeted by early people. Restoring large species has primarily been done by relocating species to a new setting where a full community is to be built. We have several examples of successfully rebuilt natural communities (Marris 2009; Ismail 2011; Taylor et al. 2019; Pettersson and de Carvalho 2021; Segar et al. 2021) including: wolves (*Canis lupus*) brought back to Yellowstone National Park (USA); houbara bustards (*Chlamydotis undulata*), sand gazelle (*Gazella subgutturosa marica*) and Arabian oryx (*Oryx leucoryx*) reintroduced to Mahazat as-Sayd reserve (Saudi Arabia); giant anteaters (*Myrmecophaga tridactyla*), tapir (*Tapirus terrestris*), collared peccaries (*Pecari tajacu*), and pampas deer (*Ozotoceros bezoarticus*) reintroduced to the Iberian wetlands (Argentina); and little-spotted kiwi (*Apteryx owenii*) and tuatara (*Sphenodon punctatus*) released in Zealandia (New Zealand). However, many large predators and herbivores are extinct and were eradicated by humans tens of thousands of years ago. Where an exact species match is not possible, rewilding promotes the use of analog species to complete a full community and reestablish ecological processes. Introducing non-native species that have strong effects on biota, and potentially the physical environment, may seem contrary to conservation practices. However, it is consistent with the core idea of rewilding that a natural environment is the product of ecological processes. This approach has been implemented with Heck cattle (*Bos taurus primigenius*) and Konik horses (*Equus ferus caballus*) to replace extinct aurochs (*Bos taurus*) and tarpans (*Equus ferus*) in Oostvaardersplassen reserve (Netherlands; Vera 2009), tundra musk oxen (*Ovibos moschatus*) have been used to replace the extinct Siberian musk oxen (*Ovibos palantis*) in the Siberian steppe (Russia; Parker et al. 2010), and Aldabran giant tortoises (*Aldabrachelys gigantea*) have replaced the extinct giant Cylindraspis tortoises (*Cylindraspis sp.*) in the Mascarene Islands (Mauritius; Griffiths et al. 2010).

One privately owned estate (owned by English multimillionaire conservationist Paul Lister), called Alladale Wilderness Reserve in the Scottish Highlands, is being managed to restore it to its natural state by increasing the tree cover of the valley, reintroducing native animals which are no longer found in Great Britain, restoring damaged peatlands, encouraging biodiversity, and promoting education through ecotourism (Alladale Wilderness Reserve 2021). Alladale Wilderness Reserve is currently the United Kingdom's largest wilderness rewilding area. It hosts horses (*Equus sp.*) as a proxy for tarpan (*Equus ferus*), bovine (*Tragelaphus sp.*) as a proxy for aurochs (*Bos taurus*), European bison (*Bison bonasus*), European elk (*Alces alces alces*), wild cats (*Felis silvestris*), red squirrels (*Sciurus vulgaris*), and has plans to reintroduce wild boar (*Sus scrofa*), bear (*Ursus sp.*), and lynx (*Lynx sp.*) (Sandom et al. 2019).

Starting in 2005, Donlan et al. (2005; 2006) focused worldwide attention on the rewilding approach by proposing a large reserve in the central United States with African lions (*Panthera leo*), African elephants (*Loxodonta africana*), and other non-native species as proxies for long lost native species. The purpose of these species' introductions was to re-create the extinct megafauna assemblages that were eradicated by the first Americans 13,000 years ago. They termed this strategy Pleistocene rewilding, and made the case that it is the only way to restore a fully natural grassland landscape (Figure 4.1). Pleistocene history and taxon substitutions can provide benchmarks for restoration not only by the presence or absence of species but by the presence or absence of species interactions, which constitutes the true functional fabric of nature. The time period of 13,000 years ago corresponds roughly with the arrival of the first Americans from Eurasia and constitutes a less arbitrary baseline than the Pre-Columbian/European standard often used. For 200 million years, large carnivores and megaherbivores were dominant features of most ecosystems. With a few exceptions, primarily in Africa, these animals became functionally extinct worldwide by the late Pleistocene. Vegetation communities have shifted and changed before and after the late Pleistocene, but the major missing component in contemporary ecosystems is large vertebrate herbivores. The Great Plains of the United States was suggested as a prime location since human populations in the region are declining and this area could support a vast, privately-owned and managed "ecological history park." The park would provide animals currently in captivity with an alternative: large natural settings, vast protected areas to roam, and live vegetation/prey. It was a bold proposal for conservation, one more optimistic than many of the proposals put forth by modern day environmentalists, and it started a debate about the rewilding concept (Rubenstein et al. 2006).

Critics of rewilding pointed out that the results of Pleistocene rewilding in North America are unknown and might well be catastrophic. Ecosystem functioning could be disrupted, native flora and fauna, including species of conservation value, could be negatively impacted, and a host of other unanticipated ecological problems could arise (Rubenstein et al. 2006). Further, how would conservationists and managers determine success, particularly if what resulted from the rewilding efforts was a novel ecosystem instead of a return to a historic ecosystem? It is a little like proposing that two wrongs will somehow make a right: both the modern-day proxy species are "wrong" (i.e., genetically different from the species that occurred in North America during the Pleistocene), and the ecosystems into which they are to be reintroduced are "wrong" (i.e., different in composition from the Pleistocene ecosystems, as well as from those in which the modern-day proxy species evolved). Pleistocene rewilding of North America will not restore the evolutionary potential of North America's extinct megafauna because the species in question are evolutionarily distinct; nor will it restore ecological potential of North America's modern ecosystems because they have continued to evolve over the past 13,000 years. In addition, there is a third and potentially greater "wrong" involved: adding these exotic species to current ecological communities could potentially devastate populations of indigenous, native animals and plant communities (Rubenstein et al. 2006).

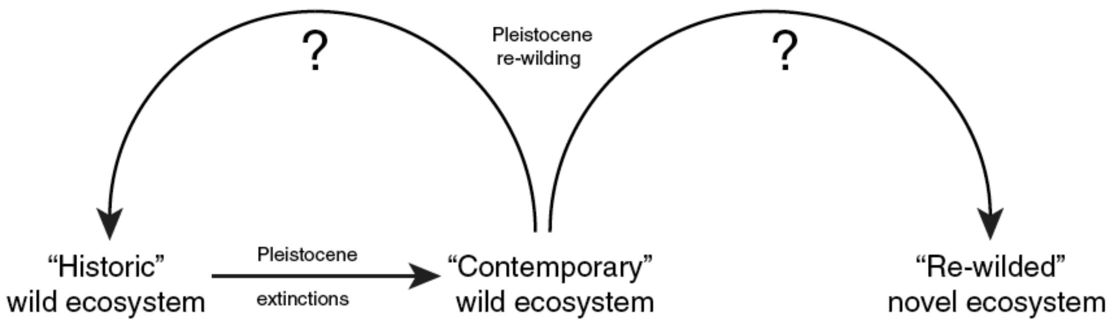

Figure 4.1: Rewilders believe that Pleistocene rewilding will lead to a restoration of the historic wild ecosystem and original ecosystem functioning. However, rewilding could result in a rewilded novel ecosystem with unique species compositions and new ecosystem functioning. Source: Rubenstein et al. 2006

Another approach that has received considerable attention is de-extinction (Novak 2018). De-extinction is an amalgum of genetic engineering, stem cell research and conservation. Essentially, de-extinction is the restoration or revival of a species by manipulation of their genetic material from an artifact source. The rationale and support for de-extinction parallel the same premises as Pleistocene rewilding: an attempt to restore ecosystems and habitats to a more natural prior state, increase biodiversity, and from a moral and ethical standpoint, undo the damage done by humans that resulted in the extinction of the species. De-extinction reinvigorates the same issues as rewilding in regard to the unintended and detrimental effects that may result from the introduction of a species that is no longer part of the current ecosystem.

A related issue and concern is: what is natural? Considering the variability and evolutionary changes that define nature, what lapse in time between between the absence of a species and reintroduction to an ecosystem is realistic? Opponents to de-extinction cite potential issues with reintroduced species exerting undue pressure on the environment and other species through competition for resources. Further, reintroduced species may serve as vectors for disease and parasites. Advocates of de-extinction support a continued focus on identifying potential studies and mitigation protocols to minimize and eliminate these risks. To date, proponents of de-extinction claim the successful creation of a frog embryo (*Rheobatrachus silus*) that has been extinct for 30 years and the genetic material of Pyrenean Ibex Capra (*pyrenaica pyrenaica*) has also been resurrected for a few minutes (Banks and Hochuli 2017). Similar attempts are underway to apply genetic technology to resurrect some of the most iconic extinct species that have been lost in recent times, including passenger pigeons (*Ectopistes migratorius*), thylacines (*Thylacinus cynocephalus*) and mammoths (*Mammuthus sp.*; Banks and Hochuli 2017). The discourse and debate on the pros, cons and feasibility of de-extinction continue.

THEORETICAL CONCEPTS BEHIND REWILDING

There is a set of principles of ecological science that support the rewilding approach. Top-down effects of predators on herbivores and beyond can have a strong influence all the way down to the vegetation cover within a landscape. The interactions of species in food chains and food webs can shape the biota, and different biota can shape a variety of ecosystem features including physical properties. Highly influential species can alter entire communities and have widespread effects on the nature of ecosystems Estes et al. 1998). Finally, the sequential effects across trophic levels can shape the biota and cascade down to the levels that alter the abiotic properties of an ecosystem (Hansen and Galetti 2009). Most rewilding efforts are predicated on the top-down trophic cascade effects on ecosystems that large carnivores and herbivores can trigger. These ecological principles complement ideas of rewilding, and set the stage for considering how to promote a natural restoration agenda.

The Hairston-Smith-Slobodkin (HSS) hypothesis (Hairston et al. 1960) posits that carnivores are responsible for a green world because predation limits the abundance of herbivores. The basis of this thesis is that any population not limited by its food supply must be limited by predators. Herbivores are commonly reduced by predation below density levels that would be limited by an adequate supply of plants. And, it is common that in today's suburban and urban landscapes, herbivores like white-tailed deer (*Odocoileus virginianus*) lack predators. Frequently, this species becomes over abundant in these areas leading to winter starvation and heavily browsed vegetation in developed settings. Thus, by consuming herbivores, predators actually benefit plant life by reducing browsing. This hypothesis has been tested repeatedly and often supported (Ripple and Beschta 2012), and has now become one of the principles of ecological science. This principle also directly supports the rewilding approach by justifying why a full complement of biotic community members is necessary to support a fully functioning ecosystem shaped by natural processes.

A meta-analysis (Schmitz et al. 2000) of 60 tests of the HSS hypothesis in terrestrial settings showed that carnivores alter plant cover indirectly through predation on herbivores. Analyses of a majority of these studies showed that plants were changed by the removal of carnivores. Though these studies included a variety of vertebrate and invertebrate carnivores, the type of carnivore did not diminish the top-down effect on plants. The conclusion, beyond supporting the HSS hypothesis, indicates that top predators have pervasive effects across an ecosystem. A clear case of this idea was shown on newly created islands in a large Venezuelan reservoir (Terborgh et al. 2001). When the reservoir was filled with water, it trapped partial communities on the islands. On predator-free islands, herbivores became hyper-abundant and not only depressed plant abundance but shifted the plant cover to taxa that were resistant to grazing. These results indicate that partial communities, especially those lacking predators, can result in biologically impoverished plant cover. Subsequently, these islands looked quite different of significant changes in vegetation cover. Additionally, top predators have been shown to be important for maintaining the diversity of smaller predators. In the absence of top predators, lower trophic level carnivores proliferate and imperil their prey (Johnson et al. 2007).Thus, the bottom line is: without the presence of a full complement of species, we will not have a natural community in ecosystems, parks, or protected areas.

The concept of the predators-herbivores-plants interrelationship represents a simple food chain. More complex is the notion of a food web, first introduced by Elton (1927) (Figure 4.2). Food webs show connections among many species through predation and competition for the same food resource (Paine 1980). These species and connections can be organized by trophic levels and linkage strengths (Paine and Schindler 2002). Species with strong interactions can significantly shape community, and their absence can result in an altered community with very different properties. Moreover, changes in communities at specific locations can alter the properties of an entire ecosystem. These strongly influential species are often labeled keystone species (Paine 1969). Links within trophic levels commonly designate competition, which can also shape community composition through time. The concepts of a food web with species interactions and keystone species point to the need to have complex communities to attain a natural environment.

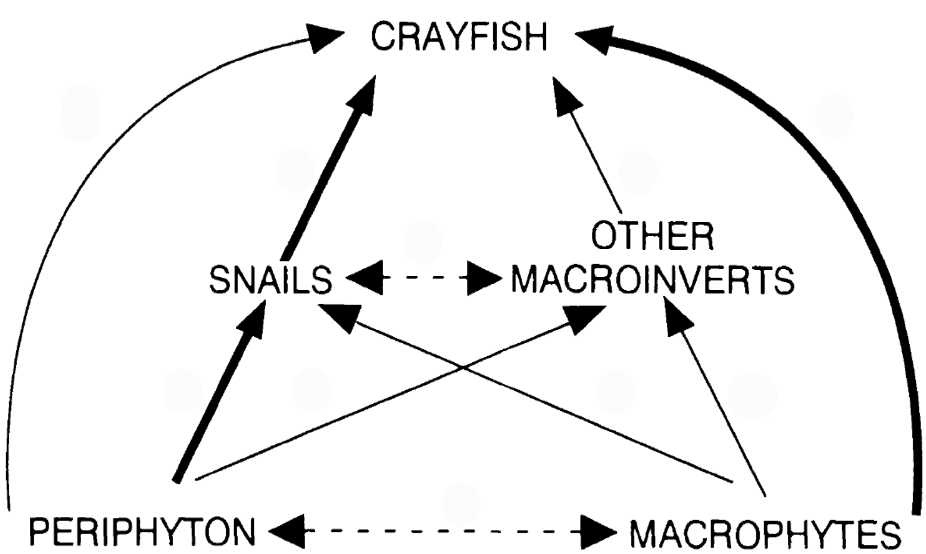

Figure 4.2: A simple littoral food web for Plum Lake, Wisconsin with three trophic levels and interactions. Strengths noted by line weights. Modified from: Lodge et al. 1994

Food chains, food webs, and keystone species represent the pervasive effects various species can have across trophic levels in a cascading manner (Carpenter et al. 1985). The term cascade is appropriate here because these effects alternate across adjacent trophic levels, and this principle is termed a trophic cascade (Figure 4.3). The effects of trophic cascades can be strong and can influence the biological organization of an ecosystem and may extend to physical and chemical changes as well. For example, introducing predatory fish in lakes has the effect of reducing planktivore abundance, increasing zooplankton, decreasing phytoplankton, and increasing the transparency of the water. This trophic cascade results in differences in macrophyte abundance, nutrient cycling, and more (Schmitz et al. 2010). Shapiro and Wright (1984) were among the first to recognize the potential of altering food webs as a management tool and termed the approach biomanipulation. Biomanipulations are intentional modifications of trophic organizations that can force trophic cascades to shape the characteristics of an ecosystem (Shapiro and Wright 1984). This lake management approach was not connected to the idea of rewilding, but again it is based on the same ecological principles and processes. Similar trophic cascades have been documented for marine ecosystems (whales–otter–urchins–kelp), small freshwater bodies (mosquitoes–protozoa–bacteria), tropical forests (beetles–ants–insects–plants) and others (Pace et al. 1999). Cascading effects across trophic levels again indicate that top predators are needed to support conservation actions that seek to restore the natural properties of ecosystems.

Further effects of altered community composition have been reported, such as proliferation of small species, chemical cycling, vegetation structure, soil properties, and others. Predator effects, food chains and webs, keystone species, and trophic cascades all support the rewilding approach to achieve natural ecosystems through species interactions within a community and its environment. Early in human history our species hunted and eliminated many large predators and herbivores. Based upon the ecological principles just discussed, we can assume these actions prominently altered most landscapes. A consequence is that we may not recognize a natural ecosystem because changes occurred a very long time ago. The rewilding approach is one strategy that attempts to recover natural ecosystems by introducing species that can reestablish the necessary relationships endemic to an ecosystem's community and habitat.

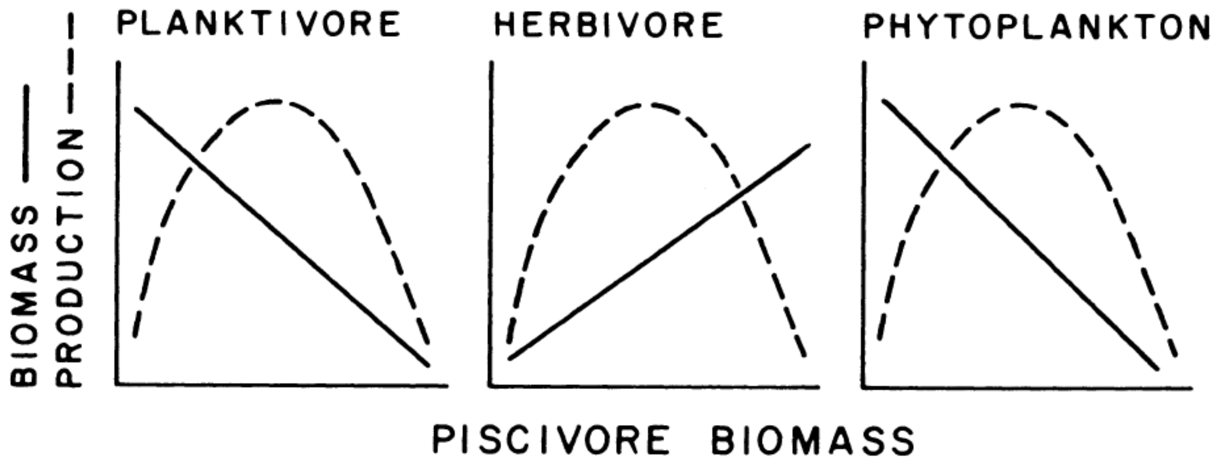

Figure 4.3: A trophic cascade showing a response by trophic level to increasing piscivore biomass. Planktivores decline, herbivorous plankton increases, and phytoplankton declines. This shows the alternating responses across trophic levels to increasing top down effects from a predator fish. This pattern is a definitive indicator of a trophic cascade. Source: Carpenter et al. 1985

IMPLEMENTING REWILDING

The conservation movement has traditionally been dominated by defensive approaches such as saving wild spaces and adding restrictions to human activities. However, biodiversity continues to decline worldwide, so more comprehensive action,and programs may be needed beyond protecting places we deem as natural. Rewilding can be considered a proactive conservation strategy that attempts to restore natural environments by reinvigoration. It is an alternative to protection because it has a focus on re-creating natural environments by putting the ecological pieces together and letting nature take over. Additionally, megafauna often hold a particular fascination for the public who recognize that most large predators and herbivores were lost long ago. Rewilding can be an affirmative move in the public arena and can result in shifting some locations to more natural environments.

Top-down effects of predators, species interactions, and trophic cascades have been documented in a wide variety of habitats and may be critical to the formation of natural and stable ecosystems (Pace et al. 1999; Donlan et al. 2006; Donlan and Greene 2010). With the rise of human populations across the world, many large species became extinct and cannot be used to reestablish natural communities. The rewilding approach advocates replacing these extinct species with existing analog species to reinstitute natural processes that may restore ecosystems (Figure 4.4). An extreme version of this was proposed by Donlan et al. (2005; 2006) to recover natural grassland ecosystems on the Great Plains of North America. Their proposal includes importing lions, elephants, camels, Eurasian horses and other large vertebrates, as taxon substitutes for long lost native species that might reestablish lost evolutionary processes and ecosystem characteristics like natural plant cover. Reaction to this idea was both strongly positive and negative in the conservation and management communities. Supporters cited a proactive conservation agenda, intrigue surrounding the idea, and interest in large animals. Opponents focused on fear of the consequences, elitism and imperialism. Regardless of any particular agenda, it does introduce the idea of biological manipulations, including the use of non-native species to recreate natural features of a landscape that have since been lost.

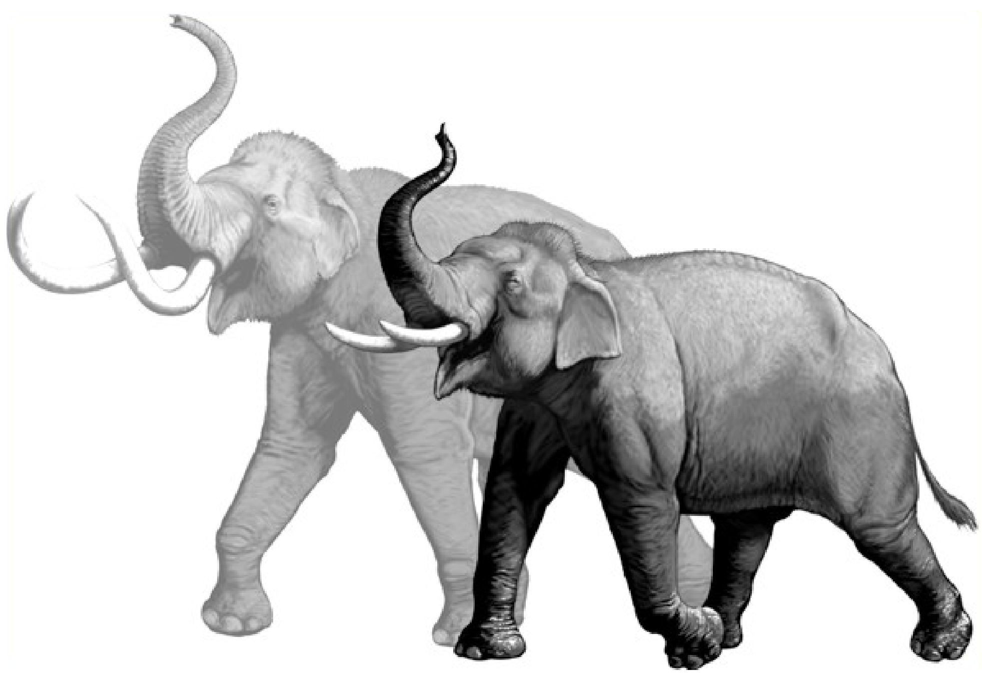

Figure 4.4: The Asian elephant (right) serves as an ecological proxy for the extinct North American mammoths (left) in an effort to restore megaherbivore function to North America. Source: Donlan et al. 2006

The rewilding strategy intends to go beyond natural properties of landscapes by including evolutionary effects on species. The effect of species interactions can limit abundances of small species, may deter invasive species, and shape the dispersion of species (Wallach et al. 2010). Species interactions like predation and competition can also shape the morphology and innate capabilities of species through natural selection. A clear example was raised by Donlan et al. (2005) for the pronghorn antelope (*Antilocapra americana*) which has incredible running speed that is no longer a fitness advantage since the demise of the American cheetah (*Acinonyx trumani*). Therefore the benefits of reintroducing species provide more than ecosystem and habitat recovery, they also influence the evolution of species characteristics, with ecotourism and extinction prevention of the American cheetah as additional benefits.

One distinctive feature of rewilding is the return of large predators to protected areas. While justified by the scientific evidence on predator effects, conservation interests often focus only on large predators. The public is often inspired by predators because of their powerful and majestic appearance. Often predators serve as flagship species in conservation programs to garner public support. These species also push conservation toward thinking about large reserves because most top predators need a large range and an expansive habitat. Thus, large predators can serve to motivate and promote conservation efforts within the public arena, and expand the scope of conservation planning.

Research supports using top predators as an indicator of high biodiversity, environmental integrity, ecosystem productivity, and resilience; yet these predators are still dependent on species in lower trophic levels and are often the first to disappear when an ecosystem is disrupted. Larger predator species are impeded by habitat alterations and fragmentation, require prey that can be specialized, and suffer other vulnerabilities such as the accumulation of toxins. In a way this is the reverse of top-down effects because ecosystem degradation passes effects up through the trophic levels to the top predators. Therefore, top predators can serve as an umbrella species for conservation since they encapsulate conditions that support many other species, and they can serve as sentinel species for ecosystem disruptions from pollutants, habitat change, and fragmentation of the landscape (Sergio et al. 2008).

Rewilding is a controversial approach in the conservation movement because of the actions it entails, such as reestablishing keystone species, predators, and non-native species. This conservation strategy would require large tracts of land and significant public involvement. Rewilding can be risky since there is always the possibility of unanticipated consequences and catastrophic effects on native flora and fauna, their habitats, and even entire ecosystems. Failures such as these would be prominent in terms of public attention and could prove damaging to broader conservation interests. Restoring the long-lost past may be impossible in the modern world, and some feel that this direction is based more on sentiment than science. Finally, questioning the effectiveness of rewilding should be expected because we do not know against what conditions of the far past we can measure success. Also, rewilding can produce ecosystems that are more a product of its own implementation details than a reflection of the original or natural ecosystem it seeks to recreate. Doubts about this conservation technique continue to echo throughout conservation circles, particularly with respect to how anyone could determine whether rewilding successfully restored a natural ecosystem. Nevertheless, it is a conservation strategy that is currently used because of its potential benefits and also because of the rewilding efforts that have yielded positive results.

CASE STUDY: REINTRODUCING WOLVES INTO YELLOWSTONE NATIONAL PARK

The gray wolf (*Canis lupus*) is one of the most dominant predators in North America because this species hunts in packs and targets large herbivores. Wolves were intentionally eradicated in the Western United States as settlers brought in cattle, horses, and sheep which are natural prey for the wolf. Yellowstone National Park was formed in 1872 and from its origin prohibited domestic livestock grazing. Wolves maintained a small presence in the park area until the mid-1920s when they were extirpated from the Western United States. Wolves were reintroduced in the Yellowstone National Park area starting in 1995 in response to their endangered species status. The restoration of wolves in the Yellowstone National Park area completed the terrestrial community of large species, but was not considered to be rewilding. Since that conservation strategy was not identified in 1995. The reintroduction of wolves to this area was well studied and the findings indicate how pervasive the effects of a large predator can be in shaping an ecosystem (Figure 4.5) (Fortin 2005; Beschta and Ripple 2010). These findings were consistent with the scientific basis of rewilding. Thus, this case is used here as an example of rewilding, and shows how communities can change to reestablish a full ecosystem by bringing back its top predator.

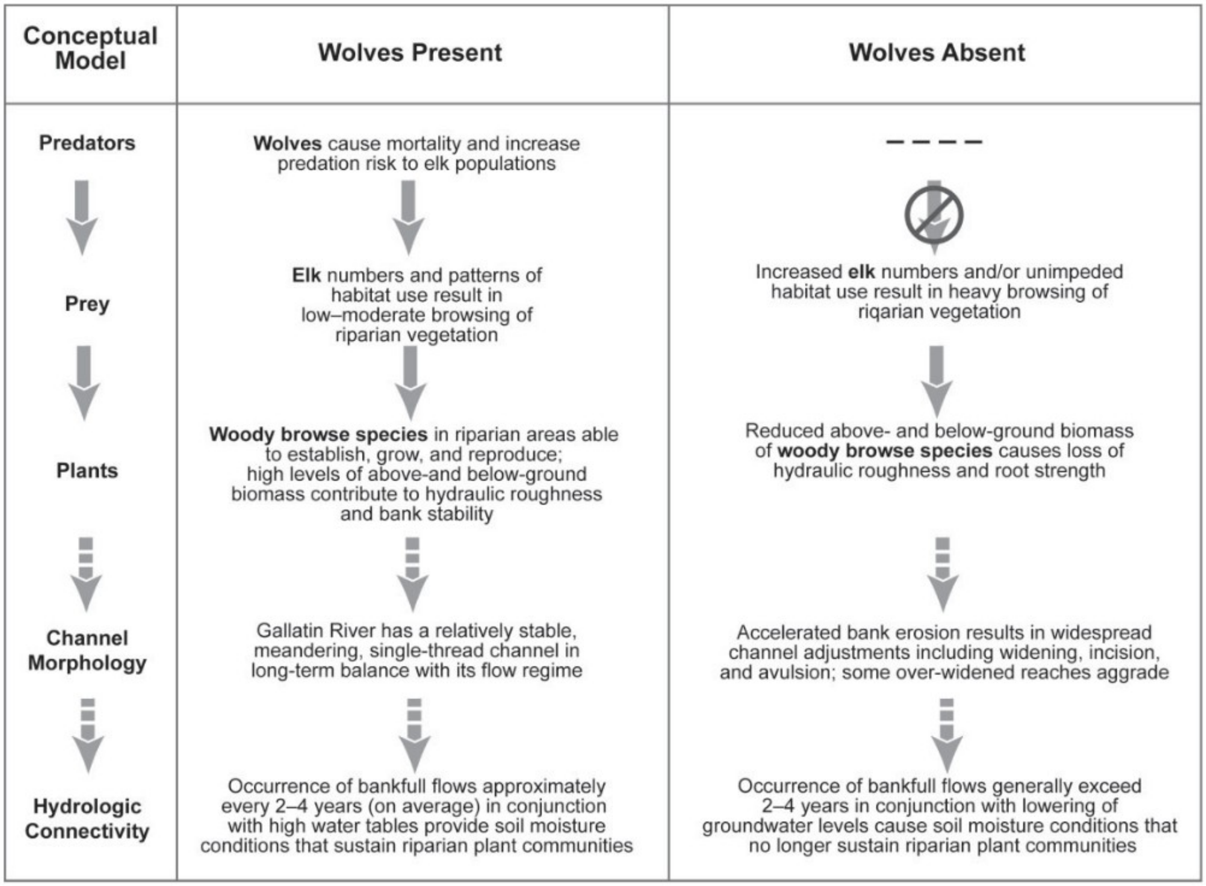

Figure 4.5: Summary of "top-down" trophic cascades (solid arrows) and hydrogeomorphic processes (dashed arrows) conceptual model with and without wolves for floodplain riparian systems in the upper Gallatin elk winter range. Source: Beschta and Ripple 2006

Adjacent to the Yellowstone National Park, the upper Gallatin River valley is under the control of the United States Forest Service which prohibits livestock grazing to help maintain satisfactory forage conditions for the elk's (*Cervis elaphus*) winter range. In 1919, when wolves were originally present in the area, the elk population was about 1,600. Then the wolf population was extirpated in the mid-1920s.

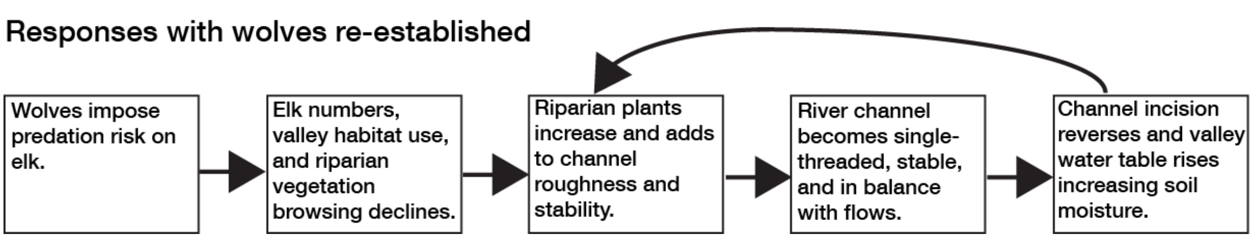

Figure 4.6: A series of responses in the biota and physical features of the Gallatin River valley with wolves reestablished. Source: Beschta and Ripple 2006

By the 1930s, elk numbers increased to around 2,500. Due to its increasing size, the elk population was reduced by culling, but this was opposed by the public. After the culling program was suspended, the herd size increased to 19,000 by 1968. At these high abundance levels, elk foraging conditions in their winter range became degraded due to high rates of herbivory. Starvation was common and the herd eventually shrank to about 1,000. Wolves were reintroduced in the park area in 1995. Many experts regard wolves as the most potent carnivore in North America. The effects of the reintroduced wolves were monitored, and many studies were conducted on all direct and indirect impacts of the wolves on the Gallatin River valley ecosystem.

The wolves initiated a trophic cascade by reducing elk numbers in the Gallatin River valley (Figure 4.6). Before wolves, elk freely grazed the riparian vegetation which was a favored food source. Once wolves returned to the valley, elk were subjected to a significant predation threat. The elk were vulnerable prey in the brushy and open terrain. Beyond reduced abundance, the elk changed their habitat use from the valley floor to the forested uplands. Behaviorally reduced use of the valley floor diminished riparian vegetation grazing. Slowly, after the mid-1990s, riparian vegetation such as aspen (*Populus tremuloides*) recovered along the Gallatin River (Figure 4.7) (Ripple and Beschta 2007).

The effects of wolf predation extended well beyond elk and riparian plants in many surprising ways. Changes to the river channel, groundwater levels, and soil conditions have been linked to reestablishment of wolves in the area. Prior to the reintroduction of wolves, heavy grazing of riparian vegetation by elk reduced the vegetative cover along river banks. That change increased bank erosion and allowed widening of the river channel. Further, without firmly established riparian plants, the river channel became unstable and incised which reduced the elevation of the water. This lower channel elevation also drained valley groundwater causing drier surface soil conditions. This further reduced valley floor vegetation cover and growth. With the return of the wolves, riparian plants recovered slowly and the natural channel structure returned, groundwater elevation rose, soil moisture increased, and the more saturated soils benefited riparian vegetation growth.

Photos (Figure 4.8) illustrate the Gallatin River valley at the time the wolves were present (Figure 4.8a), the period when the wolves were eradicated (Figure 4.8b), during the long period of wolf absence (Figure 4.8c), and after wolves were reestablished as an effective predator (Figure 4.8d). This is a clear, well-documented example of a top predator having effects that go beyond herbivorous prey and plants all the way down to groundwater conditions. This case illustrates the extent to which species interactions can change ecosystem conditions.

Further effects of wolves have been documented in the Yellowstone National Park and surrounding region. Coyote (*Canis latrans*), raven (*Corvus corax*), and grizzly bears (*Ursus arctos horribilis*) feed on elk carrion from wolf kills. This food source has improved the reproductive success and survival of these scavenger species, and changed their foraging behaviors as well (Wilmers et al. 2003). For example, grizzly bears have been known to forego hibernation altogether in Glacier National Park, Montana in favor of scavenging wolf kill sites (Wilmers et al. 2003). Without wolves, carrion availability was primarily a function of the generally severe winters when high snow levels and cold temperatures caused elk to weaken and die, usually at the end of winter. For wolves, scavenging at wolf kill sites occurs on a year-round basis. By changing the distribution and abundance of carrion availability, wolves may serve to facilitate the acquisition of food by scavengers (Wilmers et al. 2003). Wolves also decrease the year-to-year and month-to-month variation in carrion availability. By transferring the availability of carrion from the highly productive late winter to the less productive early winter, and from highly productive years to less productive ones, wolves provide a temporal subsidy to scavengers. Thus, wolves change the timing of resource availability from a pulsed resource at the end of severe winters to a more constant resource throughout the winter. This resource subsidy may in turn promote increased biodiversity and lead to larger populations of scavenger species and, in fact, studies have documented an increase in scavenger species after wolf reintroductions (Wilmers et al. 2003).

In addition, wolves may mitigate climate change effects in Yellowstone National Park. The winter period on the northern range of Yellowstone National Park has lessened since 1948 (Wilmers and Getz 2005). Evidence for winter period contraction has been seen in the form of decreased duration of snow cover, snowfall and snow depth; average temperatures increasing in late winter; and an increase in the number of winter days with temperatures above freezing (Wilmers and Getz 2005). The easing of these winter conditions implies that elk will recover sooner from the detrimental stresses of winter. Smaller snow packs allow elk easier access to food and decrease energy expenditures required for movement. Herbaceous plant growth usually begins within a few days to weeks of the last snow cover, so elk may increase the quality and quantity of food intake earlier in the year. And, elk will experience a shorter physiologically stressful winter period. These factors are likely to influence the timing and abundance of carrion as late-winter elk mortality declines. Under scenarios without wolves, scavengers could face food bottlenecks in the absence of late-winter carrion. Coyotes are highly dependent on late-winter and early-spring carrion to sustain them until late spring, when elk calves and ground squirrels become abundant. Areas without wolves will experience carrion as an increasingly pulsed resource under climate change, whereas areas with wolves will likely encounter carrion throughout the winter months. Thus wolves buffer the effects of climate change on carrion availability and allow scavengers to adapt to a changing environment over a longer time scale more commensurate with natural processes (Wilmers and Getz 2005).

Figure 4.7: August 2006 photographs of (A) recent aspen recruitment in a riparian area along the Lamar River in Northeastern Yellowstone National Park and (B) a lack of recent aspen recruitment in an adjacent upland. The dark, furrowed bark comprising approximately the lower 2.5 m of aspen boles in (B) represents long-term damage due to bark stripping by elk. Source: Ripple and Beschta 2007

In addition, increased riparian vegetation, an indirect result of wolf reintroductions, has supported a greater diversity and abundance of birds (Anderson 2007). Before the reintroduction of wolves, elk reduced willow structure (much less cover < 2 m height) and reduced the numbers of all bird species (Anderson 2007). The least sensitive species were the habitat generalists and ground nesting birds. Willow and aspen communities support a greater diversity of flora and fauna than most other habitats in the western United States; such an enhancement of biomass can subsequently increase bird diversity and abundances in the region.

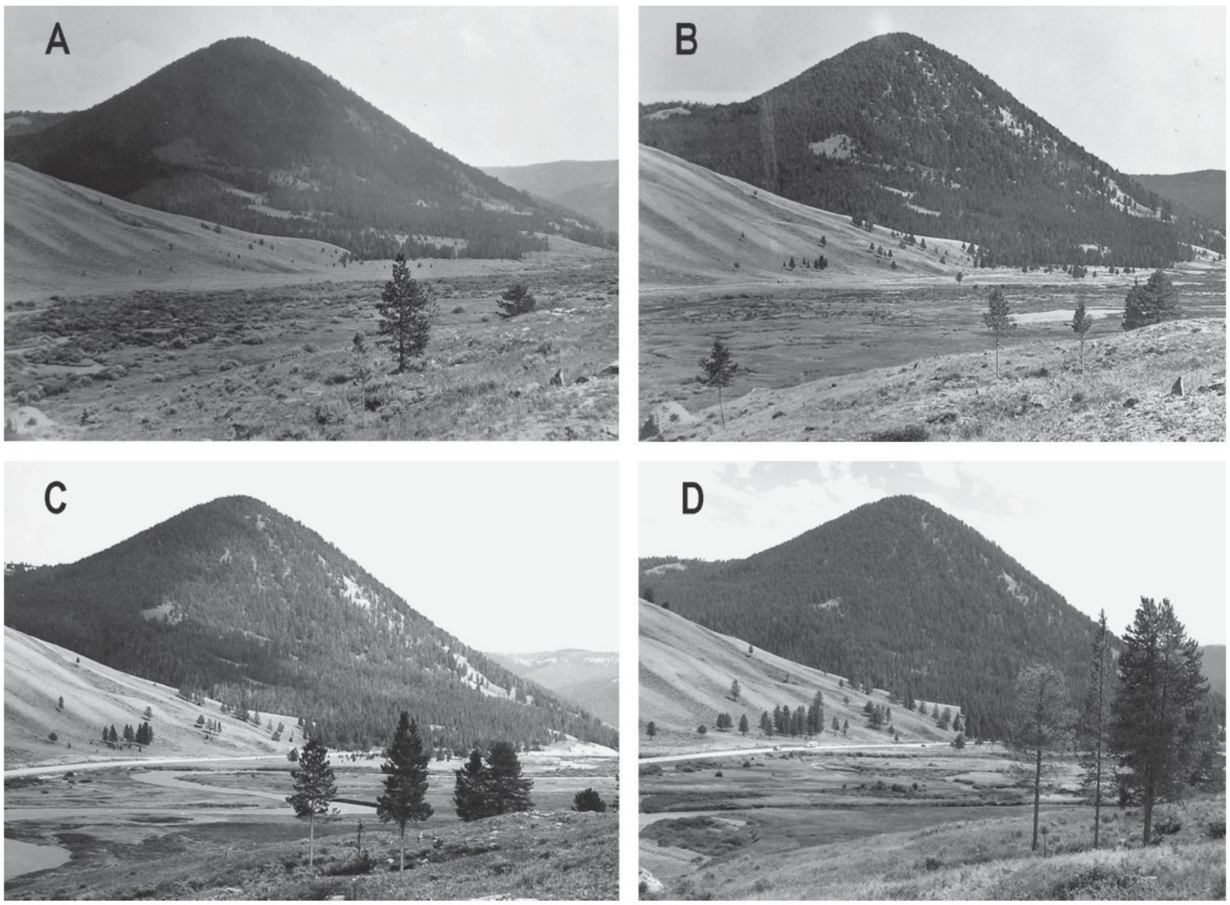

Figure 4.8: A series of photographs of the Gallatin River valley when wolves were present (A, 1924; extensive riparian willow), after they were absent (B, 1949; heavy low browsing, fewer riparian willows), when elk were abundant (C, 1961; heavy low browsing, riparian willow absent), and after wolf reestablishment (D, 2003; riparian willow returning). Dense riparian vegetation in 1924 declined by 1949 and very little remained in 1961. The 2003 photograph shows riparian vegetation partially recovered and the river channel changed shape with wolves present since 1995. Source: Beschta and Ripple 2006

Overall, biodiversity and productivity have increased in Yellowstone National Park with wolves reestablished as a top predator in the ecosystem. The extirpation of wolves had a cascading effect on lower trophic levels (first elk and then willows) along the Gallatin River. Even though the long-term trend in elk numbers was one of decline due to the annual harvest of elk via hunting, predation by other large carnivores, and the periodic occurrence of mortality during severe winters, this situation was unable to prevent the continued decimation of streamside vegetation. The heavy annual browsing of willow communities after the loss of wolves ultimately generated major changes in floodplain functions and channel morphology. This case is the first to connect a large, highly interacting carnivore to the characteristics of a river floodplain and its channel. These findings are consistent with the rewilding argument for reestablishing a natural ecosystem by assembling a full complement of species that can reinvigorate ecological processes.

SUMMARY

The case of wolves in Yellowstone National Park was not considered a rewilding case when it started, but it is consistent with the approach and demonstrates the anticipated benefits of the hypothesis. Among conservationists and managers, the rewilding approach is controversial and subsequently, is not commonly advocated. The primary issues and concerns with rewilding focus on the introduction of non-native species, the risk of unintended results that may adversely affect the ecosystem that is being recovered, and the accompanying high visibility with the public whose support is crucial for conservation programs in general. A great deal of conservation planning emphasizes conserving existing processes rather than restoring extinct species interactions. Rewilding centers on restoring ecological function which is an optimistic goal. As more cases and studies show positive results from rewilding projects, increased acceptance will fuel further utilization of this strategy in future conservation efforts.

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Biologically-Focused Techniques

Chapter 5 - Endangered Species Protection and Recovery

The next chapter in the biologically-focused techniques group centers on endangered species protection and recovery, in particular, the United States Endangered Species Act. The Endangered Species Act defines the approach to species conservation, and is a prominent part of ecological conservation in the United States. In this chapter we will cover the history and effectiveness of the Endangered Species Act, the process of listing (or delisting) species, criteria for endangerment and recovery, and will end with a case study on shortnose sturgeon (*Acipenser brevirostrum*).

HISTORY OF THE U. S. ENDANGERED SPECIES ACT

The U. S. Endangered Species Act (ESA) was enacted in 1973 in reaction to congressional findings that various species of fish, wildlife, and plants in the United States had been rendered extinct as a consequence of economic growth and development, or had been so depleted in numbers that they were in danger of or threatened with extinction (United States Code 1973; Title 16, Sections 1531-1544) . In this Act, United States Congress deemed that these species have aesthetic, ecological, educational, historical, recreational, and scientific value for the Nation and its people. So the United States pledged itself to conserve to the extent practicable the various species of fish, wildlife and plants vulnerable to extinction (United States Code 1973). Thus, the purpose of the ESA is to protect and recover imperiled species and the ecosystems upon which they depend. The ESA is administered by the Interior Department's United States Fish and Wildlife Service (USFWS) and the Commerce Department's National Marine Fisheries Service (NMFS).

Several terms are used in relation to the ESA. For clarification, "endangered" means a species that is in danger of extinction throughout all or a significant portion of its range (USGS 2021). "Threatened" means a species is likely to become endangered within the foreseeable future. "Imperiled" and "at risk" are not legal terms under the ESA. Generally speaking, these species are animals and plants whose populations are in decline and may be in danger of extinction, which can include species that are at low enough numbers to be near extinction even though they are not legally protected under the ESA (USGS 2021). All species of plants and animals, except pest insects, are eligible for listing as endangered or threatened. Congress defined "species" to include subspecies, varieties, and for vertebrates, distinct population segments. In effect, the ESA constitutes a federal takeover of species management from state-level control.

LISTING SPECIES THROUGH THE ESA

Species are listed as endangered or threatened through the ESA solely on the basis of their biological status and threats to their existence. Five factors are considered when evaluating a species for listing (United States Code 1973; section 1533):
1) Damage to, or destruction of, a species' habitat;
2) Over-utilization of the species for commercial, recreational, scientific, or educational purposes;

3) Disease or predation;
4) Inadequacy of existing protection; and
5) Other natural or manmade factors that affect the continued existence of the species.

Species are proposed for listing (or delisting) through petitions which require published, peer reviewed findings (Figure 5.1). The average time from petition to listing is 12 years, with generally higher wait times for plants (Puckett et al. 2016). The USFWS also maintains a list of candidate species. These are species for which the USFWS has enough information to warrant proposing them for listing but are precluded from doing so by higher listing priorities. These "warranted but precluded" proposals require subsequent 12-month findings on each succeeding anniversary of the petition until the USFWS either undertakes a proposal or makes a "not warranted" ruling.

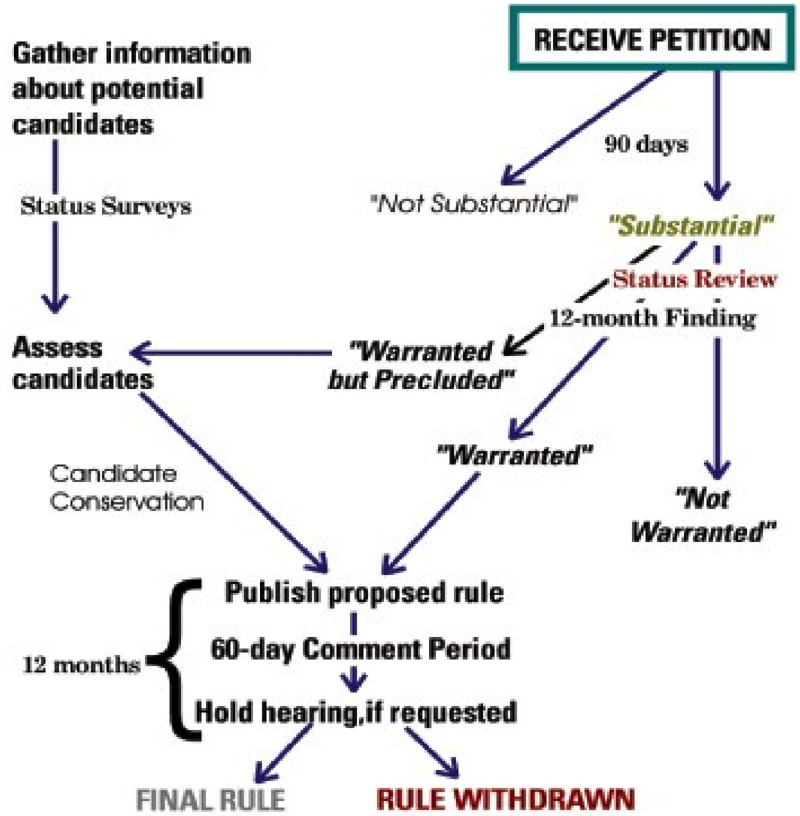

Figure 5.1: Flowchart for listing or delisting of a species. Source: United States Fish and Wildlife Service 2016

PROTECTION AND RECOVERY THROUGH THE ESA

The ESA protects endangered and threatened species and their habitats by prohibiting the "take" of listed species. Take includes activities that would harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, collect or attempt to engage in any such conduct of any listed species (USFWS 2017). This includes significant habitat modification or degradation, by private or federal entities, that actually jeopardizes the continued existence of listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering (USFWS 2017).

The goal of the ESA is to recover species so that they no longer need protection. Recovery plans describe the steps needed to restore a species to a healthy status. Agency biologists write and implement these plans with the assistance of species experts; other Federal, State, and local agencies; Tribes; nongovernmental organizations; academia; and other stakeholders (USFWS 2017).

FEDERAL AGENCY ACTIVITIES RELATED TO THE ESA

The laws of the ESA require Federal agencies to use their legal authorities to promote the conservation purposes of the ESA and work with the USFWS and NMFS to ensure that their actions are not potentially jeopardizing the continued existence of listed species. The USFWS or NMFS can make a jeopardy determination of potential actions and offer "reasonable and prudent alternatives" to avoid actions that may potentially harm a listed species (USFWS 2017). The ESA also requires the designation of "critical habitat" for listed species when "prudent and determinable." Critical habitats are geographic areas that contain the physical or biological features that are essential for a listed species' survival, even if the species is not currently occupying the area at the time of listing. Critical habitat designations affect only Federal agency actions, or federally funded or permitted activities. Federal agencies are required to avoid "destruction" or "adverse modification" of designated critical habitat (USFWS 2017). An area can be excluded from critical habitat designation if an economic analysis determines that the benefits of excluding it outweigh the benefits of including it, unless failure to designate the area as critical habitat could lead to extinction of the listed species. A process exists for exempting projects from the restrictions of the law if a Cabinet-level "Endangered Species Committee" (aka the "God Squad" due to the substantial impact of its decisions on the natural world) decides the benefits of the project clearly outweigh the benefits of conserving a species (USFWS 2017).

The Endangered Species Committee is composed of seven Cabinet-level members: the administrator of the Environmental Protection Agency, the administrator of National Oceanic and Atmospheric Administration, the chairman of the Council of Economic Advisers, a representative from the state in question, the Secretary of Agriculture, the Secretary of the Army, and the Secretary of the Interior. This committee has the authority to allow an extinction of a species by exempting a federal agency from certain requirements (known as "an exemption"). To grant an exemption, five of the seven members must vote in favor of the federal agency's project. The following conditions must be met for an exemption to receive approval (United States Government Publishing Office 1978):

- There must be no reasonable alternative to the agency's action.
- The benefits of the action must outweigh the benefits of an alternative action.
- Where the species is conserved the action is of regional or national importance.
- Neither the federal agency nor the exemption applicant made an irreversible commitment to the resources.

Additionally, mitigation efforts must be taken to reduce the negative effects on the species in question.

Since its creation in 1978, the Committee has been convened only a handful of times: for the whooping crane (*Grus americana*) and snail darter (*Percina tanasi*) in 1979 and the Northern spotted owl (*Strix occidentalis caurina*) in 1992. In cases related to the whooping crane (*Grus americana*) and Northern spotted owl (*Strix occidentalis caurina*), the Endangered Species Committee chose to favor projects over species protection (Sheikh 2017). There were three other instances (in 1979, 1985 and 1986) in which applications were filed with the committee, but these applications were ultimately withdrawn or abandoned (Sheikh 2017).

HABITAT CONSERVATION PLANS AS PART OF THE ESA

Two-thirds of federally listed species have at least some habitat on private land, and some species have the majority of their remaining habitat on private land (USFWS 2017). The ESA provides relief to landowners who want to develop property inhabited by listed species. Landowners can receive a permit to take species under an approved habitat conservation plan (HCP). HCPs include steps to minimize and mitigate any adverse impacts, as well as funding to carry out the mitigation activities (USFWS 2017). Additionally, Safe Harbor Agreements (SHAs) provide regulatory assurance for non-Federal landowners who voluntarily aid in the recovery of listed species by improving or maintaining wildlife habitat (USFWS 2017). Under SHAs, landowners manage the enrolled property and may attempt to return it to originally agreed-upon “baseline” conditions for the species and its habitat by the end of the agreement, even if this results in the incidental take of that species (USFWS 2017).

OTHER ENDANGERED SPECIES SYSTEMS

The International Union for Conservation of Nature and Natural Resources (IUCN) has become the world’s most comprehensive information source on the global extinction-risk status of animal, fungus and plant species (IUCN 2021).

The North American Native Fishes Association (NANFA) maintains a spreadsheet of endangered, threatened and other special status fishes of North America (excluding Hawaii) (NANFA 2021). The information is compiled from federal, state and provincial natural resource agencies.

Additionally, many states have their own listings of endangered, threatened and special status species.

HOW MANY SPECIES ARE ENDANGERED

Species listings and delistings fluctuate year to year but the general trend over time is toward increasing listings (Figure 5.2) (World Economic Forum 2016).

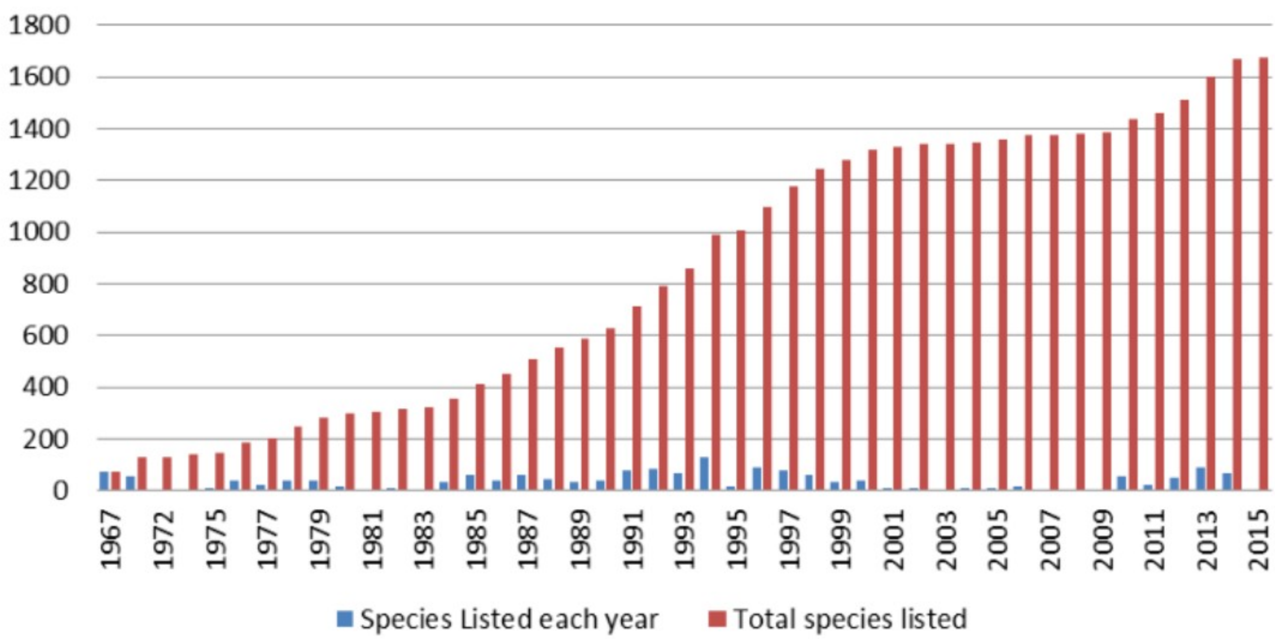

Figure 5.2: Listings under the Endangered Species Act. Source: World Economic Forum 2016

The USFWS maintains a tally of listed species, both domestic and foreign, and those with active recovery plans (Table 5.1). As of September 2021 in the United States, 1,271 species were listed as endangered (503 animals, 768 plants) and 395 as threatened (224 animals, 171 plants). In total, 1,666 species were listed (727 animals, 939 plants).

Table 5.1: Summary of United States and Foreign listed species and recovery plans as of September 2021. Twenty one animal species (13 in the United States and 8 Foreign) are counted more than once, primarily because these animals have distinct population segments (each with its own individual listing status). Source: United States Fish and Wildlife Service 2021

	United States			Foreign			Total Listings (US and Foreign)	US Listings with active Recovery Plans
Group	Endangered	Threatened	Total Listings	Endangered	Threatened	Total Listings		
Amphibians	22	16	38	8	1	9	47	27
Annelid Worms	0	0	0	0	0	0	0	0
Arachnids	12	0	12	5	0	5	17	12
Birds	76	23	99	217	21	238	337	89
Clams	76	15	91	2	0	2	93	74
Corals	0	7	7	3	15	18	25	0
Crustaceans	25	4	29	0	0	0	29	19
Fishes	94	76	170	27	9	36	206	107
Flatworms and Roundworms	0	0	0	0	0	0	0	0
Hydroids	0	0	0	0	0	0	0	0
Insects	75	14	89	4	0	4	93	48
Mammals	68	28	96	259	23	282	378	60
Millipedes	0	0	0	0	0	0	0	0
Reptiles	16	29	45	71	24	95	140	40
Snails	39	12	51	1	1	2	53	34
Sponges	0	0	0	0	0	0	0	0
Animal Totals	**503**	**224**	**727**	**597**	**94**	**691**	**1418**	**510**
Plant Totals	**768**	**171**	**939**	**1**	**2**	**3**	**942**	**728**
Grand Totals	**1271**	**395**	**1666**	**598**	**96**	**694**	**2360**	**1238**

WHY ARE SPECIES ENDANGERED

Animals and plants are endangered for a variety of reasons (Table 5.2). The most common causes of endangerment are interactions with nonnative species, urbanization and agriculture (Czech and Krausman 1997).

Table 5.2: Causes of endangerment for species classified as endangered and threatened by the United States Fish and Wildlife Service. Source: Czech and Krausman 1997

Cause	Number of species endangered by cause and rank of frequency*	Number of species endangered and rank of frequency†
Interactions with nonnative species	305 - 1	115 - 8
Urbanization	275 - 2	247 - 1
Agriculture	224 - 3	205 - 2
Outdoor recreation and tourism development	186 - 4	148 - 4
Domestic livestock and ranching activities	182 - 5	136 - 6
Reservoirs and other running water diversions	161 - 6	160 - 3
Modified fire regimes and silviculture	144 - 7	83 - 10
Pollution of water, air, or soil	144 - 8	143 - 5
Mineral, gas, oil, and geothermal extraction or exploration	140 - 9	134 - 7
Industrial, institutional, and military activities	131 - 10	81 - 12
Harvest, intentional and incidental	120 - 11	101 - 9
Logging	109 - 12	79 - 13
Road presence, construction, and maintenance	94 - 13	83 - 11
Loss of genetic variability, inbreeding depression, or hybridization	92 - 14	33 - 16
Aquifer depletion, wetland draining or filling	77 - 15	73 - 15
Native species interactions, plant succession	77 - 16	74 - 14
Disease	19 - 17	7 - 18
Vandalism (destruction without harvest)	12 - 18	11 - 17

*Including Hawaiian and Puerto Rican species. †Not including Hawaiian and Puerto Rican species.

WHERE ARE SPECIES ENDANGERED

Dobson et al. 1997 detailed the distribution of endangered species throughout the United States for plants, birds, fish and molluscs (Figures 5.3a-d). Interestingly, they found that hot spots (areas with high numbers of endangered species) for different species groups rarely overlap, except where anthropogenic activities reduce natural habitat in centers of endemism (Dobson et al. 1997). They also found through their study that the amount of land that needs to be managed to protect currently endangered and threatened species in the United States is a relatively small proportion of the land mass (Dobson et al. 1997).

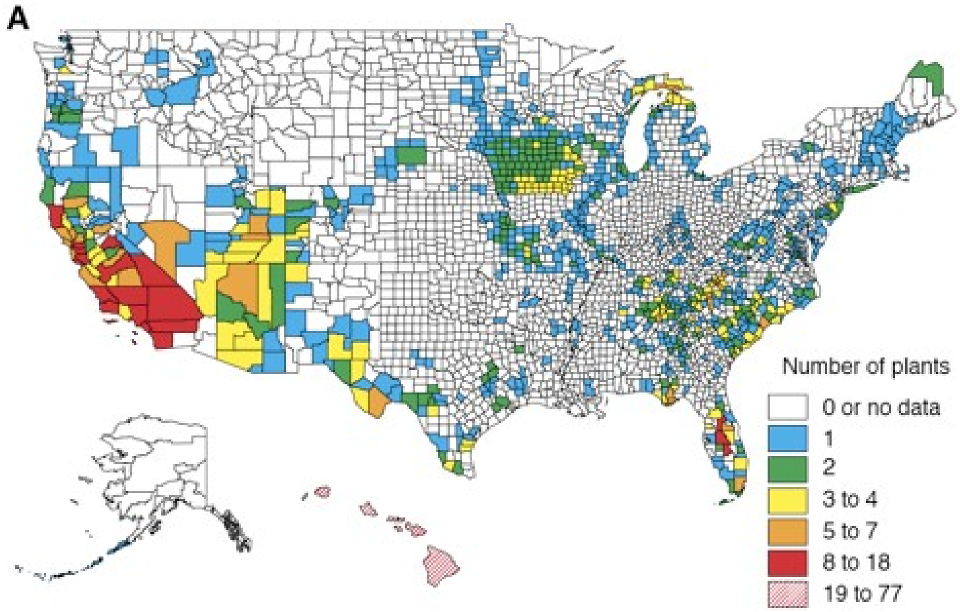

Figure 5.3a: The geographic distribution of endangered plant species in the United States. Source: Dobson et al. 1997

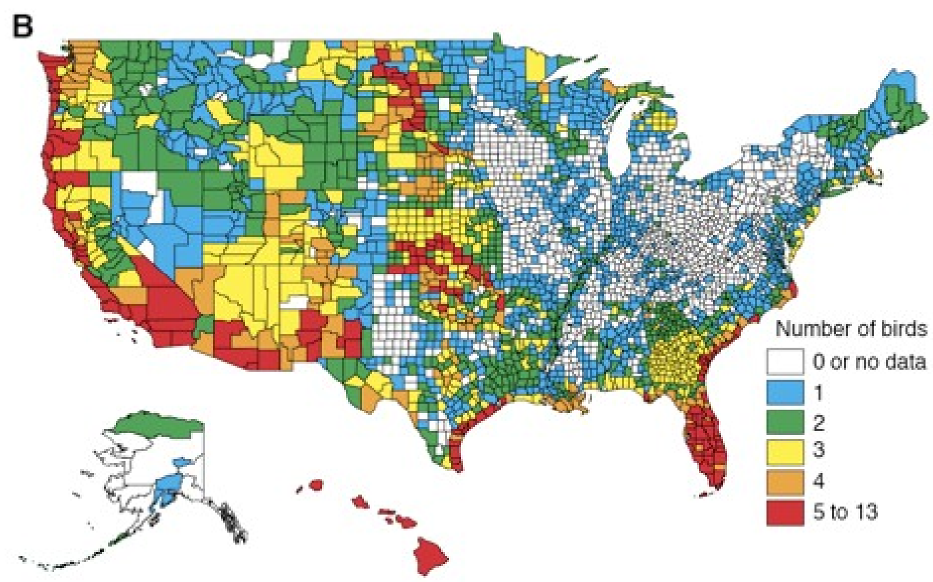

Figure 5.3b: The geographic distribution of endangered bird species in the United States. Source: Dobson et al. 1997

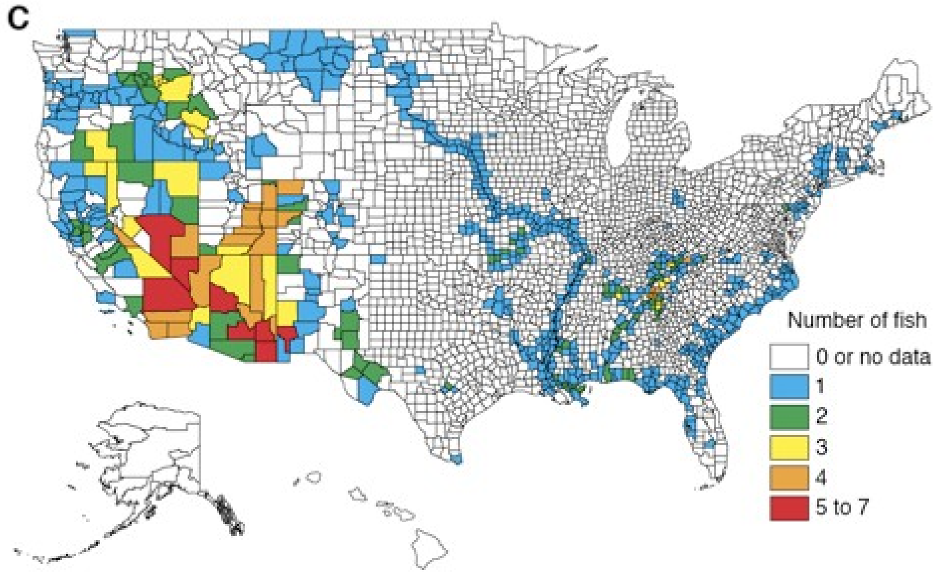

Figure 5.3c: The geographic distribution of endangered fish species in the United States. Source: Dobson et al. 1997

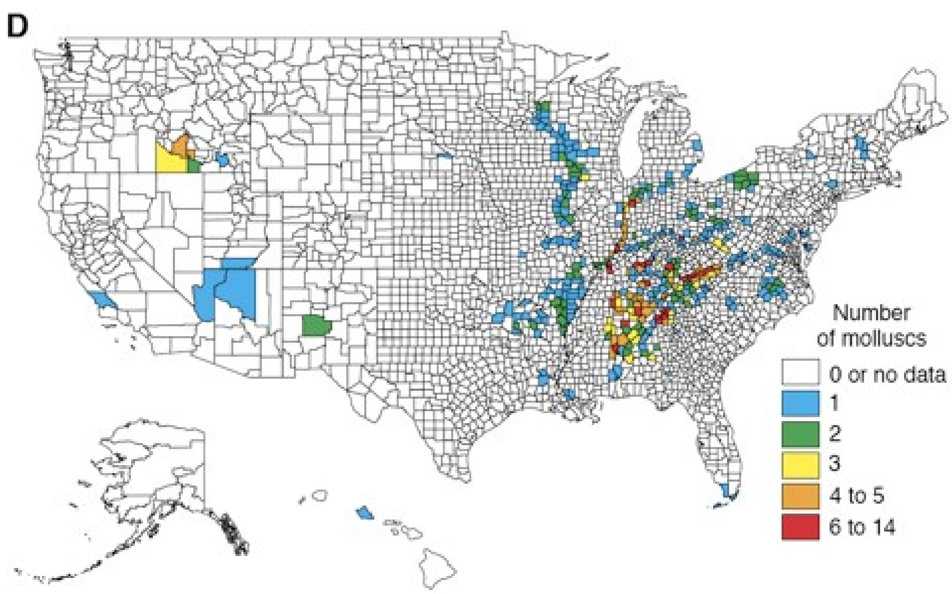

Figure 5.3d: The geographic distribution of endangered mollusc species in the United States. Source: Dobson et al. 1997

Male and Bean (2005) investigated whether the ESA was working to achieve significant results given available resources. They used data from recovery reports to the United States Congress covering the years 1988–2002 to analyze the relationship between species status and years since listing under the ESA. Using these reports, they examined the association between recovery progress and taxonomy, funding, distribution on islands, designation of critical habitat, and USFWS priorities and sought the degree to which those factors were correlated with species' declining, stable, improving or unknown status.

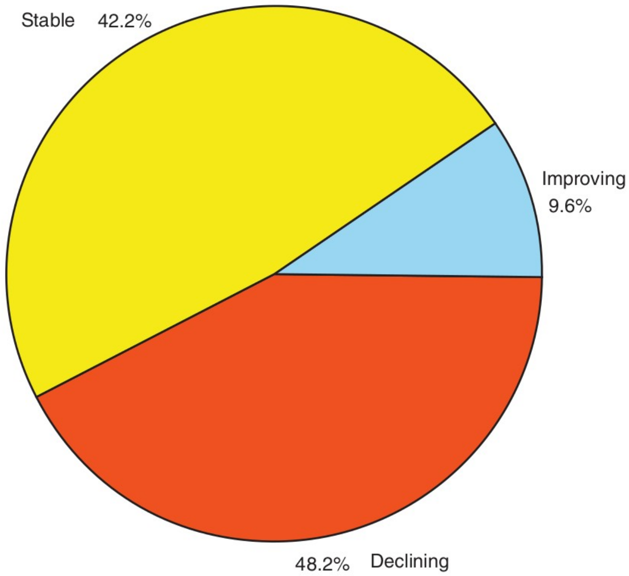

Figure 5.4: Slightly more than half of listed species were not declining or were consistently improving. Source: Male and Bean 2005

Overall they found that slightly more than half (52%) of the species examined showed repeated improvement, or were not declining over this period of time (Figure 5.4) (Male and Bean 2005).

About thirteen years after being listed, 68% of the species whose status was known were reported as having stable or improving status (Figures 5.5 and 5.6) (Male and Bean 2005). About 35% of species remained in decline. This finding suggests that many species protected by the ESA have made progress toward recovery (Male and Bean 2005).

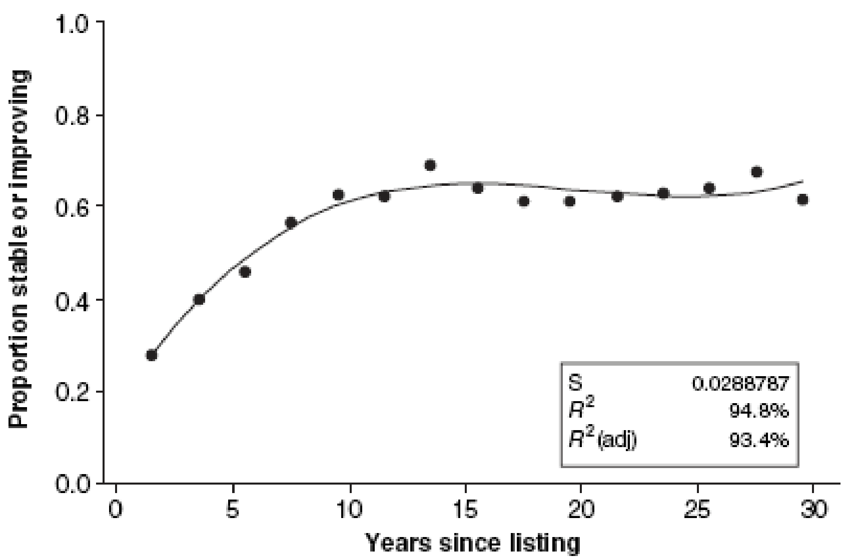

Figure 5.5: Proportion of stable or improving species by years since listing. Male and Bean 2005

Recovery progress was significantly correlated with taxonomy, funding by the USFWS and National Oceanic and Atmospheric Administration (NOAA), agency assessment of risk of extinction, and recovery potential (Male and Bean 2005).

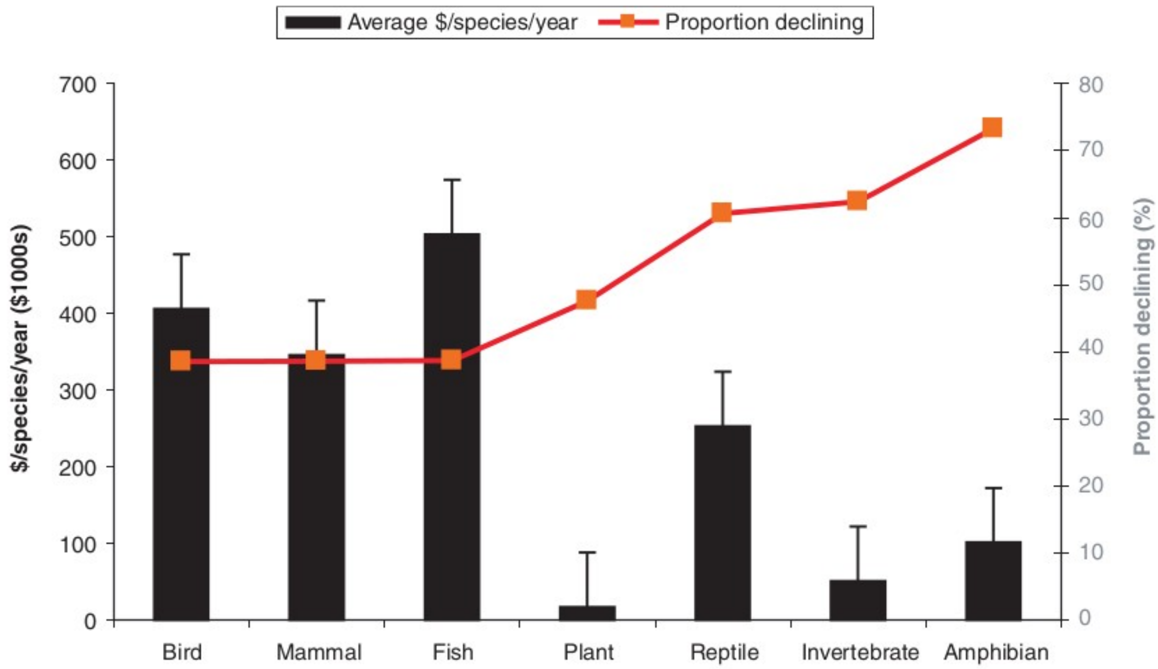

Figure 5.6: Proportion of species within a taxonomic group in decline and mean federal expenditures/species/year by taxonomic group. Source: Male and Bean 2005

SPENDING ON THE ESA

Federal spending is <$1,000 per species per year for about 275 of the listed species (Male and Bean 2005). Twenty species received 52% of USFWS ($641 million in 2005) and 69% ($2.0 billion in 2005) of NOAA funding (Male and Bean 2005). Four salmon species accounted for $806 million (36%) of the NOAA/federal agency expenditure. The bald eagle (*Haliaeetus leucocephalus*) consumed $63.1 million (8%) of all USFWS spending reported. The designation of critical habitat was not correlated with improved status (Male and Bean 2005).

Through this research, it becomes clear that endangered species status assessments provide a far more detailed picture of recovery progress than what is currently being used to inform the debates over the efficacy of the ESA. It is also clear that funding priorities are very important because funding does make a difference in recovery success. However, it can take many years to see progress so patience is needed. Further, climate change will increase both species risk and management uncertainty, requiring more intensive and controversial management strategies to prevent species from going extinct (Evans et al. 2016). Already we are seeing that ocean warming, linked to anthropogenic climate change, is having an impact on the ecology of marine species around the world. In particular, climate-driven changes in ocean circulation have altered the foraging environment and habitat use of North Atlantic right whales (*Eubalaena glacialis*), reducing the population's calving rate and exposing it to greater mortality risks from ship strikes and fishing gear entanglement (Meyer-Gutbord et al. 2021). Such a case exemplifies the increased threats to endangered species as a result of climate change, and lagging policy and economic support (e.g., financing the use of ropeless fishing gear for fishermen) to protect them.

SPECIES RECOVERY PLANS

One of three key provisions of the ESA are the recovery plans which are detailed programs for reducing the threat of extinction. These recovery plans are the only proactive part of the ESA law. Recovery criteria, the thresholds mandated by the ESA that define when species may be considered for downlisting or removal from the endangered species list, are a key component of conservation planning in the United States (Doak et al. 2015). Recovery plans for endangered or threatened species are designed in cooperation with a team of experts, not just by federal biologists.

Recovery plans have several parts. They include a review of biology, status of current populations, causes of endangerment, activities to support recovery, a schedule, and costs. Foin et al. (1998) detailed steps for improving recovery planning for threatened and endangered species. They analyzed 311 recovery plans to detect broad patterns that might increase a plan's value.

Specifically, they sought to place the management plan for each listed species into one of three categories of management intensity, ranging from lowest management intensity (habitat preservation), to greater effort (habitat restoration), to highest intensity (active management) (Table 5.3) (Foin et al. 1998). Habitat preservation ensures that adequate habitat is protected or set aside to allow for natural population recovery. Habitat preservation is appropriate in cases where species are exploited or killed (e.g., the gray wolf (*Canis lupus*), and American alligator (*Alligator mississippiensis*)). This form of management was found in 37% of recovery plans (Foin et al. 1998). Habitat restoration is suggested in cases where inadequate or poor habitat conditions exist. With improved habitat quality and quantity, natural population recovery can be expected using habitat restoration techniques. Habitat restoration is appropriate where degraded and damaged habitat exists (e.g., many plants, or desert pupfish (*Cyprinodon macularius*)), the habitat needs are clearly known, and restoration is practical to implement. This form of management was found in 21% of recovery plans (Foin et al. 1998). Active management is suggested in cases where the above two strategies are unlikely to reverse species decline. These plans often require persistent management to maintain conditions. Active management is appropriate in cases where competition exists from either invasive or native species (e.g., Delmarva Peninsula fox squirrel (*Sciurus niger cinereus*) which needs high trees with no gray squirrels (*Sciurus carolinensis*) present, Florida scrub jay (*Aphelocoma coerulescens*) which requires patchy, burned land with invasive species removed). This form of management was found in 42% of recovery plans (Foin et al. 1998).

Table 5.3: Classification of species into the three management categories. Source: Foin et al. 1998

Taxonomic group	Species listed as of 3 June 1994	Number of species covered by this analysis	Percentage of species covered by this analysis
Amphibians	9	7	78
Birds	73	62	85
Clams	40	30	75
Crustaceans	4	4	100
Fishes	63	32	51
Insects	16	13	81
Mammals	37	29	78
Plants	184	82	45
Reptiles	30	24	80
Snails	28	28	100
Total	484	311	64

Overall, it is clear from the research done by Foin et al. (1998) that habitat conservation is the dominant issue for species recovery. Habitat preservation alone will not solve most cases as more active management is needed for most species. Foin et al. (1998) warn that we must act quickly to implement recovery plans for most species since research often takes a long time to identify solutions.

Tear et al. (1993) analyzed 314 recovery plans to determine the validity of criticisms regarding the level of protection provided by the ESA. In particular, they sought to verify whether criticisms that recovery plans overprotect species and subpopulations are valid. They found that a common goal across recovery plans was to focus on a set population size. Surprisingly, 28% of the recovery plans actually specified set population levels lower than the population sizes that existed at the time of planning. Moreover, 37% of the plans specified the number of existing populations at or below levels that existed at the time of planning. In essence, 28-37% of species were being managed for extinction. Tear et al. (1993) concluded that more realistic goals were needed; specifically, they advised that policy should direct population size goals to achieve numbers which are higher than those that existed at the time a species was listed.

Doak et al. (2015) made recommendations for improving recovery criteria under the ESA. Specifically, they recommended improvements in the definition and scientific justification of recovery criteria, which addressed both data-rich and data-poor situations. Further, they recommended the use of quantitative population analyses to measure the impacts of threats, and that population status be explicitly tied to recovery criteria.

DISTINCT POPULATION SEGMENTS

In 1978, the ESA was amended to include "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature" (United States Government Publishing Office 1978; USFWS and NOAA 1996; Franklin). Notable here is the use of the phrase "distinct population segment" (DPS) as this expression is not used in science. Thus, available scientific information provides little to help in interpreting the actual meaning of DPS.

In policy however, a stock (e.g., a Pacific salmon (*Oncorhynchus Suckley*) "run") is considered a DPS if it represents an evolutionarily significant unit of a biological species. There are two criteria that must be met for a DPS to be considered an evolutionarily significant unit: 1) It must be substantially reproductively isolated from other conspecific population units; and 2) It must represent an important component in the evolutionary legacy of the species (NOAA 2021).

To be a DPS, a population or group of populations must meet two criteria: discreteness and significance (Waples et al. 2018). These criteria are identified by the following elements:

1) Regarding the discreteness of the population segment in relation to the remainder of the species or subspecies to which it belongs, a population unit can be considered discrete if it satisfies either of the following conditions: a) It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation; or b) It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist (Waples et al. 2018).

2) With respect to the significance of the population segment to the species or subspecies to which it belongs, the determination of significance may include: a) Persistence in an ecological setting that is unusual or unique for the taxon; b) Evidence that loss would result in a significant gap in the range of the taxon; c) Evidence that the DPS represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historic range; or d) Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics (Waples et al. 2018).

3) Regarding the population segment's conservation status in relation to the ESA's standards for listing (i.e., endangered or threatened?); if a population segment is deemed discrete and significant, then it meets the criteria for a DPS and is evaluated for endangered and threatened status.

In a study of 492 plants and animals listed or proposed for listing between 1985 and 1991, 20% of the taxa proposed or listed during this period were subspecies or populations rather than full species (18% subspecies, 2% populations) (Table 5.4) (Wilcove et al. 1993). The proportion of listings involving subspecies or populations differed markedly among taxa. In general, vertebrates represented a higher proportion of subspecies or populations than did other taxa. For example, 80% of the birds and 70% of the mammals that were proposed for listing or actually listed represented subspecies or populations compared with just 5% for mollusks and 14% for plants (Wilcove et al. 1993).

Table 5.4: Breakdown of United States plants and animals listed or proposed for listing under the Endangered Species Act, 1985-1991. Source: Wilcove et al. 1993

Taxonomic group	*n*	*Species*	*Subspecies*	*Populations*[1]	*% Subspecies*	*% Populations*
Mammals	23	7	16	0	70	0
Birds	15	3	8	4	53	27
Reptiles	10	6	2	2	20	20
Amphibians	3	3	0	0	0	0
Fishes	43	30	11	2	26	5
Arthropods	23	18	5	N.A.	22	N.A.
Mollusks	43	41	2	N.A.	5	N.A.
Plants	332	286	46	N.A.	14	N.A.
Total	492	394	90	8	18	2

[1] *Populations of invertebrate organisms and plants cannot be listed under the Endangered Species Act; only species and subspecies of plants and invertebrate animals are eligible for listing.*

CRITERIA FOR ENDANGERMENT AND RECOVERY

Shaffer (1981) provided an early statement of a population security goal for species conservation. He proposed that a minimum viable population (MVP) for any given species in any given habitat is the smallest isolated population having a 99% chance of remaining extant for 1000 years, despite the foreseeable effects of demographic, environmental, and genetic stochasticity, and natural catastrophes. The MVP goal was later restated as a 10% probability of extinction within 100 years as the highest acceptable risk (Mace and Lande 1991).

Formal MVP estimates take data and time to develop. Thomas (1990) reviewed MVP results, existing models, and other empirical data to provide MVP guidelines for use when needed in species conservation. He stated that a population size of 10 is too small; genetic variation will be lost rapidly, and demographic extinction is likely to be swift. For the same reasons, a population size of 100 is also too small since environmental variation and natural catastrophes could easily reduce numbers to a level from which the population cannot recover. A population size of 1000 may be adequate provided the habitat is stable and secure, and reproduction is well mixed across the population. A population size of 10,000 "should normally be sufficient to permit long-term demographic persistence and to satisfy genetic considerations" (Thomas 1990).

Mace and Lande (1991) recognized that categories of the types of threats a species may encounter (e.g., endangered, threatened, vulnerable, etc.) are widely used and have become important tools in species conservation, and yet the definitions associated with these terms are highly subjective. They proposed a system to redefine categories in terms of the probability of extinction within specific time periods based on the theory of extinction time for individual populations and on meaningful time scales for conservation action. They defined four desirable characteristics of a classification system: 1) The system should be simple, with few categories, and based on extinction probabilities; 2) It should be flexible in data requirements and able to use whatever data exists; 3) It should also be flexible in the population unit being considered; and 4) The terminology used in categorization should be appropriate and the various terms used should have a clear relationship to one another (Mace and Lande 1991).

Table 5.5: Partial decision analysis matrix showing an extinction risk analysis, based on expert judgment, by sub-population and threat category for Atlantic sturgeon (Acipenser oxyrinchus) sub-populations. Source: Patrick and Damon-Randall 2008

Sub-population	Distinct population segment	Factor A threats			
		Dams	Dredging	Water quality	Factor score
Penobscot	Gulf of Maine	2	2	3	3
Kennebec		1	3	3	3
Hudson	New York Bight	1	1	2	2
Delaware		1	3	3	3
York	Chesapeake Bay	1	1	2	2
James		1	3	3	3
Roanoke	Carolina	2	1	2	2
Tar/Pamlico		1	1	3	3
Neuse		3	1	3	3
Cape fear		4	3	3	4
Waccamaw		1	1	3	3
Pee Dee		1	1	3	3
Santee -Cooper		4	2	2	4
ACE Basin	South Atlantic	1	1	2	2
Savannah		2	3	2	3
Ogeechee		1	1	3	3
Altamaha		1	1	2	2
Satilla		1	1	2	2

Mace and Lande (1991) went on to propose the following categories of risk:
1) Vulnerable: 10% probability of extinction within 100 years. The 100 year time-span is considered workable for both planning purposes and for instances of urgency. This vulnerable designation is equivalent to the category of threatened under the ESA.

2) Endangered: 20% probability of extinction within 20 years or 10 generations, whichever is longer.
3) Critical: 50% probability of extinction within 5 years or 2 generations, whichever is longer.

Patrick and Damon-Randall (2008) created a framework based around five factors, which can be used to evaluate the status of data-poor species to determine extinction risk. They used a structured-decision approach for extinction risk assessment, which relied on expert judgment to assign a risk score to a species' probability of extinction. This method is especially useful when the species' life-history and population dynamics information are lacking. Their approach identified threats and organized them under the five factors specified in the ESA, as required for listing a species. The approach also identified populations or units of the species and made a decision analysis matrix of threats by population/unit. The cells of the decision analysis matrix were filled with scores, defined by experts, from 1 to 5 where:
1 = low risk (0–16% chance) of becoming endangered over the next 20 years.
2 = moderately low risk (17–33% chance) of becoming endangered over the next 20 years.
3 = moderate risk (34–50% chance) of becoming endangered over the next 20 years.
4 = moderately high risk; >50% chance of threats causing the sub-population to become endangered over the next 20 years, which means the sub-population should be considered threatened.
5 = high risk; >50% chance of threats causing the sub-population to become extinct over the next 20 years, which means the sub-population should be considered endangered.

Table 5.6: Overall risk score and recommendations for Atlantic sturgeon (Acipenser oxyrinchus) sub-populations extinction risk analysis. Source: Patrick and Damon-Randall 2008

Summary		
Overall risk score	Considered significant under SPOIR criteria	DPS Recommendation
3	No	Insufficient data
3	Yes	
3	Yes	Threatened
4	Yes	
3	No	Threatened
4	Yes	
3	Yes	Threatened
3	No	
3	No	
4	Yes	
3	No	
3	Yes	
4	Yes	
2	Yes	Insufficient data
3	No	
3	No	
3	Yes	
2	No	

Patrick and Damon-Randall (2008) used Atlantic sturgeon (*Acipenser oxyrinchus*) sub-populations as an example to show the results of an extinction risk analysis (Table 5.5).

Following the completion of a decision analysis matrix, discussions regarding the sub-populations which received a score of 4 or 5 in any of their threat categories begin. Team members discuss their rationale for the scores they've assigned, and are given an opportunity to adjust their recorded score based on those discussions (Patrick and Damon-Randall 2008). Median values for threats are condensed into one score for each factor using the median score, the highest score, or the elevated score due to the cumulative effects of individual threats. The final step is to consolidate scores across factors into an overall sub-population score and state conclusions (Table 5.6). The overall sub-population score is calculated using the highest of the five factor scores as the final sub-population score.

For the Atlantic sturgeon (*Acipenser oxyrinchus*) sub-populations example, the review team determined through their analysis that the 18 sub-populations should be grouped into five DPSs. The team discussed the scores of the sub-populations that made up each DPS and, consistent with ESA language, decided whether those sub-populations that had scores of 4 or 5 constituted a significant portion of the range of the DPS (SPOIR). The review team concluded that the Carolina, Chesapeake and New York Bight DPSs had sufficient data to recommend listing, and that each had a >50% chance of becoming endangered in the next 20 years. Therefore, the team recommended that these three DPSs be listed as threatened (Patrick and Damon-Randall 2008).

In some cases, recovery (delisting) is not attainable (Figure 5.7) (Scott et al. 2005). The recovery of a threatened or endangered species is often accompanied by the expectation that conservation management of the species will no longer be necessary. For many species the definition of recovery will need to include some form of active management. Recovery should be viewed as a continuum rather than a simple recovered vs not recovered condition.

Ongoing conservation management will require actions by state and local governments as well as private and governmental landowners. "Conservation-reliant species" can maintain self-sustaining wild populations with ongoing management actions (Scott et al. 2005). The criteria for assessing whether a species is conservation-reliant include:

1) Threats to the species' continued existence are known and treatable;
2) The threats are pervasive and recurrent (e. g., nest parasites, nonnative predators);
3) The threats render the species at risk of extinction, absent ongoing conservation management;
4) Management actions sufficient to counter threats have been identified and can be implemented (e. g., prescribed fires, restrictions on grazing or public access, predator or parasite control); and
5) Federal, state, or local governments are capable of carrying out the necessary management actions as long as necessary.

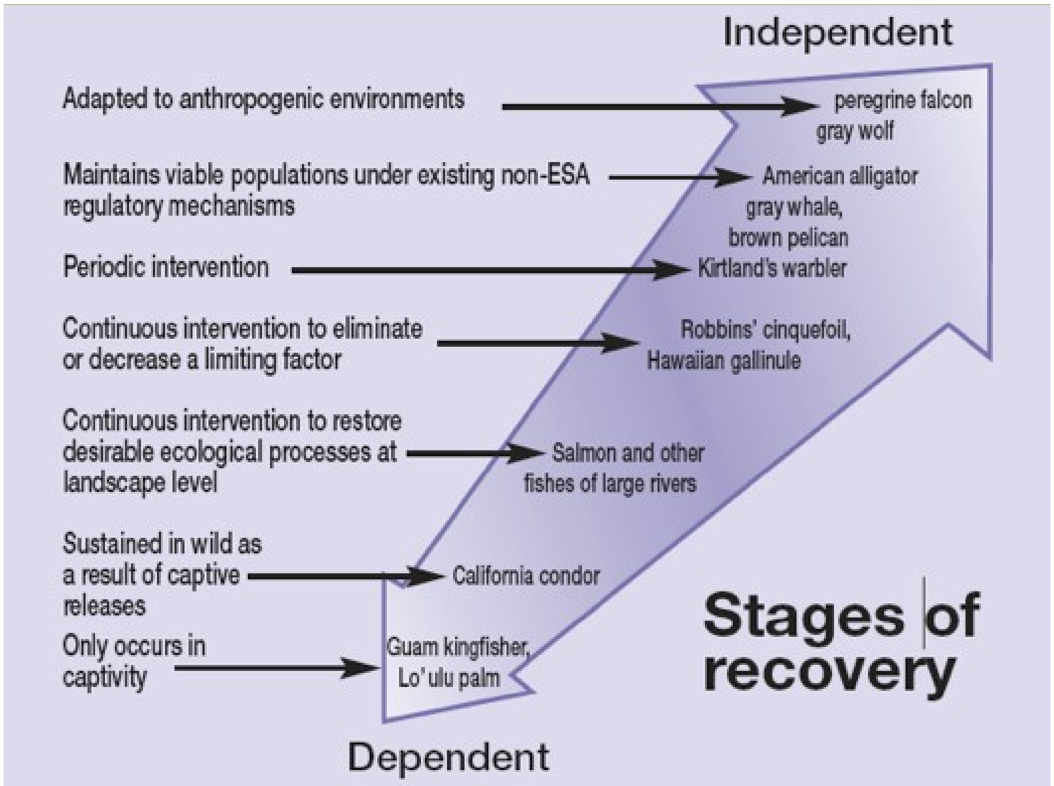

Figure 5.7: Stages of recovery of imperiled species under the Endangered Species Act. Source: Scott et al. 2005

CASE STUDY: SHORTNOSE STURGEON STATUS AND PATH TO RECOVERY

The shortnose sturgeon (*Acipenser brevirostrum*) is a diadromous fish species (Figure 5.8), with most populations living in large Atlantic coast rivers and estuaries along the east coast of North America (Kynard et al. 2016). Diadromous fish are migratory species that travel between fresh and salt water. There are no naturally land-locked populations, so all populations require access to fresh water and salt water to complete their natural life cycle (Kynard et al. 2016). River damming in the 19th and 20th Centuries extirpated some populations and caused other populations to become distinct, segmented populations (Kynard et al. 2016).

Figure 5.8: Shortnose sturgeon (Acipenser brevirostrum) held by D. Peterson at the Hudson River, New York. Source: M. B. Bain

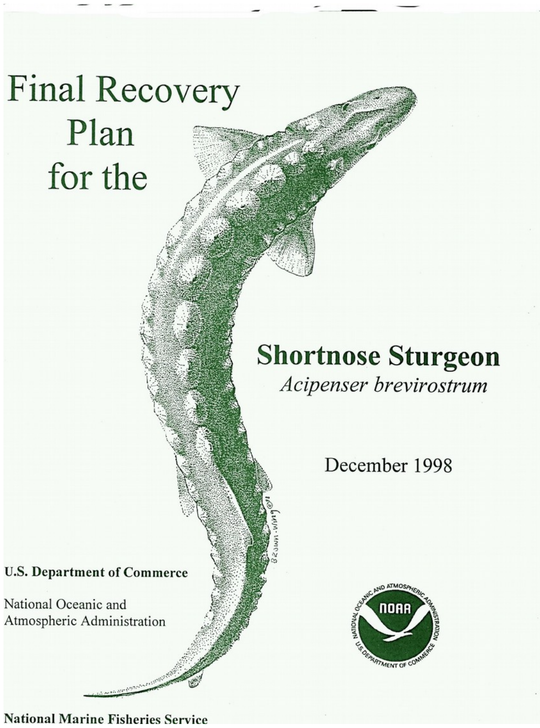

Figure 5.9: Source: Shortnose Sturgeon Recovery Team 1998

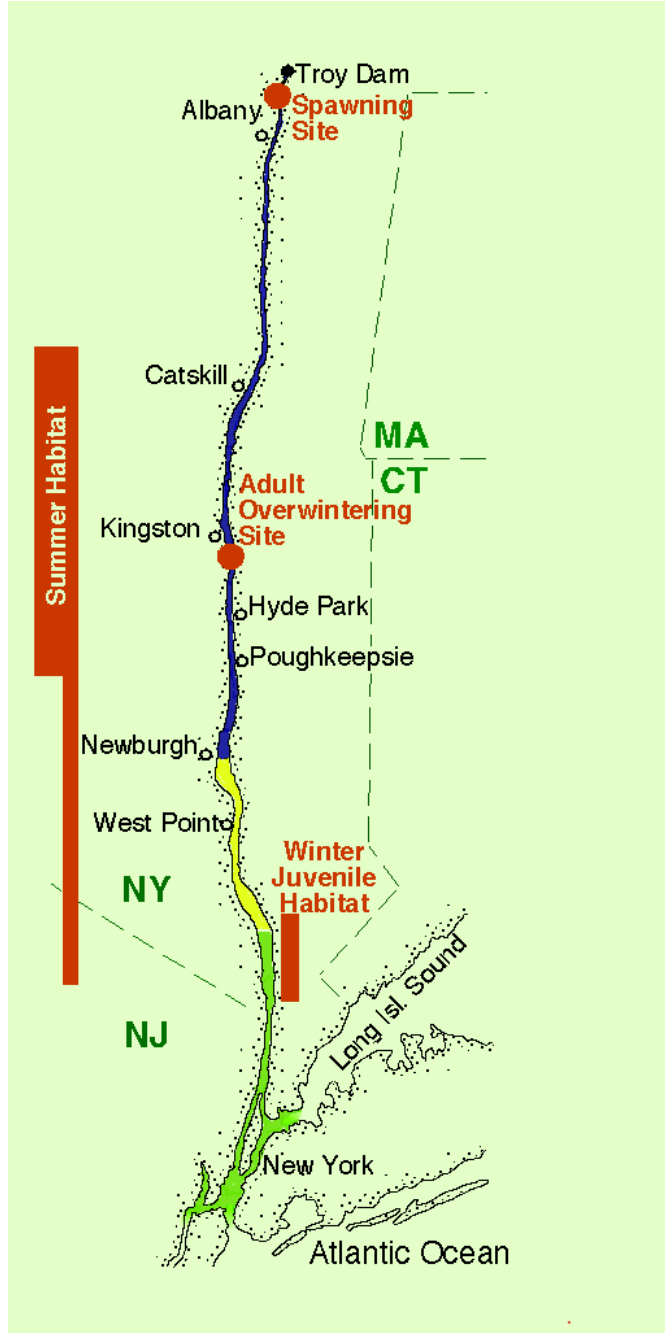

Figure 5.10: Map of the Hudson River estuary with key habitats used by shortnose sturgeon (Acipenser brevirostrum) and the salinity zones in the system. Summer habitat, winter juvenile habitat, the adult overwintering site, and the spawning site are shown. The width of the summer habitat designation corresponds with most and least heavily used sections of the river. Source: M. Bain

The shortnose sturgeon was formally protected with the passage of the 1968 United States Endangered Species Preservation Act and later designated endangered under the 1973 United States Endangered Species Act. In 1978 the ESA amended the listing for this species to include subspecies and distinct population segments. In 1987, the NMFS issued a shortnose sturgeon status review that suggested that the Androscoggin-Kennebec System supported only one population that may qualify for delisting. In 1994, Edwards Manufacturing Company petitioned the NMFS to delist the Androscoggin-Kennebec shortnose sturgeon population, citing an estimated population size of 11,000. The NMFS issued a finding in 1995 stating that the petition had merit and warranted a full review. The USFWS and NMFS issued a joint policy in 1996 on the criteria for determining distinct population segments. The NMFS issued a status review of the Androscoggin-Kennebec sturgeon, noting that there was an inadequate basis to conclude that there were two different populations.

Later in the year, the NMFS made a decision to deny the petition to delist the Androscoggin-Kennebec sturgeon stating that some threats did exist and that the actual population size was 7,222. The NMFS concluded that there may be two populations, but that there was an inadequate basis for declaring the distinction. In 1998, the NMFS published a final recovery plan detailing criteria and status of the species (Figure 5.9). In its report, the NMFS recommended that 19 rivers should be managed as distinct population segments based on the strong fidelity of shortnose sturgeon to their natal rivers. A Biological Assessment completed in 2010 reaffirmed this approach (Shortnose Sturgeon Status Review Team 2010). However, the NMFS has not formally listed DPSs under the ESA and the species remains listed as endangered range-wide in the USA (Kynard et al. 2016).

The population of shortnose sturgeon in the Hudson River is higher than in any other location along the east coast of the United States (Shortnose Sturgeon Recovery Team 1998). Shortnose sturgeon occupy the Hudson River estuary where habitats include a freshwater river channel, a low salinity fjord, and a brackish-water harbor (Figure 5.10). The 246 km Hudson River estuary is tidal and extends from New York City to the Troy Dam (upstream of Albany, New York) where the Hudson River is shallow, turbulent, and rises above sea level.

The availability and security of habitat is an important consideration in ESA listings. Random samplings from the Hudson River recorded that shortnose sturgeon were non-randomly distributed among several distinct river strata. Shortnose sturgeon were concentrated (63% of total fish catches) in the middle section of the estuary and were well represented (35% of catch) in habitats downstream to the point of persistently brackish waters (Figure 5.10) (Bain et al. 2007). The primary summer habitat for shortnose sturgeon is in the deep (regularly 13 to 42 m) tidal freshwater river channel used by commercial oil tankers and other sea ships (Figure 5.10) (Bain et al. 2007). Downstream, the estuary becomes brackish, deeper (regularly 18 to 48 m), and variable in width (Bain et al. 2007). The sections of the Hudson River primarily used by shortnose sturgeon have remained physically intact over the past century, with long-established shoreline land use that is composed of residential, historic, and some urban areas. The spawning site for shortnose sturgeon was removed from the other habitats because it was centered on turbulent river habitat between the head of tide and the Troy Dam. This section of the Hudson River was surrounded by urban areas and was immediately upstream from a river section heavily modified by industrial and shipping infrastructure.

From 1994 through 1997, gill net sampling was conducted for mark-and-recapture population estimates and a shortnose sturgeon distribution analysis (Bain et al. 2007). Sampling and marking were completed in two ways: 1) Random sampling was conducted from mid-May through early October throughout the river when the shortnose sturgeon were feeding and widely distributed; and 2) Targeted sampling of adult shortnose sturgeon, at a previously established (Klauda et al. 1988; Applied Science Associates 1999) overwintering site, was conducted in December, March, and early April, and at the spawning area from mid-April through May. Shortnose sturgeon were marked with internal (passive integrated transponder) tags and data were collected on fish length and weight (Bain et al. 2007).

In total, 6,265 individual shortnose sturgeon were captured and 5,959 of these fish were marked (Bain et al. 2007). Most (3,836) shortnose sturgeon were captured and marked at the overwintering site, high numbers (1,937) were captured and marked at the spawning site in spring, and relatively few (492) sturgeon were captured and marked in the summer random sampling that covered the estuary. From 1995 through 1997, 269 marked sturgeon were recaptured. The shortnose sturgeon captured during the targeted sampling were adults, while the summer random sampling captured a broader size range of shortnose sturgeon including some juveniles (Bain et al. 2007).

Using nine targeted sampling periods, a closed population estimate of the adults (Krebs 1989) yielded 56,708 fish with a narrow 95% confidence interval: 50,862 – 64,072 from the 1994-1997 study (Figure 5.11) (Bain et al. 2007). The mark-and-recapture estimator matched the method used in 1979 and 1980 which estimated the number of adult shortnose sturgeon at 12,669 and 13,844, respectively (Klauda et al. 1988; Dovel et al. 1992; Smith 1992; Applied Science Associates 1999). Comparing the 1994-1997 population abundance with estimates from 1979 and 1980, the Hudson River population has increased by more than 400%. Independent data from the Hudson River electric utilities trawl survey reflects approximately a 450% increase in average catch rate of mainly adult shortnose sturgeon from the 1980s to 1990s (Klauda et al. 1988; Applied Science Associates 1999).

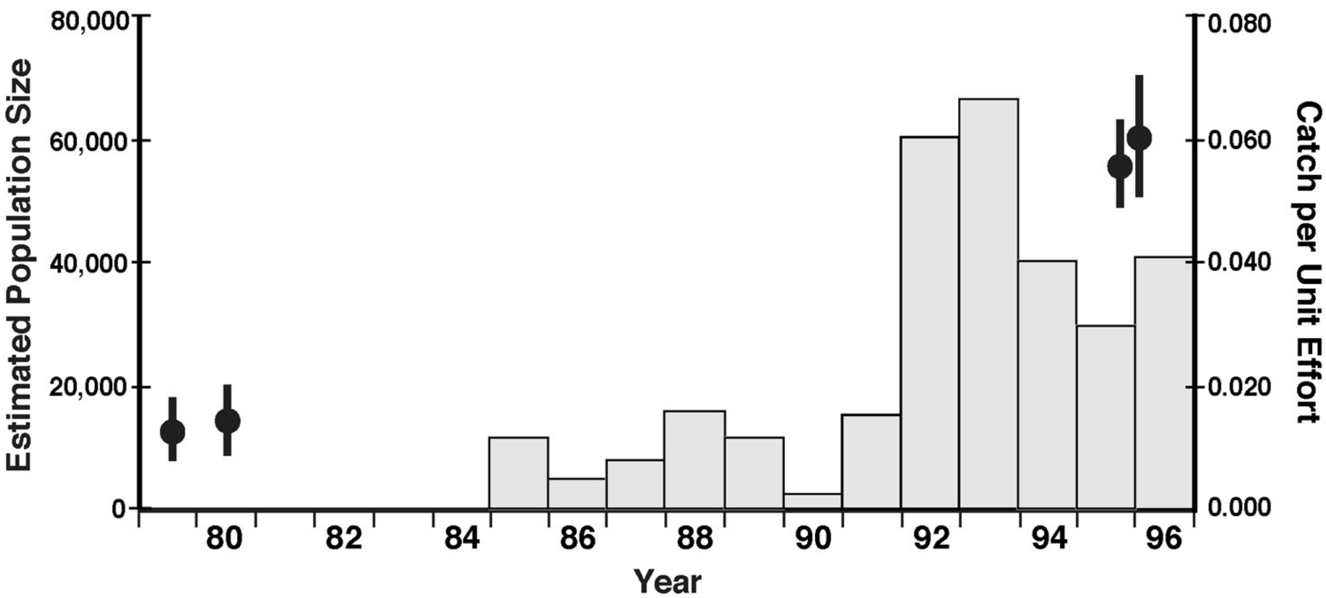

Figure 5.11: Population estimates and abundance trend for Hudson River shortnose sturgeon in the 1980s and 1990s. The paired symbols of circles (means) and heavy lines (95% confidence intervals) show the results of population estimates in the late 1970s and late 1990s. The catch per unit effort histogram bars are the average catch of shortnose sturgeon per trawl haul in a riverwide fish survey conducted annually by the Hudson River electric utilities. Source: Bain et al. 2007

The shortnose sturgeon recovery plan (Shortnose Sturgeon Recovery Team 1998) specifies three evaluation criteria: 1) A population of adequate size with a favorable trend in abundance; 2) Habitat that could sufficiently support a recovered population; and 3) Potential causes of mortality which would be insufficient to reduce the population size.

The number of sturgeon marked during the 1994-1997 study exceeded the estimated size of most other populations of shortnose sturgeon, and the population estimates were larger than the sum of all other estimated populations (seven other significant river populations) (Bain et al. 2007). Therefore, it was safe to conclude that the Hudson River supports the largest population of shortnose sturgeon, and that this system may harbor most of the individuals of this species.

A shortnose sturgeon population composed of 10,000 spawning adults has been considered large enough to be at a low risk of extinction (NOAA 1996a) and of adequate size for delisting under the ESA (NOAA 1996a, NOAA 1996b*)*. Both the total and spawning population estimates in Bain et al. (2007) exceeded this threshold by a wide margin (≥500%), clearly indicating the recovery of the Hudson River shortnose sturgeon population. The Hudson River fish monitoring and the population estimates calculated over time, indicate a positive trend in shortnose sturgeon population abundance since the 1970s. These data, including size structure and condition, suggest that the population of shortnose sturgeon in the estuary is healthy. Further, shortnose sturgeon habitat use in the Hudson River is well understood and unlikely to be physically changed, water quality is closely monitored and regulated, and the habitats themselves have remained intact enough to support the growth of shortnose sturgeon into a considerably larger population. Future causes of high mortality such as unregulated harvest, bycatch in active fisheries, and pollution stress have been and can be controlled through established fishery management and water quality regulations. Finally, non-government conservation groups in the area are engaged and well funded (Haley et al. 1996).

The National Marine Fisheries Service is the responsible federal agency for planning and implementing recovery of shortnose sturgeon under the ESA. Their approach to species recovery in the Hudson River had been to minimize interference with natural population processes, to avert habitat disruption (e.g., channel dredging, open water disposal of dredged material, and bridge construction and demolition) and direct harm to individuals by capture, handling, and disturbance. Unlike a recovery strategy based on augmenting population size through stocking or active restoration, the Hudson River shortnose sturgeon population was managed for growth within protected habitat over a long period of time (approximately 30 years). The patient and natural approach to fish species recovery succeeded for the shortnose sturgeon in the Hudson River despite the intense human use and occupation of the river and its surroundings.

SUMMARY

The ESA program was enacted to protect and recover imperiled species and the ecosystems upon which they depend. Many species protected by the ESA have made progress toward recovery though patience is needed as it takes many years to see progress. Funding has been demonstrated to make a difference in recovery success. In the future climate change will increase both species risk and management uncertainty, requiring more intensive and controversial management strategies to prevent species from going extinct.

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Biologically-Focused Techniques

Chapter 6 - Biomonitoring

The last chapter in the biologically-focused techniques group centers on the idea of using biological organisms to assess the biological integrity of a region. This method is called biomonitoring.. The scientific principles underpinning biomonitoring have been applied to a variety of environments. This technique relies on the biological community to indicate problems and needs, and it is well developed for implementation in management. In this chapter, we will review the background and reasons for implementing biomonitoring, how it works, and will end with a case study on the New York biomonitoring program.

BACKGROUND ON BIOMONITORING

Historically, policy and management goals have focused on reducing point source pollution which is the pollution that can be traced to a specific location (e.g., end-of-a-pipe). Standards have been used in the past to set limits on amounts of pollutants but there are problems with the use of standards. First, standards are not always linked with ecosystem health. Second, the goal of environmental management is to protect natural ecosystems; standards are not focused on that goal. Thus, the focus of environmental protection has broadened from the sole use of standards to restoring self-maintaining ecosystems.

One objective of the Clean Water Act (CWA) is restoring and maintaining the "...chemical, physical, and biological integrity of the Nation's waters…" (United States Code, 2021). The biological integrity mandate of the CWA depends on an overview of the entire water system, not just on water quality which historically was the focus. The biological integrity objective (Figure 6.1) encompasses all factors affecting the ecosystem and is defined as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Karr 1991). Thus, biomonitoring programs help to view rivers as living systems and to use accessible biological organisms in management programs.

The United States Environmental Protection Agency (USEPA) began developing new approaches for biomonitoring with James Karr in the 1970s. After seven years of study in creeks and rivers in Indiana and Illinois, Karr published his first article demonstrating the assessment of biotic integrity using fish communities (Karr 1981). His article emphasized the ability to sustain a balanced biotic community as this is one of the best indicators of the potential for the beneficial use of waterbodies (Karr 1981). Though Karr was trained as an ornithologist, he used fish as the biotic community to link fauna to biotic integrity. His work ultimately became the dominant concept for biomonitoring in the United States.

WHY PERFORM BIOMONITORING

Initially, there was limited use of integrative biological techniques to protect water resources. This was due to a number of reasons: 1) Dominance of the reductionist viewpoint; 2) Lack of interdisciplinary breadth; 3) Inability to deal with the whole system; 4) Limited legal and regulatory programs; 5) Fragmented responsibility (e.g., federal, state, local); 6) Few tools to evaluate regulatory programs; 7) Numerical pollution standards were clear and could be justified; and 8) Non-point source pollution was hard to link to water changes directly.

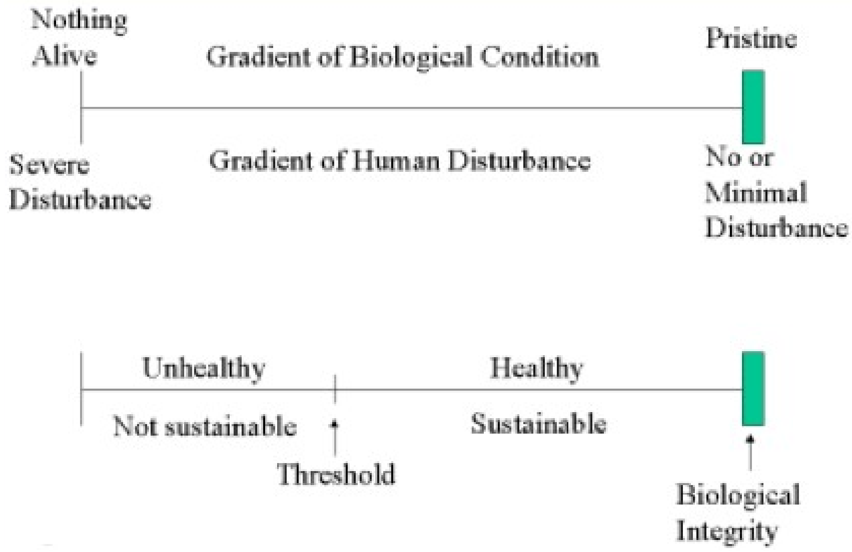

Figure 6.1: Visualizing the basis of biological integrity. Source: Karr and Chu 1999

In his paper, Karr (1981) asserts that by carefully monitoring fishes, one can rapidly and inexpensively assess the health of a local water resource, and that this process can serve as an exploratory assessment of overall water resource quality. Where impaired use is suggested by biological monitoring, a more nearly complete monitoring program can be implemented in search of the cause of the impairment (Karr 1981).

HOW BIOMONITORING WORKS

Biomonitoring (Figure 6.2) is performed using indices to assess biological integrity (called an Index of Biotic Integrity or IBI). These IBIs were developed by correlating metrics related to the biological organism being assessed (e.g., fish) with ecological health. IBIs must be able to capture the range of conditions indicating integrity. Metrics of importance vary from place to place, so IBIs are often developed for specific regions, though field methods are typically standardized.

Figure 6.2: Biomonitoring. Source: Vermont Department of Environmental Conservation 2021

Unlike other management techniques that use standards, where effluent limits are enforceable but waterways remain impaired in most cases, biomonitoring processes link field data to enforceable actions.

The fundamental objective of biomonitoring is to describe the effects of human activities on the structure and function of ecosystems and their biota (Stoddard et al. 2006).

Most biological assessments are based, either directly or indirectly, on the concept of comparing factors of current condition to natural conditions (structure, composition, function, or diversity) in the absence of human disturbance or alteration (i.e., comparison to a pristine, unpolluted, or anthropogenically undisturbed state). This natural condition is termed the reference condition.

DEFINING THE REFERENCE CONDITION

Several methods for estimating natural conditions have been developed for use as reference against collected data (Stoddard et al. 2006). The reference-site approach is by far the most common. In this approach, one would quantify the biological condition at a set of sites that are either minimally or least disturbed by human activity and these would comprise the reference (or natural) sites (Davies and Jackson 2006). Minimally disturbed condition sites are places that have escaped all but the broadest scale human disturbances and are often protected areas or forested landscapes with remnants of late-stage/old-growth watersheds, or landscapes that have substantially recovered from past disturbances (Stoddard et al. 2006). Long-term climatic, geologic, and ecological fluctuations will inevitably change the characteristics of individual sites in this group, but the range of minimally disturbed conditions should be nearly invariant, and its distribution can serve as an anchor by which to judge current condition (Stoddard et al. 2006). Least disturbed condition sites are those which have comparatively less human disturbance than other sites (Stoddard et al. 2006). These constitute the best available physical, chemical, and biological habitat conditions given the current state of the landscape. Least disturbed condition sites are ideally described by evaluating data collected at sites selected according to a set of explicit criteria defining what is "best" (i.e., those least disturbed by human activities) (Stoddard et al. 2006). The best attainable condition is equivalent to the expected condition of least disturbed condition sites if the best possible management practices were in use for some period of time. The preferred approach for estimating either the minimally disturbed condition or least disturbed condition is to use a set of criteria for site selection that exclude data on resident biota to avoid circularity. The structure of the biotic assemblage itself should not be used to classify sites as either reference or non-reference because it is important to avoid any preconceived notions about the structure and composition of biotic assemblages at reference sites (Stoddard et al. 2006).

Other methods for estimating natural or reference conditions include: 1) Interpreting historical conditions; 2) Extrapolating from empirical models; and 3) Best professional judgment. In some cases it is possible to interpret the historical condition of a site by examining documents from earlier times or through museum collections, journals, records written by early explorers, land survey notes, or early photographs (Stoddard et al. 2006). The historical condition can be an accurate estimator of natural conditions if the historical point chosen is before the start of any human disturbance. However, many other historical reference points are possible (e.g., pre-industrial, pre-Columbian). In North America, a historical period that includes the impacts of indigenous peoples, but excludes the impacts of European immigrants, has been labeled the pre-Columbian benchmark (Stoddard et al. 2006).

In situations without minimally disturbed condition sites, empirical models derived from associations between biological indicators and human-disturbance gradients can be extrapolated to infer conditions in the absence of humans (Stoddard et al. 2006). Biologists with decades of experience sampling and examining physical, chemical, and biological attributes across wide ranges of severity and types of human disturbance, may develop an empirical understanding of conditions in the absence of significant human disturbance. These judgments would constitute best professional judgment (Stoddard et al. 2006).

BIOTIC INDEXING

Rating community samples using species "tolerances" to pollution seems to have started with William Hilsenhoff, a Wisconsin state biologist, at the start of the 1980s (Hilsenhoff 1982). The system has become known as the "Hilsenoff index" or biotic index. An updated version of the biotic index was released by Hilsenhoff (1988) using family-level stream macroinvertebrates to identify the quality of stream habitat. The biotic index process involves obtaining a representative sample of stream organisms, separating the organisms by taxa, multiplying the abundance of each taxon by its pollution tolerance value to obtain a score for each taxa, summing the scores, and then dividing the final score by the number of organisms (Figure 6.3).

The resulting family biotic index value can be used to determine an overall water quality rating from very poor to excellent (Table 6.1).

PLECOPTERA—Capniidae 1, Chloroperlidae 1, Leuctridae 0, Nemouridae 2, Perlidae 1, Perlodidae 2, Pteronarcyidae 0, Taeniopterygidae 2
EPHEMEROPTERA—Baetidae 4, Baetiscidae 3, Caenidae 7, Ephemerellidae 1, Ephemeridae 4, Heptageniidae 4, Leptophlebiidae 2, Metretopodidae 2, Oligoneuriidae 2, Polymitarcyidae 2, Potomanthidae 4, Siphlonuridae 7, Tricorythidae 4
ODONATA—Aeshnidae 3, Calopterygidae 5, Coenagrionidae 9, Cordulegastridae 3, Corduliidae 5, Gomphidae 1, Lestidae 9, Libellulidae 9, Macromiidae 3
TRICHOPTERA—Brachycentridae 1, Glossosomatidae 0, Helicopsychidae 3, Hydropsychidae 4, Hydroptilidae 4, Lepidostomatidae 1, Leptoceridae 4, Limnephilidae 4, Molannidae 6, Odontoceridae 0, Philopotamidae 3, Phryganeidae 4, Polycentropodidae 6, Psychomyiidae 2, Rhyacophilidae 0, Sericostomatidae 3
MEGALOPTERA—Corydalidae 0, Sialidae 4
LEPIDOPTERA—Pyralidae 5
COLEOPTERA—Dryopidae 5, Elmidae 4, Psephenidae 4
DIPTERA—Athericidae 2, Blephariceridae 0, Ceratopogonidae 6, Blood-red Chironomidae (Chironomini) 8, other (including pink) Chironomidae 6, Dolochopodidae 4, Empididae 6, Ephydridae 6, Psychodidae 10, Simuliidae 6, Muscidae 6, Syrphidae 10, Tabanidae 6, Tipulidae 3
AMPHIPODA—Gammaridae 4, Talitridae 8
ISOPODA—Asellidae 8

Figure 6.3: Example tolerance values for arthropod families. Source: Hilsenhoff 1988

Table 6.1: Evaluation of water quality using family level biotic index. Source: Hilsenhoff 1988

Family Biotic Index	Water Quality	Degree of Organic Pollution
0.00–3.75	Excellent	Organic pollution unlikely
3.76–4.25	Very good	Possible slight organic pollution
4.26–5.00	Good	Some organic pollution probable
5.01–5.75	Fair	Fairly substantial pollution likely
5.76–6.50	Fairly poor	Substantial pollution likely
6.51–7.25	Poor	Very substantial pollution likely
7.26–10.00	Very poor	Severe organic pollution likely

The advantages of the biotic index are that it is rapid, low cost, and easy to implement, and it provides information on the health of the site. The index is based on biological measurements which provide a broad perspective of the quality of the waterbody from which they were obtained. The ratings and scores were developed based on abundant field experience by expert biologists.

INDEX OF BIOTIC INTEGRITY (IBI)

The IBI was conceived to provide a broadly based and ecologically sound tool to evaluate biological conditions in flowing waters (Karr 1991). Research has successfully linked macroinvertebrate community impairment, identified through biomoniotring protocols such as the IBI, to catchment characteristics in streams (Kennen 1999). IBIs are now widely used by states, provinces, and federal agencies (e.g., United States Environmental Protection Agency) for status assessment and monitoring.

While first used for fish (Karr 1991), the IBI has an ecological foundation that allows for its use with any taxa group, and it can be scaled up for use with multiple taxa (e.g., IBIs exist for birds (Alexandrino et al. 2017), bacteria (Li et al. 2017), phytoplankton (Ren et al. 2017), and other organisms). It uses a set of metrics that represent populations, communities, and ecosystems, and a scoring system that ranges from 1 to 5 with 1 indicating strong deviation from the reference condition, 3 denoting moderate deviation, and 5 representing a condition close to the reference condition. An innovation of the IBI is that the values used for metrics are based on comparison to a regional reference condition with little or no human influence.

The IBI is implemented by sampling a site for organisms (e.g., fish), assessing the sample in terms of the IBI metrics, giving a rating score for each metric (e.g., 5, 3 or 1; Table 6.2), summing the scores across the metrics to obtain a total IBI score, and then assessing the integrity class of the site (e.g., very poor to excellent) based on the total IBI score (Table 6.3).

Table 6.2: Metrics used to assess the biological integrity of fish communities based on an Index of Biotic Integrity (IBI). Ratings of 5, 3 and 1 are assigned to each metric according to whether its value approximates, deviates from, or strongly deviates from the value expected at a comparable site that is relatively undisturbed. Source: Karr 1991

Metrics	Rating of metric* 5	3	1
Species richness and composition			
1. Total number of fish species* (native fish species)†	Expectations for metrics 1–5 vary with stream size and region.		
2. Number and identity of darter species (benthic species)			
3. Number and identity of sunfish species (water-column species)			
4. Number and identity of sucker species (long-lived species)			
5. Number and identity of intolerant species			
6. Percentage of individuals as green sunfish (tolerant species)	<5	5–20	>20
Trophic composition			
7. Percentage of individuals as omnivores	<20	20–45	>45
8. Percentage of individuals as insectivorous cyprinids (insectivores)	>45	45–20	<20
9. Percentage of individuals as piscivores (top carnivores)	>5	5–1	<1
Fish abundance and condition			
10. Number of individuals in sample	Expectations for metric 10 vary with stream size and other factors.		
11. Percentage of individuals as hybrids (exotics, or simple lithophils)	0	>0–1	>1
12. Percentage of individuals with disease, tumors, fin damage, and skeletal anomalies	0–2	>2–5	>5

* Original IBI metrics for midwest United States.
† Generalized IBI metrics (see Miller et al. 1988).

Table 6.3: Total Index of Biotic Integrity (IBI) scores, integrity classes, and the attributes of those classes. Source: Karr 1991

Total IBI score (sum of the 12 metric ratings)*	Integrity class of site	Attributes
58–60	Excellent	Comparable to the best situations without human disturbance; all regionally expected species for the habitat and stream size, including the most intolerant forms, are present with a full array of age (size) classes; balanced trophic structure.
48–52	Good	Species richness somewhat below expectation, especially due to the loss of the most intolerant forms; some species are present with less than optimal abundances or size distributions; trophic structure shows some signs of stress.
40–44	Fair	Signs of additional deterioration include loss of intolerant forms, fewer species, highly skewed trophic structure (e.g., increasing frequency of omnivores and green sunfish or other tolerant species); older age classes of top predators may be rare.
28–34	Poor	Dominated by omnivores, tolerant forms, and habitat generalists; few top carnivores; growth rates and condition factors commonly depressed; hybrids and diseased fish often present.
12–22	Very poor	Few fish present, mostly introduced or tolerant forms; hybrids common; disease, parasites, fin damage, and other anomalies regular.
···†	No fish	Repeated sampling finds no fish.

* Sites with values between classes assigned to appropriate integrity class following careful consideration of individual criteria/metrics by informed biologists.

† No score can be calculated where no fish were found.

The main advancements of the IBI are that it provides a quantitative measure of human disturbance, incorporates attributes of biological systems, leverages professional judgment in an ecologically sound manner, is inexpensive, simple, and highly sensitive to ecosystem changes, translates ecological knowledge into statements about the health of ecosystems, uses categories of metrics to span changes expected under stress, and involves minimal information loss in its use of metrics.

CASE STUDY: NEW YORK STATE BIOMONITORING

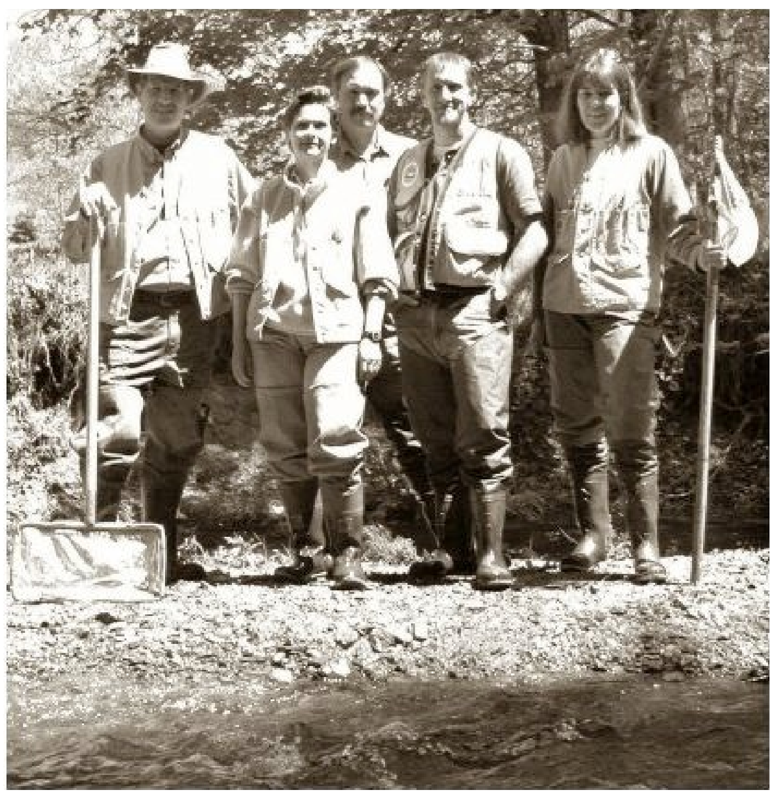

Figure 6.4: The New York State Stream Biomonitoring Unit in 2002. Left to right: Robert Bode, Margaret Novak, Lawrence Abele, Alexander Smith, and Diana Heitzman. Source: M. Bain

A biological monitoring program was initiated in New York State in May 1972 as mandated by the Federal Water Pollution Control Act Amendments of 1972 (Figure 6.4) (Public Law 92-500; United States Fish and Wildlife Service 2021). The main objective of the project was to evaluate the relative biological health of the State's streams and rivers through the collection and analysis of macroinvertebrate organisms.

Biological data from sites within the Rotating Intensive Basin Survey (RIBS) basins are combined with chemical data to make overall water quality assessments of the sites (Figure 6.5).

Stream Miles	
Total miles *(determined using a state based program)*	**52,337**
Total perennial miles	46,266
Total miles assessed for biology*	**16,000**
fully supporting for 305(b)	15,430
partially/non-supporting for 305(b)	570
listed for 303(d)	484
number of sites sampled	800
number of miles assessed per site	20

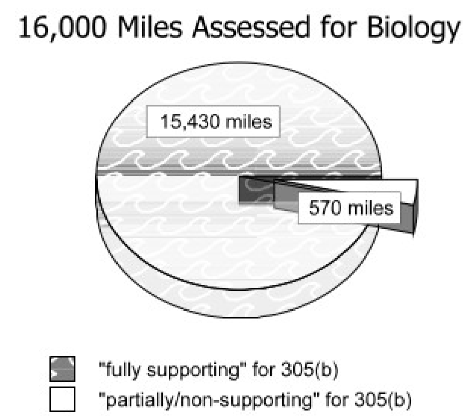

Figure 6.5: New York State stream miles assessed. Source: United States Environmental Protection Agency 2002

Figure 6.6: Dip net. Source: Maryland Department of the Environment 2021

Sampling methods involve the use of dip nets with >800 micron mesh (Figure 6.6), hester-dendy multiplate samplers (Figure 6.7), and ponar grab samplers (Figure 6.8) depending on the depth of the water.

Figure 6.7: Hester dendy multiplate sampler. Source: M. Meixler

Figure 6.8: Ponar grab sampler. Source: United States Geological Survey 2004

Sampled data can be analyzed using a variety of indices (Figure 6.9). Four of the most popular are:

1) Species richness. This is the total number of species or taxa found in the sample. Expected ranges for 100-organism subsamples of dip net samples in most streams in New York State are:
> 26 = non-impacted
19-26 = slightly impacted
11-18 = moderately impacted
< 11 = severely impacted

2) Ephemeroptera, Plecoptera, Trichoptera (EPT) value. EPT denotes the total number of species of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) found in an average 100-organism subsample. These are considered to be mostly clean-water organisms, and their presence is generally correlated with good water quality (Lenat 1987). Expected ranges from most streams in New York State are:
>10 = non-impacted
6-10 = slightly impacted
2-5 = moderately impacted
0-1 = severely impacted

3) Biotic index. The Hilsenhoff Biotic Index is a measure of the tolerance of the organisms in the sample to organic pollution (e.g., sewage effluent, animal wastes) and low dissolved oxygen levels. It is calculated by multiplying the number of individuals of each species by its assigned pollution tolerance value, summing these products, and dividing by the total number of individuals. On a 0-10 scale, tolerance values range from intolerant (0) to tolerant (10). For purposes of characterizing an organism's pollution tolerance, intolerant = 0-4, facultative = 5-7, and tolerant = 8-10 (Hilsenhoff 1987); additional values are assigned to organisms by the New York State Stream Biomonitoring Unit (Bode et al. 1996). Ranges for the levels of impact are:
0-4.50 = non-impacted
4.51-6.50 = slightly impacted
6.51-8.50 = moderately impacted
8.51-10.00 = severely impacted

4) Percent Model Affinity. Percent Model Affinity is a measure of similarity to a model non-impacted community based on percent abundance in seven major groups (Novak and Bode 1992). This index provides water quality information not entirely contained in other indices. It is based on the concept that the biological effects of pollutants can be measured by comparing the existing community with an expected community, a practice that many biologists carry out intuitively. Percent affinity is used to measure similarity to a model community of 40% Ephemeroptera, 5% Plecoptera, 10% Trichoptera, 10% Coleoptera, 20% Chironomidae, 5% Oligochaeta, and 10% Other. Ranges for the levels of impact are:
> 64 = non-impacted
50-64 = slightly impacted
35-49 = moderately impacted
< 35 = severely impacted

Each of the indices provides a quality rating of non-impacted, slightly impacted, moderately impacted or severely impacted (Table 6.4). These can be interpreted as follows:

Non-impacted: Indices reflect very good water quality. The macroinvertebrate community is diverse, usually with at least 27 species in riffle habitats. Mayflies, stoneflies, and caddisflies are well represented; the EPT value is greater than 10. The biotic index value is 4.50 or less. Percent model affinity is greater than 64. Water quality should not be limiting to fish survival or propagation. This level of water quality includes both pristine habitats and those receiving discharges which minimally alter the biota.

Slightly impacted: Indices reflect good water quality. The macroinvertebrate community is slightly but not significantly altered from the pristine state. Species richness usually is 19-26. Mayflies and stoneflies may be restricted, with EPT values of 6-10. The biotic index value is 4.51-6.50. Percent model affinity is 50-64. Water quality is usually not limiting to fish survival, but may be limiting to fish propagation.

Moderately impacted: Indices reflect poor water quality. The macroinvertebrate community is altered to a large degree from the pristine state. Species richness usually is 11-18 species. Mayflies and stoneflies are rare or absent, and caddisflies are often restricted; the EPT value is 2-5. The biotic index value is 6.51-8.50. The percent model affinity value is 35-49. Water quality is often limiting to fish propagation, but usually not to fish survival.

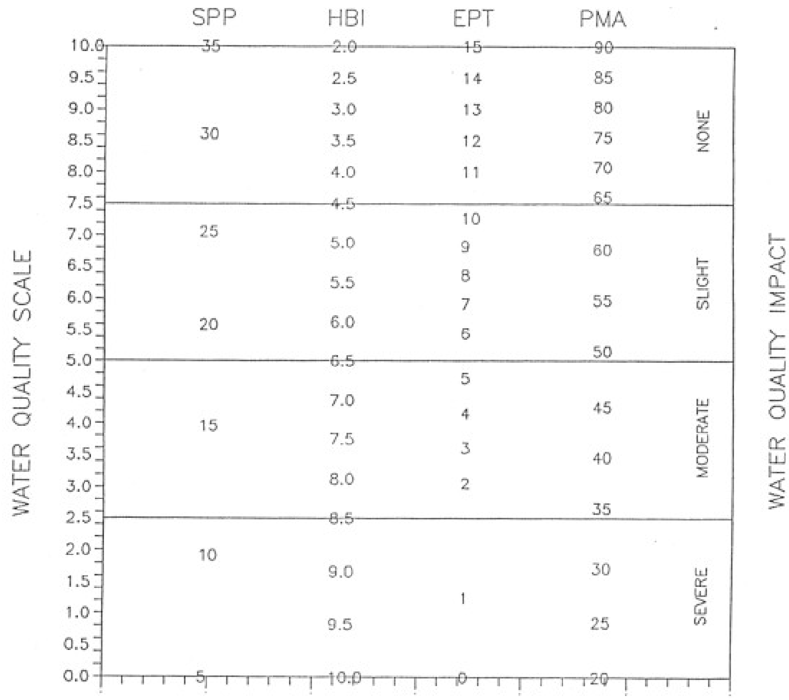

Figure 6.9: Scoring ranges for species richness, Hilsenhoff biotic index, EPT value, and percent model affinity. Source: Bode et al. 2001

Table 6.4: Water quality assessment criteria ranges for non-navigable flowing waters for species richness, Hilsenhoff biotic index, EPT value, and percent model affinity. Source: Bode et al. 2001

	Species Richness	Hilsenhoff Biotic Index	EPT Value	Percent Model Affinity#
Non-Impacted	>26	0.00-4.50	>10	>64
Slightly Impacted	19-26	4.51-6.50	6-10	50-64
Moderately Impacted	11-18	6.51-8.50	2-5	35-49
Severely Impacted	0-10	8.51-10.00	0-1	<35

Severely impacted: Indices reflect very poor water quality. The macroinvertebrate community is limited to a few tolerant species. Species richness is 10 or less. Mayflies, stoneflies, and caddisflies are rare or absent; EPT value is 0-1. The biotic index value is greater than 8.50. Percent model affinity is less than 35. The dominant species are almost all tolerant, and are usually midges and worms. Often only 1-2 species are found in the waterbody.

There are a number of advantages to using macroinvertebrates as water quality indicators (Bode et al. 1995):

1) They are sensitive to environmental impacts
2) They are less mobile than fish, and thus cannot avoid discharges
3) They can indicate effects of spills, intermittent discharges, and lapses in treatment
4) They are indicators of overall integrated water quality, including synergistic effects and substances lower than detectable limits
5) They are abundant in most streams and are relatively easy and inexpensive to sample
6) They are able to detect non-chemical impacts to the habitat, such as siltation or thermal changes
7) They are vital components of the aquatic ecosystem and important as a food source for fish
8) They are more readily perceived by the public as tangible indicators of water quality
9) They can often provide an on-site estimate of water quality
10) They can often be used to identify specific stresses or sources of impairment
11) They can be preserved and archived for decades, allowing for direct comparison of specimens across time
12) They bioaccumulate many contaminants, so that analysis of their tissues is a good monitor of toxic substances in the aquatic food chain

Some limitations also apply (Bode et al. 1995):

1) Biological monitoring is not intended to replace chemical sampling, toxicity testing, or fish surveys. Each of these measurements provides information not contained in the others
2) Substances may be present at levels exceeding ambient water quality criteria, yet have no apparent adverse community impact
3) Macroinvertebrate sampling cannot determine if water is safe for drinking

In a long-term survey of 40 years of monitoring, Smith et al. (2018) noted some successes with the use of biological monitoring. Shifts in the amount of pollutants and regulation of streams and rivers of New York between 1972 and 2012 resulted in small, incremental improvements in biological indicators, and a shift from point source dominated pollution to nonpoint sources (Figure 6.10) (Smith et al. 2018). From 1972 to 2012, 33% of the large river sites sampled had improved, while 13% had declined in biological assessment scores for water quality; 58% of the wadeable stream sites sampled showed no change (Smith et al. 2018). Macroinvertebrate community models suggest that impact sources are now dominated by nonpoint nutrient sources (Smith et al. 2018).

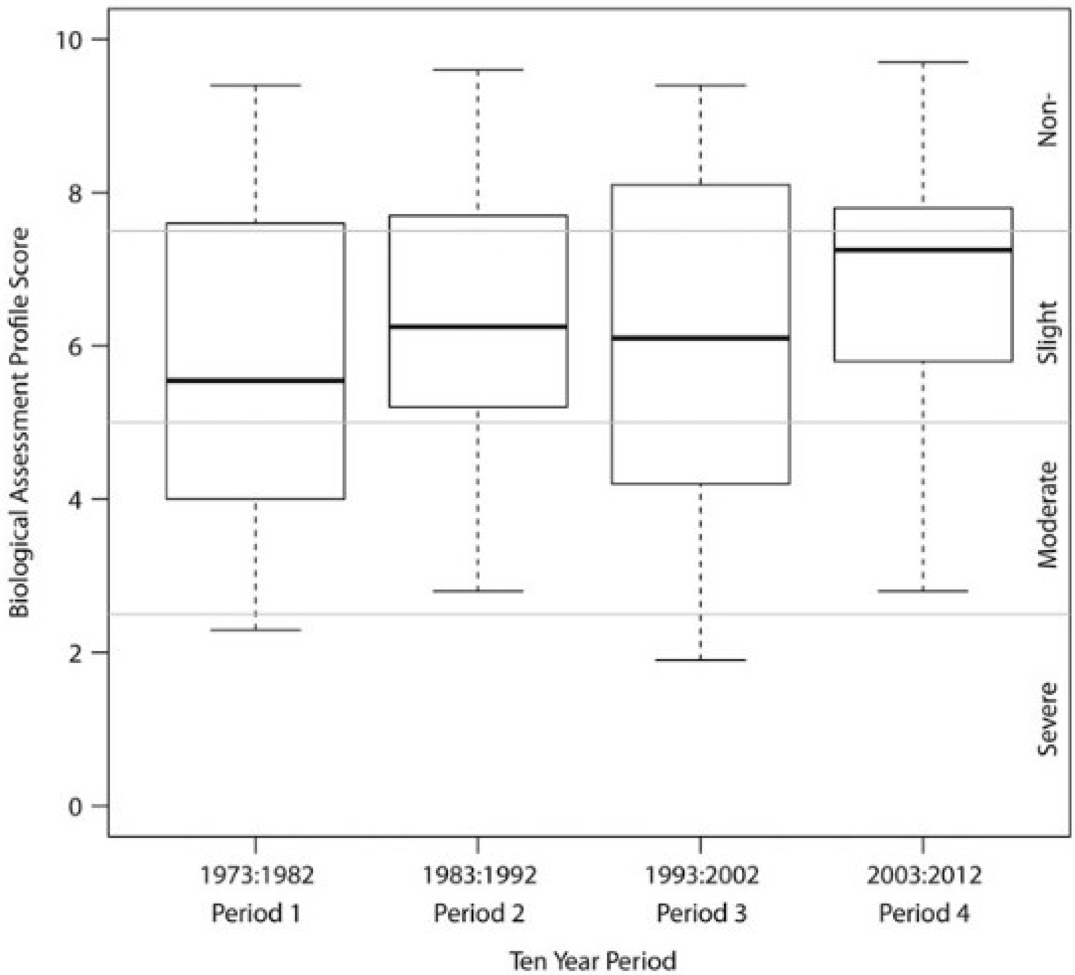

Figure 6.10: Boxplots of Biological Assessment Profile scores for large river (multiplate sample) trend sites in New York State 1973–2012, n = 30. Boxplots depict the median (black line), 25th and 75th percentiles (boxes), the 10th and 90th percentile values (whiskers), and outliers. Source: Smith et al. 2018

SUMMARY

Many states and local organizations have active and ongoing biomonitoring programs which use biological organisms to assess the biological integrity of a region. Biomonitoring can occur with a variety of different organisms (e.g., fish, macroinvertebrates, birds, frogs) and can use a variety of different indices (e.g., species richness, EPT, biotic index, percent model affinity) to evaluate ecosystem health. The field of biomonitoring has received a great deal of attention in the past and indices at the local and regional levels are continually being created and improved.

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Habitat-Focused Techniques

Chapter 7 - Habitat Assessment

The first chapter in the habitat-focused techniques group centers on the idea of habitat assessment. Many land use management plans have been based on habitat analyses to explore different future scenarios. In recent years, computing developments have enabled the emergence of landscape scale habitat modeling to explore complex options for ecosystem management. This chapter will present principles of habitat analysis and landscape scale modeling methods, which are often linked with habitat suitability for specific species, and how these ideas are used in practice to mitigate habitat losses and evaluate tradeoffs. We will end with a case study on habitat assessment in the Florida Everglades.

HISTORY AND PURPOSE OF HABITAT ASSESSMENT

Natural resource management has traditionally focused on populations. Starting in the 1970s and 1980s, natural resource management agencies increasingly employed habitat-based approaches for resource inventory and assessment (Bain and Stevenson 1999). Now, habitat is the common basis of most impact assessments, resource inventories, species management plans, and mitigation planning. Animal and plant populations fluctuate through time while habitat is much more stable. Habitat can be defined in clear and intuitive physical terms by linking habitat with specific species' known tolerances and needs. Also, the quality of a habitat can be related to species preferences. Thus, habitat is a tangible resource that can be measured and modeled for considering future scenarios of change. This resource can be quantified and used in negotiations and decision-making about development proposals. It can also be included in mitigation strategies when compensating for habitat losses. For these practical reasons, natural resource management agencies have turned to habitat as the basis for negotiating proposed developments and environmental changes.

Habitat assessment has been built into the United States' environmental laws to enhance the practicality and effectiveness of species conservation. Under the National Environmental Policy Act (NEPA), habitat is frequently assessed and is the common basis for evaluating mitigation measures for habitat loss. The Endangered Species Act (ESA) also requires protection of habitat for endangered species. Finally, the Fish and Wildlife Coordination Act (FWCA) requires that federal agencies analyze potential impacts to habitat, consider habitat when evaluating development options, and mitigate habitat losses. This Act most often engages the United States Fish and Wildlife Service (USFWS) in species conservation and requires it to consult with other federal agencies that issue permits, manage landscapes, or administer federal projects. Agencies at the state level and/or the National Marine Fisheries Service (NMFS) can also become involved. Thus, taking habitat into account is not only practical and necessary when considering environmental impacts, but its assessment is often required by some United States environmental laws.

Habitat assessment started due to the need to account for habitat losses from development and federal projects, like dams. Computer technology and geographic information systems (GIS) have expanded the use of habitat data and modeling to predict species ranges, analyze landscape transformation, and predict ecosystem change (Gurnell et al. 2002). Regional conservation planning has also extended the role of habitat to address biodiversity by seeking habitats supporting multiple conservation-priority species. Habitat fundamentals remain intact, but the expansion of habitat to the regional, state and ecosystem scales brings new opportunities for its use.

Methods for habitat assessments and modeling are important tools for many agencies and organizations with a mission for environmental conservation and management. Habitat assessments are influential in protecting some areas against forest harvesting, development, and new detrimental land uses. When habitats are quantified and found to be of high-use for valued or protected species, this information can shape other land management decisions. Also, assessment of habitat status is commonly used to set animal stocking levels, plan restorations, and determine the need for rehabilitation. Limitations on species abundance and poor habitat properties can guide habitat improvements. Predicting species distribution or estimating the suitability of habitat for a species contributes to the justification for management attention. Some state natural resource management agencies and conservation groups use this large scale information to plan land acquisitions and land-management objectives. Linking habitat models with data in a GIS is very valuable in identifying lands and waters that merit consideration for species protection. Maps of key habitats for endangered species or species popular in the public arena can inform regulatory decisions like the placement of pipelines, roads, and other relocatable infrastructure. Finally, habitat assessment results and habitat maps are often used by managers to determine specific needs, and make presentations to the public, agencies, conservation organizations, and legislators. All these uses of habitat assessments and their resulting data are now routinely incorprated into agency and organizational conservation activities.

THEORETICAL BASIS FOR HABITAT ASSESSMENT

Defining the physical conditions and limiting factors that quantify the habitat requirements for a species has a long history in ecology (Pianka 1966; Rosenzweig 1981). Habitat is tied in with the concept of a niche (Pulliam 2000; Hirzel and Le Lay 2008). Specifying a niche is equivalent to defining habitat conditions that allow a species to persist in space and time. When the habitat for a species is specified, it can be used to predict potential presence and absence. When habitat conditions are rated for quality, then predictions of species abundance can also be made. In the last century, science has made important contributions to the conservation and management of many species by defining their habitat needs (Southwood 1977). All this information regarding habitats was gathered by observing species in their natural settings and documenting the habitat conditions where they lived. This field of study was often referred to as natural history.

The term niche was given different meanings early in the 1900s. The original definition focused on the environmental requirements for a species (Grinnell 1917). Elton (1927) used a different definition of the niche, which characterized the role of a species in a community. Habitat is more consistent with the first definition of the niche, and Hutchinson (1957) elaborated on this definition. The fundamental niche is an n-dimensional hypervolume that is consistent with a range of habitat parameters. You can visualize this concept by thinking about several distinct habitat dimensions, like temperature, elevation, soil moisture, and others, that encompass the conditions needed for a species' survival. The fundamental niche often will not be fully occupied. The realized niche is the smaller portion that is occupied by a species, and is shaped by interaction with other species and limiting resources. Thus, investigations to define habitat using the niche theory most often seek to identify key variables, responses by a species to variation in variables, and the interactions among variables. The realized niche that often defines habitat is based on occupied space.

HABITAT SUITABILITY CONCEPTS

Species distribution models are based on environmental conditions in occupied habitats. Sometimes called habitat suitability maps, these models combine data on species presence or abundances with data from measurements of key habitat variables (example in Figure 7.1). Then a statistical model is used to relate the two data sets to define both tolerable habitat conditions and optimal habitats with high densities of individuals. The fundamental assumption is that individuals will select habitats most suitable to their survival. As more individuals select these habitats, the benefits of new individuals will decline because of intra-specific competition (crowding). This pattern of high abundances in the best habitats is to be expected since these habitats can support the most individuals. This pattern is termed the ideal free distribution (Fretwell and Lucas 1970, Fretwell 1972), which supports the link between high density of a species in a habitat with high suitability (Fraser and Sise 1980). Data on these high suitability habitats are collected by field measurements, meterological stations, digitized maps, aerial photographs, and satellite images. Once species presence and density data are combined with habitat data, a definition of niche and habitat requirements can be specified. These results can then be used in a GIS for predicting spatial distributions of species (Beutel et al. 1999; Guisan and Zimmermann 2000; Anderson et al. 2003; Rushton et al. 2004; Guisan and Thuiller 2005; Hirzel et al. 2006). Other analyses can be performed for assessing ecological impacts of habitat change and the effects of climate change on species distributions.

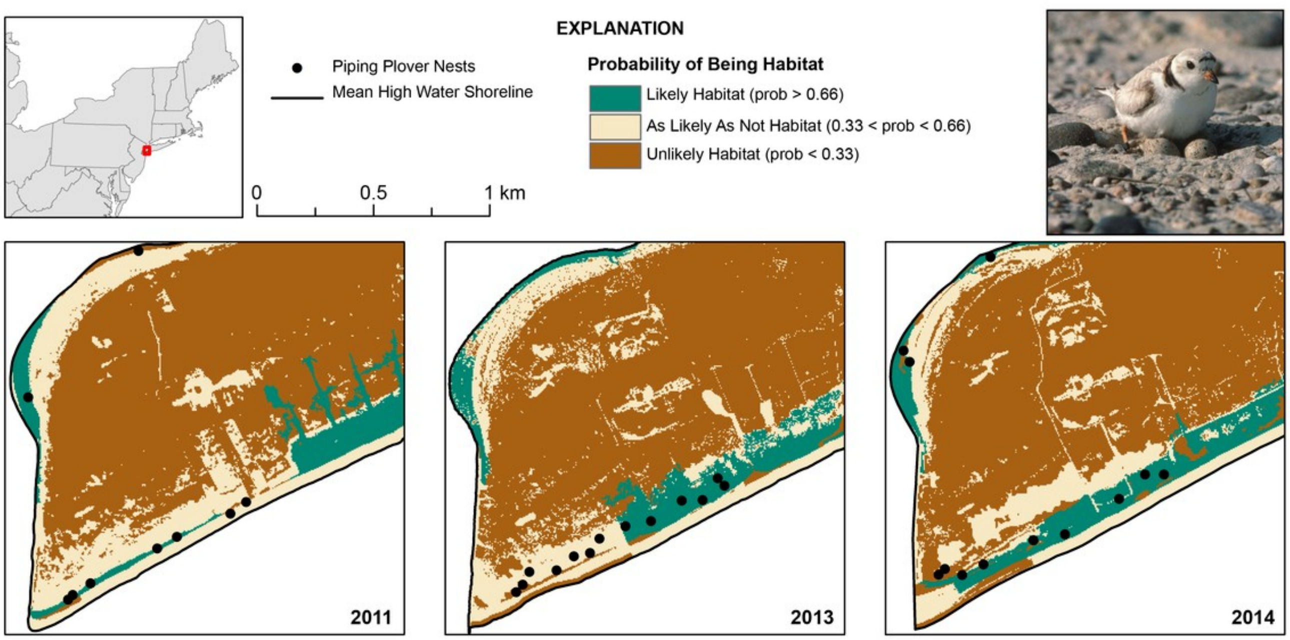

Figure 7.1: Example habitat suitability map of piping plover (Charadrius melodus) and change between 2011 and 2014. Source: United States Geological Survey 2021a

Research has shown that there can be errors in the basic study design linking presence and abundance to physical habitat conditions (Fielding and Bell 1997). Data are often collected within a narrow time frame, which can lead to bias when defining a habitat due to higher or lower population levels, climatic conditions, and resource availability levels. Competitors and predators can restrict a species' use of suitable habitat, leading it to abandon what would normally be considered suitable habitat conditions. There are questions about which spatial scale is more appropriate to use when quantifying habitats (Turner et al. 1989; knight and Morris 1996; Gustafson 1998). Often, the presence of a single individual is not necessarily a clear indicator for determining the area in which the individual may normally respond to habitat conditions. This varies by species and researchers often go by whatever works best when performing habitat data collection in the field. For species that are more flexible in their use of habitat, or generalists, it is often difficult to identify preferences for specific habitat conditions, or the physical parameters that may influence habitat use or position. By contrast, the conditions that are critical to the survival of a species with specialized habitat conditions, such as birds that nest in grasslands, are more easily identifiable. Finally, it is not always clear which physical variables may need to be accounted for when defining a habitat and assessing its suitability. Investigators often collect data on multiple habitat properties, but it is possible that some key properties might be missed.

There is a second dimension of uncertainty about linking species and habitats. False absences can occur even when suitable habitat is present because other factors may be preventing a species from occupying the area. Absence in suitable habitats can come from localized elimination of a species by things like harvest or disease, restricted dispersal by barriers, habitat that is too small to support a population, and biotic interactions that drive individuals out of their preferred habitats (Meixler 2021). These phenomena can skew the definition of suitable habitat in inaccurate ways. Likewise, false presences, or presence in unsuitable habitats, can also occur (Meixler et al. 2009). Crowding may induce individuals to inhabit fringe locations, and sub-dominant individuals can be displaced to marginal or unsuitable habitats. Movement across unsuitable habitats during migration can also happen. Also, territorial behavior can force some individuals out of highly suitable habitats. And, habitat generalists often follow their food source which can cause them to switch between habitats constantly. Finally, false presences in marginal or unsuitable habitats can alter the definition of suitable habitat and broaden the range of habitat properties considered.

HABITAT SUITABILITY MAPPING AND MODELING

Habitat quantification is vital to mapping species distribution for research on biogeography, ecology, changing climate impacts, invasive species ranges, and other investigations (Franklin 2010). Generally, species distribution maps use the fundamental niche and the widest range of suitable conditions for many detailed studies. Habitat definitions are combined with commonly available spatial data like topography, soils, climate, and vegetation maps, in a GIS to produce species distribution maps. This is a relatively novel use of habitat definitions due to the increased availability of spatial data, GIS integration of data, and mapping capability (Brooks 1997). Often, many species are mapped in this way to identify areas of high biodiversity for conservation strategy development.

The dominant demand for estimating habitat suitability was to perform impact assessments for key species. Quantification of habitat for proposed developments and mitigation needs drove applied habitat modeling. In the 1980s, the USFWS developed standardized Habitat Evaluation Procedures (HEP) to use in impact assessments, which is part of the agency’s mandate (United States Fish and Wildlife Service 1980a, United States Fish and Wildlife Service 1980b). Professional judgment and ad hoc methods were not persuasive enough to be effective in modifying the plans of other agencies, large scale developments, or for specifying mitigation measures for losses of habitat. The USFWS adopted HEP to meet several critical objectives of impact assessment. These objectives included: 1) Displaying data on present conditions and proposed development options, 2) Predicting fish and wildlife habitat changes over time with proposed changes, 3) Ensuring a method that is practical to implement using commonly available data, readily available for species habitat models, and cost effective to execute, and 4) Making sure the method is sensitive to habitat losses and the magnitudes of changes in habitat quality. The HEP fills this need and is composed of: 1) Habitat Suitability Index (HSI) models to assess habitat quality for many species, and 2) Accounting procedures to predict the impacts of proposed developments. HSI models are used to infer the ability of a habitat to support a species and are thus positively correlated with carrying capacity (Schamberger and Krohn 1982; Dussault et al. 2006). HSI models produce an index between 0 and 1, with 0 designating unsuitable habitat and 1 being optimal habitat. The HSI is calculated as study area/optimal habitat conditions, where conditions are represented by variables that include: 1) Whether the species responds; 2) If it can be readily measured; 3) If it can be predicted in the future under change; or 4) if it is influenced by projects or management (Schamberger and O'Neil 1986). The USFWS created a library of hundreds of HSI models for indicator species and other valued species (United States Fish and Wildlife Service 1981). To implement HSI models confidently, they must be tested for accuracy (i.e., validated). Validation involves testing the performance of an HSI model against independent quantitative data in space and time, such as against population density or abundance (Zorn et al. 2011; Zajac et al. 2015; Theuerkauf and Lipcius 2016).

HEP is based on combining a measure of habitat quantity (e.g., affected habitat area, in acres or hectares) with an index of habitat quality (e.g., HSI, 0 to 1 scale) to determine habitat units (dimensionless numbers) (United States Fish and Wildlife Service 1980b). The relationship is expressed as:

Habitat area x Habitat quality (HSI) = Habitat units (HUs)

The HEP have been in common use to estimate impacts to fish and wildlife species, compare development options, and set mitigation requirements (Urich and Graham 1983). Figure 7.2 shows an example in which the change in wetland habitat units are shown over time with and without a railroad construction project. The HEP are not meant to be research tools, carrying capacity estimators, population models, or be comprehensive.

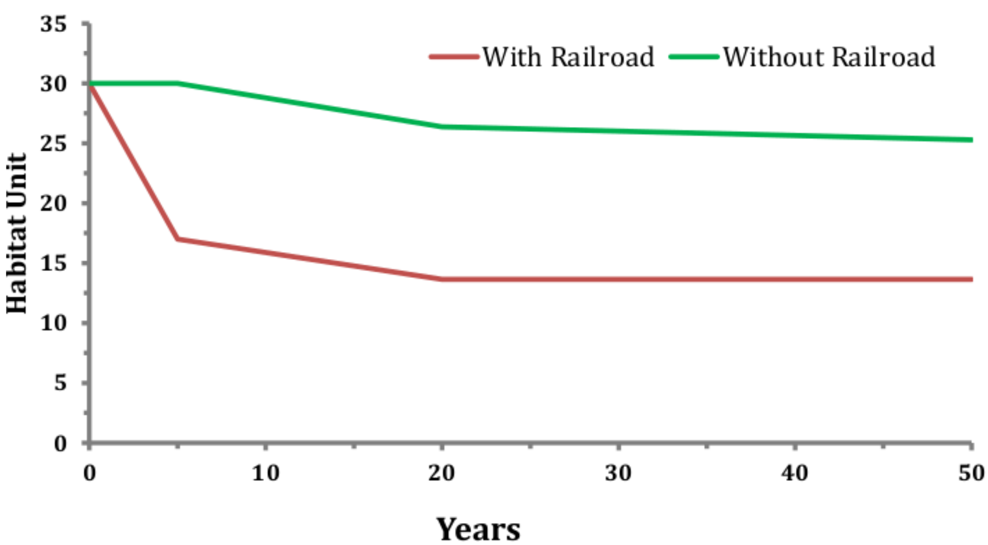

Figure 7.2: Change of wetland habitat units over time with and without railroad construction. Source: Mitra and Bezbaruah 2014

The USFWS developed a similar set of procedures for predicting habitat units in rivers and streams under different flow levels. The instream flow incremental methodology (IFIM; Stalnaker et al. 1995) was comprised of a habitat simulator for a river reach (Physical HABitat SIMulation, PHABSIM) which could be used to depict the amount and quality of habitat at different flow levels (Milhous et al. 1984). The changing spatial distributions of physical attributes of a river (e.g., water depth, current velocity, and substrate) as a result of variations in flow and the biological responses of aquatic species to these changes, provide the basis for simulating the consequences of ecosystem alteration. USFWS produced a library of suitability plots for each life stage of many valued fishes and other aquatic species. The weighted usable areas formed the basis for comparison of different flows by season, which could then be used for predicting the habitat conditions that might exist over the course of a species' life history, since different life stages of a species require different habitat conditions. The charts, depicting flow rates and usable area for a species by season, defined the basis for negotiations on how much water was needed per season.

EXAMPLE APPLICATION OF HEP

A HEP Assessment was performed on a land parcel in the Yakama Nation (Table 7.1) (Ashley and Muse 2008). A baseline HEP analysis was conducted on the Carl property (160 acres) in June 2007 to determine the number of habitat units to credit Bonneville Power Administration (BPA) for providing funds to acquire the property as partial mitigation for habitat losses associated with construction of the McNary Dam. The types of landcover present at that time were shrub/grassland (covering 99% of the area) and emergent wetland (covering 1% of the area). HEP model selection was based on the habitat types present and the species models identified in the McNary Dam Loss Assessment. The chosen species included: California quail (*Callipepla californica*), western meadowlark (*Sturnella neglecta*), mallard (*Anas platyrhynchos*), Canada goose (*Branta canadensis*), downy woodpecker (*Dryobates pubescens*), yellow warbler (*Setophaga petechia*), spotted sandpiper (*Actitis macularius*), and mink (*Mustela vison*). Sampling transects were utilized to collect field data for use in the models. The results provided information on habitat suitability (HSI) and habitat units for the land cover types in which each species resided (Ashley and Muse 2008).

Table 7.1: HSI and HU summary for Yakama Nation HEP project. Source: Ashley and Muse 2008

Cover Type	Acres	Model	Variable	SI	HSI	HUs[3]
Shrubsteppe/Grassland (includes riparian herb)	158	California Quail	V1: Percent herbaceous cover	0.88	0.30	47.69
			V2: Average shrub height	0.03		
			V3: Distance to escape cover	1.00		
			V4: Average diameter of escape cover	0.61		
			V5: Distance between escape cover patches	0.47		
		Western Meadowlark	V1: Percent herbaceous CC	0.89	0.73	114.78
			V2: Percent herbaceous CC composed of grass	0.85		
			V3: Average height of herbaceous CC	0.84		
			V4: Distance to perch	0.92		
			V5: Percent shrub CC	0.95		
		Mallard	V3: Distance between nest and emergent cover (miles)	1.00	0.83	131.23
			V4: Height of residual nesting cover	0.63		
			V5: Cover of nesting vegetation	0.87		
			V6: Human disturbance	1.00		
		Canada Goose	V1: Presence of trees	0.20	0.38	60.34
			V3: Brood areas	0.96		
			V4: Human disturbance	0.50		
Cover Type	**Acres**	**Model**	**Variable**	**SI**	**HSI**	**HUs**
Emergent Vegetation (Wetland)	2	Mallard	V7: Ratio of vegetative cover to open water	0.35	0.35	0.70
		Mink	V1: Percent of year with surface water present	0.75	0.69	1.38
			V2: Percent tree CC	0.37		
			V3: Percent shrub CC	0.20		
			V4: Percent CC of emergent vegetation	0.72		
			V5: Percent CC of trees/shrubs within 100m of water edge	0.10		
Total Acres	160				Total HUs	356.11

LIMITATIONS OF HABITAT SUITABILITY MAPPING

The application of habitat models and assessments can be unclear for cases that are controversial for regulatory and planning purposes. The real definition of habitat is dictated by the specific needs of a species, which has physical requirements for occupancy, survival, and reproduction. At the species level, habitat can be bounded and assigned a quality value using information about the species ‘ particular needs. Often classes of habitats are loosely discussed, like old growth forests, riparian zones, and grasslands, whose characteristics might be decoupled from those of the precise species under consideration (Hall et al. 1997). This approach often implies that a habitat can house a collection of species, but that is confusing and imprecise in its definition. Habitat quality should refer to the optimal physical conditions for species survival and population persistence. Habitat models often define physical conditions as high-quality habitat based on the preferred conditions of a species. These high quality habitats are considered to accommodate high densities of a particular species. Considering habitat in such a loose manner introduces ambiguity into the definitions and quality assessments of habitats, which makes it hard to pinpoint the precise effects of any given proposal. Applications of the term habitat should be based on a species-specific foundation, and backed up with physical criteria that are usable and quantifiable.

Habitat definitions and quality criteria are often used in management decisions without verification to save time and funding. The USFWS’ collection of HSI models were developed to be consistent in comparison of development options, precise in mitigating habitat losses, and transparent in regulatory decision-making. This strategy can be effective for promoting quality habitats and accounting for losses. However, there are concerns about generic HSI model applications. Individual criteria and time periods can be customized for the habitat needs of any species. Habitat generalists can be flexible in habitat choice. The habitat in question has to be measured adequately and modeled under future scenarios. Also, assessments of habitat quality require more rigorous measurements and the application of suitability models. Thus, baseline data have to be sound so projections for future options can compute habitat space and units more precisely.

HABITAT MODELING FOR BIODIVERSITY CONSERVATION

As the use of GIS became more common among natural resource management agencies and conservation organizations, there was interest in using habitat models to map species habitats and ranges, assess the hotspots of species richness, and design landscape strategies to protect biodiversity (Scott et al. 1993). This management strategy was not anticipated when agencies developed procedures to conduct habitat assessments, but it built on the fundamentals of habitat. Under the direction of the United States Geological Survey, a national program called the Gap Analysis Program (GAP) was initiated, which aimed to rapidly identify gaps in the protection of high biodiversity locations at a state level. First, the GAP program mapped vegetation types based on satellite data (Figure 7.3). Vegetation was considered a good integrator of physical and biological attributes because it includes attributes like elevation, soil types, climate, and slope. Then species-specific habitat models were developed that connect habitat types in the vegetation maps with other possible attributes in the GIS to map ranges of species. Many species of interest were mapped and locations of high biodiversity were identified. Then public lands, wildlife management areas, and parks were mapped to show gaps in the protection of biodiversity hotspots. This is the prominent comprehensive land conservation planning tool for biodiversity conservation.

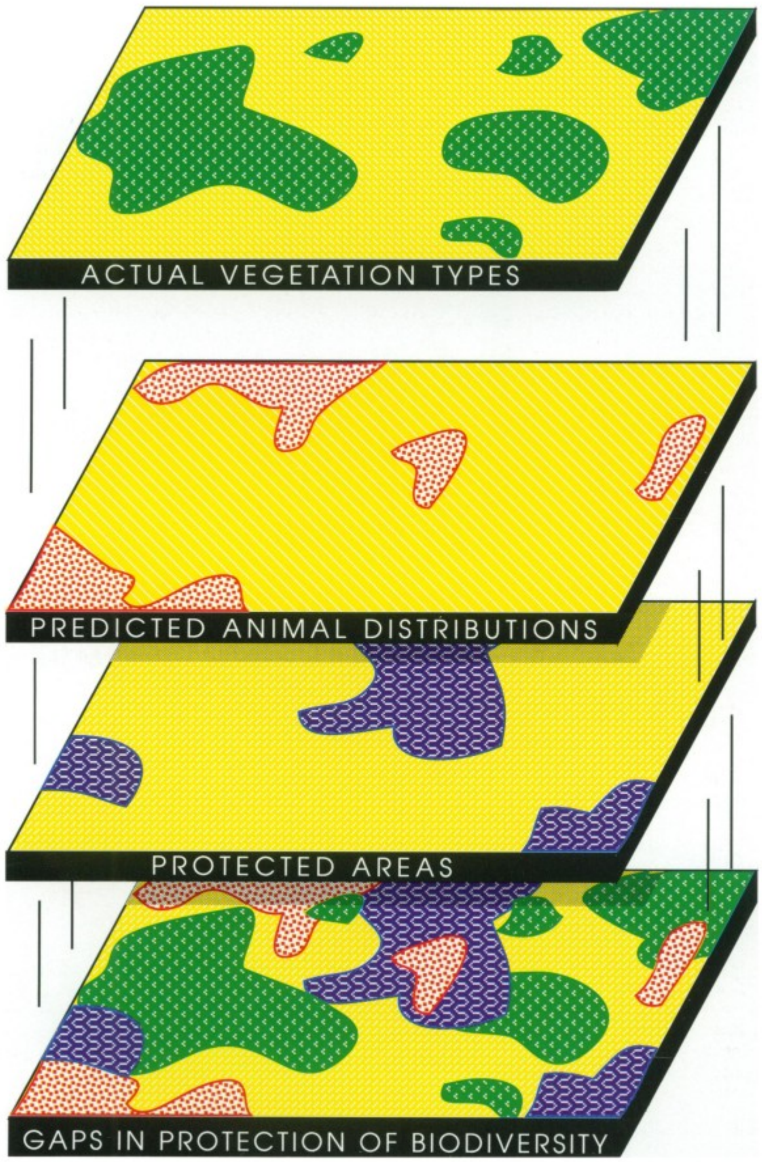

Figure 7.3: Conceptual basis for the Gap Analysis Progam (GAP) showing how GIS layers interact to show gaps in the proteciton of biodiversity. Source: Scott et al. 1993

USE OF CELLULAR AUTOMATA ANALYSIS

More recently, as larger scale environmental questions have come under analysis (e.g., whole ecosystems and landscapes), habitat modeling has been augmented to deal with larger spatial and temporal realms. Issues like change in climate, land use, and water regulation drove this endeavor to expand the way habitat is considered at much larger scales. The common approach adopted was cellular automata modeling around habitat conditions by cell (Itami 1994). Cellular automata modeling consists of a uniform grid of cells over an ecosystem or landscape, where thousands of cells can cover a study area (Figure 7.4). Each cell is connected to adjacent cells which exchange materials, like water and nutrients. Habitat models are embedded in each cell to predict the habitat status of each cell. The final result is a designation of a state or habitat type by cell, which can be used to generate a map of habitat variation across the ecosystem or landscape.

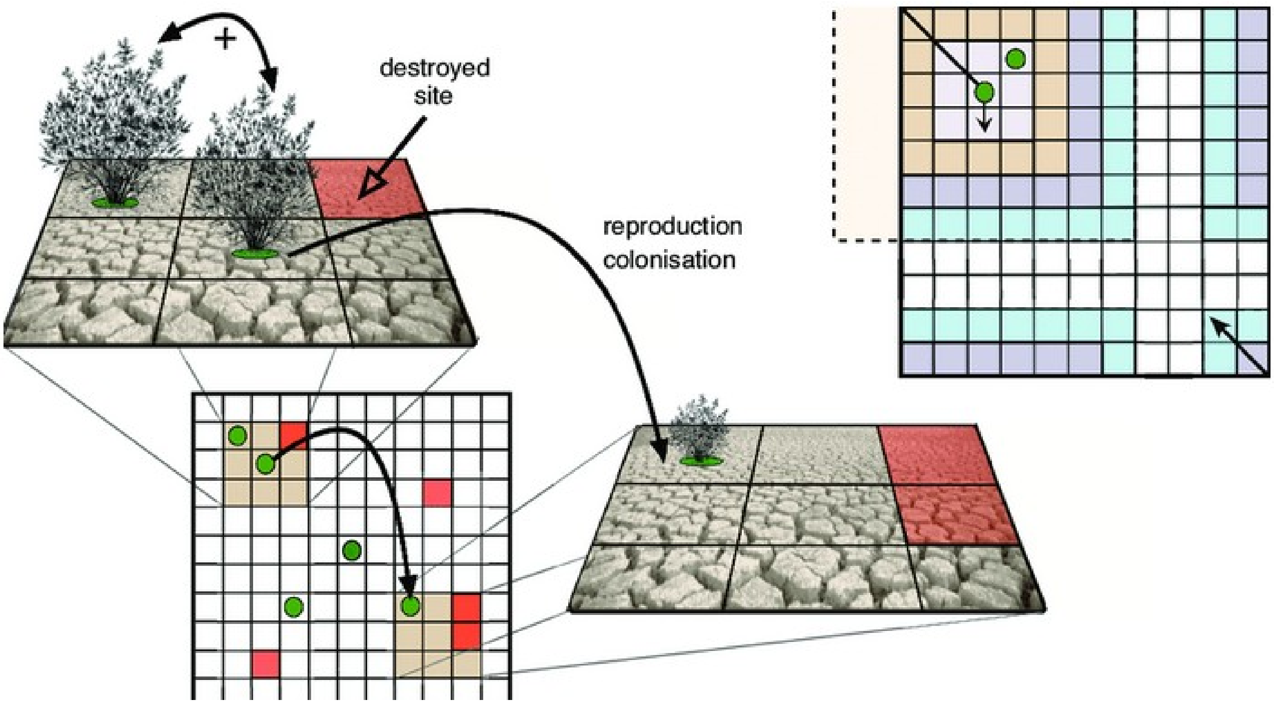

Figure 7.4: Example of use of cellular automata modeling. The image shows a schematic diagram of an ecological system with facilitation in reproduction under habitat destruction illustrated with the cellular automaton model. The individuals can grow and reproduce due to cooperation with neighboring individuals. The cellular automata is a square lattice and has three states: 1) sites occupied by an individual (green circle), 2) empty sites that can be colonized (white cells), and 3) destroyed sites (red cells) where no establishment or growth of an individual is possible. The dynamics of reproduction involves facilitation i.e., an individual can reproduce if a neighboring place is occupied. Once an individual reproduces, it can colonize another empty site of the lattice. If the colonized site is destroyed, the individual can not grow. The lattice is assumed to have toroidal boundary conditions. In the upper right corner we display two possible colonization scenarios: local (short thin arrow) and long-range (solid arrow) dispersal. Source: Sardanyés et al. 2019

CELLULAR AUTOMATA EXAMPLES

Two landscape models were developed to predict habitat changes across a wetland-dominated landscape of south Louisiana due to the effects of water flow regulation across the Mississippi River Delta and climate change. These were the Coastal Ecological Landscape Spatial Simulation (Costanza et al. 1990) and the Barataria–Terrebonne Ecological Landscape Spatial Simulation (Reyes et al. 2000). Both study teams mapped changing habitat composition in 1 km^2 cells for a portion of the delta strongly affected by controlled river flow. The cells were connected by flows of water, salt, nitrogen, sediments, and water elevation. The habitat model inside each cell included hydrodynamics, productivity, soil-dynamics, flooding, salinity, and other properties. Habitat switching was predicted using data input to the habitat models for each cell. The product resulted in a wetland landscape that could be determined under different river flow controls and climate change scenarios. The following habitat types were the results of the final classifications by cell: upland, freshwater marsh, swamp forest, brackish marsh, saline marsh, and open water. These habitat types were not defined by species requirements; rather, they were modeled using principles of habitat change. These analyses were done to cover multiple decades because the evolution of landscape properties takes many years under some scenarios. These landscape models were well developed and represent the expansion of habitat modeling in space and time.

Expansion of habitat modeling was used to address ecosystem change in the Patuxent River basin in Maryland (Costanza et al. 2002) to evaluate future scenarios of development (Figure 7.5; Table 7.2). Regional socioeconomic trends were included to predict likely development pressures and outcomes. Consequences of different development patterns were explored by modeling flows, like nitrogen and phosphorus discharges into Chesapeake Bay. Other predicted outcomes were plant growth, runoff with dissolved nutrients, and formation of soils. Cells were classified by habitat type (water, forest, agriculture, residential, and urban), and adjacent cells could exchange water, nutrients, and suspended materials. Model runs inside each cell were elaborate. When the results were tallied across cells under each development scenario, one could predict for the landscape properties, map habitat types, and see the distribution of vegetation biomass and net primary production. Again, this extension of habitat modeling across large landscapes and far into the future was an advancement from researchers' pioneering methods for larger scale habitat forecasting.

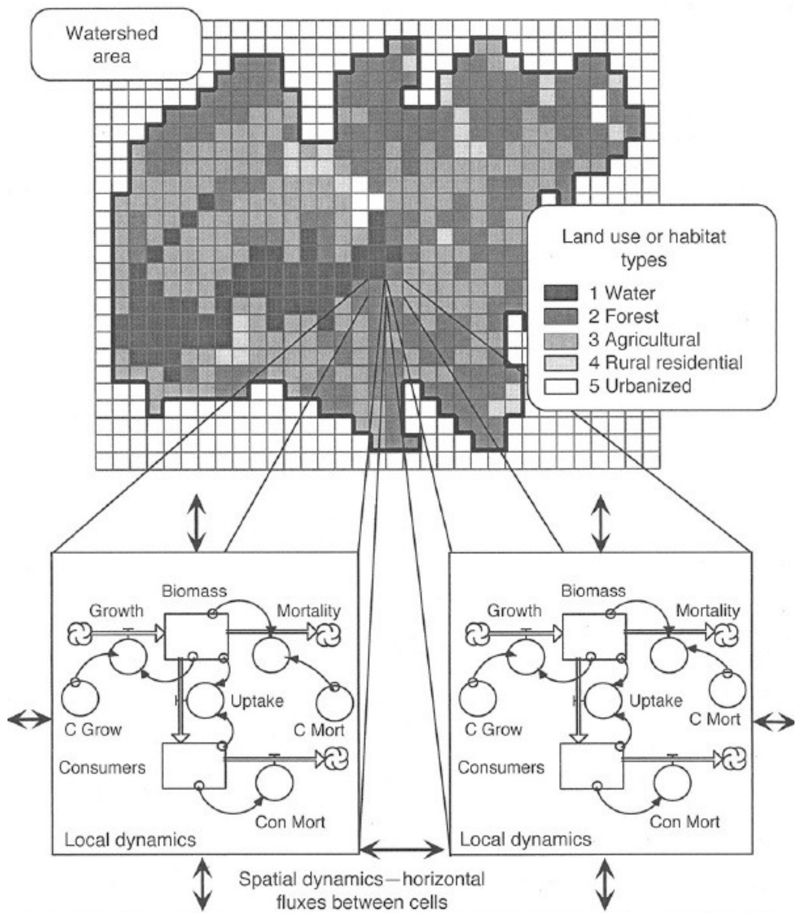

Figure 7.5: The cellular structure of the Patuxent landscape model. Each cell has a habitat type, which is used to parameterize the unit model for that cell. The unit model simulates ecosystem dynamics for that cell in the above-sediment and below-sediment subsystems. Nutrients and suspended materials in the surface water and saturated sediment water are fluxed between cells in the domain of the spatial model. Source: Costanza et al. 2002

Table 7.2: Partial results of output from Patuxent landscape model used for decision-making. Source: Costanza et al. 2002

Scenario		No. cells				Concentration ($kg \cdot ha^{-1} \cdot yr^{-1}$)				Concentration (mg/L)			
No.	Type	Forest	Resid	Urban	Agro	Atmos	Fertil	Decomp	Septic	N_{mean}	N_{max}	N_{min}	N_{gwc}
1	1650	2386	0	0	56	3.00	0.00	162.00	0.00	3.14	11.97	0.05	0.023
2	1850	348	7	0	2087	5.00	106.00	63.00	0.00	7.17	46.61	0.22	0.25
3	1950	911	111	28	1391	96.00	110.00	99.00	7.00	11.79	42.34	0.70	0.284
4	1972	1252	223	83	884	86.00	145.00	119.00	7.00	13.68	60.63	0.76	0.281
5	1990	1315	311	92	724	86.00	101.00	113.00	13.00	10.18	40.42	1.09	0.265
6	1997	1195	460	115	672	91.00	94.00	105.00	18.00	11.09	55.73	0.34	0.289
7	Buildout	312	729	216	1185	96.00	155.00	61.00	21.00	12.89	83.03	2.42	0.447
8	BMP	1195	460	115	672	80.00	41.00	103.00	18.00	5.68	16.41	0.06	0.23
9	LUB1	1129	575	134	604	86.00	73.00	98.00	8.00	8.05	39.71	0.11	0.266
10	LUB2	1147	538	134	623	86.00	76.00	100.00	11.00	7.89	29.95	0.07	0.269
11	LUB3	1129	577	134	602	86.00	73.00	99.00	24.00	7.89	29.73	0.10	0.289
12	LUB4	1133	564	135	610	86.00	74.00	100.00	12.00	8.05	29.83	0.07	0.271
13	agro2res	1195	1132	115	0	86.00	0.00	96.00	39.00	5.62	15.13	0.11	0.292
14	agro2frst	1867	460	115	0	86.00	0.00	134.00	18.00	4.89	12.32	0.06	0.142
15	res2frst	1655	0	115	672	86.00	82.00	130.00	7.00	7.58	23.50	0.10	0.18
16	frst2res	0	1655	115	672	86.00	82.00	36.00	54.00	9.27	39.40	1.89	0.497
17	cluster	1528	0	276	638	86.00	78.00	121.00	17.00	7.64	25.32	0.09	0.216
18	sprawl	1127	652	0	663	86.00	78.00	83.00	27.00	8.48	25.43	0.11	0.349

CASE STUDY: HABITAT ASSESSMENT IN THE FLORIDA EVERGLADES

Figure 7.6: Florida Everglades. Source: National Park Service 2021a

In 1947, the Florida Everglades became the first national park designated to protect an ecological system (Figure 7.6). It has also been designated a United Nations Educational, Scientific and Cultural Organization (UNESCO) World Heritage Site, an International Biosphere Reserve, and a Wetland of International Importance (National Park Service 2021a). The restoration of the Everglades is a large ecosystem management case focused on habitat conservation issues. Its management program combines traditional habitat suitability indices with large scale hydrologic modeling to compare alternative water management policies. The traditional habitat suitability indices are just like those that the USFWS developed in the 1980s. These were combined with landscape level hydrologic simulation to portray what water management can achieve. The goal of the Everglades restoration program is on "getting the water right." Over the past century, many canals were built for water drainage and about half of the Everglades has been lost to urban and agricultural conversion of land (Walker and Solecki 2004). Over time the Everglades has changed to an unhealthy ecosystem with more than 50 species on the United States Endangered Species Act list. Both Florida and the United States Congress passed Acts with the intent of restoring the Everglades to a more natural state at an estimated cost of several billion dollars. Their main focus was on quantity, quality, timing, and the distribution of water flow across this large landscape. This is a difficult and complex challenge because of the size of the ecosystem and its unique hydrology. In addition, there are many stakeholders involved who want to secure differing benefits from the Park. This is an informative case study, involving complicated ecology, hydrology, and human demands that uses fundamental habitat modeling in a new way.

The comparison of water management alternatives and planning options for the restoration of the Everglades engaged a diverse set of stakeholders, who were primarily interested in and concerned with choices. The comparison analysis needed to be easily understandable, transparent, and responsive to input. There were several models used to evaluate water management in the Everglades but they were complicated, specialized, and not easily modified (Fennema et al. 1994; DeAngelis et al. 1998). Non-specialists could not understand them, and often these models were called "black boxes." The challenge was to get these diverse parties to engage with each other in the process of finding a solution for future management. This required all parties to have an understanding of the analyses and an ability to offer informative input. This Everglades case was shaped by its focus on habitat, which was easily understood and simulated across the landscape.

Figure 7.7: American alligator. Source: National Park Service 2021b

The analysis process began by identifying habitats that could be related to water depths, flows, and timing. The selected habitats were either key landscape features, support for groups of animals, or were in need of specific species. These habitats were important attributes of the Everglades Park from the public's perspective The process identified five different habitats which characterized an ecosystem scale response to water management changes. Two landscape attributes were chosen to reflect changes in water management over decades: ridge and slough landscape habitats, and tree islands. Periphyton is a fast reacting indicator of water flows and quality, so it was included to act as an early warning signal if problems arose. One species group, fish in general, were identified to reflect conditions over time and space. These are notable biological features of the Everglades which readily respond to changes in the environment due to water management alterations. Finally, one species, the American alligator (*Alligator mississippiensis*), was identified for habitat assessment because of its ecological prominence and its recognition among the public (Figure 7.7).

Figure 7.8: Ridge and slough. Source: Tarboton et al. 2004

The ridge and slough landscape habitat is characteristic of the Everglades and is composed of sawgrass ridges and open waters (Figure 7.8). This complex habitat decreased in area during the canal building and drainage of the Everglades. Thus, it was selected as a habitat to be restored by water management, as it is considered a symbol of health for the Everglades. Tree islands are another component of the Everglades landscape (Figure 7.9). Tree islands serve as nesting sites for birds and reptiles, provide dry terrain for animals, and support more species than other habitat in the Everglades. Though a small portion (5-10%) of the Everglades landscape, they are critical for supporting wildlife. The periphyton habitat index was included as a sensitive indicator of hydrologic conditions and water quality, especially nutrients in a naturally oligotrophic system. Periphyton also has many effects on the ecosystem by influencing secondary production, nutrient concentrations, and dissolved gases. These three diverse habitats are indicative of the nature of the Everglades and characteristic of the landscape overall.

A modeling effort that combined the HSI approach and landscape simulation was developed (Tarboton et al. 2004). Fish and American alligators are both sensitive to habitat conditions. Fishes are important indicators of ecosystem status, and provide food for wading birds and alligators. When water is high, small fish flourish and become abundant, and occupy marshes where they are largely protected from predators in the vegetation. When the marshes dry, small fish are forced into confined watered areas where they are vulnerable to wading birds, alligators, and large fish. Fish abundance typically increases with time since the last marsh drying. This one multi-species group and the American alligator are interconnected, and are common images and biological features of the Everglades ecosystem.

Figure 7.9: Tree islands. Source: United States Geological Survey 2021b

The American alligator will be used to detail how habitat suitability indices are used for five habitats. Alligators are the top carnivore in the Everglades. They physically modify the ecosystem by developing water holes and trails. Water depths are important to alligators for different reasons: breeding, nest construction, nest flooding, and interaction between alligators. Four habitat factors were identified for alligators which were plotted against an influential hydrologic variable. The charts from habitat suitability indices and factors were plotted on a 0 to 1 scale, from non-habitat (0) to optimum habitat (1). This was a key decision point in the analysis process because it defined the fundamental data for comparison of water management alternatives and was used for evaluating ecosystem response. This study was done in English measurement units, so it is presented here using these same units.

For each of the five habitats, suitability indices were formed to link to hydrologic conditions (Figure 7.10). The number of days where the water was less than 0.5 feet deep influenced the proportion of adult female alligators expected to nest in a given year. This related to drought conditions that dispersed males and limited water habitats that caused physiological stress. Shorter periods of low water or marsh drying increased the suitability of the habitat for alligators. For nest construction, there was an optimal water depth during the period of mid-April to mid-May. After nests were constructed, it was important that water levels remain stable during the egg incubation period (mid-June to the end of August). Eggs were laid in the nest about 0.5 to 1.0 feet above water level, so if water levels rose more than 0.5 feet, some eggs would be flooded and not survive. During dry periods, alligators occupied water holes or other watered areas. They were crowded in these locations and small alligators were often eaten by large alligators. Therefore, alligator abundance went down because of limited water habitat in dry periods. Also, body condition decreased because of the lack of food. Optimal habitat conditions were marked by no loss of water level, as a water level reduction of 2 or 3 feet impacted alligator abundance and health. These four charts illustrate the conditions for water levels and the timing of changes (Figure 7.10). All five habitats had multiple charts like these, defining water management protocols for each primary habitat assessment feature.

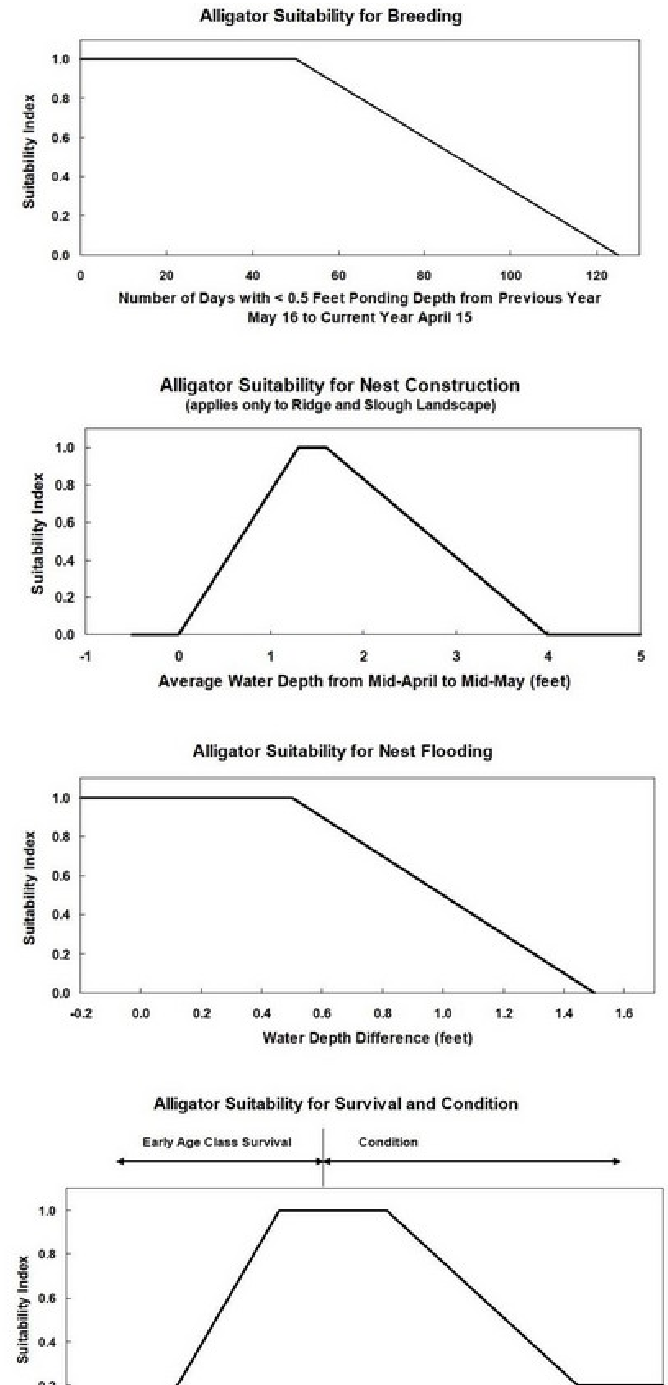

Figure 7.10: Four habitat suitability index (HSI) graphs for American alligators (Alligator mississippiensis) in the Everglades. These HSI relations were used to predict the quality of the habitat across the Everglades under different water management strategies. Source: Tarboton et al. 2004

For each of the five habitats, a consolidated index was made by doing a weighted arithmetic mean of the habitat suitability indices. The factors for the American alligator were weighted based on their expected impact on the overall population of alligators in the Everglades. The formula for alligators was:

Overall alligator suitability index = (3 x SI breeding +
3 x SI nest construction +
2 x SI nest flooding +
1 x SI survival and condition) / 9

Where SI is the suitability index of each factor.

A simulation model was then required to produce a rendering of the hydrologic properties across the Everglades. Inputs values were the habitat suitability indices simulation of water depths, flow velocity, dry periods, and the period of water level change for a daily time step and for 2-mile by 2-mile grid cells spanning the region. Then each cell had a computed overall alligator suitability index (Figure 7.11). A map was made for each water management plan to see how it affected the five habitats (Figure 7.12).

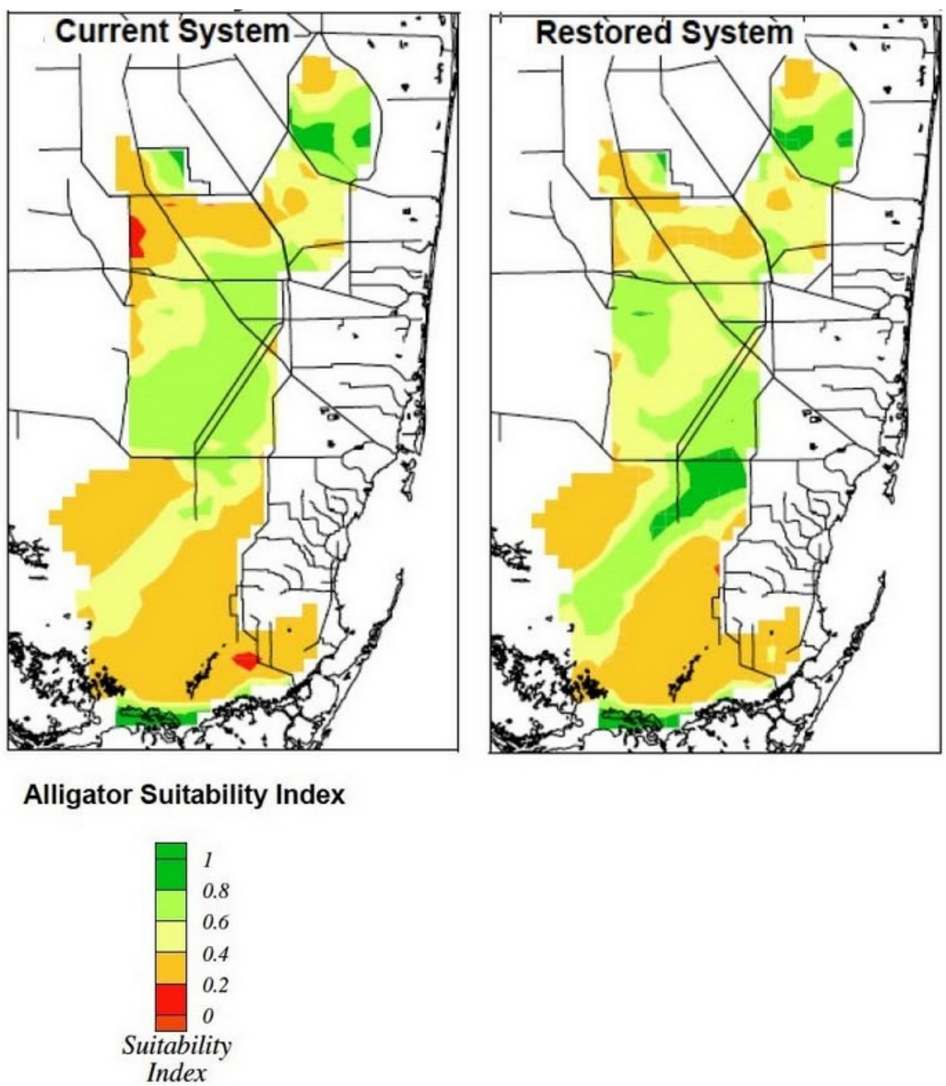

Figure 7.11: Maps of the current and restored Everglades ecosystem showing habitat suitability for American alligators (Alligator mississippiensis). Source: Tarboton et al. 2004

Each of the five habitats were mapped using simulated outcomes of each water management plan. Interaction between interested individual stakeholders and organizations led to the selection of one management plan using the habitat quality maps (Loucks 2006). Other analyses were performed, but the main result was that the restoration water management plan was selected. The analysis process was easy to understand and engaged everyone with an interest in the matter. Participants over their differences which enabled them to agree upon the selection of a single restoration plan. Again, this exercise entailed applying habitat suitability indices to a complex landscape problem which linked water management, physical conditions, and ecological restoration together within a unique and expansive ecosystem.

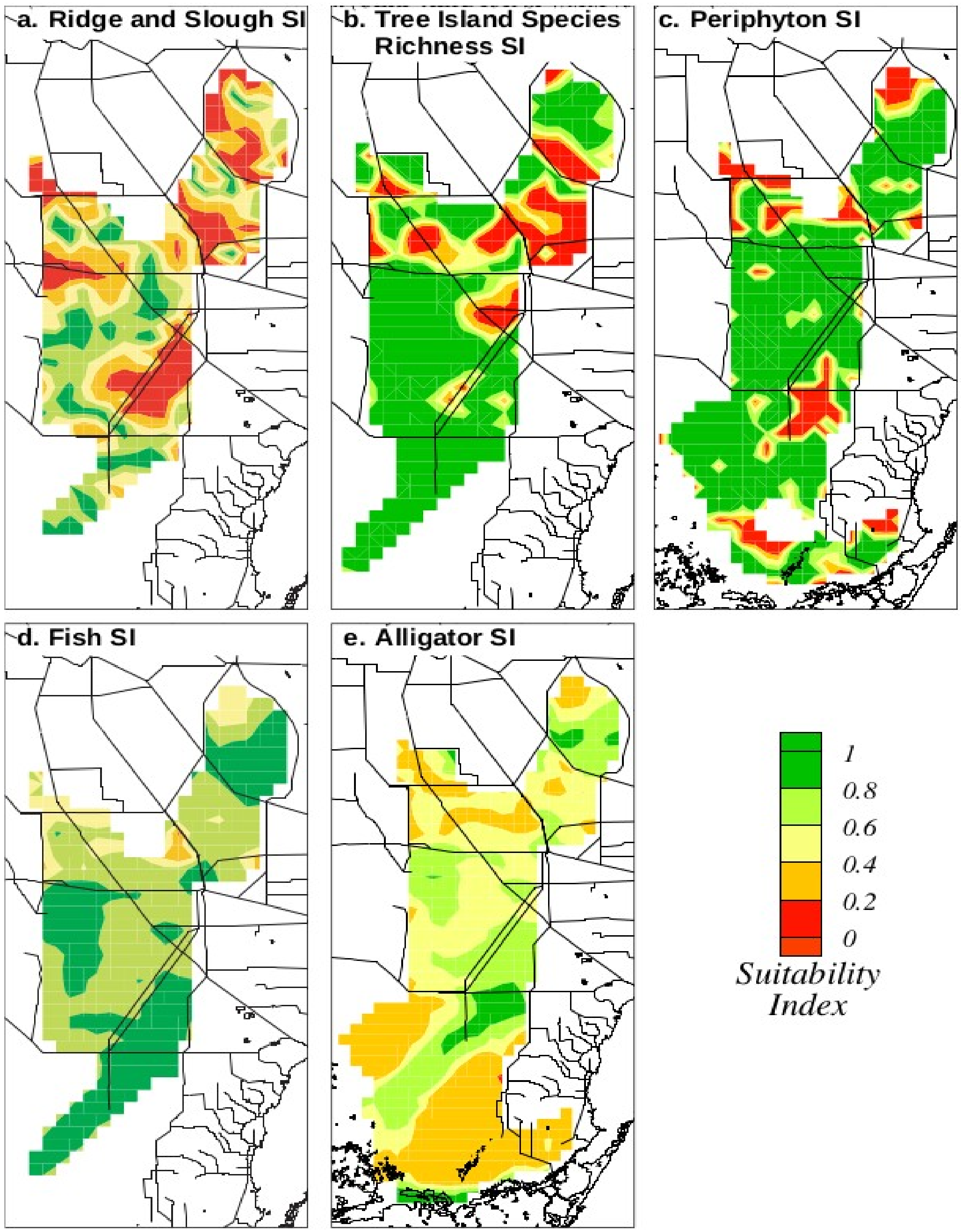

Figure 7.12: Habitat suitability indices for the five habitats under the restored ecosystem. Source: Tarboton et al. 2004

SUMMARY

Habitat assessment is used for a variety of purposes like animal stocking levels, planning restorations, determining the need for habitat rehabilitation, comparing future development scenarios, and assigning mitigation actions for habitat losses. Advancements in the science of habitat assessment and new habitat modeling options in GIS have led to better prediction of the distribution of species and design of conservation programs. These new advancements help to address many of the core issues at the heart of habitat assessment and provide the basis for stakeholder engagement and cooperation in developing solutions to changing situations on the ground.

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Habitat-Focused Techniques

Chapter 8 - Restoration

The next topic in the habitat-focused techniques group centers on the idea of restoration. The traditional definition of restoration is returning an ecosystem to its former, undisturbed state with the original functions and structure. We will explore the background of restoration, its track record, and additional details on why this has become a very active management technique. We will end with a case study on the CALFED collaborative restoration program.

BACKGROUND ON RESTORATION

Humans have caused extensive change in landscapes all over the world (Turner 2010). More recently, people have started to recognize the extent of this change and are beginning to view restoration as a way to regain natural settings from derelict lands and waters, and improve degraded habitats and settings (example of restoration in Figure 8.1) (Higgs 2003). Restoration started becoming popular around 1990 as a new way to practice environmental conservation and has been gaining interest in the last few decades. The National Research Council (1992) defined restoration as: returning an ecosystem to its former, undisturbed state with its original functions and structure. This has become a common definition for restoration. The National Research Council recommends conducting an investigation of potential restoration sites by gathering old maps, finding accounts of the area's history in newspaper articles and elsewhere, and speaking to local residents (National Research Council 1992; Jackson and Hobbs 2009). They also recommend following the practice of integrated resource management, which is management that seeks to restore the structure and function of whole ecosystems by striving to understand and respond holistically to cumulative ecological impacts. A second way is to locate a comparable ecosystem that has not yet experienced degradation and use that as a model environment (Stoddard et al. 2006). A holistic view of the structure and function of the ecosystem is needed for planning restoration activities.

Figure 8.1: Restored roadbed of the former Round Meadow loop road in Giant Forest, Sequoia National Park in 2004. Source: National Park Service 2021

In North America, we often use the Columbian landfall of 1492 as a restoration baseline (Leopold 1963). The pre-1492 landscape is thought of as a near natural environment, where native Americans were considered to be too few in number to have had a significant impact on the landscape. It was also proposed that the European colonization of North America started the major landscape changes and that they, at least in part, caused the extinction of some large animals (Martin 1973). Christopher Columbus ran into people soon after he landed on the shores of San Salvadore (Columbus and Las Casas 1989). There is ongoing debate about the time period in history when humans had the most widespread impact on the environment (Merchant 2005). The further we go back in history, the more we find human impacts on the North American environment (Grayson 2001). Nevertheless, the Columbian baseline is often used to separate landscapes that were transformed by people from those that possessed more natural landscape properties (Bjorkman and Vellend 2010).

Another way to think about restoration is as the practice of some activities that speed up ecosystem change (Hobbs and Cramer 2008). This is not returning an ecosystem to an undisturbed state with its original functions and structure, but rather improving an ecosystem to be more natural, sustainable, and self-regulating. The concept involves restoring the environment to promote more biodiversity and potentially support recovered species (Benayas et al. 2009). The focus is on undoing environmental damage as a way to recover environmental health and function (Figure 8.2). This view is different from natural restoration, which focuses on bringing an ecosystem back to a state of beneficial use, with a high potential for biological recovery. The general goal is to bring back a functioning ecosystem that is able to fit into the current landscape.

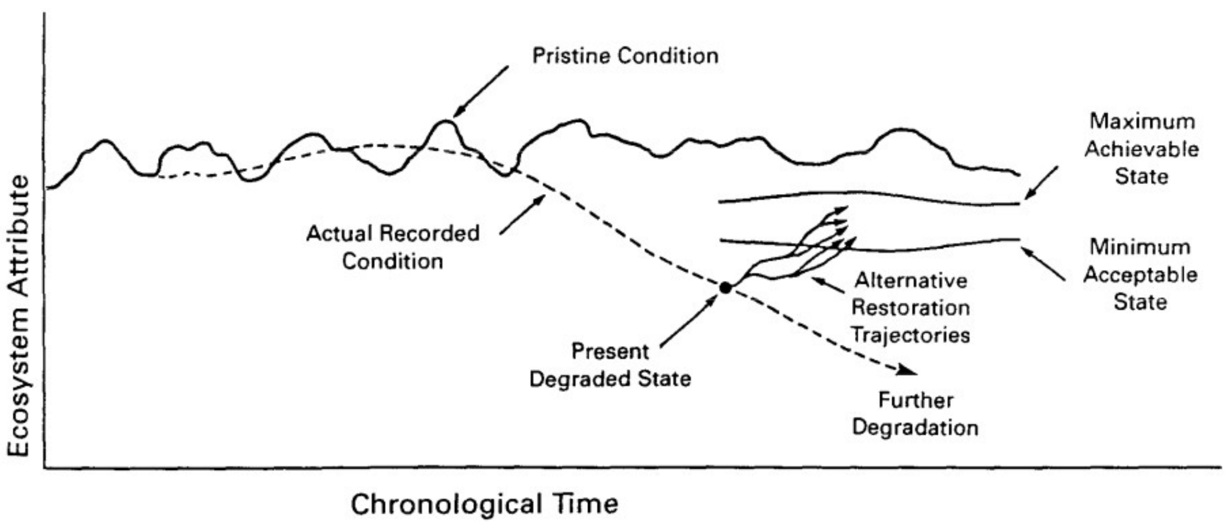

Figure 8.2: Example ecosystem health over time paired with positive impacts of restoration. Source: Cairns et al. 1993

THEORETICAL CONCEPTS BEHIND ECOSYSTEM RESTORATION

Recreating nature is often the aim of ecosystem restoration because we want the ecosystem to function naturally and maintain itself (Figure 8.3) (National Research Council 1992). Often, biological restoration is important to maintain species and assemblages of species that were once prominent. A functional biological community is seen as the primary goal of restoration. If the ecosystem restoration site is isolated from colonization and dispersal sources, then human-assisted transport and release is required. The stability of the community and persistence of its member populations signal restoration success. It is important to reestablish key species that can shape the ecosystem (e.g., herbivores) and interact with a wide range of community members. The presence of herbivores will often shape their habitat's plant cover, which can then affect other functions of the ecosystem. And, carnivores can influence herbivore density and behavior. Also, high species richness can accelerate successful ecosystem restoration (Williams et al. 2017). Increased ecosystem function, like primary and secondary production, is another clear sign of restoration success (Cortina et al. 2006). In addition, natural disturbances (e. g., floods and fires) need to be acceptable because natural disturbances alter the mix of species and foster higher biodiversity through time (Nilsson et al. 2001). Natural disturbances often favor some species while reducing the abundance of others. This process can reset the dominance of competitive species. The emergence of a new biological community, colonization, human-assisted return of species, key species that shape the ecosystem, stability of community, and natural disturbance are all important factors of restoration.

Figure 8.3: Restoration of the Linville River in 2012. Source: North Carolina Forest Service 2021

AN ALTERNATIVE VIEW OF RESTORATION

Traditionally, restoration is aimed at returning an ecosystem to its original, undiminished natural state. Another view of restoration embraces rebuilding ecosystems that have been damaged or destroyed thereby creating ecosystems that are sustainable, which can then return benefits to people (Alexander et al. 2016). This is very different from the traditional view of restoration, but this perspective is gaining acceptance because of climate change, the worldwide transport of species, and altered landscapes. This perspective is focused on recreating an ecosystem which has renewed biological function and ecosystem services. The idea is not to restore the ecosystem's original condition, but to improve it by assisting its recovery (Jones and Schmitz 2009). Ecosystems are highly dynamic and change over time. Thus, the aim is to create a novel ecosystem at the original location, but one which differs from its previous state. The idea is to make the new ecosystem stable and highly resilient. Restoration plans and environmental managers would need to consider what is practical for the site, and develop a realistic plan to improve the ecosystem based on informed conclusions.

Ecosystems are not static, and often change. These changes may be brought about by external factors like climate, and they can also arise internally from different mixes of species. Change is an important ecosystem dynamic because it frequently favors some species over others which can alter the mix of species and promote greater biodiversity. These effects on the mix of present species promotes high biodiversity. Much of restoration planning views the ecosystem as static and is backward looking in that it aims for a fixed, desirable state from the past. The common expectation is a return to a set state. Note, it is not feasible to restore some settings to their original condition, such as mined lands and urban brownfields. In reality, societal interests shape restoration endeavors and ultimately determine the amount of funding and effort expended on the site. Consequently, societal interests must be addressed when planning an ecosystem-scale restoration effort since biocultural and eco-societal interests are an important part of restoration (Cairns 1995). Most restorations are financed through government programs, and the need for public support is essential. Also, public engagement and an image of the desired restoration endpoint help to generate and sustain effective restoration efforts (Palmer et al. 1997; Cairns 2000; Hobbs and Harris 2001; Hobbs 2007; Miller and Hobbs 2007; Hobbs et al. 2010). However, the public often finds it difficult to accept traditional restoration goals like structural properties, biological composition, and functional durability. In public discussions, a healthy environment is often the aim, but the attributes of a healthy environment such as vigor, organization, and resilience can seem arcane and challenging to explain. Some restorations fail because it may take years to determine whether changes are progressing in the intended direction. Many years may be needed to achieve that vision of restoration success, and along the way, deviations from the projected course can occur. At the same time, the public and local leaders are often too impatient to see tangible results from the restoration process. For restoration efforts to be effective, integrative, multi-disciplinary practices are needed and such practices must recognize and account for the social dimensions that inevitably attend such endeavors (Zweig and Kitchens 2010).

THE ROLE OF HABITAT IN RESTORATION

Recreating and diversifying habitat has become a leading strategy for ecosystem restoration. Habitat can be designed to support specific species, have a precise form, and create a visual image (Figure 8.4). Restoration activities create habitats, which are tangible resources. Diversifying habitat is believed to enhance biodiversity (Kremen 2020). Habitat restoration targets biological improvement which can increase biodiversity. However, there has been an inconsistent record of habitat restoration supporting improved biological communities, especially in rivers and streams (Scrimgeour et al. 2013). Maximizing habitat heterogeneity also has a mixed record in enhancing biodiversity (Denslow 1995; Palmer et al. 2010).

A common belief is that the general surroundings of the restoration site limit what can be realized with habitat improvements. Urban restoration is popular and typically includes stream and riparian habitats (Ingram 2008). However, little evidence exists that urban restoration can return an ecosystem to a healthy state. The limitations on biota in these ecosystems come from the surrounding area, like air and water pollution, erosion, quick runoff events, and the restrictions imposed by structural habitat. More emphasis needs to be placed on restoration site selection. The site should not be limited by its surroundings, and should be easily linked to a source of colonists which can respond to the habitat. Creating habitat to restore ecosystems may be alluring but full restoration may ultimately be beyond what can be accomplished. Habitat is required but not entirely sufficient to restore an ecosystem. More planning is required to find holistic solutions to environmental degradation at the site and its surroundings.

Figure 8.4: Before and after photos of a Partners for Fish and Wildlife prairie seeding project in Arkansas. Source: United States Fish and Wildlife Service 2021

THE ESTUARY RESTORATION ACT AND ITS FOCUS ON HABITAT

In 2000, the United States Congress approved the Estuary Restoration Act, which engaged many agencies active in coastal waters (United States Code 2021). The current lead agency is now the National Oceanic and Atmospheric Administration (NOAA). The purposes of Act are to: 1) Promote the restoration of estuary habitat; 2) Develop and implement a National estuary habitat restoration strategy; 3) Provide federal assistance for estuary habitat restoration; and 4) Enhance monitoring and research capabilities. The implementation of the Estuary Restoration Act is an ecosystem-scale restoration effort to improve our estuaries for productive aquatic life and merge agency agendas around a goal.

NOAA promoted an organization called “Restore America’s Estuaries” (Restore America’s Estuaries 2002) and helped pair it with a scientific group called the “Coastal and Estuarine Research Federation” (Coastal and Estuarine Research Federation 2021). The planning process developed by these groups had three main parts (Figure 8.5). First, they planned to evaluate each estuary and its watershed to identify losses in habitat and opportunities for restoring habitats. Second, they planned to establish restoration priorities by identifying habitat needs, linking habitat to species benefits, and incorporating public interests and the economic value of the species. Finally, they sought to develop a plan for restoration which included stakeholder viewpoints, public input, conservation benefits, clear goals, and evaluation. Water quality issues are common challenges in estuaries, and inland dams and diversions can block the spawning of fish and other taxa. A broader view is needed, but promoting restoration is easier with tangible resources ahead of the argument.

NOAA and the United States Congress focused on habitat as a clear path to improvement. For estuaries, restoration is really about habitat. Habitat is generally thought of as the physical conditions needed for a species. It is possible to look at, photograph, and inventory habitat, so it is considered a tangible outcome of restoration activities. Habitat is linked to species and assemblages of species that live in the same region. Thus, it is important to target the needs of species or assemblages of species for restoration. Another common idea is that we can create a set of habitats that are persistent within the estuaries. However, this approach is not holistic and ecosystem based, so it would not return the environmental system to a sustainable condition.

SUMMARY OF PLANNING PROCESS

Evaluating a Watershed or Estuary

1. Evaluate current status of habitat
2. Describe causes and rates of decline in habitats
3. Identify services provided by habitat
4. Evaluate opportunities to restore habitats in the system

Establishing Restoration Priorities

1. Severity of need (scarceness of habitat/threat to habitat or species)
2. Ecological benefits provided by the habitat or species
3. Chances of successfully restoring the habitat or species
4. Public support for restoration of the habitat or species
5. Social and economic benefits provided by the habitat or species

Establishing a Plan for Restoration

1. Consider multiple stakeholder viewpoints
2. Establish an open and public process
3. Make a strong link to conservation and protection efforts
4. Document restoration goals—identify areas, habitats and species in the region for priority restoration and protection (identify how ongoing restoration programs and efforts can be linked together)
5. Revisit and revise the plan as needed after monitoring

Figure 8.5: Restoration planning process of Restore America's Estuaries. Source: Restore America's Estuaries 2002

EVALUATING RESTORATION SUCCESS

River and stream restoration has become a world-wide phenomenon as well as a booming enterprise (Palmer et al. 2005). Billions of dollars are being spent in the United States alone. Although there is growing consensus about the importance of river restoration, agreement on what constitutes a successful restoration project continues to be under discussion. Thus, much research on ecosystem restoration has been aimed at evaluating its success or failure (Ruiz-Jaen and Aide 2005; Zedler 2007). It is hard to measure whether restoration reestablishes the state, structure, and functions an ecosystem formerly possessed. Methods are being developed to determine what ecosystem restoration is accomplishing. However, data on the original ecosystem state rarely exist. Therefore, historical investigation processes are being used with paleoecological methods to reconstruct information about past ecosystems and create measures for ecosystem stability (National Research Council 1992). A second path is to find a reference ecosystem that has not been changed (Stoddard et al. 2006) and use that ecosystem as a model for planning the restoration project, and afterward for its evaluation. This option allows the restored ecosystem ‘s structure and function to be measured and characterized against benchmarks from the reference ecosystem. Then, the target ecosystem can be evaluated for needs and a plan developed for restoration activities. The problem with a simple reference is that it represents a single state or expression of ecosystem attributes. Either way, the focus is on emulating past or reference ecosystem conditions. Evaluation to document success at restoration is complicated and research is ongoing to create standardized evaluation criteria for the measurement of success.

Palmer et al. (2005) proposed five criteria for evaluation of ecological success by restoration efforts (Table 8.1). First, a guiding image of an ecological endpoint can be developed by historical reconstruction, reference site emulation, and expert formulation. Second, ecosystems are improved by clear progress toward the guiding image. Improvements can come in a range of levels and affect different ecosystem components, but real progress is linked to restoration activities. Restoration goals have been attained when defined ecological and stakeholder outcomes have been met and future efforts benefit from the understanding gained. Third, resilience is generally increased in order to achieve self-sustainable ecosystems. Measures of resilience are debated in the field of ecology but with time, resilient ecosystems maintain their properties and functions. Restoration measures should show a capacity to remain much like the guiding image despite stress and disturbance. Unless some level of resilience is restored, projects are likely to require on-going management and repair, the very antithesis of self-sustainability. Fourth, no lasting harm is done by restoration activities. Restoration is an intervention that causes impacts to the system, which may be extreme (e.g., changes in soil conditions, invasive plants, alterations of the surface topography). Finally, an ecological assessment is completed and data show improvements to the ecosystem. Well-documented projects that fall short of objectives may contribute to the future health of our landscapes through learning. These five measures are important to judge whether restoration activities have improved the ecological status of the ecosystem.

Table 8.1: Criteria for ecologically successful river restoration. Source: Palmer et al. 2005

Criteria	Guidelines
Guiding image of dynamic state	The guiding image should take into account the average condition, range, or some fixed value of key system variables; an ecological endpoint has been identified
Ecosystems are improved	Appropriate indicators of ecological integrity or ecosystem health should be selected based on relevant system attributes.
Resilience is increased	System should require minimal on-going intervention and have the capacity to recover from natural disturbances.
No lasting harm	Pre- and post-project monitoring of selected ecosystem indicators should demonstrate the impacts of the restoration intervention.
Ecological assessment	Ecological goals should be clearly specified, with evidence available that post-restoration met the goals.

The Society for Ecological Restoration International (Science and Policy Working Group 2004) defines restoration as a process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. They proposed nine attributes to use in determining if restoration has been accomplished (Table 8.2). First, the restored ecosystem has a characteristic biological assemblage such as grasslands, antelope, or prominent herbivores. Second, the ecosystem consists of indigenous species to the greatest extent possible. Native species should be easily seen and common. Third, all functional taxon groups for development and stability of the ecosystem are present. Carnivores have been shown to reduce the densities of herbivores which then shifts the plant cover of the ecosystem (Ripple and Beschta 2007). Carnivores are a functional group that has to be maintained to promote the plant cover that is characteristic of the ecosystem. Fourth, the physical environment of the restored ecosystem is capable of sustaining reproducing populations of the species necessary for the species' continued persistence in the ecosystem. Fifth. the restored ecosystem functions normally and any signs of dysfunction are absent such as non-native dominated plant cover. Sixth, the ecosystem is suitably integrated into the larger landscape and interacts with it through abiotic and biotic flows and exchanges. The general idea is that the restored ecosystem fits into the landscape and exchanges animals, plants, water, and other chemicals with the landscape. Seven, potential threats to the integrity of the restored ecosystem from the surrounding landscape have been eliminated or reduced. Threats include activities like collecting plants and animals, spraying insecticides broadly, groundwater depletion, and others. Eight, the ecosystem is sufficiently resilient to maintain normal properties following periodic stress events like floods, high temperature periods, and droughts. Finally, the restored ecosystem is self-sustaining and has the potential to persist indefinitely under existing environmental conditions. Characterization of the nine attributes can show whether or not restoration progress is being made. If the site is showing clear signs of progress for all criteria, the restoration has accomplished its goal of improved environmental conditions for the ecosystem.

Table 8.2: Criteria for determining when restoration has been accomplished. Source: Science and Policy Working Group 2004

Criteria	Guidelines
Characteristic assemblage	Restored ecosystem contains appropriate community structure.
Indigenous species to the greatest practicable	Native species are common, and no abundance of ruderals and segetals plants.
Functional groups necessary for the continued stability	Functional groups are present or they have the potential to colonize.
Ecosystem is sustaining reproducing populations	Species necessary for its continued stability or development along the desired trajectory.
Ecosystem functions are normal	Signs of dysfunction are absent.
Ecosystem is integrated into a larger landscape	Interacts through abiotic and biotic flows and exchanges.
Improved environmental health and integrity.	Potential threats from the surrounding landscape have been eliminated or reduced to negligible.
Ecosystem resilient to normal periodic stress events	Ecosystem is able to maintain its integrity.
Ecosystem is self-sustaining	Potential to persist indefinitely under existing conditions.

PROGRESS OF RESTORATION EFFORTS

Several reviews of restoration programs provide some insight into the progress of restoration efforts (Bernhardt et al. 2005; Bernhardt and Palmer 2011). A National river restoration synthesis (Bernhardt et al. 2005) was developed which included a database of more than 37,000 restoration projects (Figure 8.6). The United States spent $15 billion to restore streams and rivers during the period from 1990 to 2003, which is over $1 billion a year. The review discovered that funds came from many sources and that substantial societal investment in restoration was occurring. The review only covered rivers and streams though restoration efforts were occurring in other ecosystems. The most common goals for river restoration improvements were for enhanced water quality, management of riparian zones, improved stream habitat, fish passage, and bank stabilization. Approximately 20% of the projects had no goals, and many more projects stated such brief purposes that it was hard to determine whether projects were undertaken to restore stream ecosystems or were merely river manipulation projects (e.g., bank stabilization). Most restoration programs were very small with projects costing less than $45,000, thus many of these projects had narrow interests. Most projects (90%) did not have a plan for evaluation of the effects of restoration activities (e.g., no assessment or monitoring plan). This account gives a picture of what restoration is like in practice because it spans many projects over a decadal time period.

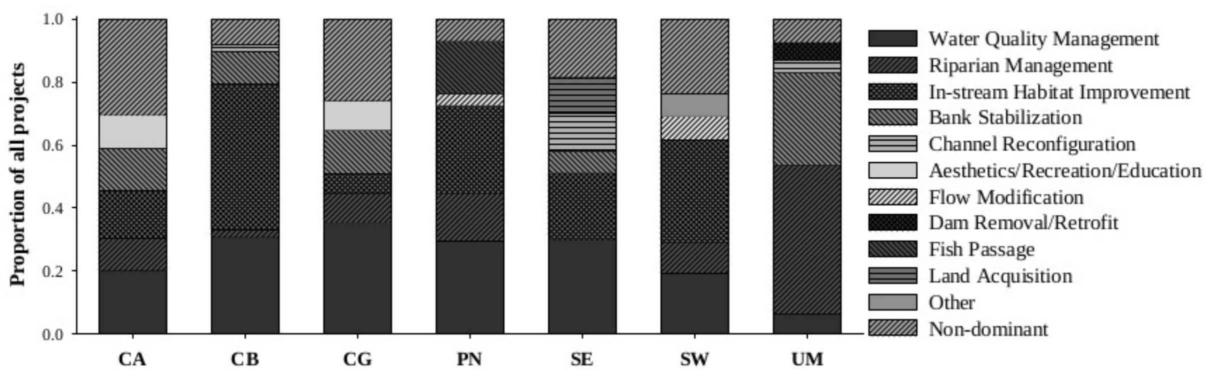

Figure 8.6: Regional differences in the distribution of types of restoration efforts. To facilitate visual comparison only the top five intent categories for each node are shown in each stacked column. All other "non-dominant" intents are summed as part of the "non-dominant" category. Source: Bernhardt et al. 2005

The Natural Heritage programs at the state level often aim to bring back endangered species or rare communities (e. g., marsh birds). Restoration of species is the central target of 25% of the projects; 30% address ecosystems and landscapes (Ehrenfeld 2000). The remaining 45% cover a variety of goals for eliminating limitations on species and the diversity of communities (Ehrenfeld 2000). Thus, the public has to be invested in restoration and the mission has to be broader. The restoration of numerous species at a site requires a diversity of habitats. A broad agenda for restoration supports the idea that the entire ecosystem needs to be healthy to support a diversity of species (Stranko et al. 2012).

Many gaps still exist in terms of restoration progress (Young et al. 2005; Christian-Smith and Merenlender 2010). Ladouceur and Shackelford (2021) issued a call for a global collation of restoration data so that knowledge gaps could be addressed and data synthesized to advance toward a more predictive science that could inform restoration efforts and assist in achieving more consistent restoration success.

CASE STUDY: CALFED COLLABORATIVE RESTORATION PROGRAM

California and the United States Federal government came together to form the CALFED Bay-Delta Program in 1994 (CALFED Bay-Delta Program 2021). CALFED was an ambitious, collaborative, ecosystem restoration program. At the point of its inception, it was the largest and most comprehensive collaborative water management program in the United States. It was centered where the Sacramento and San Joaquin rivers come together at sea level and form a delta that flows into the San Francisco Bay (Figure 8.7). The mission of the CALFED Bay-Delta Program was (CALFED Bay-Delta Program 2021): to develop a long-term comprehensive plan that will restore ecological health and improve water management for beneficial uses of the Bay-Delta system.

The delta is at the hub of the California water system where the northern waters meet the southern waters. Much of the delta's waters are pumped from the delta and sent south for municipal and agricultural use. The delta has intensive agricultural land uses and a diverse recreational economy, and the area is urbanizing. The plan for CALFED came from an environmental impact statement and report to meet the requirements of the National Environmental Policy Act and the California Environmental Quality Act. The record of decision (CALFED Bay-Delta Program 2000) identified a plan and objectives for managing the bay-delta environment and water together. There were four central objectives for CALFED to pursue (Table 8.3). First, to restore the health of the estuary ecosystem. Second, to improve water supply reliability. Third, to improve water quality in the delta waters. Fourth,

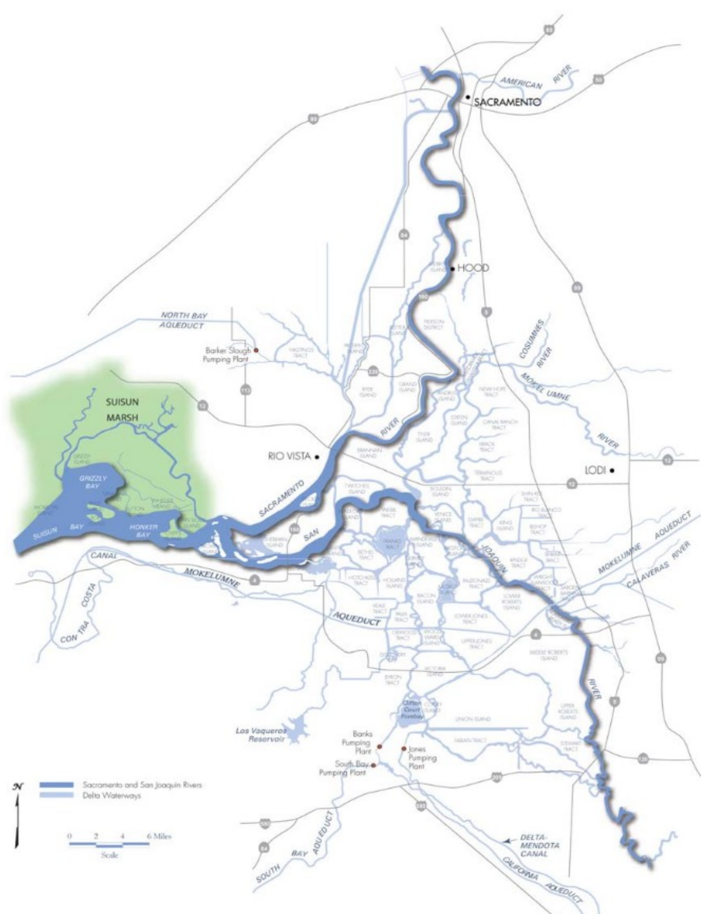

Figure 8.7: Map of the Sacramento-San Joaquin River Delta showing islands, waterways, and significant infrastructure. Source: Delta Vision Blue Ribbon Task Force 2008

to maintain levees. The distinct feature of this ecosystem restoration effort was to balance the water needs of the environment with those of the people.

Many Californians depend on water supplies from the delta. The San Joaquin Delta's freshwater furnishes municipal water supplies for approximately 22 million Californians (Dutterer and Margerum 2015). The California central valley has some of the most productive farmland in the world, and the area which relies on delta waters is close to 3 million acres (Healey et al. 2008). These numbers were hard to grasp, and California and the federal government prepared to invest billions of dollars in CALFED activities. There are 1,700 km of earthen levees (Figure 8.8) in the delta that control channel dimensions and water flows, and protect land that's used for agriculture or urban development (Sherman et al. 2004). Most precipitation in California falls north of the delta, while most water use is well south of the delta. Improving levees were an objective for CALFED because they are vulnerable to flooding, earthquakes, and sea level rise. Mean sea level has gone up, and a warming climate is making the high elevation snow melt earlier (Gornitz 1995; Stewart et al. 2004). There are reservoirs well north of the delta to store water and pumps to extract water in the southern part of the delta that discharge into canals to take the water even further south. All these numbers supported the big challenge the CALFED objectives were intended to meet.

Figure 8.8: An example of farmland in the bay-delta area. Note all land in this image is surrounded by levees. Source: Google Maps 2011

Table 8.3: The four objectives of the CALFED assignment and the drivers of change in the Bay-Delta that CALFED was charged with accommodating. Sources: CALFED Bay-Delta Program 2000 and Mount et al. 2006

CALFED assignment (CALFED Bay-Delta Program 2000)	
	Restore the ecological health of the Bay-Delta estuary
	Improve the water supply reliability for California's farms and cities
	Protect the drinking water quality
	Protect the Delta levees that ensure its integrity
Bay-Delta drivers of change (Mount et al. 2006)	
	Subsidence: Gradual sinking of landforms.
	Sea Level Rise: Sea level has been rising and is expected to continue to rise.
	Regional Climate Change: California has been warming and will continue to warm.
	Seismicity: Earthquake activity or the occurrence of earth tremors.
	Exotic Species: Bay-Delta is one of the world's most invaded estuaries.
	Population Growth and Urbanization: California's population is growing.

Geology, climate, and human activity make the delta environment an ever changing place. Mount et al. (2006) characterized the drivers of change in the ecosystem (Table 8.3). The gradual process of subsidence of the land has progressed in some areas as much as 5 m below sea level. Sea level rise aggravates this problem and has been increasing (Gornitz 1995), making the water nearer to levee capacity and altering mixing by changing the tidal processes and channel hydrodynamics. Levees are vulnerable to the slower process of subsidence and sea level rise, and are very vulnerable to instant change from earthquakes as the delta is a high seismic activity area (Service 2007; Burton and Cutter 2008). In addition, the climate has been warming in California, which changes the timing of snow melt in the Sierra Nevada Mountains which in turn affects agricultural schedules (Stewart et al. 2004). Climate warming also has the potential to disrupt current water use (Cloern et al. 2011). People like being connected with area waters and channels. Consequently, the delta has been rapidly urbanizing (Figure 8.9) and this process changes the land cover, runoff rates, and the pollutant concentrations of surface waters (Jordan et al. 2014). The final alarming change is the rise in number of exotic species in the area (Cohen and Carlton 1998). California has relatively newly settled land and waters in comparison to patterns worldwide. Therefore, the state was depauperate in terms of biota, especially in the freshwaters, and new species invaded rapidly. In the delta, non-native species comprised 40-100% of common species, 97% of total animal numbers, and 99% of the biomass (Mount et al. 2006). All these factors indicated a novel ecosystem that needed to be restored to accommodate an estuary where change was anticipated.

CALFED faced many challenges in restoring the delta ecosystem to a better status. Many of the issues were compiled into nine clusters related to: water quality and pollution, flows, water use and storage, levee functionality and sea level rise, habitat restoration, invasive species, pelagic organism decline, migratory birds, and economic development (Table 8.4). Changes in the delta meant that it was hard for CALFED to predict management outcomes. CALFED promoted watershed conservation and best practices for agricultural lands in the area to improve water quality and reduce mercury coming from tributary watersheds. The purpose was to meet standards for drinking water, agricultural use, and ecosystem needs. Issues concerning in-flowing waters to the delta were integrated into the management plans for water across the basins. Fish migrations and spring delta water outflows needed to be higher, and channel and estuary water circulation within the ecosystem needed to be improved. Water management procedures involving storage, conveyance, and pumps posed a threat to aquatic life. The Sacramento-San Joaquin Bay-Delta water system was likely at or near its capacity. Thus, tradeoffs that benefited the environment would end up reducing water volumes for human use, making it hard to decide which outcome to prioritize. Levee vulnerability needed attention as well since breaks in the system protecting farmland and infrastructure could impact the regional economy. The habitat creation plan incorporated space and flow requirements to address the lack of interconnections, as well as accommodate both tidal and floodplain needs. Science provided the knowledge that was needed to plan for these actions, but new areas of uncertainty then emerged. A species of fish using open pelagic waters sharply declined, though the cause was not clear (e.g., contamination, an invading species, entrainment in water pumps). The CALFED agencies needed to integrate the objectives, activities, and anticipated outcomes of the project to manage the whole ecosystem.

An independent review was done by Lund et al. (2007) to see how CALFED was executing the assigned objectives. The delta environment was not serving most stakeholders well, and it was vulnerable to change. The CALFED organization tended to go along with the consensus and not make big changes in operations that would disappoint some stakeholders. The lack of bold decision-making meant that alternative operations which were predicted to accomplish some objectives went overlooked. CALFED was commonly seen as failing to meet all of its objectives. And, its failure to address the challenges introduced an element of risk from imminent changes in this complicated ecosystem. After extensive monitoring, many scientists noted that they still did not understand many of the ecological trends in the San Joaquin Delta (Dutterer and Margerum (2015), and environmental problems remained (Lubell et al. 2013).

Among its successes however, CALFED initiated new communications systems, more integrated management techniques, significant restoration projects, an increase in monitoring and data collection activities, and some significant shifts in conceptualizing water management (Dutterer and Margerum 2015). CALFED components like the Environmental Water Account and Science Program have been deemed relatively successful (Lubell et al. 2013).

In total, the CALFED program spent $3 billion on research, environmental restoration, and administration before dissolving in 2007 (Dutterer and Margerum 2015). A meta-analysis of CALFED sought to determine the reasons for its dissolution (Dutterer and Margerum 2015). Their findings identified limitations related to problem, societal, and policy context; also highlighted were different interpretations about politics, leadership, and governance arrangements (Nawi and Brandt 2008; Dutterer and Margerum 2015). The lessons from CALFED include the limitations of adaptive management, the risk of dependence on political leadership, the challenges of an informal structure, and the flaws in CALFED's efforts to create a more formal structure (Dutterer and Margerum 2015).

Table 8.4: Issues for CALFED.

Watershed conservation and restoration Improve water quality for drinking water, agriculture, and ecosystem Reduce mercury entering the Delta from tributary watersheds Support best agriculture practices on Delta farmlands
Provide adequate flows at the right times to support fish migrations Restore Delta flows and channels to support a healthy estuary Increase spring Delta outflow Reconfigure Delta geometry to increase estuarine circulation patterns Justify environmental water account
Improving water supply reliability Reduce fish kills in Delta pumps Relocate water intakes away from sensitive habitats Storage projects Conveyance improvements Water use efficiency promotion
Levee system integrity Levee improvements Protect grasslands and farmlands throughout the Delta Sea level rise accommodation
Restore large areas of interconnected habitats Restore tidal habitats Support rearing habitat for resident native fish Increase floodplain inundation and establish new floodplains
Control harmful invasive species
Reverse pelagic organism decline
Accommodated millions of migrating birds for pass-through or over-winter
Integrated agency actions Regional economic development plans

The CALFED case study involved many scientists, from a wide range of disciplines, who were engaged in planning a true ecosystem restoration and other tasks. Their ecosystem project was complex with a lengthy list of issues. Their mandate to balance the water use needs of people with those of the environment proved challenging. Also, this ecosystem is changing gradually and has generated surprises, like a sharp decline in pelagic fishes. The delta restoration was supposed to function naturally and maintain itself as an estuary ecosystem. Some of those goals were met while others were not. Overall, restoration, especially for an entire ecosystem, is complex and time is often needed to accomplish its objectives.

SUMMARY

Humans have caused extensive change and sometimes damage in landscapes all over the world. Restoration helps to return these landscapes to more natural, undisturbed states with their original functions and structure. Many years may be needed to achieve the image of restoration success, and deviations along the way may occur. To be effective, integrative, multi-disciplinary practices are needed along with stakeholder input.

Figure 8.9: An example of an urbanizing area in the bay-delta. Source: Google Maps 2011

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Habitat-Focused Techniques

Chapter 9 - Ecological Engineering

The last topic in the habitat-focused techniques group centers on the idea of ecological engineering. Ecological engineering is different from restoration because it is focused on designing and reconstructing environments consistent with ecological principles, but it also attempts to integrate the needs of human society within its natural environment. We will explore the background of ecological engineering within a framework of ecological stress and health, and delve into the ideas that set ecological engineering apart from restoration. We will end with a case study on the Chicago waterfront ecological engineering project.

A SHIFT IN THINKING FROM RESTORATION TO ECOLOGICAL ENGINEERING

Human domination of the earth has become widespread and pervasive (Vitousek et al. 1997). Alteration of ecosystems by direct changes such as land cover replacement, introduction of new species, pollution, and development has resulted in new ecosystems unlike those in their original state (Ellis 2011). We have come to recognize that indirect human-motivated changes such as climate change, water use and rerouting, and the decline of biodiversity are changing the natural world (Braun 2020). Generally, these direct and indirect influences of humans are creating new ecosystems that are novel in properties, possess new biological communities, and support people. Much of earth is already a merger of natural and human processes, and we must understand and manage these novel ecosystems for both natural and human benefits (Hobbs et al. 2006). These new ecosystems have to be understood and often recreated from damaged and abandoned lands and waters.

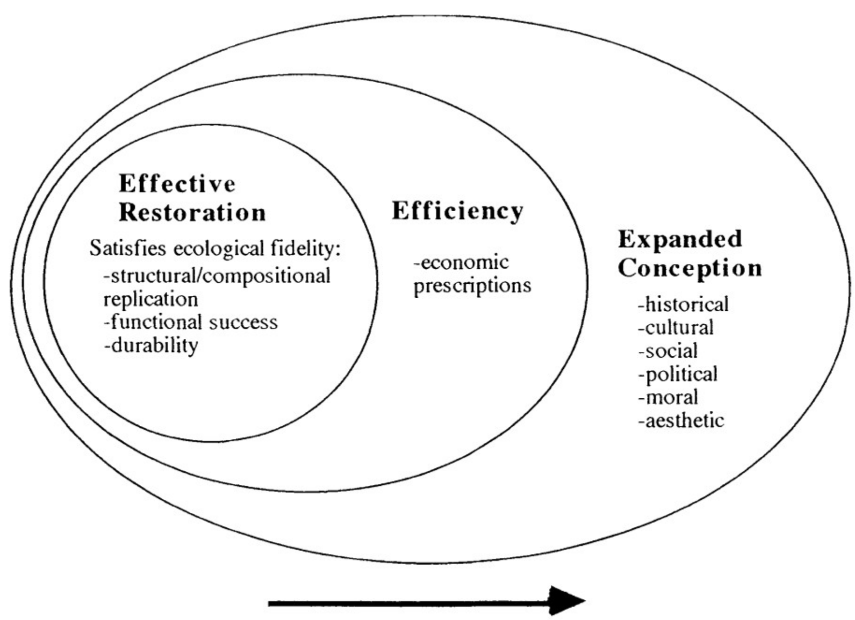

Figure 9.1: Shift from restoration to ecological engineering. Source: Higgs 1997

Restoration has been the typical method used in the past to improve degraded ecosystems. Restoration applies a static approach to dynamic ecosystems to shift the site to a fixed state (e.g., an ecosystem or habitat that once existed in the past). The field of restoration has received criticism from within and beyond the professional restoration community as being impractical because of irreplaceable losses, irreversible changes, and for being backward looking. Further, restoration often progresses on an ad hoc, site- and situation-specific basis and proceeds with considerable subjectivity in determining restoration goals. Goals are often idealistic and not feasible under prevailing economic, social, and political circumstances. Restoration is viewed as an " art " rather than a science and often relies upon intuition rather than a documented knowledge base (Choi et al. 2008).

More recently there has been a shift away from site- and situation-specific restoration projects to a broader approach (Hobbs 2004). The shift (Figure 9.1) has been toward forward-looking restoration practices, acknowledging a changing and unpredictable environment, assuming ecological communities are dynamic in nature, connecting to landscape elements, and seeking public support and community participation in setting realistic goals (Choi et al. 2008). The new processes driving ecological engineering build on direct links to human interests for support from people, and are economically, ethically, socially, and politically acceptable, and well-justified in these terms (Choi 2004; Choi et al. 2008).

Ecological engineering has been going on for years under specialties titled landscape rehabilitation, nature engineering, renaturing, ecotechnology, biomanipulation, ecological restoration, reconciliation ecology, designer ecosystems, invented nature, and artificial ecology. In general, natural-human ecosystems have not been studied much, although there is now a strong trend in this direction.

There are common elements in ecological engineering such as a move away from the thinking that people and nature are incompatible. Instead, human interests and ecological benefits are used to define objectives for restoring a site to meet cultural and ecological interests. Social and aesthetic interests often influence the design objectives through community participation in planning. One common goal is to restore healthy relationships between residents and natural spaces by creating harmony between human and ecological activities. The kinds of features that are often restored are initially raised by interested stakeholders. Aesthetic qualities are important in strengthening public support for maintaining the restoration. A good ecosystem plan may rely on providing natural services for human activities, and maintaining desired properties in support of societal values.

The process of ecological engineering helps to set realistic objectives that address ecological, economic, and social benefits where copying attributes of a natural system has little relevance. Creative practices are especially needed in highly stressed ecosystems that are not expected to return to a near-natural state. Plans commonly consider how the ecosystem will function and persist in the near-term and long-term future. Merging ecological and social considerations with engineering practices differentiates this approach from past restoration approaches with respect to damaged or abandoned environmental settings.

DEFINITION AND HISTORICAL BASIS FOR ECOLOGICAL ENGINEERING

Ecological engineering has been defined in various ways but the core concepts involve designing and reconstructing environments for nature and people (Mitsch and Jørgensen 1989; Jackson et al. 1995; Mitsch and Jørgensen 2003; Odum and Odum 2003). The sustainability of these novel environments is also a priority (Costanza 2012). The goal is to achieve persistent ecosystems that serve the needs of both people and nature in an efficient manner. The most common definition of ecological engineering was provided by Mitsch (1996) as the design of sustainable systems, consistent with ecological principles, which integrate human society with its natural environment for the benefit of both. The key elements of the definition are that engineering practices need to be grounded in ecological science, focused on human interactions with the natural ecosystem, designed for both people and nature, and take into account both human and natural values (Figure 9.2). Overall, ecological engineering has grown in importance in environmental management because of the increasing need to address human welfare and the natural environment in settings that are damaged or dominated by humans (Kareiva et al. 2007).

Figure 9.2: Application of ecological engineering principles. In the desert southwest, natural stream flow (top) varies but may increase substantially after large summer rainfall events. A common past solution has been to convert stream channels to concrete culverts (middle). This reduces economic loss from flooding but provides few other ecological, social, or economic benefits. An alternative to concrete is an ecological engineering solution, such as Indian Bend Wash in Scottsdale, AZ (bottom), in which vegetated pathways and wetlands minimize flood damage, improve water quality, enhance surrounding land values, and create a park-like environment for recreational activities. Source: Palmer et al. 2004

This relatively new approach to the practice of environmental management was introduced in the United States during the 1960s by Howard T. Odum and in China by ecologist Ma Shijun. Both scientists based their original ideas on maintaining ecosystem dynamics and cycling of energy and substances. Twenty years later these ideas also emerged in Europe as a technological approach for ecosystem management with a strong basis in ecological understanding. This field is a good fit for both ecologists and engineers that are designing and constructing ecosystems in human-dominated settings, such as natural resource specialists, environmental and civil engineers, agroecologists, and landscape planners. The field also provides ample material for scientists and managers who are striving to restore environments that have been substantially degraded by human activities, and environments that must blend human and ecological values.

FUNDAMENTAL ECOLOGICAL ENGINEERING PRINCIPLES

Research aimed at ecological engineering has been less developed than other techniques used for environmental management and has been scattered across different fields. Ecologists tend to focus their research on natural or wild ecosystems that can provide information on the dynamics of nature. Some applied sciences such as forestry, agriculture, and landscape planning focus on the control and maintenance of productive ecosystems for direct human benefits. In the middle are ecosystems that are neither completely natural nor entirely controlled for commodity production. These are the types of ecosystems that are of interest in ecological engineering. These are ecosystems that may have been damaged so much that they can no longer revert to a natural state, become self-organized into an ecologically desirable condition, or be used for human commodity production without substantial efforts. Other ecosystems that are of interest to ecological engineering are those that have the capacity to support natural communities with mitigation activity, highly managed systems for waste treatment, abandoned lands and waters, newly constructed habitats such as wetlands, and environments constructed for housing people which still retain some natural attributes. The science aimed at studying these ecosystems tends to focus on ecosystem stress and environmental health, independent of natural conditions. Concepts that can define ecosystem quality are resilience, resistance to change, self-organization capacity, and a -diverse structure. There is a need for a framework that defines the successful creation and improvement of ecosystems that provide persistent benefits for both nature and people. This has not been accomplished yet but researchers' interest in pursuing this vision is now growing.

The incorporation of ecological principles into ecosystem engineering is what distinguishes this environmental management technique from standard engineering practices (Example in Figure 9.3). Engineering designs traditionally aim for a static end product that will provide benefits to people and maintain operations. Ecosystems are dynamic, meaning they vary through time, but persist within a range that defines system behaviors. One principle of ecological engineering is shifting from classic engineering designs to a looser ecological design that recognizes a flexible functional space (Bergen et al. 2001). That is, ecological engineering has to acknowledge that ecosystems do need space to vary, but not so much that the ecosystem transitions to a new stable status. Wide tolerances and multiple natural components have to be specified, as well as measures that keep the system's dynamics within a range that is seen as acceptable. Redundancy in ecosystem structures (e.g., food chains) is one well recognized way to provide built-in resilience to stressors (Costanza and Mageau 1999). In traditional engineering, redundancy is not seen as needed. Thus, the specifications required for ecosystem engineering have to: include varying community composition, ensure that tolerant or undesirable species do not dominate the landscape, provide space and benefits for people, and incorporate redundant natural components for security.

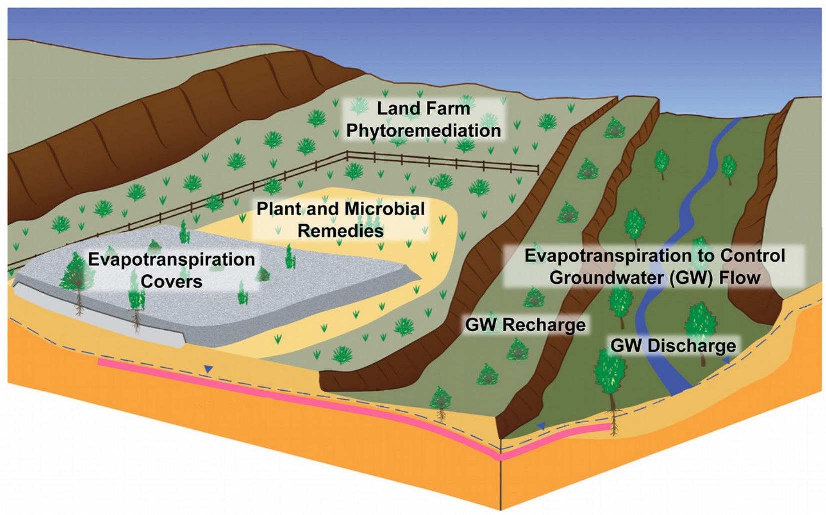

Figure 9.3: Ecological engineering remedies at a uranium processing site to address groundwater contamination issues. Source: United States Department of Energy 2021

Another ecological principle that should be followed is to design and manage projects with efficiency, low-energy inputs, and sustainability in mind. Ecosystems are like any other system which has internal components that interact to shape properties and self-organize. Relying on ecosystem self-organization leads to sustainability, because an emergent system is more likely to remain in an approximate configuration. By contrast, high energy inputs can certainly be used to maintain ecosystem properties, but they are energy intensive, use regular stocking of species, require heavy landscape maintenance, involve retention and diversion of water, utilize extensive construction, and other strategies. Working with natural processes to maintain approximate ecosystem conditions can be less energy intensive, less costly, and promote sustainability in the long-term. This strategy differs from standard engineering practices and calls for a shift in thinking in terms of how natural processes can be used to create and maintain a desired ecological state.

A final principle based on the science of ecosystems and their maintenance focuses on place. Designs should be very context sensitive. Specialized species and habitat-linked communities need full consideration. The physical constraints on what could persist in the ecosystem need attention. Culture, aesthetics, and human values are equally important because local people will be part of the ecosystem and internally shape the outcome through time. Public support and natural constraints are associated with the local context and define what can be achieved and maintained in an ecosystem with diverse benefits.

One criticism of the ecological engineering field is that the strategy or evaluation steps are rarely published to convey lessons for planning future ecological engineering projects. These trends cause this environmental management technique to be poorly documented and to omit common practices. There is a need to learn from past cases, and develop principles for future practices.

ECOSYSTEM STRESS

Stress on ecosystems is an external force that disrupts organization, processes, and functions (Rapport et al. 1985). The effects are internal to the ecosystem but are a response to an external pressure. While not all stresses elicit the same responses, common symptoms have been recognized as attributes of degraded ecosystems. Energetics of ecosystems tend to be balanced with the rate of production and respiration. Under stress, respiration rates commonly increase, and the rate of both production and respiration often increase relative to biomass of plants and animals. Further, energy inputs to the ecosystem become more important, and export of primary production tends to increase. Overall, this is a pattern of less efficient use of internal energy in an ecosystem, and an increased loss of energy beyond the ecosystem. Nutrient cycling shows a similar pattern, where nutrient turnover increases, cycling is disrupted, and nutrient loss increases. Under ecosystem stress, biological community structure changes with a trend toward decreasing organism size, shortening life cycles, and increasing r-strategists (organisms that are opportunistic, fast reproducing, and have a high capacity for population increase). Consequently, food chains simplify, species diversity tends to go down, and dominance of tolerant taxa increases (Schindler et al. 1985). Overall stress on ecosystems commonly simplifies ecosystem structure and functions, increasing the loss of energy and nutrients that power the system. Frequently stressed ecosystems appear in the early stages of building complex biological communities ,with weaker interactions among species, less organized communities, and less diverse flora and fauna. These attributes can mark an ecosystem as in poor condition and in need of mitigation measures to reverse these trends.

Table 9.1: Ecosystem properties expected when normal and stressed. Source:Modified from Pratt 1990

Ecosystem properties	Normal status	Under stress
Structures	Stable r-strategists abundance (fast reproducing, opportunistic)	r-strategists increase in abundance
	Size distribution and life spans of organisms stable and fit theoretical forms	Size distribution and life spans of organisms shifts to smaller size and shorter life cycles
	Food chains and interactions of species are complex and extensive	Food chains and interactions of species become simplified and fragmented
	Species richness as expected for the setting	Species richness is reduced
	Native, sensitive, and specialized species commonly detected	Native, sensitive, and specialized species are in reduced abundance and uncommon
		Generalist and tolerant species increase and become dominant
Functions	The balance in the rates of photosynthesis and respiration are maintained	Respiration increases and becomes higher than the rate of photosynthesis
	Rates of photosynthesis and respiration relative to biomass are stable and in balance	Rates of photosynthesis and respiration relative to biomass increase
	Primary production, energy, and nutrients are efficiently cycled	Primary production, energy, and nutrients increasingly leave the ecosystem and are loosely cycled

Drawing from representative empirical studies about the impacts of stress on the structure and function of ecosystems, Rapport et al. (1985), Odum (1985), Schindler (1987), Schaeffer et al. (1988), Pratt (1990), and Davies and Jackson (2006) identify a number of common stages of ecosystem responses to stress (Table 9.1). *The first signs of ecosystem disruption are often seen as a reduction or loss of long-lived, large species* and dominance by short-lived and opportunistic species. *These species are most susceptible to habitat loss and fragmentation, prey reductions, and the accumulation of toxins.* Additionally, there will be a marked change in community size structure and a loss of sensitive and specialized species. Second, the ecosystem will experience changes in primary productivity linked to nutrient availability. Afterward, redistribution of abundance and biomass will occur in some groups of taxa linked to altered habitats, and species diversity tends to be reduced. Often, tolerant and generalist species increase at the expense of sensitive and specialized species. In the end, the ecosystem will break down and transition to a new system with different properties. These signs of change in ecosystem properties signal degradation of the living environment, and a likely transition to a new ecosystem state.

An analytic model of ecosystem change in response to stressors has been developed by an expert group led by Davies and Jackson (2006). Called a biological condition gradient, this model organizes changes in ecosystem structure and function to characterize the ecosystem status (Figure 9.4). The model synthesizes ecosystem properties into six phases ranging from undisturbed, or natural ecosystem conditions, to severely altered environments with major loss of ecosystem structure and function. This method builds on the characteristics of stressed ecosystems that have been long recognized but not organized into step-wise phases. The sequence of changes starts with reductions in sensitive and specialized species followed by a clear increase in tolerant and opportunistic species. As this community transitions from native and sensitive species to tolerant species, the change greatly alters community composition and leads to new food webs, species interactions, community size structure, and life cycles. These changes result in altered ecosystem functions like energy and nutrient cycling, biomass accumulation, and efficiency of resource retention. This sequence of changes in ecosystem structure and function depicts the degradation of ecosystems, and the decline of environmental quality which should be avoided.

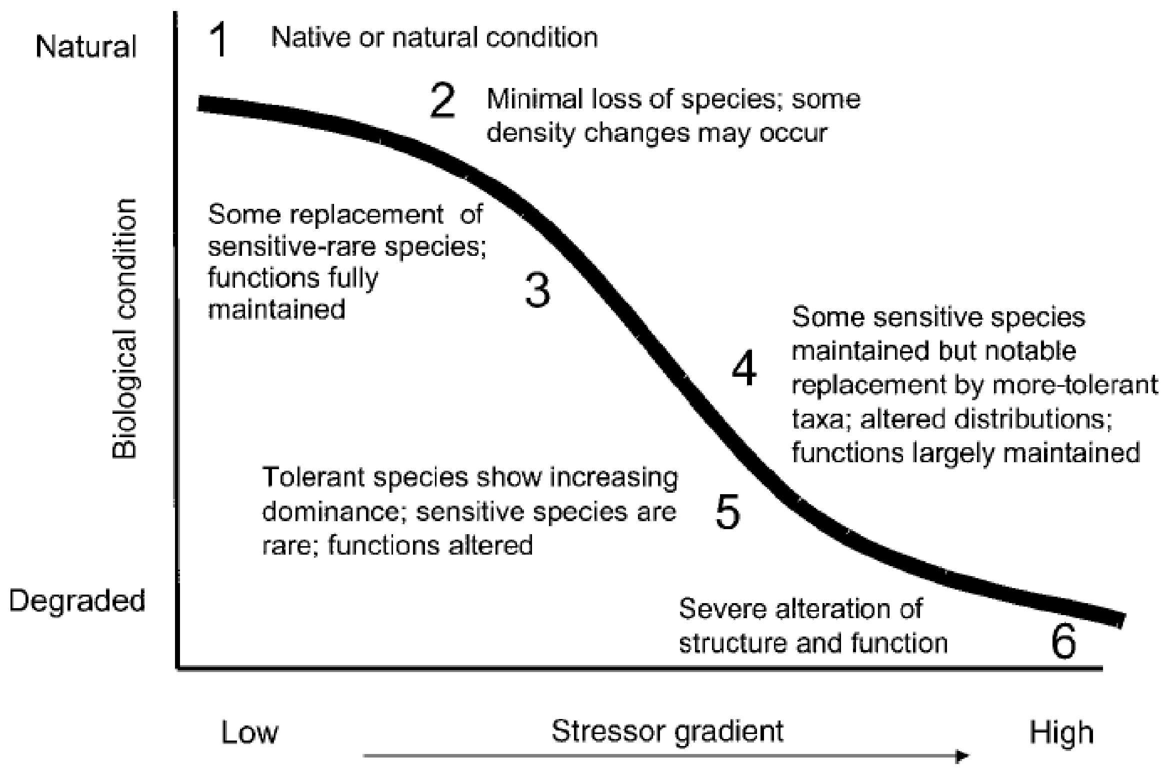

Figure 9.4: The biological condition gradient that shows the progression of ecosystem changes with increasing stressors resulting in a degraded ecosystem. Source: Davies and Jackson 2006

ECOSYSTEM HEALTH

With the recognition of the stress and degradation of ecosystems, ecological engineering needs to remedy such environmental distress to foster healthier landscapes. Ecosystem health is a term that is convenient because it conveys good conditions in the environment (Schaeffer et al. 1988; Rapport et al. 1998a; Rapport et al. 1998B; Costanza and Mageau 1999; Rapport and Singh 2006; Costanza 2012). The public often says they want a healthy environment, so maintaining ecosystem health is a good aim for ecosystems that are degraded and can not revert to a natural state. It is easy to make the analogy with human health when talking about ecosystems. However, this analogy is not appropriate because there is no environmental or ecosystem homeostatic process that maintains consistent internal properties in healthy ecosystems. Rather, ecosystem health refers to stress responses, damage, and altered properties and processes (Lackey 2001). Diagnosing and treating stress and damage are the central aims of ecological engineering, although undoing the undesirable ecosystem status for the benefit of nature and people is very challenging.

An ecosystem can be considered healthy when free of stress symptoms, resilient to external pressures, and sustainable (Costanza and Mageau 1999). Ecosystems are dynamic but persist within a range that lacks stress responses indicating sound status. Stable interactions between human and natural properties are also the aim of ecological engineering. A healthy status of an ecosystem includes a stable organization among both human and natural components, very active productivity, and persistence with occasional external stresses. Health can be judged based on ecosystem characteristics, behavior of the system through time, and desired services it provides to people and nature.

ECOLOGICAL ENGINEERING IN PRACTICE

Identification of public priorities requires a broad participatory process that defines objectives and prioritizes benefits. The key issue under discussion is what should be created and sustained in the conservation design. The lead organization or local government must somehow balance competing values and desires, include scientific information, and promote the benefits that are selected for a plan. Commonly in ecological engineering cases, there are no reference sites to copy, standards to be achieved, or science that defines a target ecosystem state. Therefore, a lead organization must decide which properties are desired and which are unimportant, and then what can be achieved through feasible practices.

Although the public always wants a healthy environment, this is difficult to transform into clear public policy for environments that are to be newly formed and merge the interests of people with nature. Making the tradeoffs among ecological engineering alternatives and finding compromises across high and low benefit priorities is necessary and difficult. Benefits to people and nature that command top attention in an ecological engineering plan have to be formed collaboratively. The process for changing damaged properties to desirable conditions also has to be identified. Science can support these actions by helping people understand how ecosystems work, and how to assemble ecological components that will interact. An ecosystem has a dynamic nature and can be unpredictable through time. Science can portray the possible variations in conditions for public acceptance. In the end, the local public has to support a plan. Also, the engaged public has to recognize that ecological engineering aims for a broad target with natural, social, and economic aspects of developing a new ecosystem that are feasible.

Ecosystems are considered healthy by the public when they support natural resources, absorb nutrients and pollutants, and persist in a desired state despite natural perturbations. Also, it is common to design ecosystems that will not require intensive maintenance and inputs of energy, water, and cost. In short, healthy ecosystems are persistent, support high biodiversity, resist human impacts, and are productive for people and wild species (Evers et al. 2018). Also, the design of a new ecosystem for human benefits often includes sustaining agriculture, allows space for people to live and enjoy nature, resists easy overharvesting of wild species, needs little water, is not conducive to overabundant non-native species, and maintains other similar attributes. These benefits and others are expected to persist in time and be desirable for future human inhabitants. Targeted benefits should not rely on surrounding lands, people, and ecosystems. Sustainability can be designed as a product of ecosystem processes rather than energy intensive maintenance. Finally, any design is a product of societal values and ecological processes which aims for a positive change in an environmental setting (Cairns 1995).

Creating novel ecosystems can include new species for a location, new assemblages of species, different ecosystem functions, compatible human activities, but should not require continual maintenance of the setting by humans. The need for designing and engineering novel ecosystems can result from the regional extinction of species, which may require the replacement of the natural communities for persistent new assemblages of species. Interior urban settings, and intensively cultivated or permanently degraded landscapes can restrict ecological processes and recolonization by wild species of plants and animals. Finally, new ecosystems are often required for locations that have had major changes in the abiotic environment, such as mined lands, brownfields, and areas with extensive soil depletion. In these settings, engineered ecosystems are aimed at providing ecological, social, and economic benefits. Some of the common goals for establishing new ecosystems are to support greater biodiversity, integrate human activity, and provide sustainability through internal system processes. A central challenge in designing new ecosystems is to provide both human and natural services that are maintained by the novel ecosystem.

Within the conservation community there is a counter view to ecological engineering. The central concept of this view is that an ecosystem that is natural or wild is authentic, and if it has human elements in it, it is less respectable. This view hinges on the concept that “natural and wild” is one option, and “human and anthropogenic” is another option. Ecological engineering that mixes people with nature diminishes natural values. Designing ecosystems can be like art forgery, producing faked-nature as the outcome. When human design and domination of nature is attempted, then nature is destroyed. We can achieve a pleasant natural environment but that can be illusory and a false reality. These views can seem polarizing and extreme, but there is a segment of the population that promotes these ideas. They run counter to ecological engineering and there is a practical argument that must be made to improve the environment when it cannot be restored to its natural state. This debate is increasingly active as we learn more about how humans altered the earth in the past, and appreciate the worldwide scope of change humans have caused starting thousands of years ago (Martin 1973; Westphal et al. 2010).

CASE STUDY: CHICAGO WATERFRONT ECOLOGICAL ENGINEERING PROJECT

Ecological engineering is especially relevant for urban landscapes that need rehabilitation for enhanced cultural and natural benefits. Urban lands are often isolated and heavily degraded making creative planning essential to address physical, biological, and social elements (Bhattacharyya and Mahanta 2014). Experts that are park planners, landscape architects, ecological restorationists, historical preservationists, and others are often engaged. As important as diverse expert input, is broad citizen involvement is also essential to capture their own interests and perspectives. This case study illustrates this process and demonstrates challenges in restoring urban lands (Gobster and Barro 2000; Gobster 2001).

Figure 9.5: Montrose Point in Chicago, Illinois. Source: Google Maps 2021

Montrose Point is a 4.5 ha (11 acres) artificial land extension into Lake Michigan made from landfill in the northern portion of Lincoln Park in Chicago (Figure 9.5). Lincoln Park itself is Chicago's largest park and one of the largest city parks in the United States at over 485 ha (1200 acres) (Schweitz 2017). Lincoln Park is a central lakefront location with a variety of natural and developed settings which makes it an extremely popular recreation destination, with an estimated 20 million visitors annually (Gobster 2001). A plan for Montrose Point was drafted in 1938 by Alfred Caldwell who aimed for a natural landscape design that had elements of midwest prairies, savannas, and woodlands. However, before his plan was developed the United States army took possession of Montrose Point for a radar station and Nike missile launch site to protect Chicago in the 1950s. Thus, this parcel of land is entirely non-natural and was initially used by the military. In the 1970s the Point was returned to the Chicago Park District and little happened on the Point until the 1990s when attention was focused on the entire Lincoln Park.

From the 1970s to the 1990s, the land was idle and significant growth of non-native vegetation developed. A large hedgerow of honeysuckle became established, and attracted a wide variety of birds. Bird watchers named this the “Magic Hedge” because it was common to see as many as 200 species of birds during fall and spring migrations. The Magic Hedge on Montrose Point gained national and international recognition for birding. In the late 1990s, the Chicago Park District and a non-governmental organization called the Lincoln Park Advisory Council started the Montrose Point Restoration Project to gather different visions of nature by stakeholders, design cultural and natural features for Montrose Point, and identify points of consensus and conflicting interests of stakeholders. Stakeholders included birders, environmentalists, historic preservationists, landscape architects, passive users, volleyball players, anglers, and yacht club members. A series of focus-group discussions were held and recorded to identify various interests for the design of a new Montrose Point. This process was viewed as a complex challenge linking physical, biological, and social aspects of design.

Focus-group participants overwhelmingly valued Montrose Point for its natural qualities and nature was a key element in their enjoyment of the place. There was broad agreement on improving the natural attributes of Montrose Point for plants, animals, and enjoyment by people. However, there were different views on the ways in which the Point should serve public uses, and the different user groups each had their own perspective on what was essential and important. Gobster (2001) characterized four different visions of nature on Montrose Point and summarized the perspectives based on purpose of nature (function), character of desired landscape (structure), natural and cultural significance (values), intended enjoyment (use), and symbolic features of the Point (icons) (Table 9.2). Landscape icons embodied natural and cultural features that defined what each stakeholder group strongly desired and considered essential.

Table 9.2: Summary of four visions of nature expressed by Montrose Point stakeholders.

Criteria	Designed landscape	Critical habitat	Recreation	Pre-European settlement
Function	Aesthetic experience, sense of creativity, separate from city	Primary focus on birds	Nature as substance and backdrop	Emulate pre-settlement ecosystems and processes
Structure	Native plants, multi-layered arrangement	Food and cover as bird habitat	Natural appearance	Native plant communities
Values	Landscape art	Unique birding experience, bird diversity	Nature appreciation, isolation from city, special place	Biodiversity, endangered species, natural experience
Use	Passive-appreciative	Limit use except for birders	Balance nature with use	Learn and experience restored vegetation and land
Icons	The meadow, the long view out onto the lake	The Magic Hedge	Beach, harbor, "hook", revetment	Entire landscape

Designed landscape - The group that identified with Alfred Caldwell’s naturalistic design for Montrose Point was largely comprised of landscape architects and historic preservationists. Their vision centered on aesthetic experiences and design as historic landscape art, and was shaped by views distinctly separate from the shoreline and city. This perspective saw an expansive meadow as an analogy to midwest prairies, multi-level vegetation surrounding the meadow as isolating walls, and a long view out to Lake Michigan. Proponents of this vision saw the short-grass meadow and long view as essential aspects that needed to be maintained. The intended uses of the Point were for passive appreciation and included relaxing, sitting, walking, and watching nature and people. The defining icons of the designed landscape perspective were the meadow and long view.

Critical habitat - This group promoted a vision of nature as habitat mainly for birds, but also other wildlife such as small mammals and butterflies. Their intent was to improve the habitat along the Lake

Michigan shoreline since most of the city waterfront is constructed and heavily used by people. The main mission of their view of ecological engineering was to maintain and enhance the Magic Hedge. Many of the honeysuckle plants were reaching maximum life expectancy, thus requiring active planting of a diversity of grasses, forbs, and shrubs. This group also promoted the creation of wet areas on the Point and the idea of letting aquatic vegetation and woody debris accumulate along the lake shore. The strongest advocates of this vision were birders and they argued that Montrose Point is a special place for birds and birding. This group did not want to see meadow mowing, dog walkers, volleyball players, mountain bike riders, picnickers, anglers, sail surfers, and jet skiers. Essentially the Magic Hedge was the icon, and habitat for birds was the top priority.

Recreation - This group saw Montrose Point as a natural backdrop for recreational activities such as walking, dog walking, picnicking, biking, league volleyball, sailing, and fishing. Values that recreationists raised were: a sense of nature, isolation from the city, fresh air, and a sense of a special place. Important from a practical perspective were picnic facilities, restrooms, and parking. Different users valued additional items such as beach space for volleyball, harbor space for yachts, and the long breakwater called the “Hook.” Passive recreationists saw the stone revetment as important for lounging and enjoying the shoreline. Thus icons for recreationists varied by intended use.

Pre-European Settlement - The vision of nature for this group included the landscape attributes likely present prior to the substantial European inhabitation of Chicago. This corresponds with the common notion of a pre-Columbian baseline as the initial era of the decline of North American natural landscapes. This group advocated for creating a mosaic of prairie, savanna, and shoreline habitats with indigenous community types, enhanced biodiversity, and native plants. In conflict with this view was the notion that Montrose Point did not exist prior to European colonization. However, this group wanted to use this sizable artificial landscape to sustain exclusively native plants and animals in close proximity to the city. Eradication of invasive and non-native species was one proposed strategy for transitioning the Point to a natural landscape. Rare and high priority conservation species could be re-established on Montrose Point such as searocket (*Cakile edentula*), wild mustard (*Synapis arvensis*), and trailing juniper (*Juniperus horizontalis*). Advocates wanted to focus on the restored natural landscape, and many other uses of the Point would be restricted. No icon existed on the present Montrose Point for this group, but the future icon would be the restored landscape.

The four visions of nature for Montrose Point revealed the differences in stakeholder thinking about the future of this site. Also there were differences in what should be done to enhance benefits to people and nature. The challenge was not to pick what vision was best for the Point, but to see how to integrate the perspectives and desires. The icons of three of the groups form what is needed to satisfy their desires for the future of the Point. Ultimately, the approach that was used was a hybrid vision that led to a culturally sustainable future for Montrose Point. Elements built into the resulting ecological engineering plan included Caldwell’s design of a central meadow that would be planted as a prairie, not a mowed grass field. Views would be created for aesthetic appreciation of the lake. The Magic Hedge would be expanded and maintained. Finally, a perimeter pathway around the Point would allow access for different user groups and allow people to experience the land and lake. Integration of these different visions of Montrose Point provided diverse benefits for people and nature, and was consistent with the aims of ecological engineering.

This case showed that many different ideas of "nature" exist among the public, agencies, and organizations. The successful ecological engineering plan of Montrose Point depended not on choosing the "right" nature, but instead on integrating the diverse values of stakeholders regarding culture and nature. A plan was formed that excited and encouraged historic preservationists, ecological restorationists, birders, and other recreationists to work toward restoring Montrose Point and its landscape icons as symbols of nature and culture in an urban setting. Integration is key in creating more natural places that attract the attention of people and inspire public care and admiration. The Montrose Point project is a nice example of design precipitating community involvement to create ecologically and culturally sustainable landscapes (Page 2016).

SUMMARY

As a field, ecological engineering focuses on alleviating ecosystem stress responses as these are symptoms of poor ecological health. In practice, there seems to be more emphasis on collaborative goal-setting that results in consensus on a vision for ecosystem design. The practical need to gain public support for a new ecosystem is seen as essential for long-term sustainability. The underlying goal of ecological engineering is to improve degraded and abandoned environments to provide benefits to both people and nature and this goal figures prominently in the case study. While this environmental management technique lacks an established track record and principles for success, there appears to be general consistency in many of the key features of this approach.

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Chapter 10 - Ecosystem-Based Management

The first topic in the holistic techniques group is ecosystem-based management. Holistic environmental management was proposed decades ago but has only more recently become a common technique under the heading of ecosystem-based management. Agencies like the United States National Oceanic and Atmospheric Administration and others have developed frameworks for ecosystem-based management and these frameworks are being used as fundamental processes for managing the environment. In this chapter, we will cover the background and justification for ecosystem-based management, discuss implementation , and use a case study to demonstrate application of ecosystem-based management principles in the New York ocean and Great Lakes.

HISTORY AND MOTIVATIONS OF ECOSYSTEM-BASED MANAGEMENT

The idea of managing ecosystems in a holistic way and at a large scale goes back to Victor Shelford in his Ecological Society of America Nature Sanctuary Plan (1933), Aldo Leopold in his Sand County Almanac (1949), and a few others. Government agencies began implementing ecosystem management in the late 1980s and early 1990s because of broad public controversies about management of western forests, increased attention on large mammals in national parks (e.g., Yellowstone and its grizzly bears *(Ursus arctos*)), and the general acknowledged decline in biodiversity.

The motivations for implementing ecosystem scale management in the 1990s were many. Aside from the aforementioned biodiversity decline and public awareness of management issues in particular ecosystems, there also existed a widespread lack of progress in addressing environmental deterioration, an increasing focus of people on nature, development of the field of conservation biology, support for increased management through environmental laws, federal mandates for diverse-interest management, and a disappointing trend of delays in environmental management decision-making due to litigation (Grumbine 1994). Along with these issues, at the forefront of society's attention was an increased awareness among scientists, academics, politicians, and appointed officials of the need for management of ecosystems as a whole (Lackey 1998). This idea was embraced as a bold new concept and a potentially better way to achieve conservation goals (Figure 10.1).

DEFINITION OF EBM

Ecosystem-based management (EBM) is defined as an integration of scientific knowledge, based on ecological relationships within a complex sociopolitical and values-oriented framework, with a focus on the general goal of protecting ecosystem integrity over the long term (Grumbine 1994). Brunner and Clark (1997) provide a simpler definition of EBM as a philosophy or paradigm of natural resource management intended to sustain the integrity of ecosystems. Finally, Lackey (1998) called EBM the careful and skillful use of ecological, economic, social, and managerial principles in managing ecosystems to produce, restore, or sustain ecosystem integrity and desired conditions, uses, products, values, and services over the long term. Essentially, the idea is to restore and maintain the health, sustainability, and biological diversity of ecosystems while supporting sustainable economies and communities. An important distinction of EBM is that it involves a plan to manage ecosystems to provide for all associated organisms, as opposed to a strategy or plan for managing individual species. In 2006, Arkema et al. analyzed a variety of definitions for EBM. Their analysis yielded 17 criteria for EBM.

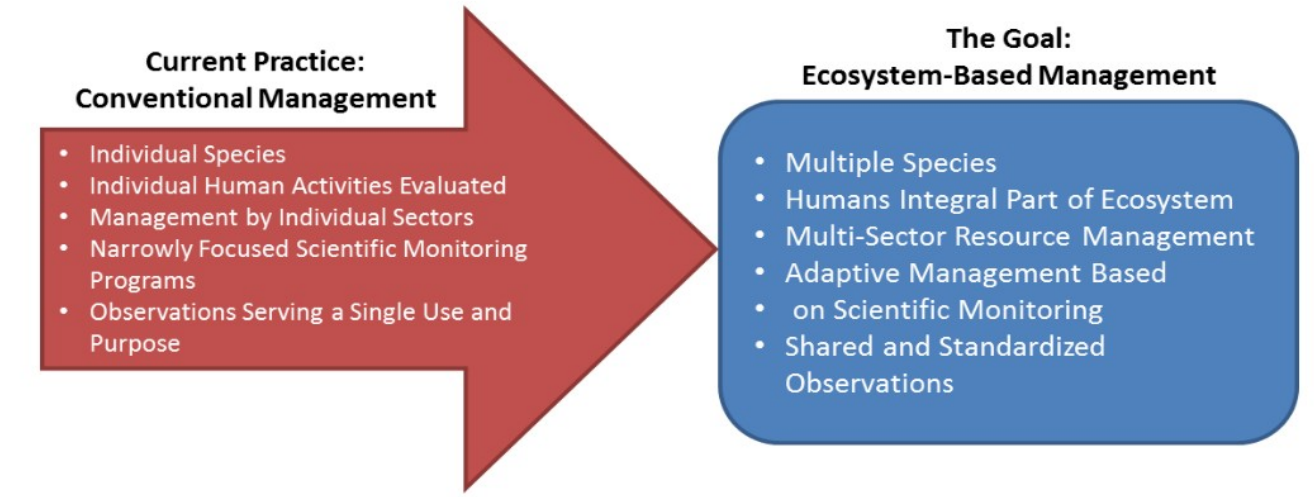

Figure 10.1: The differences between conventional management and ecosystem-based management. Source: NOAA 2021a

General criteria including sustainability, ecological health and inclusion of humans were important (Yaffee 1996). Sustainability emphasizes maintenance of one or more aspects of the ecosystem. Ecological health includes non-specific goals for ecosystem health or integrity. Inclusion of humans recognizes that humans are elements in an ecosystem and their education and well-being are important components of management decisions.

Ecological criteria such as complexity, temporal and spatial scales were also important. Complexity, meaning the linkages between ecosystem components, such as food web structure, predator–prey relationships, habitat associations, and other biotic and abiotic interactions, should be incorporated into management decisions. Temporal scale incorporates time and the dynamic character of ecosystems. Spatial scale incorporates ecosystem processes which operate over a wide range of locations.

Human criteria included ecosystem goods and services, economic factors and stakeholders. Humans use and value natural resources, such as water quality, harvested products, tourism, and recreation, and these are classified as ecosystem goods and services. Economic factors take into account the costs of ecosystem goods and services. Stakeholders are the varied parties engaged in the management planning to find common solutions.

Finally, management criteria included science-based decisions, boundaries, technology, adaptive management, co-management, a precautionary approach, an interdisciplinary approach, and monitoring. Science-based decisions involve management decisions based on tested hypotheses. Boundaries define the spatial extent to which management decisions apply. Science and technology are used to monitor ecosystem and management actions. Adaptive management improves implementation through systematic evaluation. Co-management promotes shared responsibility between governments and stakeholders. Precautionary approaches manage projects conservatively when uncertainty exists. Interdisciplinary approaches utilize science from several disciplines. Finally, monitoring tracks changes in biotic, abiotic, and human ecosystem components.

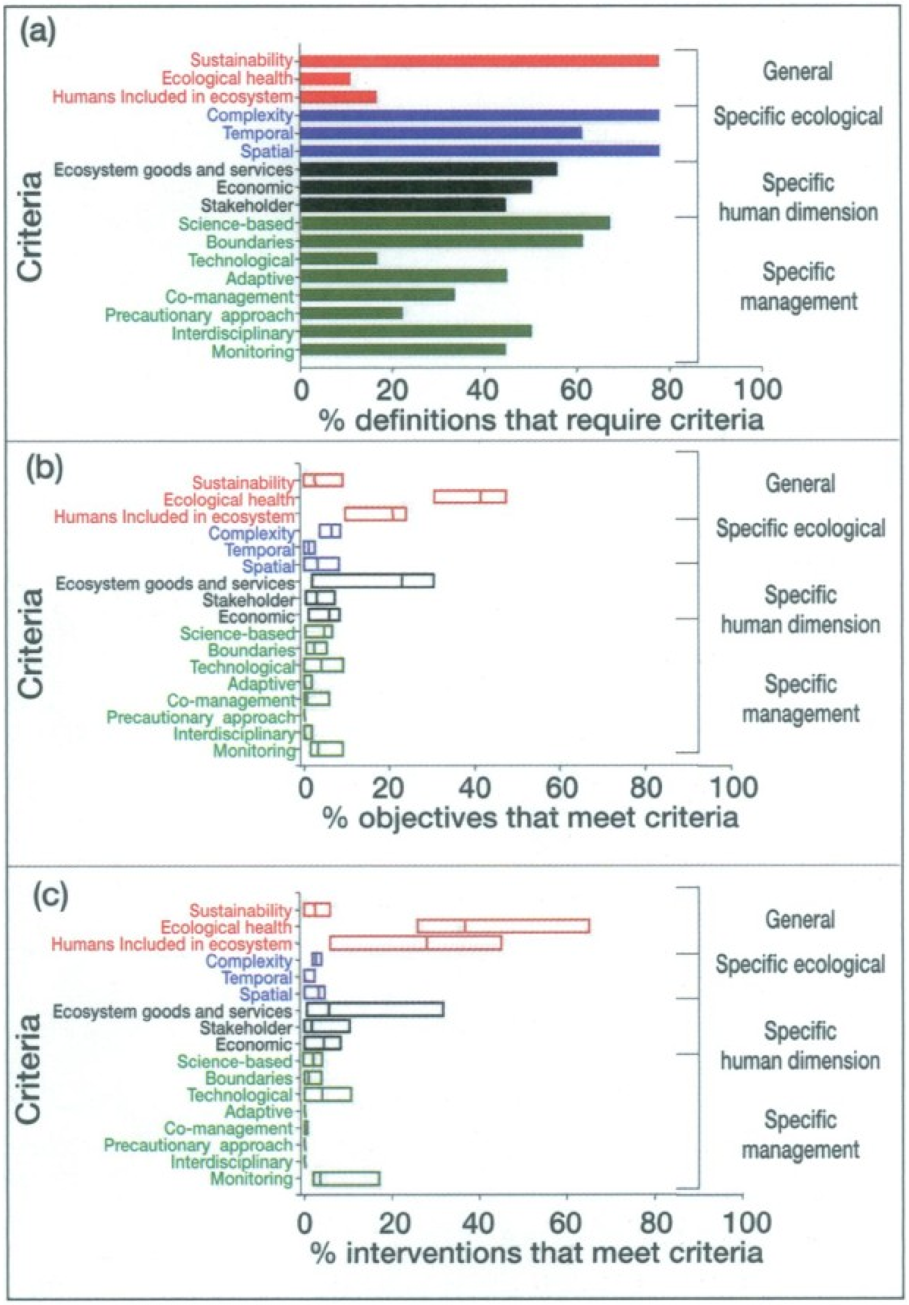

Figure 10.2: Comparison of EBM criteria among definitions and management plans. (a) Percentage of definitions (n=18) that include the criteria scientists use to describe EBM. (b) Percentage of management objectives averaged among sites (n=8) that address EBM criteria. (c) Percentage of interventions averaged among sites (n=8) that address EBM criteria. Source: Arkema et al. 2006

IMPLEMENTATION OF EBM

Policy makers, management agencies, and academic scientists have shown increasing interest in implementing EBM (National Research Council 1999). Arkema et al. (2006) analyzed the application of EBM principles in management by reviewing 49 management plans for eight large marine and coastal ecosystems. They applied their 17 EBM criteria to each of the management plans to investigate the consistency with which the EBM criteria were applied in each of the ecosystems. Results from their analyses showed that most of the emphasis within the management plans was placed on ecological criteria (Figure 10.2a). However, the majority of definitions included at least one specific human-dimension criterion, such as ecosystem goods and services. When looking at management objectives addressing EBM criteria, they found that ecological health was a major focus and that ecosystem goods and services were also important (Figure 10.2b). However, specific ecological and management objectives were few. The objectives were mostly aimed at generalized goals (Figure 10.2c). When investigating the percentage of actions addressing EBM criteria, they found again that management actions were largely aimed at general goals, however a little more attention was paid to the human aspects of the ecosystem.

Arkema et al. (2006) concluded from this study that scientists and managers see EBM differently. They found that management objectives and interventions tended to miss critical ecological and human factors emphasized in the academic literature. Academic thinking is more oriented toward ecological and human criteria. Thus, moving forward, scientists and managers need to work collaboratively to generate realistic methods for applying EBM principles, thereby helping to overcome barriers between the scientific concept of EBM and its implementation.

IMPROVING IMPLEMENTATION OF EBM

Brunner and Clark (1997) evaluated three major approaches for improving the principles of EBM. First, they focused on the principle of clarification of the goals of EBM. Researchers and practitioners generally feel that setting clear goals is crucial to the success of EBM. However, goal setting is not enough. Commonly cited goals would not be sufficient for many EBM decisions, even if they were clarified, as judgments are needed in almost every context. EBM must integrate multiple, often incompatible or incommensurate, goals. Political differences must be reconciled as practitioners need to consider how to appeal more effectively to others on their own terms. Second, Brunner and Clark (1997) focused on the principle of needing a better scientific foundation for management decisions. A scientific foundation is often considered a prerequisite for EBM. However, a better scientific foundation of relationships is not necessary for the practical purposes of EBM since scientific generalizations cannot be considered universal from a practical standpoint. Instead, people need to focus on using the existing science in more creative ways to meet our evolving management needs. Management methods need to be contextual, integrative, and interpretive. Third, Brunner and Clark (1997) focused on the principle of implementing a practice-based approach. Practitioners must make interpretations and judgments that function as maps for their decisions. Practitioners review past successful EBM cases for use as standards of good practice from which to distill general principles. However, practitioners also need to identify and address constraints on good practice that recur across cases in EBM and review, design and test innovative models (aka prototypes) that address critical problems within particular ecosystems.

Overall, the practice-based approach recognizes that moral, scientific, and practical considerations are integrated implicitly or explicitly into EBM decisions (Brunner and Clark 1997). The practice-based approach recognizes that such decisions are human factors that most directly affect the integrity of ecosystems, either by sustaining or degrading them. Practice provides a reality check on the considerations integrated into decisions, the best opportunity for learning from experience, and the only reliable gauge of progress in EBM. Clearer general goals and a better scientific foundation are the means for improving decisions on behalf of ecosystem integrity, but they are not ends in themselves.

THEMES OF EBM

Figure 10.3: Word cloud created by participants of a workshop highlighting the many aspects of ecosystem-based management. Source: NOAA 2021b

There are ten dominant themes of EBM as explained by Grumbine (1994; 1997) (Figure 10.3). First, EBM involves a *hierarchical context,* which is contextual or big-picture thinking. Systems problems require systems thinkers who can work across disciplines and be imaginative and integrative, flexible and adaptive. Second, a starting step in EBM is to bring all interested parties together to define common problems and boundaries of concern, also called *ecological boundaries.* Third, *ecological integrity* needs to be maintained in the ecosystem. This includes maintaining viable populations of native species, representation of all ecosystem types, maintaining ecological processes, management over the long term, and accommodating human uses. Fourth, practitioners are tasked with finding, collecting and using the best available *scientific information* (e.g., data), including the human aspects of a system. Fifth, practitioners are also tasked with *monitoring* the system to gain information on progress toward providing benefits and self-maintenance. Sixth, *interagency cooperation* among all parties is important, as both a mindset and standard operating procedure, to manage problems and stay focused on the protection of ecological integrity as opposed to social issues. All parties must share power and share in defining the problem and setting key goals. Seventh, we have a long cultural tradition of viewing humans apart from nature. EBM runs counter to this tradition by *embedding humans in nature.* Eighth, EBM practices *adaptive management,* which is a process of continuously gathering data on the success of previous actions. This allows the incorporation of feedback from those results to help managers remain flexible, be responsive to change, adapt to uncertainty, and institutionalize learning. Ninth, nature is nonlinear and full of surprise, and management is aimed at balanced, linear, and predictable scenarios. Thus, organizations must attempt to transform themselves to become more flexible through a process called *organizational change*. People in various roles from top-level decision-makers to mid-level managers to field-level implementers must be supported by organizations as flexible as the complex tasks of EBM require. Finally, people make commitments based on *values* as much if not more than on facts and logic. Generally, resource allocation decisions, which relate directly to resource use, are matters of political struggle rather than technical fact and as such are more about manipulating human behavior rather than physical things. As managers learn to accept the role of human values explicitly in their management of resources, the success of EBM will become more likely.

APPLICATION OF EBM TO MARINE RESOURCES

The dire state of marine fisheries, oceans, and coasts has reached a level of general public alarm in recent years (Duarte 2002; Lewison et al. 2004; Limburg and Waldman 2009; Eddy et al. 2021). A series of national and international assessments were conducted and the major reports recommended the use of EBM (Pew Oceans Commission 2003; United States Commission on Ocean Policy 2004). Both commissions called for a more comprehensive, integrated, ecosystem-based approach to address the current and future management challenges of restoring and protecting our oceans.

McLeod et al. (2005) compiled the consensus views from the marine scientific and management community. The main points of the consensus statement are: 1) The key challenges are to refine EBM, and develop a set of principles to guide management and policy; 2) EBM is the application of ecological principles to achieve integrated management of key activities affecting the marine environment; 3) EBM explicitly considers the inter-dependence of all ecosystem components, including species, both human and nonhuman, and the environments in which they live; and 4) The EBM goal for oceans is to protect, maintain, and restore ecosystem function in order to achieve long-term sustainability of marine ecosystems and the human communities that depend on them.

There are four main aspects of scientific knowledge regarding marine ecosystems: 1) The key interactions among species within an ecosystem are essential to maintain if ecosystem services are to be delivered. Some species' interactions strongly influence the overall behavior of ecosystems. Small changes to these key interactions can produce large ecosystem responses. EBM therefore entails identifying and focusing on the role of key interactions, rather than on all possible interactions; 2) The dynamic and complex nature of ecosystems requires a long-term focus and an understanding that abrupt, unanticipated changes are possible. The abundances of species are inherently difficult to predict, especially over longer time periods, in part because they may change abruptly and with little warning. Management must thus anticipate and be able to adjust to these changes; 3) Ecosystems can recover from many kinds of disturbance, but are not infinitely resilient. There is often a threshold (i.e., tipping point) beyond which an altered ecosystem may not return to its previous state. Features that enhance the ability of an ecosystem to resist or recover from disturbances include the full natural complement of species, genetic diversity within species, multiple representative stands (i.e., copies) of each habitat type, and a lack of degrading stressors from other sources; and 4) Ecosystem services are nearly always undervalued. Although some goods (e.g., fish and shellfish) have significant economic value, most other essential services are neither appreciated nor commonly assigned economic worth. Examples of services that are at risk because they are undervalued include protection of shorelines from erosion, nutrient recycling, control of disease and pests, climate regulation, cultural heritage, and spiritual benefits.

Key elements of an EBM in the marine environment would: 1) Emphasize the protection of ecosystem structure, function, and key processes; 2) Be place-based in focusing on a specific ecosystem and the range of activities affecting it; 3) Explicitly account for the interconnectedness within systems, the import and export of larvae, nutrients, and food, and the importance of interactions between many species or key services and non-target species; 4) Acknowledge interconnectedness among systems, such as between air, land and sea; 5) Integrate ecological, social, economic, and institutional perspectives; 6) Consider cumulative effects of different activities on the diversity and interactions of species; 7) Incorporate measures that acknowledge the inherent uncertainties in ecosystem-based management and account for dynamic changes in ecosystems (i.e., precautionary management); 8) Create complementary and coordinated policies at global, international, national, regional, and local scales, including between coasts and watersheds; 9) Maintain historical levels of native biodiversity in ecosystems to provide resilience to both natural and human-induced changes; 10) Require evidence that an action will not cause undue harm to ecosystem functioning before allowing that action to proceed; 11) Develop multiple indicators to measure the status of ecosystem function, service provision, and effectiveness of management efforts; and 12) Involve all stakeholders through participatory governance that accounts for both local interests and those of the wider public.

Ruckelshaus et al. (2008) defined six basic principles for the application of EBM to the management of resources in marine environments. First, they defined the spatial boundary of the system. The spatial extent of the ecosystem determines which species, other ecosystem attributes, and human activities are the focus of management. Next, they developed a clear statement of the objectives. This included determining which biological and social values were desired from the ecosystem. Potential objectives included maximizing the overall ecosystem harvests and benefits to society, targeting levels of ecosystem services such as nutrient cycling or toxin filtering, and/or increasing ecosystem properties such as resilience, biodiversity, redundancy, and modularity (Levin and Lubchenco 2008). Then they included humans in characterizing ecosystem attributes and indicators. This is an important step as including human uses of and interactions with natural resources improves the likelihood of achieving desired outcomes. Next, they used strategies to hedge against uncertainty in ecosystem responses to EBM. This included building learning into strategy development and adopting an approach that could become more prescriptive over time as information about the system increased. This also included a diversity of regulation, reward, and other incentives for human behaviors consistent with the objectives of the process. Then they used spatial frameworks to coordinate multiple sectors and approaches. This was important to help manage competing uses and authorities from such sectors as fisheries, recreation, research, conservation, and shipping. Finally, they linked the governance structure with the scale of the EBM project, since management decisions, monitoring, and authorities should be governed at the scale of the ecosystem.

CASE STUDY: NEW YORK OCEAN AND GREAT LAKES EBM

In 2006, the New York Ocean and Great Lakes Ecosystem Conservation Act was passed (New York State Senate 2021). This Act declares that: 1) New York's coastal ecosystems are critical to the state's environmental and economic security, and integral to the state's high quality of life and culture. Healthy coastal ecosystems are part of the state's legacy, and are necessary to support the state's human and wildlife populations; 2) The policy of the state of New York shall be to conserve, maintain and restore coastal ecosystems so that they are healthy, productive and resilient and able to deliver the resources people want and need; 3) The governance of coastal ecosystems shall be guided by the following principles:

a. Activities within and uses of the coastal ecosystem are sustainable;
b. Ecological health and integrity is maintained;
c. Ecosystems' interconnections among land, air and water are recognized;
d. Understanding of coastal ecosystems is enhanced;

e. Decisions are informed by good science;
f. When risks are uncertain, caution is applied; and
g. Broad public participation occurs in planning and decision-making

Figure 10.4: New York ecosystem-based management activities. Source: The Nature Conservancy 2009

The Act is directly responsible for the establishment of two demonstration areas, the Great South Bay on Long Island and the Sandy Creeks Watershed on the eastern shore of Lake Ontario, to gain on-the-ground experience in applying EBM principles.

In addition, New York developed an EBM concept (Figure 10.4) with the definition that EBM is an emerging, integrated technique that considers the entire ecosystem, including humans, to achieve improved environmental conditions and sustained ecosystem services that support human needs and social goals (New York Ocean and Great Lakes Ecosystem Conservation Council 2009; Southern Tier Central regional planning and development 2013). Some principles that generally guide New York's EBM program are protection of ecosystems, place-based action, interconnectedness within systems, interconnectedness among systems, integration of a variety of perspectives, collaboration, and adaptive management.

Many New York State agencies' programs to manage human activities had already incorporated EBM principles. For instance, the New York State Department of Environmental Conservation's (NYSDEC) mission embodies the principles of EBM: to conserve, improve, and protect New York's natural resources and environment, and control water, air, and land pollution, in order to enhance the health, safety and welfare of the people of the state and their overall economic and social well-being (New York State Department of Environmental Conservation 2021). In following this mission, the NYSDEC has created an Office of Climate Change, with units holistically focusing not only on “command-and-control” approaches, but on science, policy, outreach, and partnerships. They have also created the Pollution Prevention Institute, which is designed to complement NYSDEC’s existing regulatory approaches to chemical policy with technical assistance, green business support, green chemistry research, and partnerships between academia, state government, and local industries.

New York State also developed agency guidelines to integrate principles of EBM. The NYSDEC expanded the capacity of the observer network to conduct monitoring and tracking of environmental conditions; developed EBM goals for the Long Island Sound, South Shore Estuary, Peconic Estuary and New York and New Jersey Harbor; created an ecosystem monitoring and assessment program based on indicators that inform adaptive management decision-making; conducted targeted natural resource inventories to identify the location and condition of key habitats and associated species to prioritize the implementation of conservation strategies; and utilized professional literature and existing programs to evaluate potential impacts of climate change on our natural resources which include habitat loss, habitat degradation, change in the timing of biological functions, and harm to populations of fish and wildlife.

Likewise, the Office of Parks, Recreation and Historic Preservation has integrated EBM into their master plan and Statewide Comprehensive Outdoor Recreation Plan (Bogan and Cady-Sawyer 2021). They wish to expand stakeholder involvement in planning and evaluations, provide targeted training in EBM to staff, adopt policies that provide direction for present and future agency decisions, implement the Oceans/Great Lakes Literacy Project through educational kiosks, better integrate planning and management programs, and enhance water quality monitoring at state park beaches and lakes.

Also, the State University of New York (SUNY) has implemented elements of EBM by supporting research efforts of SUNY faculty to improve knowledge of ecosystems and EBM. SUNY also supports and implements the recommendations in the Scientific Advisory Group’s Research Priorities statement, and supports, through housing and leadership, initiatives of the Great Lakes Research Consortium and the New York Marine Sciences Consortium. This support led to the development of the Great South Bay Modeling project which built an ecosystem model of the Great South Bay, including temporal, spatial and food web components, for use in evaluating and guiding restoration efforts (one example from the work is Hinrichs et al. 2018). Support from SUNY also helped lead the Scientific Advisory Group (SAG) charged with the development of a New York Ocean and Great Lakes Ecosystems Research and Monitoring Agenda. And, finally, their support led to the establishment of the New York Marine Sciences Research Consortium to serve as the voice for marine research and education and advance marine research priorities in the State.

New York has successfully implemented a number of EBM goals since establishment of the New York Ocean and Great Lakes Ecosystem Conservation Act of 2006. There is more work to do however. Additional priorities in New York to achieve healthy ecosystems through EBM include managing multiple uses of offshore environments, using regional approaches to establish place-based ecosystem goals, enhancing local planning and protection in coastal transition zones, minimizing the effects of upland development, protecting sensitive coastal and offshore habitats, restoring marine and Great Lakes fisheries, managing Great Lakes water levels, managing invasive species, reducing point and non-point source pollution, and implementing riparian buffers.

SUMMARY

EBM involves managing ecosystems in a holistic way and at a large scale. Clear goal setting, use of good scientific foundations for management decisions, employment of a practice-based approach, and acknowledgment of the role that human values play in management of resources all help in the achievement of a successful implementation of EBM. Governmental agencies have developed frameworks for EBM and these frameworks are increasingly being used in managing the environment.

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Holistic Techniques

Chapter 11 - Adaptive Management

The second topic in the holistic techniques group is adaptive management. Adaptive management is a systematic approach based on the idea of improving management by learning from outcomes. This technique includes iterative adjustments in plans over time based on knowledge gained during the process. In this chapter, we cover details of the process of adaptive management, its benefits and limitations, and end with a case study of the Glen Canyon dam adaptive management program.

BACKGROUND ON ADAPTIVE MANAGEMENT

Adaptive management is a technique that befits important situations where information is inadequate or incomplete for use in making confident decisions. Adaptive management is best used for recurrent decision-making in which uncertainty about the decision is reduced over time through comparison of outcomes predicted by competing models against observed values of those outcomes (Moore et al. 2011). The strategy includes interdisciplinary teamwork to develop management options, models, hypotheses, monitoring, periodic assessment of management outcomes, and adjustment in management plans. The process is ongoing because it relies on repetition of the process to learn from management performance in a cyclic manner. The prominent benefits of adaptive management practices are integration of efforts and expertise of managers and scientists who learn from management performance.

Learning from management has a basis in science and is not meant to partition the roles of science and decision-making. Information generated by this process is seen as a benefit to both management and science, and the new knowledge can be applied in an orderly way to advance management effectiveness. With each iteration of the process, the adaptive management team explores alternative ways to achieve objectives, makes predictions of the intended outcomes, monitors successes or failures, and then reconsiders objectives and plans.

Adaptive management was introduced to the environmental field in 1978 with a book produced by C. S. Holling. He stated the original definition of adaptive management as “an integrated, multidisciplinary and systematic approach to improving management and accommodating change by learning from the outcomes of management policies and practices.” While this concept had been known since 1978, agency and ecological conservation programs have only recently begun to adopt this process (McFadden et al. 2011). Some notable cases drove this transition because of their progress (e.g., the mitigation of impacts in the Grand Canyon, waterfowl management across the continent, managing forests across eastern Canada, and restructuring water flows across the Everglades ecosystem). The scientific community played a direct role in these applications, and many scientists were involved in the adaptive management teams. Also, scientists are increasingly promoting the adaptive management process because it has a sound scientific basis and yields benefits for developing research (Haney and Power 1996). In recent years, adaptive management has been growing in the scholarly published literature (McFadden et al. 2011). The core ideas of learning, teamwork, and dealing with uncertainty are appealing to both scientists and practitioners (Johnson 1999; Medema et al. 2008; Smith 2011). The promise of adaptive management is substantial for complex ecological conservation problems, but the track record of its application is mixed (McLain and Lee 1996). This may be due to inconsistencies in use of the concept (Allen et al. 2011). It also maybe be due to the challenges of changing our society into one that values reflection and rewards thinking, sharing, humility and understanding.

THE ADAPTIVE MANAGEMENT PROCESS

Often adaptive management is called *learning by doing* (Walters and Holling 1990), or sometimes *trial and error* management. The U. S. Department of Interior Technical Guide for adaptive management (Williams et al. 2009) defines the process as a systematic approach to improving environmental management by learning from outcomes. The adaptive management process in Williams et al. (2009) is illustrated as a six-step iterative process (Figure 11.1) that includes exploring management alternatives (assess problem), predicting outcomes from current information (design), selecting one or more alternatives (implement), measuring outcomes (monitor), determining success or failure (evaluate), and updating management actions (adjust). This is the general format of adaptive management.

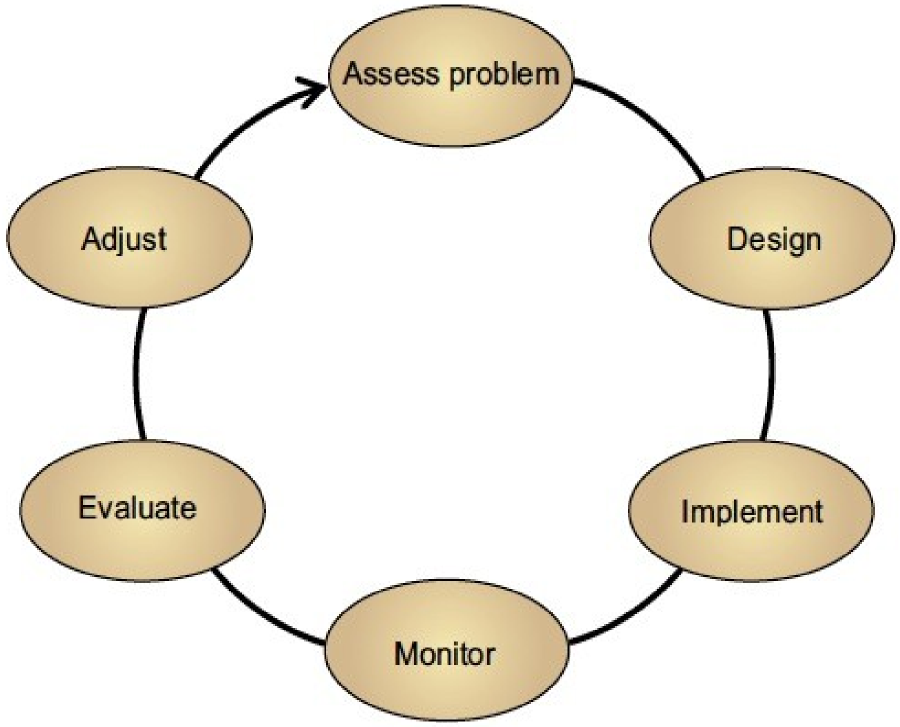

Figure 11.1: Illustration of adaptive management as a six-step iterative process. Source: Williams et al. 2009

The six-step process is consistent with the spirit of adaptive management and includes the learning from outcomes basis, but lacks emphasis on experimentation, hypotheses, and modeling to predict outcomes. Some organizations have created expanded adaptive management processes (Figure 11.2) which include more emphasis on tasks such as modeling (Delta Stewardship Council 2019).

UNDERLYING THEME OF EXPERIMENTATION FOR INTERDISCIPLINARY MANAGEMENT BY RESEARCHERS AND MANAGERS

Management of the environment must deal with the ever changing nature of environmental systems and with the uncertainty that poses (Kato and Ahrn 2008). The dynamic nature of the environment puts managers and scientists under the same challenges because each wants to know what to expect from management policies. Thus came the idea that management can be treated as an experiment. Like any experiment, adaptive management is bounded in time, requires data collection, and has a stage where findings are used to evaluate hypotheses. Adaptive management tends to be adopted as agencies and managers shift to ecosystem-scale challenges like the response of fauna and flora to water management, climate change, and landscape change (Woods 2021) or management of phragmites in the Great Lakes (Figure 11.3) (Great Lakes Phragmites Collaborative 2016).

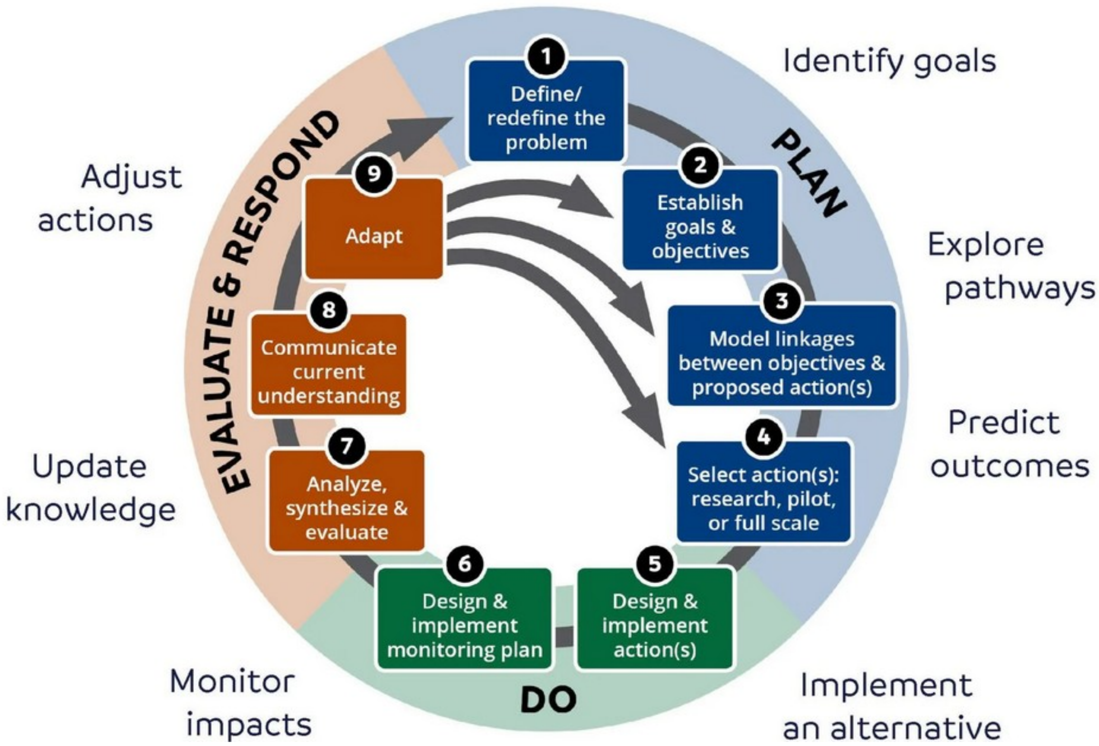

Figure 11.2: Example of a more extensive adaptive management cycle that includes modeling. Source: Delta Stewardship Council 2019

Research and management do not naturally go together. Under adaptive management, they are integrated because management actions are treated as experiments. Scientists do not specify the experiment, but they do work with the practitioners. As a team they design management options, then develop predicted outcomes, and identify measures of success or failure. Then scientists and managers work together to readjust management actions, building on the understanding they have obtained.

The interdisciplinary focus of adaptive management was a new concept when it was introduced (Dreiss et al. 2017). Managing complex environments on a large scale requires broad thinking from a team that has mixed perspectives and is willing to raise imaginative options. Adaptive management is best used when choices are difficult or uncertain for decision-making (Kato and Ahern 2008). The traditional thinking that a one-time assessment study can resolve what to do does not fit these situations (Walters 1986). Beyond a set of policy options, creative work has to be done on objectives, models, and designing a course of action as an experiment. Teamwork is needed to craft a policy direction and develop the expected outcomes and measures which can then be used to test predictions. Adaptive management is a big switch from a process where administrators select management directions based on their experience, to a team-developed strategy with explicit tradeoffs and predicted outcomes.

MODELING IN ADAPTIVE MANAGEMENT

The repeating process in adaptive management involves: modeling, hypothesis testing, monitoring, and a set schedule for reevaluation. Modeling is fundamental to adaptive management, but the use of modeling is not to identify an optimal solution. The pursuit of a single, best solution to meet management objectives is not the goal of adaptive management. Instead, simulation modeling is aimed at predicting outcomes from a set of viable management plans. Often in adaptive management cases, there are substantial uncertainties and a weak understanding of the key drivers. Incomplete or erroneous models can provide false predictions, and that is part of this process. Predictions that prove inaccurate are a clear signal to change management direction, and they can provide lessons on the need for management change. Finally, the process of modeling also reveals data gaps, lack of understanding, and uncertainties. This is an important part of the process as these issues help to identify what knowledge may still be needed and provide learning experiences.

Modeling is used for predicting management outcomes and for developing options for action. Team-developed models integrate perspectives, expertise, and new ways of considering management alternatives, and they generate expected outcomes, hypotheses to be tested, and specific measures for testing. Model results are used to craft new hypotheses and design monitoring protocols that will yield data indicating whether a management strategy produced the expected outcomes. This comprises a step in the creative process of adaptive management. There is some risk that multiple models will be debated, which can stall progress in the adaptive management process. The

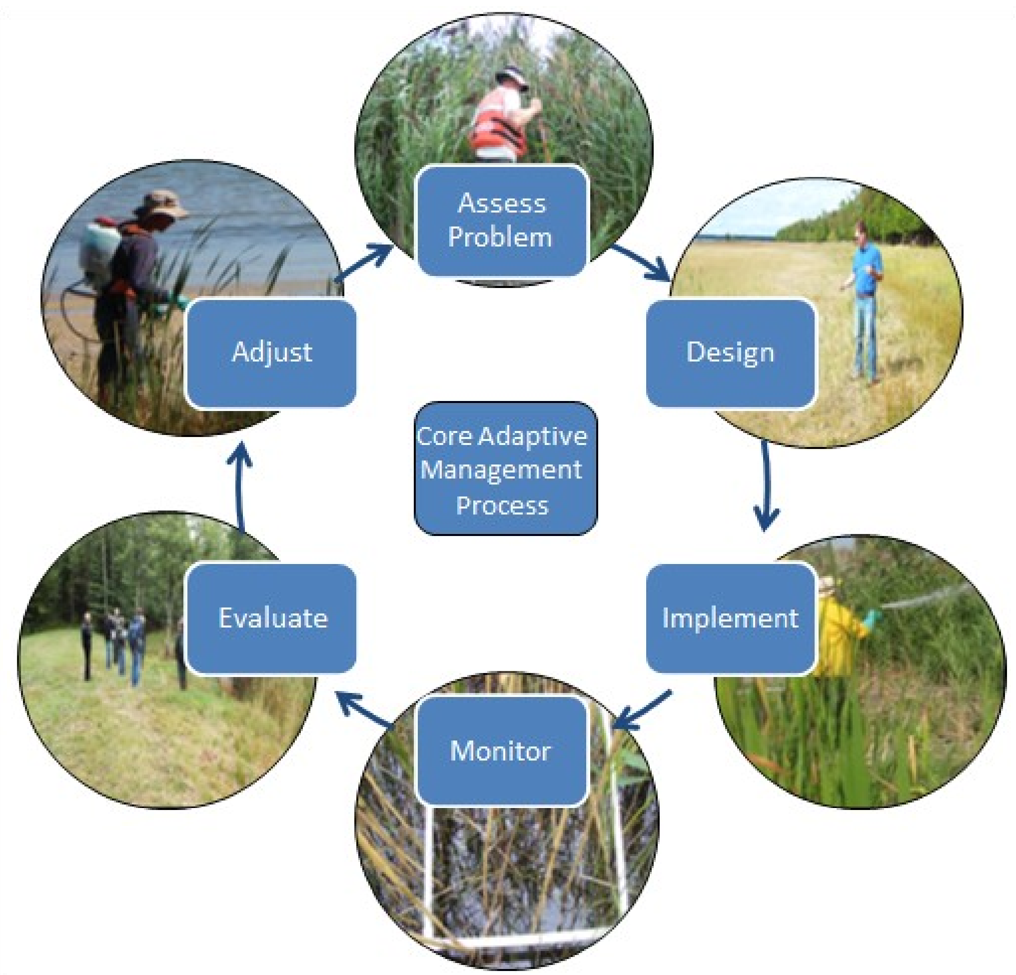

Figure 11.3: Example of adaptive management in action for phragmites management in the Great Lakes. Source: Great Lakes Phragmites Collaborative 2016

prediction-test-readjust progression is iterative so the best policy choice is not necessary at the start of the adaptive management process. At the start, a management direction that has a reasonable chance of succeeding can be tested and possibly be replaced by a new management strategy. This raises one issue for the team: in the public policy arena, are risky decisions possible or desirable? Failed management strategies may provide valuable information and better management outcomes in time, but parties outside the process could blame agencies for taking too many risks and causing failure. Managers avoid this very strongly. However, if modeling indicates potentially successful options and reasons why their application can work, then the adaptive process is working and builds support for these choices.

The use of models in a team context suggests that simulation models need to be simple in structure and operation. This simplification allows a diversity of participants to understand model structure, key variables, and how predictions were made. Also, key drivers often shape outcomes whose details often do not matter in the larger context of complex ecosystem-scale management cases. Models for creative uses in defining a management plan and conveying the key drivers to the stakeholders and the public should be easily understandable. The point of modeling in adaptive management is not to produce accurate and precise predictions of outcomes. Instead, models are used to focus deliberations on options, hypotheses, and design measurements and monitoring for validating or refuting management outcomes. Over time, with numerous iterations of adaptive management, models should become more accurate, precise, and supportive of management directions.

HYPOTHESIS TESTING IN ADAPTIVE MANAGEMENT

Learning under adaptive management comes from experimentation with management plans. Hypotheses are crucial for experimentation, so there is a need to be specific about them. Also, hypotheses should define monitoring specifics and be feasible for testing. Management choices are the basis for experimental opportunities. Traditional experimentation is reductionist, testing very narrow hypotheses to gain an understanding of specific mechanisms and questions. Under adaptive management, hypotheses are treated holistically and aimed to perform objectively. Thus hypotheses should be focused at the systems level and integrate a wide variety of system properties. Once a management plan is developed and the objectives defined, then management actions and monitoring should commence with the aim of detecting compliance with objectives. With the natural dynamics of environmental systems, it is important to define monitoring measures and schedules to detect the effects of management separately from the natural dynamics of the system.

MONITORING IN ADAPTIVE MANAGEMENT

In management programs, monitoring is often seen as open ended with ongoing costs, and thus managers tend to avoid this activity. However, monitoring is of central importance in adaptive management because of the orientation toward treating management practices as experiments. Monitoring environmental properties serves three purposes under adaptive management. First, monitoring has to be aimed at collecting data to test the predictions of management outcomes. This is critical to iterations of adaption in management. Second, monitoring builds knowledge of responses by the environmental system. This is important to the learning process and is one of the benefits of the adaptive management strategy. Third, over time, monitoring builds a database of the performance of alternative management policies and actions. Monitoring should not be broad and aimed at measuring everything. Instead it should target management outcomes from model predictions, and be very specific to measures that can be used to test predictions. Over time, such monitoring will evolve with adjustments in management policies and plans. Because management effects can take years to materialize, monitoring requires careful planning and institutional arrangement to maintain the program for some time. Overall, monitoring has to be efficient and effective for testing predictions of management outcomes.

EVALUATING PERFORMANCE IN ADAPTIVE MANAGEMENT

The adaptive term in the adaptive management technique indicates that management changes have to be considered. Once a management plan has been adopted with predicted outcomes and test data collected, it is time to evaluate management performance. The timeline should be set at the beginning of the process so that the experiment is defined and a schedule set to reevaluate management. Then the team needs to review the data and evidence on management performance and determine what changes are needed. This is the feedback stage where results from the first round of adaptive management are considered, and new management is proposed. Changes in management should be larger in early iterations to explore management options rather than refine the current management approach. This can be controversial because for managers it is often safer to continue past practices instead of making large changes and taking risks. However, the advantages of alternative management become apparent when different policies are pursued. The team of managers and scientists learn more by making changes and taking some risks. Also, because the environment is dynamic and uncertain, different options may be informative through time and truly improve management. To be really adaptive, management changes are necessary and fundamental to the learning process.

WHAT MAKES SOME PROJECTS APPROPRIATE FOR ADAPTIVE MANAGEMENT?

Environmental problems that are conducive to the adaptive management technique are the ones that are typically viewed as critical and require action. There has to be a mandate for acting on the best course of action under limited knowledge and uncertainty. In other words, the problem must be important enough to require action. These cases have a high learning potential and opportunities to apply learning, and those aspects of adaptive management are seen as promising benefits to long-term management effectiveness. There should also be institutional capacity and commitment, funding to set up a monitoring system, analytic expertise on the team, and experts on the key issues. Key decisions in important cases have to be developed by a thorough and structured process considering a diversity of perspectives. The direction of management needs to be explicitly justified and accountable, and this requires analyses, debate, and consensus building. The management plan adopted has to have clear and measurable objectives and criteria for success. These attributes of ecological conservation support the lead agency or institution because the information, management experience, and success are all highly valued. Engagement of scientific expertise, high-level managers, and financial resources for maintaining the process are vital despite being more costly than top-down administrative decisions. In short, adaptive management is often selected for high profile, complex, costly, and controversial ecological conservation challenges that are being watched by the public, elected officials, and diverse stakeholder groups.

Conversely, there are situations in which adaptive management would not be appropriate. Adaptive management would be difficult to apply in a complex legal environment, in a situation where there are unresolvable conflicts in defining explicit and measurable management objectives or alternatives, where institutional reluctance to change is strong, when risks associated with learning-based decision-making are too high, when decisions that affect resource systems and outcomes cannot be made, and where decision-making occurs only once. Additionally, adaptive management is not appropriate when monitoring cannot provide useful information for decision-making. Monitoring may be ineffective when the frequency of data collection is too low to keep pace with changes in the natural system, there are significant time lags between management actions and their impacts, a monitoring plan cannot be designed to test hypotheses, a firm commitment to funding and institutional support for monitoring is lacking, or when not enough data can be collected to evaluate progress.

IMPLEMENTING ADAPTIVE MANAGEMENT

There are three forms of implementation of adaptive management (Walters and Holling 1990; Allan and Curtis 2005). One is evolutionary, or "trial and error," in which early choices are essentially haphazard, while later choices are made from a subset that gives better results. Second is passive adaptive, where historical data for each time period are used to construct a single, best estimate or model for response, and the decision choice is based on assuming this model is correct. Third is active adaptive, where the data available at each time period are used to structure a range of alternative response-models, and a policy choice is made that reflects some computed balance between expected short-term performance and long-term value of knowing which alternative model is correct.

BENEFITS AND SUCCESSES OF ADAPTIVE MANAGEMENT

The primary benefits of the adaptive management technique are many. The process encourages long-term collaboration among stakeholders, managers, scientists, and policy-makers (Williams et al. 2009; Williams and Brown 2014). Sharing understanding of the problem and a teamwork setting for finding solutions help to overcome divisions among involved parties. Also, the stakeholders learn management outcomes and the responses of the environmental system. The sharing of perspectives and learning among diverse partners can bring about consensus on management policies that solve conflicts. Acting under uncertainty can be seen as risky, but adaptive management encourages action and has a philosophy that accepts low risk (Figure 11.4) (Allen and Gunderson 2011). The expectation of selecting a management plan with limited information favors learning (Williams et al. 2009). A team effort to explore options and choose a course of management action, forces careful consideration of the objectives, why certain decisions were made, and when results are to be expected. This builds focused decision-making, detailed justifications, and documentation on debates and resolutions (Williams et al. 2009). This also enhances information flow to policy-makers and administrators running complex agencies or organizations. Finally, the concept that management can and should change when more information is available allows decisions to be seen as non-permanent, flexible over time in the face of uncertainty, and iterative (Williams et al. 2009).

Moore et al. (2011) evaluated the success of United States government programs in implementing adaptive management at scales ranging from small, single refuge applications to large, multi-refuge, multi-region projects. Their evaluation suggested three important attributes common to successful implementation: a vigorous multi-partner collaboration, practical and informative decision framework components, and a sustained commitment to the process. Successful application of adaptive management also requires building a thorough understanding of the various elements of the process through cumulative experience (Gerber et al. 2007).

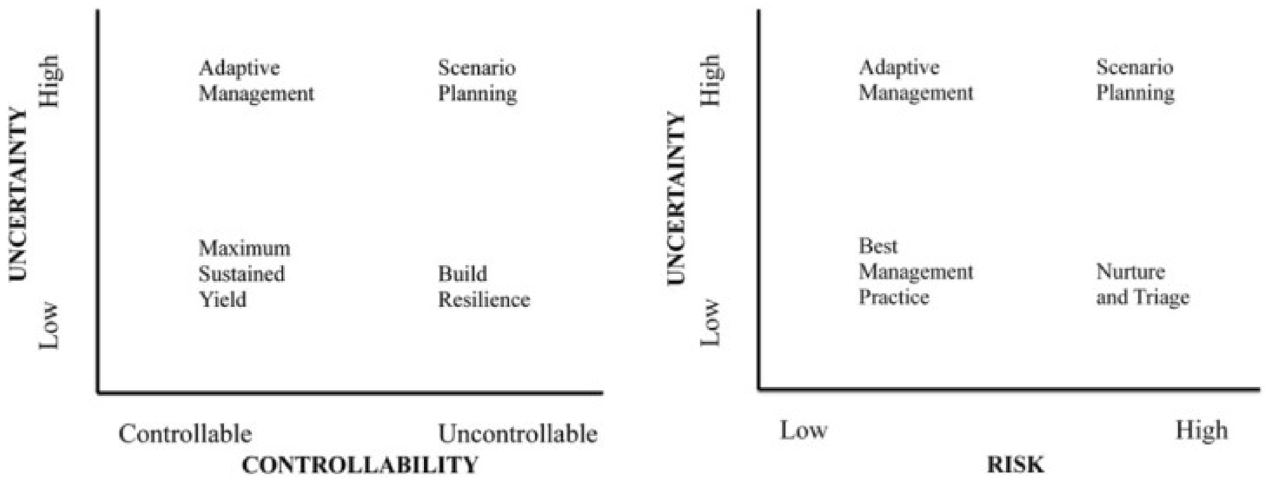

Figure 11.4: Conditions under which adaptive management could be most effective. Source: Allen and Gunderson 2011

LIMITATIONS IN APPLYING ADAPTIVE MANAGEMENT

There are situations where adaptive management is not appropriate. Polarized perspectives on management alternatives, complex litigation settings, legal mandates for specific actions, and institutional reluctance for change may make adaptive management impossible to accomplish. The experimental nature of adaptive management can be another limitation. If there are substantial time lags (e.g., decades) between management actions and system responses, the iterative cycle of adaptive management can be too long. Also, there is a possibility that the environmental setting is rapidly changing, resulting in failed monitoring of the system. Finally, it is possible that the experimentation cannot be properly evaluated because so many confounding factors shape outcomes. These are some examples of situations in which adaptive management can be too difficult to implement and maintain for agencies to support the process.

A key tenet of adaptive management is that collaboration among scientists, managers, and stakeholders results in creative exploration of management options and selection of innovative paths for improved management. However, at times the outcome from a collection of diverse collaborators can stymie truly innovative management options due to the need to achieve consensus across participants. Incompatible perspectives and goals within a group can limit the breadth of considerations and be restrictive of management options. For example, in California, adaptive management of the Sacramento-San Joaquin Bay-Delta did not consider reducing water use by people because it was impossible to reach consensus on this approach (Kallis et al. 2009). Thus tension in collaboration can stifle flexibility, creativity, and adaptability.

A second limitation of collaborative development of management is the temptation to avoid risky options. Basic disagreements on management outcomes among collaborators can limit risk-taking because of the fear of failure. Managers also will be held accountable for failure even though management can be improved by learning and trying new ideas. The process of adaptive management increases transparency in decision-making and adds public and stakeholder attention to management plans. This can elevate the attention on likely management benefits and chances of failure. Many managers avoid bad news coverage, and can resist options debated during the extended experimental period. Adaptive management can be effective for those involved in the process, but for the public the management decisions will require detailed explanations.

Adaptive management can be a costly process. Expenses are generated by ecosystem scale experiments, collaboration, information gathering, modeling, and monitoring. This process takes time and financial resources from agencies, and can be hard to maintain for the long-term, when adaptive management benefits are ultimately realized. Sometimes the adaptive management process is associated with inadequate support, resulting in weak application of science, superficial modeling, and ineffective monitoring. A lead agency can weaken the process for cost and effort reasons, making experimentation and learning less effective across the management options. In practice, adaptive management can miss the mark on the theory of the process and result in a compromised program.

There have been some reviews of adaptive management cases. For example, Walters (1997) investigated 25 adaptive management cases and found only two were effectively executed. Most others ended with no product, little learning, and no improved management. One trap noted was protracted model development and refinement. Overall, there are few adaptive management cases where management was improved by the process. Adaptive management demands fundamental changes in practices and agency operations and an extended period of commitment to realize the gains. New ways of decision-making, information sharing, and learning by experiments take many years of commitment and investment by scientists and managers. Admitting a lack of understanding, the need to learn, and the sharing of information can be difficult for organizations to embrace. Government and agency leadership often changes faster than the time period that the adaptive management process requires. It is often said that history repeats itself, and learning from experience can be slow, unrecognizable, or lost by organizations.

CASE STUDY: GLEN CANYON DAM ADAPTIVE MANAGEMENT PROGRAM

Glen Canyon Dam is directly upstream of Glen Canyon and the Grand Canyon National Park (Figure 11.5). Completed in 1963, this dam and its operations have profoundly changed the nature of the Colorado River in the Grand Canyon and upstream canyons. Scientific evidence gathered during Grand Canyon environmental studies indicated that significant impacts on downstream resources were occurring due to the operation of Glen Canyon Dam. Some of the major effects on the river were cold water temperatures, termination of down river sediment transport, clear water releases, cessation of seasonal high flows and flooding, species endangerment, proliferation of non-native species, dense vegetation cover on shoreline and sand bars, altered channel morphology, and more (Stevens and Waring 1985; Webb et al. 1999). These findings led to a July 1989 decision by the Secretary of the Interior to prepare an environmental impact statement to reevaluate dam operations. The purpose of the reevaluation was to determine specific options that could be implemented to minimize, consistent with law, adverse impacts on the downstream environment and cultural resources, as well as Native American interests in Glen and Grand Canyons. The United States Congress passed the Grand Canyon Protection Act in 1992 with a mandate to modify dam operations to “protect, mitigate adverse impacts to, and improve the downstream resources of the Grand Canyon National Park and the Glen Canyon National Recreational Area” (United States Congress 2021). The Colorado River in the Grand Canyon cannot be fully restored to pre-dam conditions, and there was substantial uncertainty about the environmental and river responses to modified dam operations.

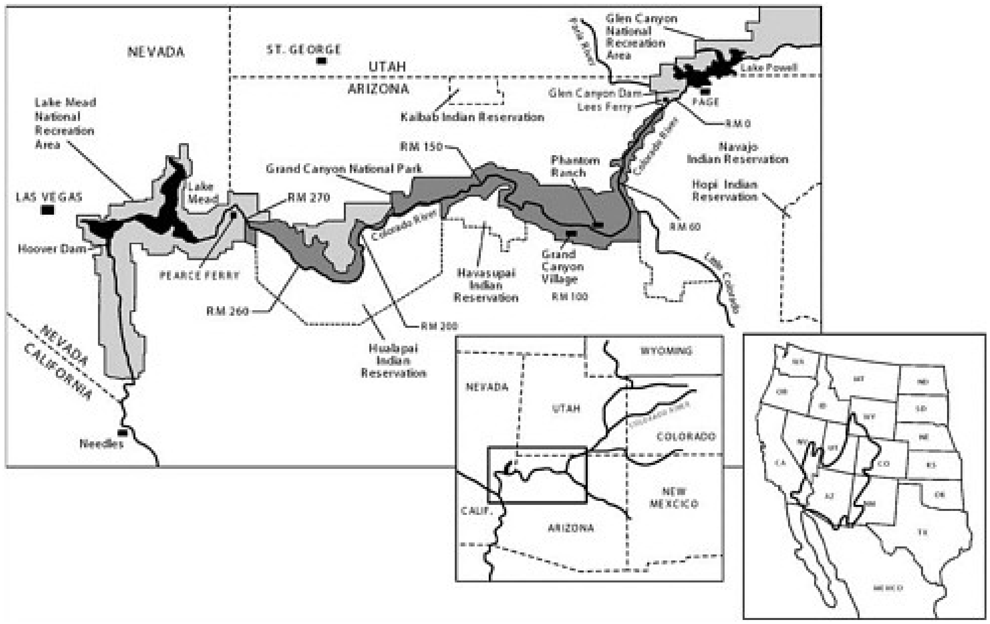

Figure 11.5: Placement of Glen Canyon Dam in the Grand Canyon River Ecosystem and Colorado River Basin (Inset). Source: Boesch and the National Research Council Panel on Adaptive Management for Resources Stewardship 2004

An environmental impact statement was issued in 1995 and included information on beaches, endangered species, ecosystem integrity, fish, power costs, power production, sediment, water conservation, air quality, rafting and boating, and the Grand Canyon as wilderness (United States Department of the Interior 1995). The United States Secretary of the Interior made a decision to adopt modification of Glen Canyon Dam operations including establishing an adaptive management program (United States Department of the Interior 1995). The adaptive management program was adopted to deal with the uncertainty of dam operations on the canyon environments (Wieringa and Morton 1996). Also, the program was implemented to conduct management experiments to fulfill obligations under the Grand Canyon Protection Act. Experimental dam operations, long-term monitoring, and extensive research on options for additional dam operation modifications were seen as necessary. Environmental commitments made by the Secretary also included building beaches and habitats with flow events, protection of cultural resources, increasing flood frequency, and recovery of the endangered humpback chub (*Gila cypha*). The adaptive management program was charged with:

1. Developing models to predict effects of policies, activities, and operations that are being considered,
2. Formulate hypotheses on outcomes of dam operations and management actions,
3. Conduct experiments to test hypotheses,
4. Monitoring and evaluations of management activities, and
5. Integrate new knowledge and information in management options and recommendations to the United States Secretary of the Interior.

The tasks identified for this adaptive management program are very consistent with the concept of adaptive management. Also, this case shares many attributes of feasible adaptive management cases because of its importance, long-term investment, and high-level government support.

The Adaptive Management Work Group was appointed by the United States Secretary of the Interior with representatives of the federal agencies, states, Native American tribes, environmental conservationists, recreational organizations, and electric power user groups. This Work Group led the adaptive management program for protecting and mitigating adverse impacts to the Grand Canyon National Park and the Glen Canyon National Recreational Area. The responsibilities of the Adaptive Management Work Group were to annually review monitoring data and hypothesis tests to determine if objectives were being attained, develop recommendations to the United States Secretary of the Interior for modifying dam operations, and facilitate input from interested parties. Issues of interest spanned natural canyon properties like open beaches and shorelines and the recovery of endangered species as well as non-natural features like the trout fishery below Glen Canyon dam. Therefore, the Adaptive Management Work Group had to develop management options for a novel ecosystem and consider human uses of the National Park and National Recreational Area.

A prominent modification of Glen Canyon dam operations promoted by the Adaptive Management Work Group was periodic controlled floods. An experimental flood was conducted in 1996 with an increased water release of 1,274 m^3/s from the dam for seven days (Figure 11.6). The test flood was needed to test the hypothesis that the dynamic nature of fluvial landforms and aquatic and terrestrial habitats can be wholly or partially restored by short-duration dam releases substantially greater than power-plant capacity. This controlled flood was predicted to restore open sandy beach, scour riparian vegetation, and create shoreline nursery habitats for an endangered fish. The controlled flood was recognized as much smaller than pre-dam floods that ranged from 3,000 to 8,500 m^3/s. Field studies were executed to test predictions of environmental benefits from restoring flood flows to the canyon. This modification of dam operations succeeded in building sandbars by moving sediments from the channel to the river shorelines, creating higher but not wider sandbars (Figure 11.7) (Collier et al. 1997). Sediment buried some riparian vegetation but was insufficient to scour perennial riparian vegetation, especially woody species. There was no detectable harm on endangered species, but there were clear improvements for recreational rafters because of new camping beaches (Stevens et al. 2001). Lake Powell above the dam dropped 1.1 m and $2.5 million in hydropower revenue was lost (Patten et al. 2001). Research costs were $1.5 million for a total financial investment in this operational modification of about $4.0 million (Patten et al. 2001).

The conclusions of the controlled flood experiment were that restoring high flows to the canyon altered the ecosystem in beneficial ways, improved understanding of varying flow rates, and that high flow periods (part of the natural flow regime) should be part of the operations of the dam. The 1996 controlled flood received substantial press coverage, and now this practice has been implemented on many rivers throughout the world and again from the Glen Canyon dam with different river volumes and durations (Figure 11.8) (Melis et al. 2010; Topping et al. 2010; United States Department of the Interior 2011; Yao and Rutschmann 2015). Monitoring data and hypothesis tests improved sediment modeling, and this improvement allowed managers to determine the best frequency, timing, duration, and magnitude for future controlled floods. Varying the specifications for controlled floods is an example of enhanced management actions being developed by the adaptive management process.

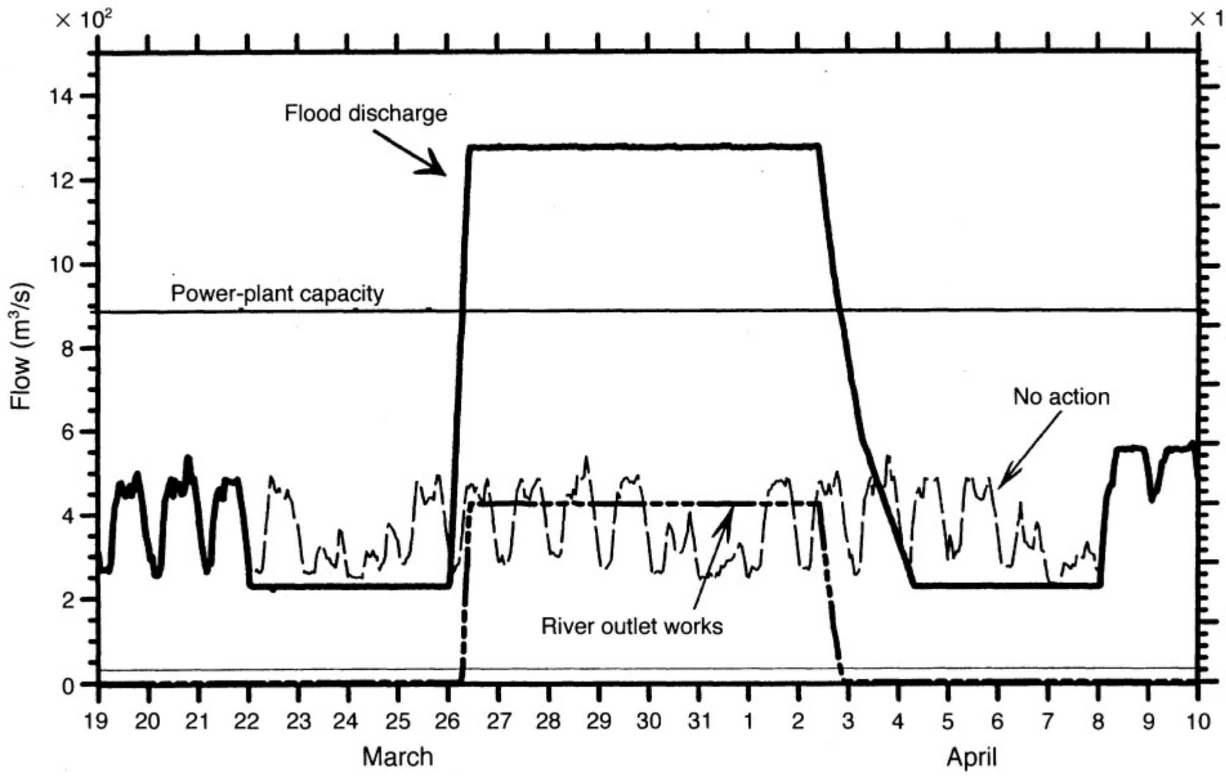

Figure 11.6: Hydrograph of the Colorado River showing flows from Glen Canyon Dam in 1996 when the first controlled flood was implemented. Source: Patten et al. 2001

Integration of knowledge and perspectives by scientists and managers is fundamental to improving decisions under adaptive management. The Adaptive Management Work Group included scientists that are experts in fields important to Grand Canyon issues. That built credibility for expensive and novel dam operation changes. Managers engaged in Grand Canyon issues pressed scientists to be focused on management needs. Monitoring and hypothesis testing of new dam operations enabled managers to learn how modifications of dam operations could influence ecosystem properties. The controlled floods illustrated how science, modeling, and public use of the Grand Canyon were combined to change dam operations. The diverse Adaptive Management Work Group dealt with difficult trade-offs and reached a compromised strategy that improved management. This was seen from the cooperation and planning involved in the flood events as they were a big departure from standard dam operations and involved working with special interest groups that were strongly resistant to change. Developing consensus among stakeholders on the use of scientific information and managed floods for sediment and resource management remains a primary challenge to the Adaptive Management Work Group.

Figure 11.7: A sandbar in Grand Canyon that was created by the 2008 controlled flood. To the left of the sandbar is a newly created backwater used by endangered humpback chub (Gila cypha) young as rearing habitat. This sandbar also provides beaches for camping by hikers and whitewater rafters. Source: Melis et al. 2010

Commonly, with formal environmental impact assessments under the National Environmental Protection Act, the adoption of the preferred management alternative is viewed as a "forever" decision. The Department of Interior's 1995 impact statement included adaptive management in the final decision. This inclusion opened the way for modifying dam operations through time to improve the Grand Canyon ecosystem and allow for management flexibility with new knowledge. It also allowed science, monitoring, and hypothesis testing to continue with the purpose of improving both management and the ecosystem. The experimental results from the Glen Canyon Dam program represent scientific successes in terms of revealing new opportunities for developing better river management policies (Melis et al. 2015). This is a case where a United States national treasure was degrading and Congress acted to mandate that the impacts on it be mitigated and the ecosystem improved.


Figure 11.8: Glen Canyon Dam in March 2008 during a controlled flood with increased release of water. Source: Melis et al. 2010

SUMMARY

In truly important cases, like the Glen Canyon Dam reviewed here, all of the adaptive management program attributes were implemented in a manner that matched the concept of adaptive management: assessing the problem, predicting outcomes based on current information, implementing a plan, monitoring the outcome, evaluating success or failure, and adjusting management actions. There are few cases like this; many applications of adaptive management have failed (McLain and Lee 1996). However, when the government and agency investment in the process is significant, it appears that the benefits of adaptive management can succeed over time and truly improve holistic ecological conservation (Bormann et al. 2007).

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Chapter 12 - Ecosystem Services

The third topic in the holistic techniques group is ecosystem services. Natural ecosystems provide humans with many and diverse benefits and products. These benefits are called ecosystem services. In theory, recognizing the value of nature through the services it provides should greatly increase investments in conservation, while at the same time fostering human well-being. In other words, if we align economic forces with conservation principles that explicitly link human and environmental well-being then theory can become practice. In this chapter we will cover the background and justification for emphasizing ecosystem services as a technique for conservation, and the attempts made to place economic value on those services. We will end with a case study on payments for ecosystem services to ranchers in Central and Southern Florida.

BACKGROUND ON ECOSYSTEM SERVICES

Natural ecosystems provide humans with many diverse products and benefits (Daily 1997). Recognition of these products and benefits, known as ecosystem services, is considered important for increasing investments in conservation (De Groot et al. 2010). Often in the process of engineering ecosystems for valued products like food, wood, and fiber, many other less valued benefits become diminished or eliminated. The importance of ecosystem services has been growing in science, conservation, and government policies since the late 1990s with the publication of Nature's Services: Societal Dependence on Natural Ecosystems, edited by Gretchen Daily of Stanford University (1997). This book made the argument that society should invest in the conservation of ecosystems to secure a diversity of services that support human well-being. Fish (2011) expanded on this idea by saying that we need to think holistically about how any given project, proposal or plan impacts the provisioning of ecosystem services and human well-being. Altogether, the basic idea is to identify and assign values to ecosystem services for justifying conservation efforts, and making the protection of natural ecosystems important and appealing to the public, businesses, and the government. Or, put more simply, that "nature provides humans with benefits" (Persson et al. 2015).

Ecosystem services may be defined in slightly different ways, but the central idea is that natural ecosystems support human well-being (Millennium ecosystem assessment 2005). Ecosystem services may be used actively or passively to benefit people (Fisher et al. 2009). These services depend on ecosystem organization, functions, and products. Some common services, such as timber, water supply, and recreation, already have market value, so the benefit can be easily estimated. Other services like air pollution removal, carbon sequestration, soil development, and local-climate mitigation are challenging to convert to a monetary value, but they are nevertheless seen as important benefits to people (Example in Figure 12.1).

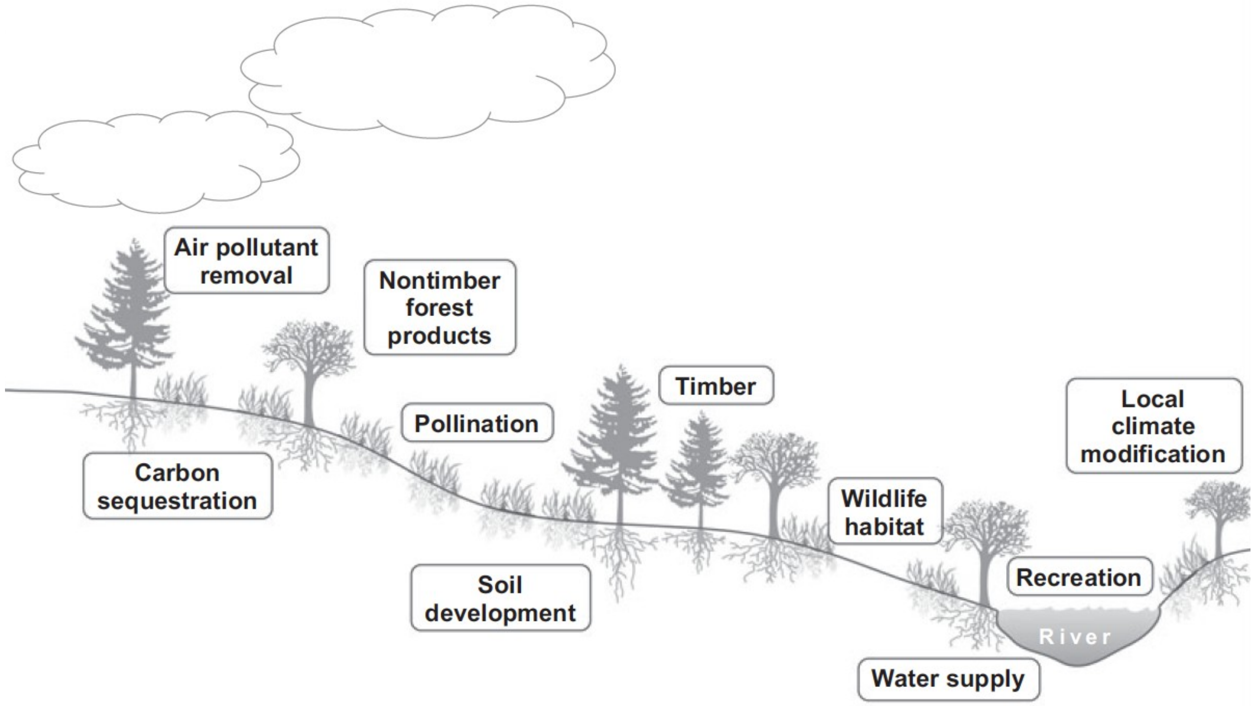

Figure 12.1: Illustration of ecosystem services across a forested river valley. Source: Brauman et al. 2007

The ecosystem services technique for ecological conservation (also termed integrated conservation–development, and community-based natural resource management) includes concepts from both ecology and economics to define benefits, determine valuations, and guide investments in conservation (Braat and De Groot 2012). Capital is defined as a stock of materials or information that exists at a point in time. There are three forms of capital: natural, manufactured, and human. Natural capital can be seen as trees, minerals, clean air and water, and natural lands. Natural capital, or an intact ecosystem, provides a diverse flow of ecosystem services (Fisher et al. 2013). Changes in natural capital will change the benefits to people, and can be thought of as an alteration of the flow of services. Human capital is defined as people that have the capacity to produce valued items. Manufactured capital are things such as machines, buildings, factories, and equipment. Services are flows that transform materials, or the spatial configuration of materials, to enhance the welfare of humans. Ecosystem services consist of flows of materials, energy, and information from natural capital stocks, which combine with manufactured and human capital services to produce human welfare (Costanza et al. 1997). Changes in the particular forms of natural capital and ecosystem services will alter the costs or benefits of maintaining human welfare.

Political leaders, conversation organizations, and scientists around the world are increasingly recognizing ecosystems as natural assets that supply life-support services of tremendous value, and are striving to merge conservation with economic systems (Guerry et al. 2015). The ending statement from a paper by Daily et al. (2009) concisely summarizes the approach and aim: "If we can get the price closer to being “right,” everyday behavior and decisions will be channeled toward a future in which nature is no longer seen as a luxury we cannot afford, but as something essential for sustaining and improving human well-being everywhere." This fundamental concept of the ecosystem services technique is now working to attract the public, policy-makers, and scientists to advance conservation on an ecosystem scale.

CLASSIFICATION OF ECOSYSTEM SERVICES

Table 12.1: Ecosystem services used in two studies and one review of policy cases. Sources: Costanza et al. 1997, Millennium Ecosystem Assessment 2005, and Fisher et al. 2009.

Study purpose and source	Ecosystem Services
Assess ecosystem change worldwide (Millennium Ecosystem Assessment 2005)	<u>Provisioning services</u>: food, fiber, fuel, genetic resources, biochemicals, natural medicines and pharmaceuticals, ornamental resources, fresh water <u>Regulating services</u>: air quality, climate, water, erosion, water purification, disease, pest, pollination, natural hazards <u>Cultural services</u>: cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, recreation and tourism <u>Supporting services</u>: soil formation, photosynthesis, primary production, nutrient cycling, water cycling
Estimate values of biomes worldwide (Costanza et al. 1997)	Gas regulation, climate regulation, disturbance regulation, water regulation, water supply, erosion control and sediment retention, soil formation, nutrient cycling, waste treatment, pollination, biological control, refugia, food production, raw materials, genetic resources, recreation, cultural
Review of 34 cases where ecosystem services were used in policy making (Fisher et al. 2009)	<u>Water</u>: regulation, supply, quality, cycling, provision, flood control, drought prevention, watershed protection, salinity mitigation, saltwater intrusion <u>Forests</u>: timber, non-timber forest product, firewood, charcoal, fire protection <u>Cultural</u>: recreation, tourism, social benefits, values, ornamental and ceremonial use, aesthetic values <u>Climate</u>: carbon sequestration, regulation, oxygen <u>Natural threats</u>: hurricane mitigation, tsunami defense, coastal protection <u>Erosion</u>: control, shore stabilization, sedimentation <u>Products</u>: food, agriculture, plantations, fish, fisheries, hunting, trapping, oil and gas, hydroelectricity, raw materials, medicinal plants, construction materials <u>Biodiversity</u>: provision of habitat, nursery habitat, endangered species <u>Ecosystem processes</u>: nutrient cycling, nitrogen fixation, disease control

The United Nations launched the Millennium Ecosystem Assessment in 2001, and this effort became a standard-setting application of the ecosystem services concept (Millennium Ecosystem Assessment 2005). The goal was to assess the consequences ecosystem change can have on human well-being (Figure 12.2) (Alcamo 2003). Thousands of scientists from all over the world were involved, and together they developed a methodology of ecosystem services analysis. This framework for ecosystem services is commonly used today, and sets the standard for identifying and categorizing these services. Ecosystem services were organized into three classes, with a fourth class that supports the production of services (Table 12.1). The first class is provisioning services that are directly used by people such as food, fiber, fuel, clean water, space for recreation, and others. These goods and services are commonly marketed, and estimations of their values are easily calculated. A second class is cultural services that include religious values, heritage, social relations, aesthetics, and other societal values. These services benefit people in many ways, but are largely noneconomic and their monetary values are often difficult to estimate. The third class is regulating services, which shape and modify local climate, erosion, pests, water quality, and other ecosystem features. The benefits provided by regulating services can be hard to estimate and quantify, so ascribing a monetary value for them is challenging. The final category is supporting services, which create the conditions necessary for the production of the other three classes of services that people directly use. Most of these supporting services involve cycling or production of biologically-linked compounds like water, nutrients, oxygen, biomolecules, and biomass. These are essential components of ecosystems, though assigning monetary values for them can be diffi-

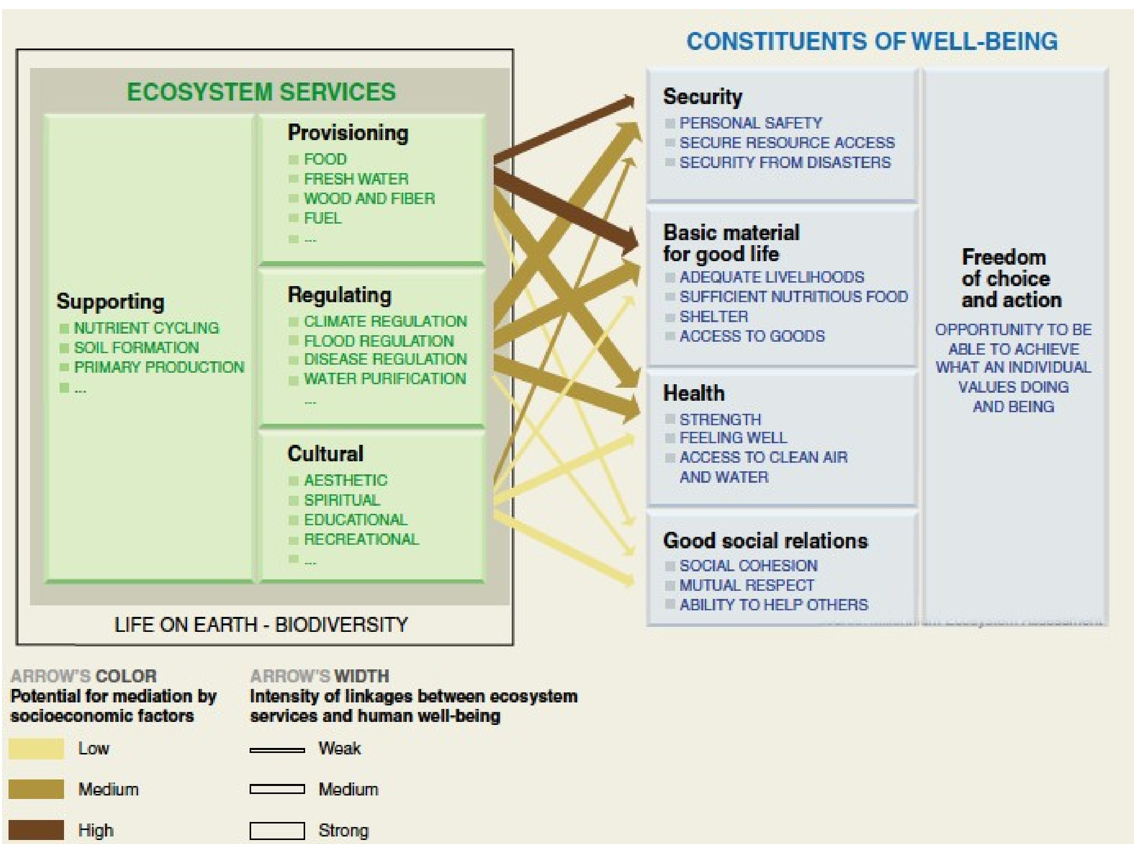

Figure 12.2: Linkages between ecosystem services and human well-being. Source: Millennium Ecosystem Assessment 2005

cult. The relationship between these services with human well-being has been described, which is informative as to why these services are important (Millennium Ecosystem Assessment 2005). Monetary values were not featured in the Millennium Ecosystem Assessment reporting, but increasing and decreasing trends in services production were identified to communicate the impact of ecosystem change on human well-being. Overall, the Millennium Ecosystem Assessment stimulated efforts to include ecosystem services practices in ecological conservation.

A few other applications of the ecosystem services technique sought to identify and classify services. Costanza et al. (1997) estimated the value of biomes across 17 ecosystem services (Table 12.1). This list of ecosystem services predated the Millennium Ecosystem Assessment framework and is simpler by comparison. Fisher et al. (2009) reviewed thirty-four cases where ecosystem services were included in environmental management policy-making (Table 12.1). They classified ecosystem services into broad categories of: water, forests, cultural, climate, natural threats, erosion, products, biodiversity, and ecosystem processes. Shepherd et al. (2016) charted the global trend (positive trend, negative trend, no trend, ambiguous trend, or no indicators detected) for a variety of provisioning, regulating, and cultural service (Table 12.2). These applications of ecosystem services highlight potential services for consideration in actual conservation cases. Examples such as these have been useful to practitioners (wetland example in Figure 12.3).

Table 12.2: Global status of ecosystem services. Source: Shepherd et al. 2016

Ecosystem service	State	Benefit	Access	Overall
Provisioning services:				
Food	↘	↗	↗	(↘↗)
Raw materials	↘	↗	–	(↘↗)
Fresh water	↘	↘	–	↘
Medicinal resources	↘	↗	–	(↘↗)
Regulating services:				
Local climate and air quality	–	(↘↗)	→	(↘↗)
Moderation of extreme events	→	(↘↗)	↘	(↘↗)
Waste–water treatment	↘	–	–	↘
Erosion prevention and soil fertility	–	↘	→	↘
Pollination	↘	↗	–	(↘↗)
Biological control	–	↗	–	↗
Cultural services:				
Recreation and physical and mental health	–	↗	–	↗
Aesthetic appreciation and inspiration	–	–	–	–
Spiritual experience and sense of place	–	–	–	–

↗ positive trend; ↘ negative trend; → no trend; (↘↗) trend ambiguous;—no indicators identified.

MANAGEMENT OF ECOSYSTEMS FOR SERVICES

For effective management and conservation of ecosystems that provide natural services, there is a need to engage the local society in the planning process. The approach for how to address this need has not yet been established for ecosystem services support. First, the ecosystem has to be assessed to identify ecosystem services, beneficiaries, and responsible management organizations. These issues must be resolved to allow the ecosystem services process to be incorporated into local land use planning. How should an ecosystem's structure and function be maintained in order to sustain the flow of services that are in demand? The valuation of services can be important for assessing tradeoffs related to the costs of ecosystem maintenance and the production of services. Then the engaged group of stakeholders, managers, conservationists, and the public can debate the possibilities and limitations of what can be done to maintain the ecosystems and their valued services. This step starts the planning of a vision and strategy for protecting a natural ecosystem and the services it provides. Also, an agenda of objectives and actions are needed to form policies and practices to maintain the ecosystem. With the management plan in place and with local support, the process of coordinating actions across parties to pursue the vision and strategy for ecosystem protection can begin.

While there has been much interest in ecosystem services for ecological conservation, it has not received much attention in the scientific literature (Laurans et al. 2013). This begs the question: to what degree is it actually used in mainstream applications? The impact of ecosystem services concepts on environmental decision-making is not evident, with marketed goods and services being the more predominant factor in most cases (National Research Council 2005). Ecosystem services ideas have more recently found their way into discussions of conservation and management programs. Non-governmental conservation organizations have been out in front of this issue by adopting the spirit of ecosystem services. The true merger of ecosystem ecology and economics has begun, and there have been several prominent applications of ecosystem services practices with the vision of ecosystems benefiting mankind.

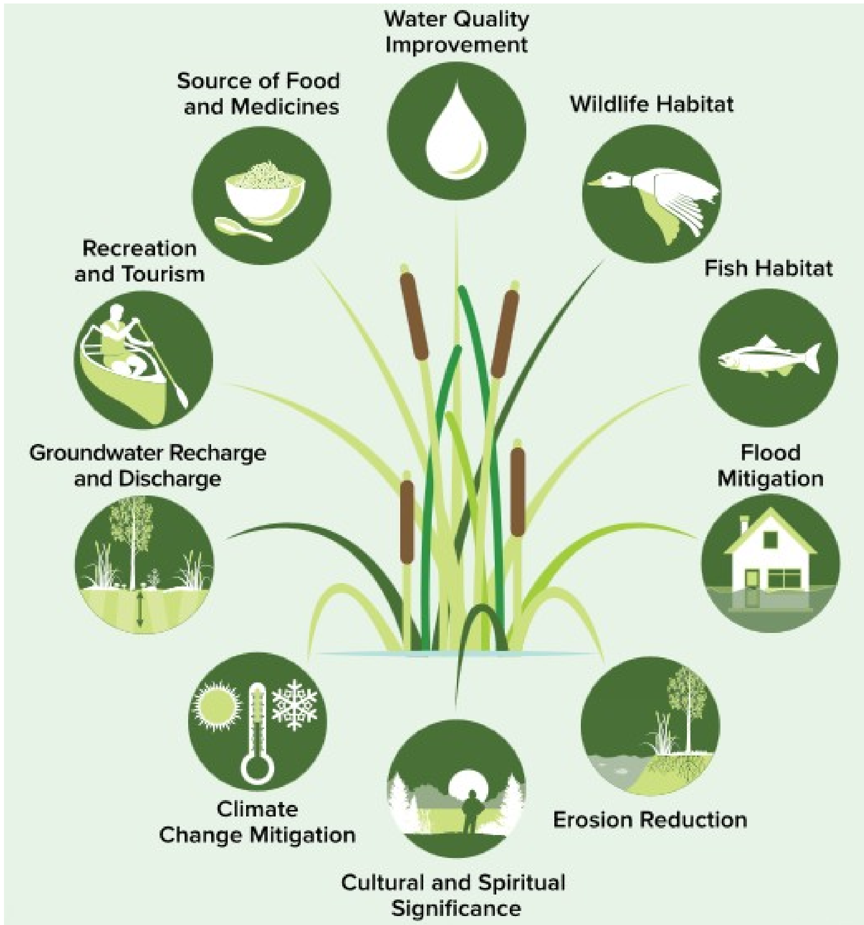

Figure 12.3: Wetlands provide a multitude of ecosystem services. Source: Ontario Ministry of Natural Resources and Forestry 2017.

INCREASING RESEARCH FOCUS ON ECOSYSTEM SERVICES

Scientific investigation into ecosystem services has increased greatly in recent years (Cowling et al. 2008; McDonough et al. 2017). One avenue of research has focused on the need to learn more about the species and properties of ecosystems that are important for producing ecosystem services (Kremen 2005). There is also a need to characterize and specify the conditions that are important for supporting individual services. And, there is a need to understand how to protect service providers within an ecosystem. There can be multiple species that are important for providing services, and we need to know how redundancy of service providers works. Drivers of ecosystem function may provide services with similar dynamics, making it easier to manage the flow of services. If that commonly holds, then ecosystem services could be managed together as components of an ecosystem. The structure of ecosystems which support a variety of services needs to be identified in order to maintain efficiently functioning ecosystems, stabilize the flow of services, and enhance resilience of the ecosystem to disturbances. Much research is focused on these points to understand the dynamics of ecosystems and the services they provide (Costanza and Farber 2002).

At the ecosystem scale, there is a concern that optimizing some services can change ecosystem functions, leading to a transition to a new ecosystem with much different service production (Elmqvist et al. 2003). For example, converting a prairie landscape into a largely agricultural production zone will terminate many ecosystem services. This scale of service reorientation results in a new domesticated ecosystem that lacks a high variety of services. More subtly, the concern is that managing ecosystem services could alter the ecosystem through time and reduce levels of other desirable services. Research on this concern needs to identify thresholds or tipping points that are associated with a transition to a new ecosystem with a different set of services. The goal of this research is to consider managing ecosystem services in a way that does not diminish the resilience of the ecosystem to maintain its basic nature.

SPATIAL AND TEMPORAL FACTORS SHAPING ECOSYSTEM SERVICES

At the ecosystem services level, information is being generated on ecosystem attributes that shape the distribution, production, and persistence of services. Much attention has been aimed at spatial distribution of ecosystem services and when service benefits are generated in time (Naidoo et al. 2008; Fisher et al. 2013). With this information, conservationists and environmental managers can specify landscape locations and times that produce specific services and maintain the requirements for service production. Also, management organizations must be matched to the ecosystem scales that are important for maintaining services production. Mapping of ecosystem service locations and times has been a research priority for maintaining service hot spots in a landscape context (Schröter and Remme 2016).

Some studies have related ecosystem services to land cover classes in an effort to map ecosystem services patterns across a landscape (Chan et al. 2006; Koschke et al. 2012). This is a way to design conservation plans which maintain land cover classes in an effort to sustain service production. A thorough study of this type was done by Chan et al. (2006) across the Central Coast Ecoregion of California. Six ecosystem services were mapped (carbon storage, crop pollination, flood control, forage production, recreation, and water provision) and planning units of 500 ha were tracked across the ecoregion. Almost all spatial correlations among ecosystem services were low, indicating that services were not sharing the same spatial patterns on the landscape. One moderate correlation was between carbon storage and water provisioning, since higher elevations were forested and received more precipitation. Cities and human population centers influenced the distribution of demand for some services, and shaped some service maps. This study and others point to the need for multidisciplinary teams in planning ecosystem service strategies due to the complex pattern of service production and its inconsistent distribution. For planning ecosystem services across large landscapes, there have to be broad conservation goals and a variety of stakeholders and experts involved.

A current priority for science is to improve our understanding of the relationships among ecosystem services within an ecosystem (Bennett et al. 2009). Recent studies indicate that many services respond differently to changing conditions and will not respond in synchrony (Raudsepp-Hearne et al. 2010).

Thus, there is a real risk that managing ecosystem to maximize some of the more desirable services will lead to tradeoffs in other services (Abson and Termansen 2010). In contrast, there are some findings that indicate that some ecosystem services responses are linked together with some drivers, which raises the prospect of synergies among sets of services (Raudsepp-Hearne et al. 2010). A better understanding of service tradeoffs and synergies can yield efficiencies in managing sets of services and avoiding undesirable tradeoffs. Ecosystem services that have common response patterns are often called service bundles (Raudsepp-Hearne et al. 2010). These service bundles have been identified by both spatial distribution and homogeneous response patterns, indicating coincidence of both location and behavior. This knowledge can help to support better management and conservation efforts, since bundling ecosystem services broadens and simplifies planning for maintaining the flow of services.

ECONOMIC VALUATION OF ECOSYSTEM SERVICES

Research on the methods for economic valuation of individual ecosystem services is ongoing (Costanza et al. 1997; Pimm 1997; Mendelsohn and Olmstead 2009). This level of investigation requires merging ecology and economics, which is a new challenge. Also, estimates of services valuations need to be presented to decision-makers and the public, and explained in a way that justifies efforts to conserve these services. The greatest challenge is for non-market based ecosystem services, as these need novel and creative methods for determining valuations. Research on monetary valuation of ecosystem services is ongoing and is critical to the science-based agenda for promoting this ecological conservation technique.

The economic justification for maintaining ecosystem services is the other dimension of this ecological conservation technique. Mechanisms for incentivizing ecosystem conservation are often based on economic values (Jack et al. 2008). Natural ecosystems are commonly exploited and greatly altered for production of goods and services for markets (Farnworth et al. 1981). However, research by Balmford et al. (2002) on the values of goods and services delivered by a relatively intact biome, and one which has been converted to typical forms for human use, showed that the loss of non-marketed goods and services may commonly surpass the economic worth of the marketed products (Figure 12.4). For example, the research showed that there were high initial benefits when destructive fishing techniques were used, such as blasting, but sustainable fishing yielded benefits over the long-term (Balmford et al. 2002). The social benefits of sustainable exploitation (e.g., coastal protection and tourism) were also lost through blasting. The economic value of retaining an essentially intact reef was almost 75% higher than that of destructive fishing (at \$3300/ha compared with \$870/ha) (Balmford et al. 2002). This example argues for better economic valuation of natural ecosystem services which confronts short-term private gains with long-term public loss of natural services. There are three rationales that underlie this short-term perspective on economic gains: 1) Lack of fair valuations for ecosystem services leads to domination of market-based actions; 2) Ecosystem conversions and degradation are commonly justified by tangible local-scale gains rather than losses to society at an expansive scale in space and time; 3) Finally, government policies often encourage short-term economic benefits. The private benefits of conversion are often exaggerated by interventions such as government tax incentives and subsidies. These pressures promote the loss of natural ecosystems, foil sustainable use of natural landscapes, and encourage decision-makers to favor programs which support near term economic gains (Balmford et al. 2002). In short, "Our relentless conversion and degradation of remaining natural habitats is eroding overall human welfare for short-term private gain" (Balmford et al. 2002). An effective merger of ecological science with economics may stop or slow these trends and justify new thinking about ways we approach the conservation of natural landscapes and ecosystems. Retaining as much as possible of what remains of nature through a combination of sustainable use, conservation, and compensation for attendant opportunity costs, makes economic as well as moral sense (Balmford et al. 2002).

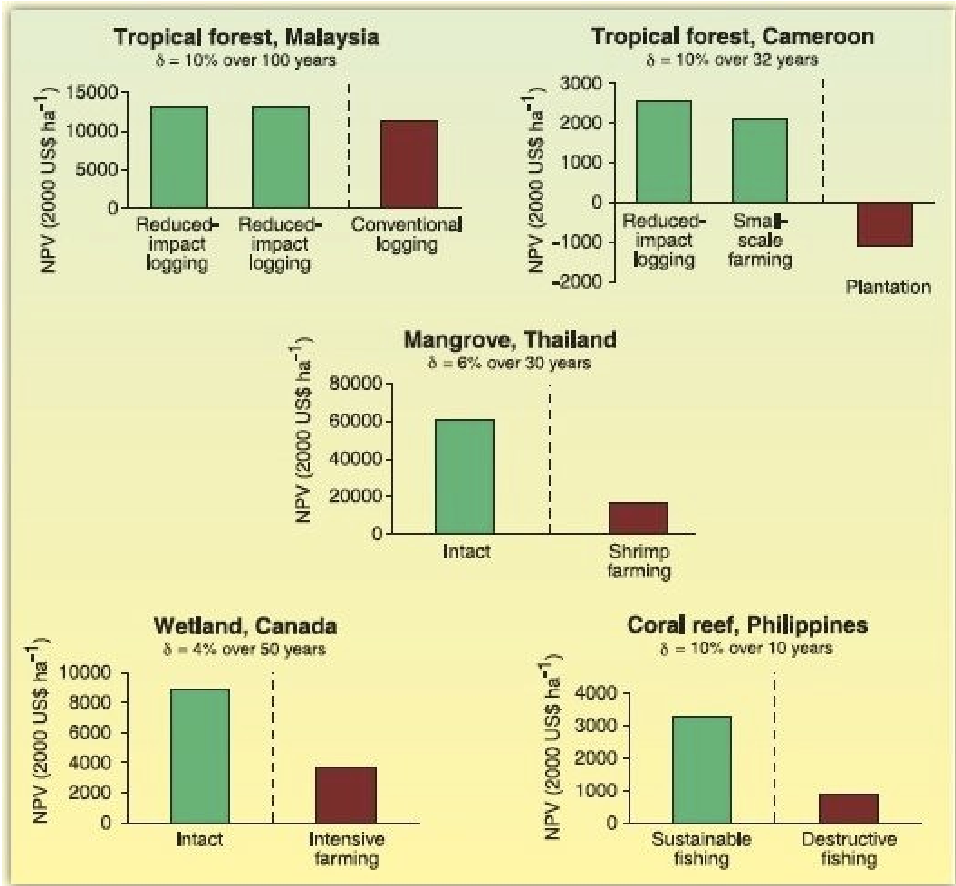

Figure 12.4: Loss of non-marketed services (green) outweighs the marketed benefits of conversion (red), often by a considerable amount. Source: Balmford et al. 2002

There are many challenges to fully valuing a wide range of ecosystem services. Methods are available for valuing both marketed goods and direct benefits to people. However, most services do not contribute directly to human needs (e.g., food production) because most services have indirect benefits (e.g., nutrient cycling). These indirect benefits are diffuse and usually not directly used by people. Such services are often time delayed, and difficult to quantify. Another challenge is that valuations are often specified and considered as stable so they are viewed as independent of the dynamics of ecosystems. Another issue is the risk that follows from the loss of some services. For example, mangroves provide flood protection from coastal storms, but their removal makes space for shrimp farms. Floods are considered rare events, but when they occur the safety and protection that mangroves provide is a clear benefit. Thus, the timing of benefits are often distant from decisions and threats. Placing valuations on cultural benefits poses another challenge because they are difficult to quantify. There are often mismatches between the temporal and spatial scales of ecosystem services and the scales that human institutions follow. All these issues diminish confidence in many of the valuations placed on ecosystem services, and make this a challenging part of the ecosystem services technique.

To accomplish ecosystem services valuation, there are many needs. Data should be available on the timing of service benefits, location of benefits, rates and flows of benefits, demand for human use, relevant government policies and incentives, and importance to human well-being. The economic framework used to determine values for ecosystem services must also meet these criteria. Values can be ascribed to or associated with a service. How can values be estimated under varying supply and demand conditions (i.e., marginal valuation)? Can the benefits be exchanged with other more confidently valued benefits? What is the cost of maintaining ecosystems to provide a set of services? Currently there are scientists and economists taking on the challenge of accurately assessing diverse ecosystem services valuations.

Valuing ecosystem services is not easily accomplished (Balmford et al. 2002), yet information on the economic estimates of benefits is seen as important for justifying protecting natural ecosystems (Cimon-Morin et al. 2013). Several conventional economic valuation tools are being used to estimate the monetary values of these services. Revealed-preference approaches are tools that can be used to estimate the amount paid for goods and services that directly shows their value (Lovett 2019). Travel costs indicate how much monetary value is incurred to enjoy services that are associated with an ecosystem. Hedonic methods are used to estimate economic values for ecosystem services that directly affect market prices. Hedonic methods are most commonly applied to variations in housing prices that reflect the value of local environmental attributes. Production costs indicate the value of services that can be used to increase output as in farming. Stated-preference approaches ask people about either their willingness to pay for an ecosystem service (contingent-valuation method), or their choice among scenarios with different services and costs (conjoint analysis) (Lovett 2019). Cost-based approaches can also be used to estimate the value of some services (Lovett 2019). The cost to replace a habitat that provided a needed service, such as a riparian buffer that provided water filtration from agricultural fields. Avoidance or insurance costs can be used to estimate the monetary value of services that, if removed or reduced, would increase exposure to harm, like coastal flooding when marshes are removed. These estimation methods have been used for natural resource services to estimate costs of damage to public goods, and for crafting policies that are environmentally efficient. They can also be applied to ecosystem services for estimating values that can be used to justify protecting natural ecosystems (Cimon-Morin et al. 2013).

Acceptance of the use of techniques for determining monetary values of ecosystem services has been slow (Gómez-Baggethun et al. 2010), but other values can be attributed to ecosystem services. The importance of ecosystem services to the public can be ascertained through focus groups, stakeholder engagement, rating and voting actions, and expert opinions (Christie et al. 2012). Results from these estimates of importance can be used to draw attention of decision-makers to ecosystem services. Monetary valuation can be helpful, but often this information is not heavily used in public policy debates (Fisher et al. 2008). Often leaders do not like to rely entirely on money in controversial public issues. Importance estimates can be used to support conservation policies that protect natural ecosystems much like monetary valuations can support natural ecosystems benefits (Christie et al. 2012).

PAYMENTS FOR ECOSYSTEM SERVICES

One application of valuing ecosystem services is to compensate people or communities for undertaking conservation actions that protect the flow or increase the provision of ecosystem services (e.g., water purification, flood mitigation, carbon sequestration; example in Figure 12.5) (Jack et al. 2008; Redford and Adams 2009; Farley and Costanza 2010; Hein et al. 2013). Payments could come in the form of lump sums for their efforts (e.g., planting a buffer strip), or a set rate for scalable actions (e.g., number of trees planted). These payments for ecosystem services are a practical way to incentivize the maintenance of ecosystem services (Nelson et al. 2010). Payments can address market deficiencies where there are non-marketed services that do not provide an economic benefit for maintaining these services. Local people that do not have options to promote ecosystem services are often attracted to payments as inducements to practice conservation. The payments can help promote protection of some ecosystem services, and payments can be tied to ecosystem service production levels.

Funds that are used to pay for ecosystem services in general depend on the demand for those services. Recipients of the services can be charged for access to the services, or their willingness to pay can monetarily support the charges. However, in general, donations and voluntary purchases have not generated funding close to the level at which the services are valued. Funds for payments can be raised by taxes, user fees, fees on development rights, and public subsidies (Farley and Costanza 2010). Also, tradable permits for development can include fees or mitigation arrangements that can pay for ecosystem services (Farley and Costanza 2010). Environmental groups and government agencies can arrange for development mitigation programs that generate funds or conservation actions for protecting the ecosystems and the services provided. These methods work by making developers pay to set aside land in one location in exchange for development rights elsewhere. Also, ecolabeling can reduce market friction by providing information about the origin of products. Demand-driven benefits of services require complex and variable methods of payment.

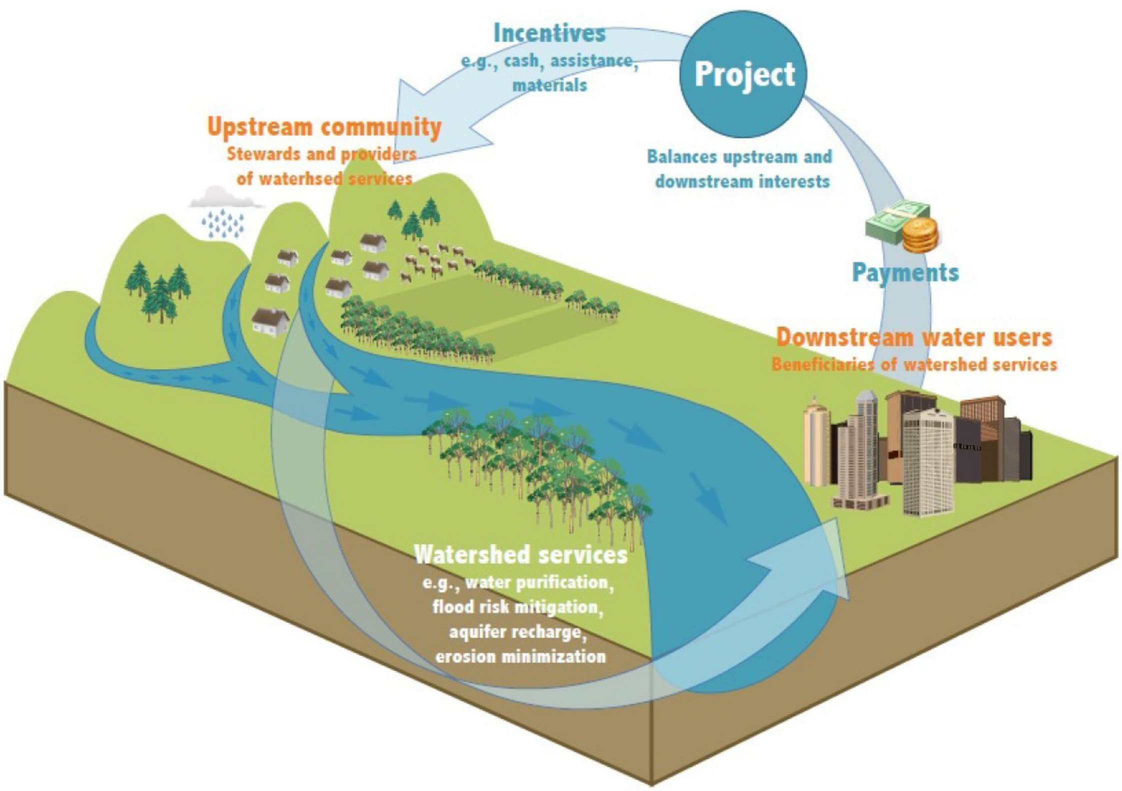

Figure 12.5: Representation of a payment for ecosystem services scheme in which downstream water users pay upstream land owners to provide watershed services. Source: Wagner et al. 2019

Incentive programs for promoting natural ecosystems and the services they provide can be complicated. Often secondary measures for service benefits are used because they are easier to estimate and track. For example, forested riparian zones are often counted as water purification systems in agricultural settings, but the actual service is clean water. Secondary measures need to be well understood and effective for estimating actual service benefits. When incentive programs address many ecosystem services, then an agency or managing organization must issue rules and criteria for payments. Also, ecosystem protection where many people live, like a community, adds complexity and calls for a central authority to arrange incentives. That leads to top-down control of ecosystem protection and diminishes community-based strategies. Surrogate measures, a multitude of services, and involvement of numerous people can lead to complicated incentive and conservation programs. These complications could stifle innovative methods and increase the cost of administration.

There are some conservationists opposed to the idea of merging economics and ecology to maintain natural ecosystems which provide services for nature and people (McCauley 2006; Sandbrook et al. 2013), yet the idea has received considerable interest in recent years (Salzman et al. 2018) . Paying for ecosystem services can be interpreted as payment for not damaging nature and curtailing bad behavior. They feel that landowners should be expected to support society and natural features without compensation. There are risks associated with paying for ecosystem services. Markets exist for some goods and services and they can command a large share of attention because they are easy to value. Easily valued services can outweigh other services, and lead to a diminished scope of ecosystem service benefits. Engineered ecosystems may be better at producing select valued goods and diverse services. Ecosystem services have been promoted on the notion that everyone comes out ahead, and little debate has been conducted about the consequences of the technique.

EXAMPLES OF PAYMENTS FOR ECOSYSTEM SERVICES PROGRAMS

The United States Department of Agriculture's Conservation Reserve Program provides payments to farmers to take highly erodible and environmentally sensitive land out of production and undertake resource conserving practices (e.g., planting permanent vegetation on environmentally sensitive cropland) for 10 or more years (Stubbs 2014; Conservation Reserve Program 2021). Even though this program was established in the mid-1980s prior to the concept of ecosystem services, the aim was to restore agricultural lands for production of a variety of ecosystem services. The program is large and has paid more than $1.8 billion to take 36 million acres out of agricultural production (Nelson et al. 2008). In addition, grassland signups are increasing (Conservation Reserve Program 2021). The Conservation Reserve Program promotes ecosystem service benefits like restoring natural habitats and carbon sequestration from restoring forests. Payments are important for getting private landowners in the Conservation Reserve Program, but the program is unclear on what ecosystem services are attained. Monitoring and evaluation of restored agricultural lands are needed to demonstrate the gains in specific ecosystem services.

MIXED SUCCESS AT IMPLEMENTING VALUATION OF ECOSYSTEM SERVICES

The World Bank has a lengthy record of designing development projects that are aimed at improving both economic and environmental conditions for people. Success on both goals is termed a "win-win" outcome. Tallis et al. (2008) reviewed 32 World Bank projects that had a goal of "win-win" between 1993 and 2007. Only five of 32 had clear gains in terms of environmental conservation and poverty alleviation, thus indicating a very low success rate (Figure 12.6). A full accounting of ecosystem services might improve evaluation of both human and ecosystem well being. However, there are complications in doing this. Most World Bank projects focused on one environmental benefit at a time, rather than a whole ecosystem service agenda. Economic returns respond quickly, but ecosystem changes may take many years before benefits are visible. Also, different ecosystem services respond on different spatial and temporal scales making a comprehensive accounting difficult. World Bank development projects that address conservation and human benefits could take into account the use of ecosystem services, tradeoffs among services, and economic returns from service markets. This strategy fits with the concept of integrating human and ecosystem processes for mutual benefit (Farber et al. 2006).

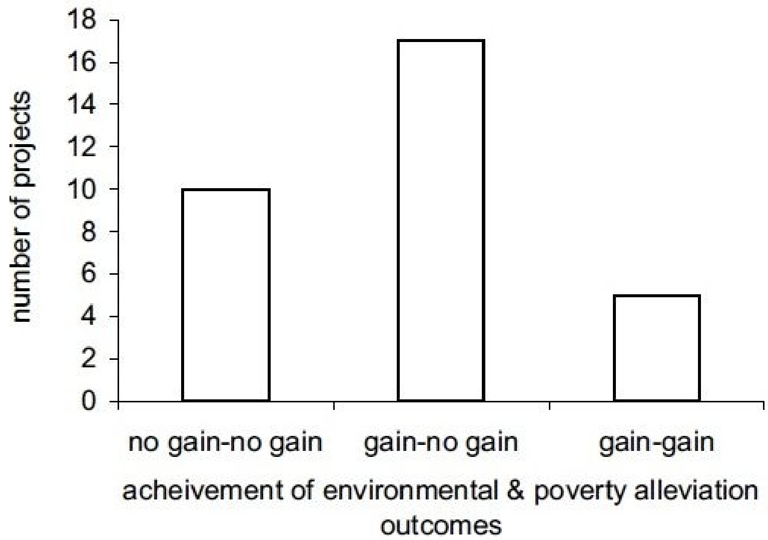

Figure 12.6: Only five of 32 projects (16%) had substantial gains in terms of their stated environmental and poverty alleviation outcomes. Source: Tallis et al. 2008

Costanza et al. (2017) published a paper titled: "Twenty years of ecosystem services: How far have we come and how far do we still need to go?" The authors reviewed the history leading up to two 1997 publications on ecosystem services, outlined subsequent debates, research, and institutions they triggered, summarized lessons learned during the twenty years since 1997, and provided recommendations for the future of research and practice. The authors concluded that "the substantial contributions of ecosystem services to the sustainable well-being of humans and the rest of nature should be at the core of the fundamental change needed in economic theory and practice if we are to achieve a societal transformation to a sustainable and desirable future" (Costanza et al. 2017).

CASE STUDY: PAYMENTS FOR ECOSYSTEM SERVICES TO RANCHERS IN CENTRAL AND SOUTHERN FLORIDA

Central and Southern Florida has been transformed from a landscape that was dominated by wetlands (Everglades) to an intensively developed region (Anderson and Rosendahl 1998). Most land not in parks or preserves has been converted to agriculture and urban or suburban development. With the demand for developable land, a massive engineering effort was built to drain land and move water to the coast and Lake Okeechobee. Water control structures were common, an extensive canal system was built, and then flood control structures were needed. Fast flowing water carries high nutrient concentrations into Lake Okeechobee and nearby coastal waters. Lake waters have doubled in phosphorus concentration since the 1970s (Bohlen et al. 2009). This led to eutrophication and increased algal blooms that degraded waters for aquatic life and recreational use. The lake drains southward into the Everglades and has changed in flood flows, low flows, and nutrient concentrations. The Comprehensive Everglades Restoration Plan (United States Department of the Interior 2021), developed in 2000, aimed to restore, protect, and preserve the water resources of Central and Southern Florida, including the Everglades. The projected cost was estimated at about $7.8 billion of public funds from the United States government and the State of Florida to enact 68 projects over 36 years (Carter and Sheikh 2003). Many of the canals and flood protection barriers were slated to be removed and modified to restore more natural water flows across the landscape. The restoration effort also included buying land to restore wetlands for treating land drainage to remove phosphorus, constructing reservoirs to retain water and slowly release it, and developing aquifer storage wells. In short, the aim was "getting the water right" at a great cost to the public (Clarke and Dalrymple 2003).

Large cattle ranches dominate the landscape north of Lake Okeechobee and their runoff drains rapidly to the lake (Flaig and Reddy 1995). These ranches have changed the land cover and disrupted the water regime with drainage canals. In 2005, Florida Ranchlands Environmental Services Project (FRESP) was established to develop a cost effective approach for ranch owners to produce ecosystem services that would retain water on their property and reduce nutrient concentrations (The Florida Ranchlands Environmental Services Project 2011). The project was initiated through a partnership between The World Wildlife Fund and a regional government agency (South Florida Water Management District) which jointly recognized that existing approaches to water quality management were not delivering desired water quality outcomes in Lake Okeechobee and downstream estuaries in Florida (Lynch and Shabman 2011). The vision of the FRESP was to attract ranch owners with service payment contracts to modify water management on their properties for storage and nutrient load reductions (Lynch and Shabman 2007; Bohlen et al. 2009). The buyer was the state agency and the sellers were ranchers who were willing to modify the structure and management of existing water control devices. Modifications allowed higher water retention on fields and wetlands, and prevented phosphorus runoff (Wainger and Shortle 2013). The program's administrative objectives were to be cost-effective for governments, profitable for ranch owners, provide needed ecosystem services, and feasible to administer. The FRESP included cattle ranchers, environmental organizations, academic scientists, and agencies of the United States government and the State of Florida. The potential environmental benefits were intended to contribute to efforts to restore major waterways in Central and Southern Florida, serve the interests of ranching businesses, and serve as a model for cost-effective provisioning of ecosystem services.

Ranchers who joined the FRESP worked to meet the needed services on their ranches. Drained wetlands were restored, and canal water was pumped into wetlands for natural nutrient reductions. Pastures were used to store water (Figure 12.7), and minor water retention structures were built to impound surface runoff. The FRESP measured ecosystem service performance to ensure payments were justified. Ranch lands were not taken out of production, and payments contributed to the financial stability of the ranch. One aim was to retain ranchlands in operation because other developments were often more environmentally damaging to waterway protection. Also, compatible water conservation practices on ranches were less costly to the public, and maintained agricultural production for economic benefits.

The FRESP was successful in its recruiting of participating ranch owners and subsequent implementation of water management practices (Cheatum et al. 2011; Meyer et al. 2016). Getting the interested people involved in FRESP was essential. Ranchers that were interested in Florida's environment were critical in exploring the program's benefits and practices. Agency leaders that were critical to the FRESP were the ones that departed from normal agency practices and expanded their methods. Also, scientists were important for designing evaluation methods for ranch practices and program documentation. However, some ranchers resisted involvement in the FRESP, because the program required additional practices above and beyond current practices for water and waste management. Some ranch owners wanted to concentrate on intensive production, which would interfere with water storage and elevated nutrient runoff. Finally, Florida was experiencing rapid population growth and some ranchers were interested in selling land for development. For success, the interested people had to be engaged, though not all parties went along with the FRESP.

Figure 12.7: Wet prairie pasture used to store water in South-Central Florida. Source: Bohlen et al. 2009

FRESP needs were diverse and it was a challenge to get it established (Wainger and Shortle 2013). Program leaders had to depart from normal practices and face policy and regulatory issues. Also, leaders were responsible for political support and initial startup financing. The payments had to be justified, and an evaluation system was needed to document ranch-generated environmental benefits. There were state and federal permit issues that needed to be resolved, and initial cost-sharing investments. Record keeping was novel for ecosystem service payments but required. Negotiating and executing contracts was new, and these often ranged from 5 to 20 years to accommodate wet and dry years for steady ranch payments. The processes for establishing prices for ecosystem services was new and demanded accountability. Finally, financing for the long-term needed to be secured.

The FRESP achieved two important goals. The program demonstrated that public investment can be cost effective for water retention and nutrient treatment on agriculturally productive ranches. The program also contributed to economic sustainability of cattle ranching in a region that was under intensive development and posed great threats to Florida's waterways. This program became a role model for other payments for ecosystem services programs in the United States (Shabman and Lynch 2013). The program demonstrated that this ecological conservation technique can be practical and effective when ecosystem services are truly needed.

SUMMARY

Common study topics related to ecosystem services are patterns of the response of services to change, distribution of service flows in space and time, conditions that promote stability of services, tradeoffs and synergies among services, and resilience of ecosystems when managed for some services (Carpenter et al. 2009). Valuation of ecosystem services requires collaboration among ecologists and economists and holistic thinking. Some notable conservation efforts and analyses have used the ecosystem services technique. The priority has been to identify a broad range of ecosystem services that benefit people, and to ascertain practical measures of service benefits. Payment for providing ecosystem services has been implemented to promote conservation, and provide direct benefits to local people who control ecosystems. The record on payment success has been mixed (Bussiere et al. 2015), but there is optimism that this strategy can work for conservation and people. Overall, ecosystem services as a conservation strategy has potential but, as expected, the challenge is working through impediments (Daily and Matson 2008).

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Holistic Techniques

Chapter 13 - Sustainability

The last topic in the holistic techniques group is sustainability. Sustainability is a popular concept for current management. The definition of sustainability varies but generally includes aspects of maintaining options for future generations, interaction between humans and the environment, and interdisciplinary collaboration to solve problems. In this chapter we define sustainability and provide examples of sustainable actions, present information on recent developments in the field, and examine successful applications of sustainable principles. We end with a case study of San Francisco, a city with a long history of sustainable practices.

DEFINITIONS AND OBJECTIVES OF SUSTAINABILITY

The concept of sustainability first appeared in the early 1970s and 1980s as a method for managing interactions between nature and society. Despite the intervening decades, sustainability has not yet become a rigorously defined term. Most definitions include human needs for survival and natural needs. The meaning of sustainability is variable in different contexts and to different people. The most established definition is the one that spans generations and time. By this definition, sustainability involves development that meets the needs of the present without compromising the ability of future generations to meet their own needs (World Commission on Environment and Development 1987). However, there are many that hold to a perspective more focused on preserving nature. This view results in definitions that emphasize meeting fundamental human needs while preserving the life support systems of the planet (Kates et al. 2001). Some definitions include the equitable distribution of resources between present and future generations of all human beings (Weiss 1990), and the use of these resources in a way that will not jeopardize the continued persistence of the planet's biodiversity and ecosystems (Chapin et al. 2010). Another dimension of sustainability is that it applies to human needs, and also results in a balance of nature and society (Castree 2017). A final definition for sustainability is one that espouses peace, freedom, better living conditions and a healthy environment (National Research Council 1999).

There are few common concepts across all definitions of sustainability. Definitions include aspects of an intergenerational nature (transference from one generation to another); level of scale (multiple scales are involved); domain (multiple domains participate including at the least economic, ecological, and socio-cultural); and interpretation (there are a multitude of interpretations of the meaning of sustainability) (Martens 2006). At its core, sustainability is a fundamentally holistic technique (Hani et al. 2003).

The sustainability concept is not new to resource harvesting, especially fisheries and forestry (Clapp 1998). From the early 1900s to the present, theoretical and empirical studies have been undertaken to identify maximum sustained yields (e.g., catches or harvest) for valuable resources that could be sustained at some level of effort (Nielsen 1976; Larkin 1977; Luckert and Williamson 2005). Yields were solely based on the biological properties of species and population processes, and were not subject to societal interactions or political influences. In practice, pure sustainable management rarely occurred and most fisheries and forest resources were commonly overexploited (Ludwig et al. 1993) due to societal, economic and industry pressures. In the 1970s, professional organizations of fishery managers adopted "optimum sustained yield" as the paradigm for management (Bennett et al. 1978). This philosophy recognized the role of non-biological factors in management decisions and became the start of sustainability as a union of economic, social, and public interest factors; a balance of forces linking the biotic resource and human needs. This was the start of sustainability becoming a union of economic, environmental, and social aspects. This concept has been termed the Three Pillars of Sustainability (Hansmann et al. 2012). Unfortunately, overexploitation continues to occur in many fisheries (Ye and Gutierrez 2017) and forested areas (Islam and Bhuiyan 2018).

WHAT DOES IT MEAN TO BE SUSTAINABLE?

Sustainability focuses on natural features and human needs. The natural features include life support systems and biodiversity. Human needs target people, the economy, and community. Reducing the impact on the environment at the same time as increasing food, income, and health is a fundamental challenge (Table 13.1).

Table 13.1: What is to be sustained and what would be increased. Source: adapted from Parris and Kates 2003

Sustained	Increased
Nature Earth Biodiversity Ecosystems	People Child survival Life expectancy Education Equity Equal opportunity Health Poverty income Food availability
Life support Ecosystem services Resources Environment Climate	Economy Wealth Productive sectors Communications Energy availability
Community Cultures Groups Places	Society Institutions Social capital Regions Scientific information

There are many aspects to enacting sustainability. At the individual level, enacting sustainable practices can mean driving less, eating more locally-produced and more plant-based foods, setting the thermostat higher in the summer and lower in the winter, avoiding single use plasticware, reducing waste and more. At the corporate level it could mean switching to using recycled paper, striving for net-zero emissions from buildings, installing solar panels on rooftops, encouraging worker well-being, and more. At the regional, state and national scale, it can mean establishing policies and regulations to reduce pollution, encourage conservation, and shift the way people think about all aspects of life in a way that fully encompasses sustainable principles.

SUSTAINABILITY SCIENCE

Sustainability science has emerged as a distinct research program (Clark and Dickson 2003; Clark 2007; Barrett 2021). The aim of the program is to advance our understanding of the interactions between society and nature to manage the transition to an increased use of sustainable principles for managing the earth’s resources. The incorporation of sustainable principles in resource management is a big change for most societies around the world and, as such, pulls together a diverse array of disciplines (Aronson 2011). Scientists promoting sustainability have needed to engage in research ranging from complex systems theory to cultural and political ecology. Combining these different theoretical approaches is a big challenge because it requires scientists to get into policy and engage decision-makers and the public. The research itself must be focused on the character of nature-society interactions, on our ability to guide those interactions along sustainable trajectories, and on ways of promoting the social learning that will be necessary to navigate the transition to sustainable practices (Kates et al. 2001). Another dimension of sustainability science is to solve problems at the societal-ecological interface. Sustainability science is problem-driven and interdisciplinary oriented. The hope is that stakeholders with diverse experiences will discuss key questions, appropriate methodologies, and institutional needs, and that outcomes from these discussions will provide applications that lessen the human impact on the natural world and simultaneously support human needs (Kates et al. 2001). This is more in the realm of traditional science because it is narrower and problem-oriented. Sustainability science needs to be connected to a political agenda to engage national and state leaders as a priority issue. If applied, all of these aspects will help to manage nature-society interactions to successfully transition to a greater use of sustainable principles.

BEST PRACTICES FOR TRANSITIONING TO MORE SUSTAINABLE PRACTICES

Sustainability requires an enthusiastic agenda that brings together academics, agencies, and institutions that can take action, consider global and local perspectives, and derive information from the environment, society, and the engineering and health care sectors (Figure 13.1). Cash et al. (2003) performed historical analyses of environmental issues, from initial scientific discov-

Figure 13.1: Sustainability word cloud. Source: Town of Maynard, MA 2021

ery to high-level policy agenda. They used scientific input to assess how these issues were defined and framed, which options were considered, and what actions were taken. They discovered that for big policy ideas (e.g., green revolution, aquifer depletion on the central United States, El Niño forecasting, ocean fisheries, and transboundary air pollution), it takes a decade or more to reliably evaluate the impact of science on policy (Cash et al. 2003). The impact of scientific information on policy and public action depends heavily on the perceptions of stakeholders and involves three key factors: salience, credibility, and legitimacy (Cash et al. 2003; Cash and Belloy 2020). Sustainability must be relevant to the people involved (e.g., salience). The arguments for focusing on sustainability must be supported by technical evidence (e.g., credibility). The discourse must be respectful, unbiased, and fair to divergent values and beliefs (e.g., legitimacy). The public, concerned about transitioning to a program that includes sustainability, must be convinced that without such a transition, they might lose valuable materials and experiences. Scientists and leaders must make the argument that they are trying to avoid future problems.

The act of mobilizing science for sustainability requires that the boundaries between knowledge and action be managed for salience, credibility, and legitimacy of the information produced. This is often termed "boundary work," - work that is carried out at the interface between communities of experts and communities of decision-makers (Cash et al. 2003). The three functions that contribute most to boundary management are: communication, translation, and mediation. Communication requires active, iterative, and inclusive exchanges between experts and decision-makers. Communications experts can translate among scientists, decision-makers, and the public to overcome impediments. Translation involves linking knowledge to action and requires that participants understand each other. Mutual understanding between experts and decision-makers is often hindered by jargon, language, past experiences, divergent values, and presumptions about what constitutes a persuasive argument. Active mediation of conflicts makes the boundary between experts and decision-makers selectively porous (i.e., open to certain purposes but closed to others; for example, getting data to researchers but keeping politics out of the scientific process). The boundary-management functions summarized above (communication, translation, and mediation) can be performed effectively through various organizational arrangements and procedures. These functions can be institutionalized in boundary organizations mandated to act as intermediaries between the arenas of science and policy for the purposes of: 1) Organization for managing the boundary; 2) Responsibility and accountability to social arenas on opposite sides of the boundary; and 3) Provision of a forum in which information can be co-produced by actors from different sides of the boundary. Those groups that made a serious commitment to managing boundaries between expertise and decision-making effectively linked knowledge to action (Cash et al. 2003). Such groups invested in communication, translation, and mediation, and thereby more effectively balanced salience, credibility, and legitimacy in the information they produced.

CHALLENGES

Cash et al. (2003) also identified a number of challenges, particularly in the way different stakeholders viewed the process of moving toward more sustainable action. Mobilizing science toward sustainability requires performing tasks not conventionally associated with research, leading many scientists, not surprisingly, to see participating in knowledge systems for sustainability as at best uncomfortable and at worst inconsistent with real scholarship. Reciprocally, many managers and decision-makers view the process as at best an expensive time investment with uncertain returns and at worst a risk to their perceived autonomy and independence. The focus on multiple, interacting perturbations and stressors, attention to coupled human–environment systems, and place-based analysis in the context of large scale change demanded a recasting of the interactions between scholar and practitioner. Effective processes were characterized by multiple boundary organizations, or multiple organizations that performed specific functions in managing the boundaries of complex systems. Often, single individuals played key boundary-spanning functions, independent of their particular organizational affiliations, thus there was a need to harness the boundary spanning potential of individuals and organizations. The new ideas for projects being called for in many sustainability discussions needed to be viewed as truly radical; these were not just individual studies or projects, but ideas to shift whole professional careers.

SUSTAINABILITY NEEDS TO BE PRACTICED WORLDWIDE

Unsustainable activities are degrading the planet's ability to support humans (Chapin et al. 2011). The switch to sustainability could secure the Earth into the future. Nearly a quarter of the world's human population is living in poverty (Human Development Initiative 2018), and sustainability should address this problem. Additionally, about 10% of people worldwide lack a secure connection to electricity (International Energy Agency 2019). Globally, energy generation produces a lot of emissions which change our climate (Davis et al. 2010). Hunger and malnutrition are widespread and food production should be addressed to feed the world's human population. We need to rethink food production, and move away from our heavy reliance on oil and fertilizers to make agriculture more sustainable (McKenzie and Williams 2015). Global governance is necessary, one which embraces a platform of trust between regions and nations.

GLOBAL AGENDA FOR SUSTAINABLE DEVELOPMENT

To address global challenges, all United Nations member states (193 countries) committed in 2015 to make progress toward achieving seventeen United Nations Sustainable Development Goals (SDGs) (Figure 13.2). They created a document called the 2030 Agenda for Sustainable Development, which provides a shared blueprint for peace and prosperity for both people and the planet, now and into the future (United Nations 2021a). At the core of the agenda are the seventeen SDGs which provide a call for action by all countries - developed and developing - in a global partnership. The SDGs recognize that ending poverty and other deprivations must go hand-in-hand with strategies that improve health and education, reduce inequality, and spur economic growth – all while tackling climate change and working to preserve our oceans and forests (United Nations 2021b).

The SDGs themselves are the following (Figure 13.2): 1) No poverty; 2) Zero hunger; 3) Good health and well-being; 4) Quality education; 5) Gender equality; 6) Clean water and sanitation; 7) Affordable and clean energy; 8) Decent work and economic growth; 9) Industry, innovation, and infrastructure; 10) Reduced inequalities; 11) Sustainable cities and communities; 12) Responsible consumption and production; 13) Climate action; 14) Life below water; 15) Life on land; 16) Peace, justice and strong institutions; 17) Partnerships for the goals (United Nations 2021b).

Many of these goals interact with other goals to various extents (Figure 13.3). For instance, clean water and sanitation (goal 6) strongly relates to responsible consumption and production (goal 12) and benefits can be harnessed for both goals (e.g., co-benefits) by addressing these issues. Conversely, climate action (goal 13) and life below water (goal 14) have significant tradeoffs for one in tackling the other, and these will need to be addressed going forward.

Figure 13.2: The seventeen Sustainable Development Goals (SDGs) developed by the United Nations. Source: United Nations 2021b

Each SDG has a set of targets and related indicators with differing numbers of targets (from 5-19) for each goal depending on its complexity. Targets are the concrete actions that each SDG is striving to achieve. For example, let's examine the SDG Climate Action: take urgent action to combat climate change and its impacts. Its five targets are: 1) Strengthen resilience and adaptive capacity to climate-related hazards and natural disasters in all countries; 2) Integrate climate change measures into national policies, strategies and planning; 3) Improve education, awareness-raising and human and institutional capacity on climate change mitigation, adaptation, impact reduction and early warning; 4) Implement the commitment, undertaken by developed-country parties to the United Nations Framework Convention on Climate Change, to a goal of jointly mobilizing $100 billion annually by 2020 from all sources to address the needs of developing countries in the context of meaningful mitigation actions and transparency of implementation, and fully operationalize the Green Climate Fund through its capitalization as soon as possible; and 5) Promote mechanisms to increase the capacity for effective climate change-related planning and management in least developed countries and small island developing States, including focusing on women, youth and local and marginalized communities.

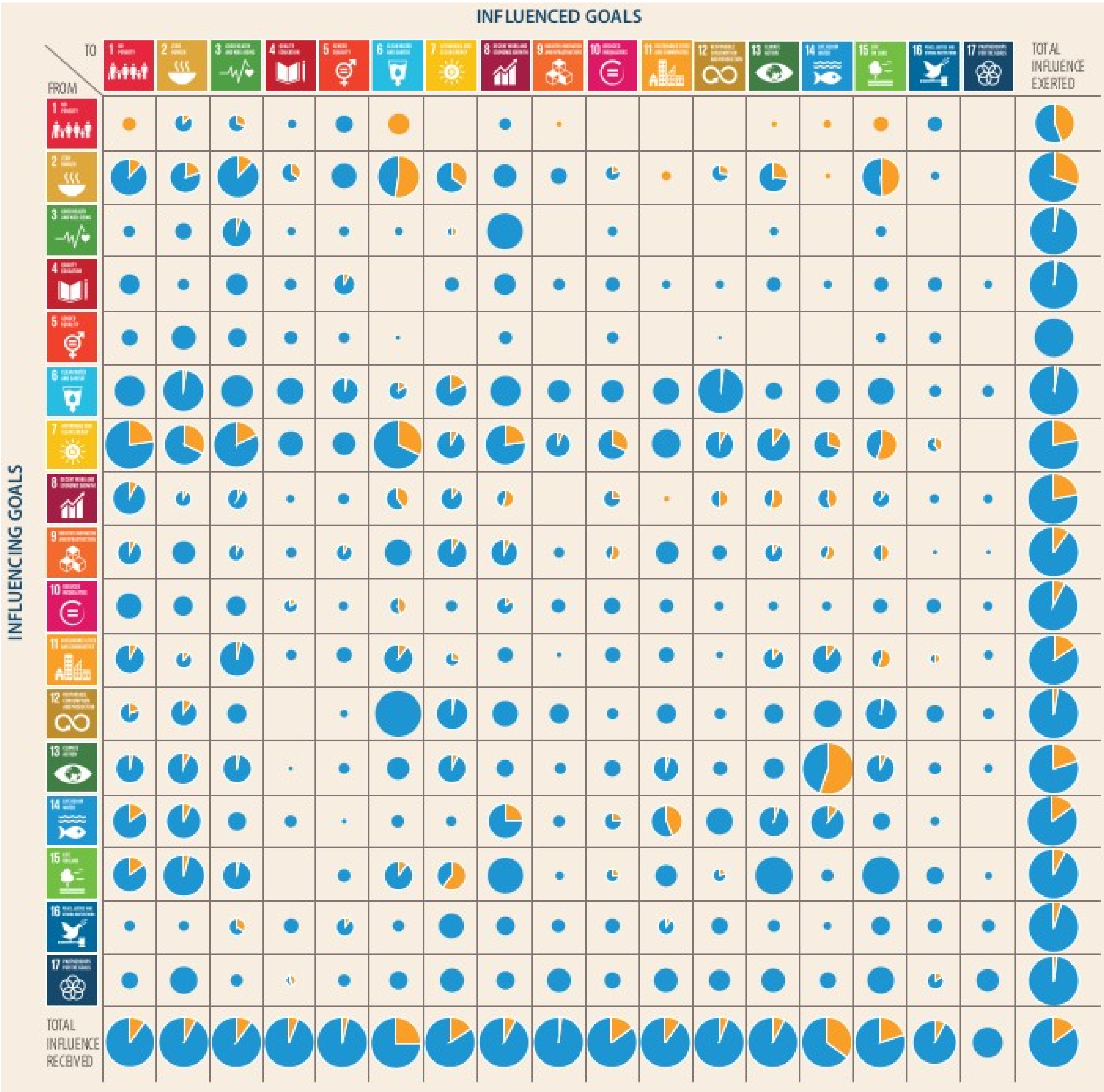

Figure 13.3: Interactions among Sustainable Development Goals (SDGs). Source: Messerli et al. 2019

A single target can have multiple indicators. Indicators are the metrics used to determine if a target was met. The indicators relating to each of the Climate Action targets, respectively, are: 1) Number of deaths, missing persons and persons affected by disaster per 100,000 people; number of countries with national and local-disaster risk-reduction strategies; and proportion of local governments that adopt and implement local-disaster risk-reduction strategies in line with national-disaster risk-reduction strategies; 2) Number of countries that have communicated the establishment or operationalization of an integrated policy/strategy/plan which increases their ability to adapt to the adverse impacts of climate change, and foster climate resilience and low greenhouse gas emissions development in a manner that does not threaten food production; 3) Number of countries that have integrated mitigation, adaptation, impact reduction, and early warning programs into their primary, secondary and tertiary curricula; and number of countries that have communicated the strengthening of institutional, systemic and individual capacity-building to implement adaptation, mitigation and technology transfer, and development actions; 4) Amount of United States dollars mobilized per year, starting in 2020 accountable towards the $100 billion commitment; and 5) Number of least-developed countries and small island developing States that are receiving specialized support, and the amount of that support, including finance, technology and capacity-building, for mechanisms that raise capacities for effective climate change-related planning and management, including focusing on women, youth and local and marginalized communities.

Additionally, the United Nations tracks events, publications, news, and actions related to each SDG. The United Nations also bridges a variety of needs in reaching SDGs by supporting policy analysis; capacity development; inter-agency coordination; stakeholder engagement, partnerships, communication, and outreach; and knowledge management (United Nations 2021c).

ASSESSING PROGRESS TOWARD SUSTAINABLE DEVELOPMENT GOALS

Global progress toward SDG fulfillment is monitored by 231 unique socio-ecological indicators spread across 169 targets. The United Nations Global Sustainable Development Report 2019—The Future is Now: Science for Achieving Sustainable Development (Messerli et al. 2019) concluded that, despite initial efforts, the world is not yet on track for achieving most of the SDG targets (Figure 13.4).

Good health and well-being (Goal 3) and Quality education (Goal 4) are closest to meeting some of their targets. No poverty (Goal 1), Zero hunger (Goal 2), Quality education (Goal 4), Clean water and sanitation (Goal 6), Affordable and clean energy (Goal 7), and Industry, innovation and infrastructure (Goal 9) are within 5-10% of meeting some of their targets. The majority of the SDGs are greater than 10% from meeting their targets. Disturbingly, some aspects of Zero hunger (Goal 2), Reduced inequalities (Goal 10), Responsible consumption and production (Goal 12), Climate action (Goal 13), Life under water (Goal 14), and Life on land (Goal 15) have been trending away from their targets (Messerli et al. 2019). In 1999, the National Research Council stated that it will take two generations to adopt a serious sustainability need (National Research Council 1999) and that seems to be playing out over perhaps an even longer time scale (Tibbs 2011).

Some positive news is that cross-national flows of information, goods, capital and people have all increased dramatically in the last few decades, underpinning a world that is more interconnected than ever (Figure 13.5). These flows overlap and interconnect and link the development of nations and regions across North and South, global and local, current and future. The flows produce many benefits. For example, through remittances, finances are transferred from richer parts of the world to poorer ones, and use of the Internet can give small entrepreneurs and artisans access to the global marketplace.

Conversely, the flows can also propagate negative impacts, such as deepening inequalities, unfair competition, resource depletion and environmental pollution and destruction (Messerli et al. 2019).

GOAL	WITHIN 5%	5–10%	>10%	NEGATIVE LONG-TERM TREND
Goal 1		1.1. Eradicating extreme poverty	1.3. Social protection for all	
Goal 2		2.1. Ending hunger (undernourishment)	2.2. Ending malnutrition (stunting) 2.5. Maintaining genetic diversity 2.a. Investment in agriculture*	2.2. Ending malnutrition (overweight)
Goal 3	3.2. Under-5 mortality 3.2. Neonatal mortality		3.1. Maternal mortality 3.4. Premature deaths from non-communicable diseases	
Goal 4	4.1 Enrolment in primary education	4.6 Literacy among youth and adults	4.2. Early childhood development 4.1 Enrolment in secondary education 4.3 Enrolment in tertiary education	
Goal 5			5.5. Women political participation	
Goal 6		6.2. Access to safe sanitation (open defecation practices)	6.1. Access to safely managed drinking water 6.2. Access to safely managed sanitation services	
Goal 7		7.1. Access to electricity	7.2. Share of renewable energy* 7.3. Energy intensity	
Goal 8			8.7. Use of child labour	
Goal 9		9.5. Enhancing scientific research (R&D expenditure)	9.5. Enhancing scientific research (number of researchers)	
Goal 10			10.c. Remittance costs	Inequality in income*
Goal 11			11.1. Urban population living in slums*	
Goal 12				12.2. Absolute material footprint, and DMC*
Goal 13				Global GHG emissions relative to Paris targets*
Goal 14				14.1. Continued deterioration of coastal waters* 14.4. Overfishing*
Goal 15				15.5. Biodiversity loss* 15.7. Wildlife poaching and trafficking*
Goal 16			16.9 Universal birth registration **	

Figure 13.4: Projected distance from reaching selected targets (at current trends). Source: Messerli et al. 2019

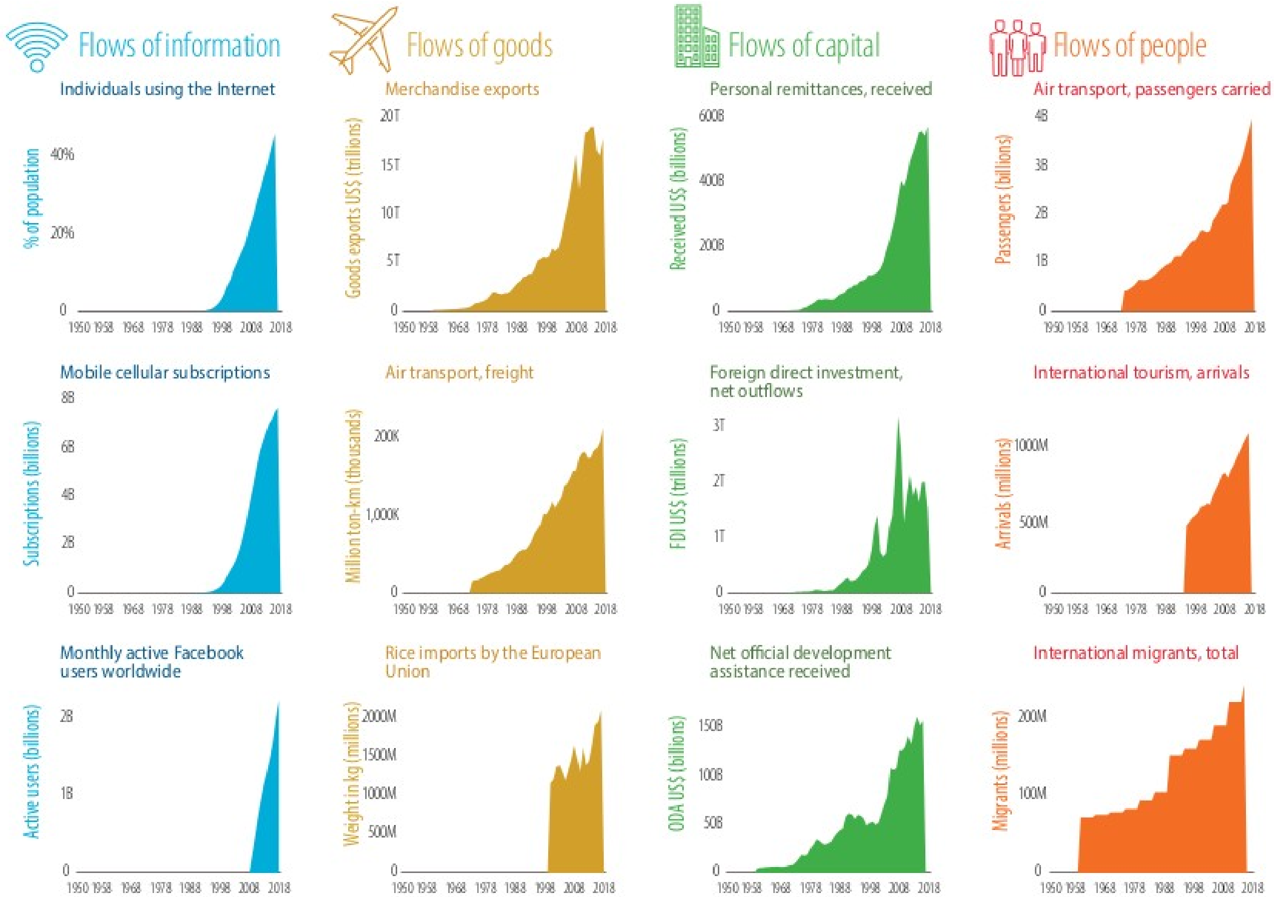

Figure 13.5: Cross-national flows of information, goods, capital and people. Source: Messerli et al. 2019

CASE STUDY: SAN FRANCISCO - SUSTAINABLE CITY

San Francisco adopted a plan to become a sustainable city in 1997. The plan, which later became a City document, was drafted by a community collaboration in which City staff contributed on equal footing with members of other sectors of the community including representatives from the City Planning Department, the Bureau of Energy Conservation, the Recreation and Parks Department, and the Solid Waste Management Program, businesses, environmental organizations, elected officials, and concerned individuals. In all, nearly 400 people worked on the plan. The plan was aimed at changing long-standing environmental practices and consisted of goals, actions, and objectives to be achieved. The aim of the plan was to “begin to fulfill our responsibility to our own futures and that of our children” (Sustainable City 2021).

Although there was remarkable unanimity among the plan drafters about the basic attributes of a sustainable society, as would be expected in any exercise of this size and scope, participants did not always agree on the best strategy for achieving goals. Some felt strongly that the plan did not go far enough and contained too many compromises; others felt that it had gone too far and was unrealistic.

Nonetheless, the document provided the rough game-plan that was necessary for a concerted effort to achieve a sustainable society, an effort that had been orchestrated by as broad a cross-section of the community as possible.

Sustainability can be divided into manageable sections with specific strategies proposed for action. Topics addressed in the plan were divided into two main categories: 1) Specific environmental topics and 2) Topics that span many issues. Specific environmental topics included: air quality, biodiversity, climate change, energy, food and agriculture, hazardous materials, human health, ozone depletion, parks and open spaces, solid waste, transportation, and waste and wastewater (Sustainable City 2021). Topics that spanned many issues included: the economy and economic development, environmental justice, municipal expenditures, public information and education, and risk management (Sustainable City 2021).

Each topic had specific goals associated with the issue. We will describe the plans for two of these topics here: biodiversity, and water and wastewater. San Francisco is a heavily urbanized city, which nonetheless has a rich variety of plant and animal communities. Thus, the strategy to increase biodiversity had five goals: 1) To achieve a greater understanding of biodiversity, its importance, how it is threatened, and how to protect and restore it; 2) To protect and restore remnant natural ecosystems; 3) To protect sensitive species and their habitats and support their recovery in San Francisco through reintroductions of extirpated species and habitat management; 4) To maximize habitat value in developed and naturalistic areas, both public and private; and 5) To collect, organize, develop and utilize current and historic information on habitats and biodiversity. The following indicators were used to assess progress toward biodiversity goals: 1) Number of volunteer hours dedicated towards managing, monitoring, and conserving San Francisco's biodiversity; 2) Number of square feet of the worst invasive species removed from natural areas; 3) Number of surviving indigenous native plant species planted in developed parks, private landscapes and natural areas; and 4) Abundance and species diversity of birds, as indicated by the Golden Gate Audubon Society's Christmas Bird Counts.

A water policy that creates sustainable water use balances the needs for protection of the environment and public health, while not compromising the ability of future generations of San Franciscans to procure water to meet their basic needs. A sustainable water policy also creates a shift from the traditional view of water as a commodity managed solely for the convenience of humans to a more balanced effort to maintain the water needs of the entire ecosystem, of which humans are only a part. San Francisco is fortunate in having a source of high quality drinking water which comes from the headwaters of the Tuolumne River in the Sierra Nevada Mountains. The Tuolumne River is captured behind O'Shaughnessy Dam and diverted to San Francisco via the Hetch Hetchy system. The strategy to increase water and wastewater sustainability had fifteen goals: 1) To maximize recovery and reuse of resources from wastewater; 2) To maximize water conservation and minimize water use and waste; 3) To minimize storm water flows into the combined sewer system; 4) To eliminate contaminants in supply and receiving waters; 5) To discharge only wastewater that does not impair receiving water; 6) To ensure a sustainable and adequate water supply; 7) To maximize protection of public health by providing safe drinking water and the safe handling of wastewater; 8) To ensure fair and effective permit and enforcement procedures; 9) To create a water and wastewater policy that reflects true environmental costs and benefits; 10) To restore and enhance ground-water supply; 11) To achieve long-term enhancement and restoration of local marine and fresh-water habitats; 12) To create an inclusive community of environmental stewards; 13) To repair, replace and upgrade infrastructure; 14) To include alternative water, wastewater and storm water policies; and 15) To create drinking water and wastewater standards that protect local and regional natural resources and public health. The following indicators were used to assess progress toward water and wastewater goals: 1) Per capita water consumption measured by the San Francisco Water Department; 2) Mass of pollutants in wastewater; 3) Mass and frequency of combined sewer overflows; 4) Recycled water use; and 5) Acres of habitat restored.

To begin to achieve these goals San Francisco created a new Department of the Environment, held meetings for public comment, sought and gained endorsement of the plan by City leaders, and began the long process of creating a healthy society that respects the needs of all its members, and the needs of the natural systems of which they are a part. They created various projects to plant trees on public schoolyards, ban plastic bags (in 2007), shift public transportation in the city toward zero emissions, and encourage residents to conserve water (Djoulakian 2015). On several of these issues, San Francisco was the first city in the nation to enact such projects or policies.

How well did San Francisco do? According to a 2011 Siemens/Economist Intelligence Unit study released at the Aspen Institute in Munich, San Francisco is North America's greenest city, beating out other sustainable cities such as Vancouver, New York, and Seattle (Roggenbuck 2011). San Francisco took one of the top five spots for the categories of energy use, water quality, and air quality; second place for building standards and transportation; and ***first place for waste management*** (Figure 13.6) (Roggenbuck 2011). However, biodiversity was not among the categories for which San Francisco was highly ranked.

Figure 13.6: San Francisco's iconic three-stream waste collection program. Source: United States Environmental Protection Agency 2021

What factors account for this success? First, political will and supportive voters were needed to pass sustainable legislation, and San Francisco had both. Voters passed, by wide margins, measures such as the 2001 Proposition H, which set the stage for community choice aggregation (Hess 2005), and the 2003 Proposition K, which continued a sales tax to fund socially and environmentally motivated transportation projects (County of San Francisco 2011). Additionally, San Francisco became the first United States city to mandate solar and living roofs on most new construction (Sustainable City 2018). Second, San Francisco's experience with alternative energy helped it become a leader in solar energy use, and the city has completed a number of successful solar projects (Figure 13.7). Finally, the city has strong environmental planning due in part to its robust sustainability plan (Diamond 2011). San Francisco continues to work toward its goals and increasingly become more sustainable each year.

Figure 13.7: Solar panel installation in San Francisco, CA. Source: Bay Area Rapid Transit 2021

SUMMARY

The transition toward the inclusion of sustainability principles in ecological conservation is a challenge and faces many hurdles. Efforts have to address multiple scales, interests, and shortcomings to eliminate impacts to natural environments and maximize human benefits. The transition to the incorporation of holistic, sustainable principles in managing the earth's resources is anticipated to take many decades because different ways of thinking have to be adopted across the world. However, the United Nations SDGs are inspiring, and progress is being made every day toward the achievement of these targets.

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