Monitoring Ecosystems

Interdisciplinary Approaches for Evaluating Ecoregional Initiatives

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CHAPTER 13

Setting and Monitoring Restoration Goals in the Absence of Historical Data: The Case of Fishes in the Florida Everglades

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In designing and conducting ecosystem monitoring, one must ask to what end are we monitoring? The usual answer, to document the status of trends of biological or physical indicators toward some restoration goal, presupposes that well-defined goals have been established. However, setting goals and assessing success of ecological restoration are among the greatest challenges of the restoration enterprise (Yount and Neimi 1990). By definition, restoration entails regaining some aspects of the ecological system that have been lost (Gore 1985; Bradshaw 1987). However, quantitative biological data on historical ecosystems are often limited in quantity or quality, or are completely absent (Cairns et al. 1977). Compounding the challenge, ecological restoration is at times controversial, and goals and assessments may come under intense scrutiny, even taking center stage in court proceedings (e.g., Rizzardi 2001). Thus, a substantial burden may rest on ecologists to justify assumptions and methods used in establishing and monitoring restoration targets.

In many ecological management and restoration cases, recapturing historical or "natural system" conditions for an entire ecosystem is neither practical nor desirable. In the Florida Everglades, for example, endangered species management and natural system management at times appear to be in conflict (e.g., management for snail kites versus wood storks [Banner 1990]). Targeting portions of a regional ecosystem for restoration of one or more endangered species or some ecosystem function, while other portions are identified for "natural-system" management, may be the most common circumstance facing managers.

Ideally, before beginning a restoration action, historical ecosystem conditions should be delineated by data collected prior to human intervention in the ecosystem. Such data are rare, and generally occur in a narrative rather than quantitative framework (Egan and Howell 2001). Use of contemporary reference areas for comparison begs the question of whether those areas are themselves unaltered. Human alterations of ecosystems are pervasive and often act over large spatial scales affecting habitats not directly manipulated. For example, extirpation of large predators and herbivores by humans has probably dramatically altered most, if not all, natural ecosystems (e.g., Jackson 2001). Such effects may extend back into ancient times when human activities such as burning grasslands or overharvesting animals undoubtedly altered ecosystems dramatically (Jackson et al. 2001). Thus, historical conditions inferred from techniques of paleoecology may still reflect the effects of early human cultures on ecosystem composition and processes. Identification of targets must be explicit with regard to what time period of ecosystem history is to be restored.

Scientists are forced to recreate historical conditions by analogy through description of reference areas or by analytical tools such as simulation models because little or no historical data are typically available in planning restoration (Clark et al. 2001). Reference or background sites are a preferred approach in restoration efforts, but true historical references seldom exist. More critical, it is generally questionable that nearby regional ecosystems are truly comparable, even if relatively pristine sites exist (see Hargrove and Pickering 1992). As Marjory Stoneman Douglas wrote in her book *River of Grass*, "[t]here is but one Everglades." To which we might add, "Or Colorado River or Konza Prairie, or Olympic Peninsula." Thus, the premise of our chapter is that regional ecosystem initiatives must depend heavily on developing a predictive ecological understanding (or perhaps "post-dictive") of the ecosystem being restored to set restoration targets. Most importantly, a critical part of developing this understanding generally includes data gathered by retrospective monitoring (Noon, chap. 2). The process of moving from retrospective monitoring to prospective monitoring should include development of assessment tools to recreate "natural system" conditions that can serve as targets. We will illustrate this premise through our monitoring effort of fishes inhabiting the Florida Everglades.

Performance Measures: Targets for Monitoring

Assessing the progress of ecosystem restoration requires measurement of ecological parameters (physical or biological) that both are indicative of contemporary ecosystem conditions and trends, and can be projected to the idealized end points of the restoration as targets. A diversity of these so-called "performance measures" must be identified to allow assessment of ecosystem function or to measure progress toward specific targets, possibly for societal goals. For example, the number of nesting pairs of wading birds in the Florida Everglades serves both goals: as top carnivores they are indicative of food-web integrity *and* their presence and status is of great aesthetic interest to the public (Frederick and Ogden, chap. 12). Also, performance measures should be selected to reflect ecosystem responses across a range of time scales because important ecosystem processes may operate with multiyear time lags. For example, large predatory fishes require several years to reach sexual maturity and may induce important regulatory effects on communities of their prey. The exclusive use of ecosystem metrics that respond rapidly, such as periphyton or bacterial communities, would not indicate whether top trophic levels have been restored in an aquatic community. Finally, to be of greatest utility, it must be possible to make defensible projections of values for each performance measure in a restored ecosystem. Attainment of the projected values for monitoring indicators is the target for judging restoration success.

Several approaches can be employed to validate performance measures for a natural-system scenario. When time and monetary resources are limited, managers may be forced to rely on the opinion of experts deemed most familiar with the ecosystem—opinions usually termed "best professional judgment." At its best, this approach is an informal form of more-sophisticated model building; however, it suffers from obvious limitations of subjectivity and repeatability. Most significant for managers, the prospects of this approach withstanding an aggressive legal challenge are dubious. At the other extreme, application of quantitative historical data could provide the strongest basis for choice of restoration targets. Unfortunately, historical data are often lacking, and historical data collected by past generations of scientists often prove difficult to apply because of changing techniques and expectations of data-quality control (e.g., wading birds: Frohring et al. 1988, Ogden 1994; fishes: Miller 1990, Trexler 1995).

A third approach for setting restoration targets is the use of reference or background sites as benchmarks for desirable ecosystem management. This approach uses existing areas for identification, analysis, and documentation of performance measures and their targets. There are limitations to this approach, in addition to concerns discussed earlier that no "pristine" habitats remain to provide baseline conditions, particularly with removal of large grazers or carnivores, or deposition of airborne toxics, such as mercury (Zillioux et al. 1993). Karr and Chu (1999) make a compelling case for the use of baseline knowledge of geophysical setting and undisturbed biological condition to identify metrics for monitoring. A presumption of this view is that extensive baseline knowledge has already been established to permit the development of relatively generic measures of ecosystem health or integrity. Unfortunately, many regional ecosystem initiatives are in unique habitats that lack such detailed knowledge and may include all remaining examples of the endangered habitat, all of which may have experienced some level of human disturbance. Thus, setting targets for indices based on modern conditions in habitats perceived as pristine may undervalue the resources of the historical ecosystem (Jackson et al. 2001)—Daniel Pauly's (1995) "shifting baseline syndrome."

Ideally, development of simulation models for exploration of management scenarios (Fitz et al. 1996; DeAngelis et al. 1998; Curnutt et al. 2000; Sklar et al. 2001) brings together all three previous approaches and makes explicit the assumptions and data leading to restoration targets (DeAngelis et al., chap. 6). In doing so, simulation models formalize best professional judgment and extrapolate historical and contemporary knowledge of ecosystem state and function to a common framework. Models can also explore the implications of hypothesized species extinctions and anthropogenic effects through scenario building. Of course, validation of those scenarios may be possible only indirectly, but, short of time travel, they provide a powerful tool to peer into historical ecosystems. In the remainder of this chapter, we will use our studies of fish communities in the Florida Everglades to illustrate the linkage of ecological research, monitoring, and model building in a process leading to the development of restoration targets.

Monitoring and Model Development

Over the past century, goals for the Everglades have changed in terms of their relative degree of public support. Throughout much of this period, monitoring of the ecosystem's physical and biotic factors has contributed to the dialogue about such goals.

History of Monitoring

Alteration of the natural southerly flow of water through the Everglades, begun in the 1890s, was well underway by the early 1900s (reviewed in Blake 1980). Drainage was accelerated after two major hurricanes flooded

Lake Okeechobee in 1926 and 1928, killing thousands of people. The major canals and levees blocking natural flow patterns from the north were in place by 1930. Although formation of a national park in the Everglades region was suggested as early as 1905, the Everglades National Park (ENP), located at the southern end of the ecosystem, was not officially dedicated until 1947. The Tamiami Trail and Canal, completed in 1928, marked the northern boundary of the park and limited the southerly flow of water into it until large plugs were opened in the mid-1960s (S-12 structures; fig. 13-1). By that time, there was widespread agreement that the Park was not receiving adequate water supplies to fulfill its conservation mission. A federal commitment to re-think the south Florida water-delivery and drainage system, developed over the preceding one hundred years, was codified in a report from the U.S. Army Corps of Engineers in 1968 (Blake 1980, 191). This culminated in the Comprehensive Everglades Restoration Plan (CERP), a major restoration initiative passed by Congress in 2000 (Anonymous 2000).

Quantitative collections of marsh fishes and aquatic invertebrates from the Everglades are relatively recent and, prior to the 1950s studies, were

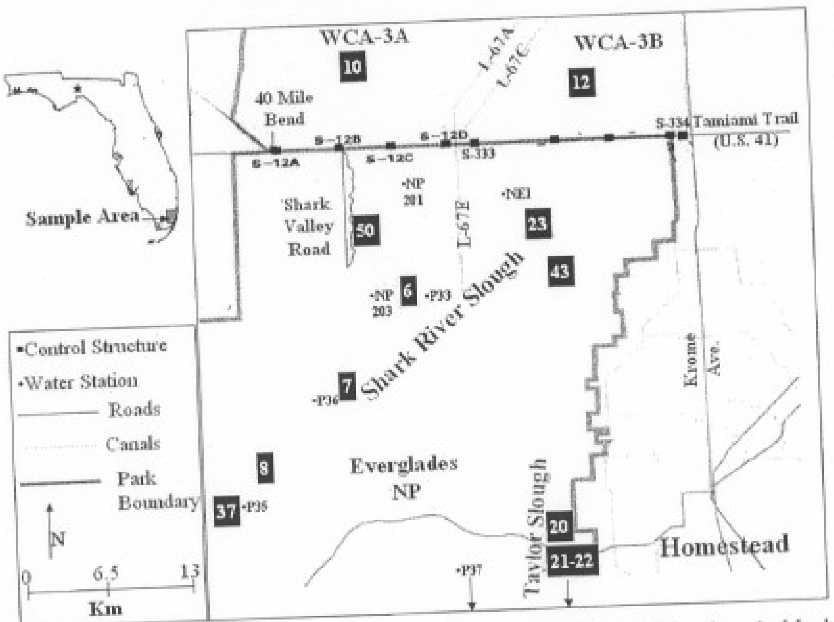

Figure 13-1. Map of the study sites in the Shark River Slough. Numbers in black squares indicates study sites. Sites 6, 23, and 50 are discussed in the text. Three plots, not shown, separated by approximately 1 kilometer of marsh habitat, were present at each study site.

entirely qualitative. The first quantitative studies of southern Florida freshwater animals were conducted in the early 1950s–1960s (Hunt 1952; Reark 1960, 1961; reviewed in Loftus and Eklund 1994). Reark (1960, 1961) was the first to collect fish and invertebrate density and biomass data with relation to vegetation cover. He compiled the only database, albeit very limited, from Shark River Slough (the primary drainage of the southern Everglades) before the operation of the S-12 water-control structures (fig. 13-1). Those flood gates placed control of southern marsh hydrological patterns in the hands of water managers, exacerbating departures in the timing, quantity, and spatial distribution from natural water flow in the Everglades National Park brought about by the Tamiami Trail. Thus, all data on aquatic animals from the Everglades south of the Tamiami Canal, and probably all data on the Everglades generally, were collected long after marsh hydrology was disturbed by the construction of the drainage system.

From 1965 to 1972, the U.S. Geological Survey (USGS) was contracted by Everglades National Park to collect data on aquatic-animal community composition and population variability related to hydrology in Shark River Slough (Higer and Kolipinski 1967; Kolipinski and Higer 1969). Those data all appeared in the peer-reviewed literature over a period of years (fishes: Kushlan 1976, 1980; apple snails [*Pomacea paludosa*]: Kushlan 1975; crayfish [*Procambarus* spp.] Kushlan and Kushlan 1979; and freshwater prawns [*Palaemonetes paludosus*] Kushlan and Kushlan 1980). In 1976, the research division of Everglades National Park began a long-term commitment to study the aquatic ecosystem. ENP supported an eight-year investigation of fish dynamics with relation to hydrology, using improved techniques (a 1-square-meter throw trap) developed and tested by Kushlan (1981; Loftus and Eklund 1994). Information generated by that eight-year study was used to design the next study that ran from October 1985 to 1992 (Loftus et al. 1990). In that study, a new sampling design was initiated to test hypotheses about the effect of marsh hydroperiod on community parameters. Collections at two of the long-term sites (6 and 23) were continued in this study, and a new site (50) added (fig. 13-1).

Following Hurricane Andrew in 1992, the original ENP-funded program was supplemented with support from the U.S. Army Corps of Engineers and the South Florida Water Management District. The new funding sources related to Everglades restoration projects, and the transfer of program personnel to the newly formed National Biological Service in 1993, resulted in major changes to the monitoring program. The increased funding also provided the first opportunity to build a larger regional sampling program by expanding the ENP program (fig. 13-1). That program expanded the system of monitoring sites in Everglades National Park and Water Conservation Area 3A and 3B to a total of twenty sites. A collaboration of Florida International University, ENP, and USGS personnel now share the sampling responsibilities. The most important aspect of the expanded program is that it employs a standardized and consistent sampling design to enable data comparisons across the region (Jordan et al. 1997a). Several reports and papers resulting from this long-term program have been published (Trexler et al. 1996a,b, 2000, Trexler and Loftus 2001; Loftus et. al. 1990, 1997; Turner et al. 1999).

Scope of Existing Monitoring

Monitoring aquatic animals in the southern Everglades has focused on the ridge-and-slough habitat that dominates the central Everglades ecosystem (Gunderson 1994). Sampling depends heavily on use of a 1-square-meter throw trap and is limited to spikerush-dominated (primarily *Eleocharis cellulosa*) prairie and slough habitats. The throw trap was described in Kushlan (1981), and the method has been well supported by a literature on sampling bias and effectiveness through the range of conditions where it is employed for Everglades monitoring (Kushlan 1981; Jacobsen and Kushlan 1987; Chick et al. 1992; Loftus and Eklund 1994; Jordan et al. 1997a). Limitations of the technique include that it does not sample large fishes (longer than 8-centimeter standard length) or small macroinvertebrates (able to pass through a 2-millimeter sieve) well. Large fishes are not enumerated well both because of trap avoidance and because they are too sparse to be collected effectively at the 1-square-meter scale (though sampling juveniles of these species is not biased by avoidance). Although crustaceans (crayfish and grass shrimp), dragonfly naiads, and several groups of aquatic bugs (e.g., Belostomatidae and Naucoridae) are sampled effectively, the majority of Everglades macroinvertebrates are too small for effective enumeration by field sorting (Turner and Trexler 1997). Since 1997, throw-trap sampling has been supplemented by boat electrofishing for larger species (Chick et al. 1999) to provide a more comprehensive monitoring of fish communities. Smaller macroinvertebrates have largely been ignored in landscape-scale monitoring of the Everglades (McCormick et al. 2001), but an aquatic invertebrate monitoring program has been started in the Everglades National Park. A separate fish-monitoring effort more than ten years in duration is ongoing in the mangrove zone at the southern edge of the freshwater Everglades. That study uses permanently installed drop-nets because of the challenges presented for sampling fishes among the prop roots of red mangrove tress (Lorenz et al. 1997; Lorenz 1999). Fishes are also sampled by ENP and USGS personnel in the Rocky Glades and other short-hydroperiod areas of Park by the use of wire minnow traps and throw traps (see also Kobza 2001; Trexler et al. 2001).

Special monitoring initiatives are in place for documentation of mercury in fishes, though long-term funding has not been identified. Eastern mosquitofish (*Gambusia holbrooki*) and largemouth bass (*Micropterus salmoides*) have been the targets of those efforts. Biologists from the U.S. Environmental Protection Agency and Florida International University have estimated total mercury burden in mosquitofish sampled from over seven hundred locations selected by a stratified random design to cover the entire Everglades marsh ecosystem. Their collections were made from 1995 to 1999 (Stober et al.1998, and subsequent reports). The sampling was completed in four synoptic sampling events in April 1995, May 1996, May 1999, and September 1999.

Clearly, not all ecosystem elements can be selected for monitoring. Each choice must be justified based on its responsiveness to management actions and Everglades restoration goals. Fishes, as well as aquatic invertebrates and frogs, have been chosen because of their critical roles in the Everglades food web and because they are important in the diets of charismatic species of wading birds and alligators. For example, most wading bird species found in the Everglades consume fishes as a portion of their diet, and many species, such as white ibis, also consume large quantities of crayfish. Ornithologists believe that wading bird nesting success is largely limited by food availability (Frederick and Spalding 1994; Ogden 1994). Apple snails serve as food for a number of birds, notably the limpkin and endangered snail kite. Fishes, crayfish, frogs, and apple snails are major diet items of alligators in the Everglades (Fogarty 1984; Barr 1997). In addition, pig frogs are economically important in the Water Conservation Areas, where they are harvested for commercial sale. Finally, the species or groups selected for monitoring are known to be sensitive to changing patterns of hydrology and water quality, responding to management actions over a range of time periods from months to years.

Performance Measures and Targets

The Comprehensive Everglades Restoration Plan is in its early phases and work on its monitoring program primarily involves development of restoration targets for performance measures (see Busch and Trexler, chap. 1), tools for assessment of those measures, and assimilation of data needed to complete assessments (Ogden et al., chap. 5). Performance measures and restoration targets are being set separately for several different habitats in the ecosystem. Targets for fish communities in ridge-and-slough habitats illustrate this process.

Four types of performance measures have been proposed for fishes that inhabit the ridge-and-slough habitat from a series of workshops employing best professional judgment. These four metrics are abundance, size distribution, relative abundance, and contaminants (Ogden et al., chap. 5). It is proposed that the recovery targets for these ridge-and-slough performance measures will be set by reference sites in Shark River Slough or patterns predicted from hydrological information from a natural system model (NSM). The hydrological NSM estimates water depth for 6.4-square-kilometer grid cells based on actual rainfall data across the freshwater ecosystem at a daily time step under natural landscape patterns, including the removal of all canals and levees (Fennema et al. 1994). Performance measures and their targets that are presently under consideration for fishes include (1) *Abundance.* In response to restoration of lengthened hydroperiods, patterns of fish population dynamics should include increases in marsh fish numbers (measured as density) and biomass. (2) *Size distribution.* In response to lengthened hydroperiods, the range of biomass and body length of marsh fishes should increase by increasing the frequency of large-bodied species. (3) *Relative abundance.* In response to lengthened hydroperiods, the relative abundance of centrarchids and chubsuckers should increase. (4) *Nonnative species.* An additional goal is to maintain the low frequency of nonnative fishes currently observed in the interior of the ridge-and-slough system. (5) *Contaminants.* Levels of mercury and other toxins in marsh fishes should be reduced.

Several challenges to the process leading to these performance measures have emerged. Criticisms of this process include that a rigorous conceptual and empirical basis for the performance measures has yet to be developed. Also, forcing the ecosystem into discrete habitat categories, such as ridge and slough or marl prairies, creates problems because the habitats are not uniform and distinctions emerge gradually along hydroperiod gradients. Spatially referenced simulation models can overcome these difficulties, but spatially explicit conceptual models become confusingly complex as the basis for discussion among agency personnel. Thus, the current conceptual models have emerged as an expedient mechanism to facilitate communication among groups with wide-ranging experience and interests. It seems reasonable that developing targets will be an evolutionary process, with modeling becoming critical to establish an objective foundation for targets in the long run.

Application of Performance Measures

Assessing restoration targets is a four-step process: (1) collect data indicative of fish communities under the range of present-day field conditions,

(2) develop analytical tools to predict values of performance measures from hydrological parameters (stressors), (3) use NSM hydrological data and analytical tools to derive restoration targets, and (4) once restoration begins, compare observed values for performance measures to targets. This process need not be complex if adequate data are available, as illustrated by our work with monitoring data from Everglades fishes. We have used statistical analysis of long-term data records from ridge-and-slough communities to develop functions that predict fish density given two hydrological parameters: days since an area last dried (water depth less than 5 centimeters) and average water depth during the thirty days prior to the assessment (fig. 13-2). The density of small fishes generally increases as time passes after a dry-down event because of population growth and re-colonization of the dried area. However, some species are always most abundant at short-hydroperiod sites (illustrated by flagfish in fig. 13-2), and these temporarily increase in relative abundance at long-hydroperiod sites following a dry-down event. The eastern mosquitofish is a unique species in re-colonizing the marsh rapidly following a dry-down event, and displaying little or no long-term trend with respect to time since dry down (fig. 13-2). Its relative abundance drops as time passes after a dry-down event because species characteristic of long-hydroperiods (illustrated by bluefin killifish in fig. 13-2) increase in density over time, while mosquitofish density remains relatively constant.

We estimated the functional relationship of fish density and days since marsh drying at twenty existing monitoring locations from WCA-3A, 3B, Shark River Slough, and Taylor Slough. The quality of the estimates varied among those sites related to the length of the existing data record and whether the site dried during the period when data were collected. Fortunately, the functions estimated from most locations were fairly similar in shape, so the results appear robust with some important exceptions. Data from one of our monitoring sites located near natural creeks in southern Shark River Slough (site 37, fig. 13-1) indicated a much more rapid recovery from dry-down events than for central marsh sites. This illustrates that assessment models must be spatially explicit and incorporate landscape features such as dry-season aquatic refuges (DeAngelis et al. 1997). It also provides a caveat for simplistic application of habitat characterizations.

The existence of long-term records for fishes in the Everglades makes the development of predictive models of performance measures possible. ENP personnel have collected records of fish density for over twenty years at two locations in Shark River Slough, one in the main slough south of the S-12C structure (Site 6; fig. 13-1) and one in the northeast Shark River Slough east of the L67E levee (Site 23; fig. 13-1). Time since the

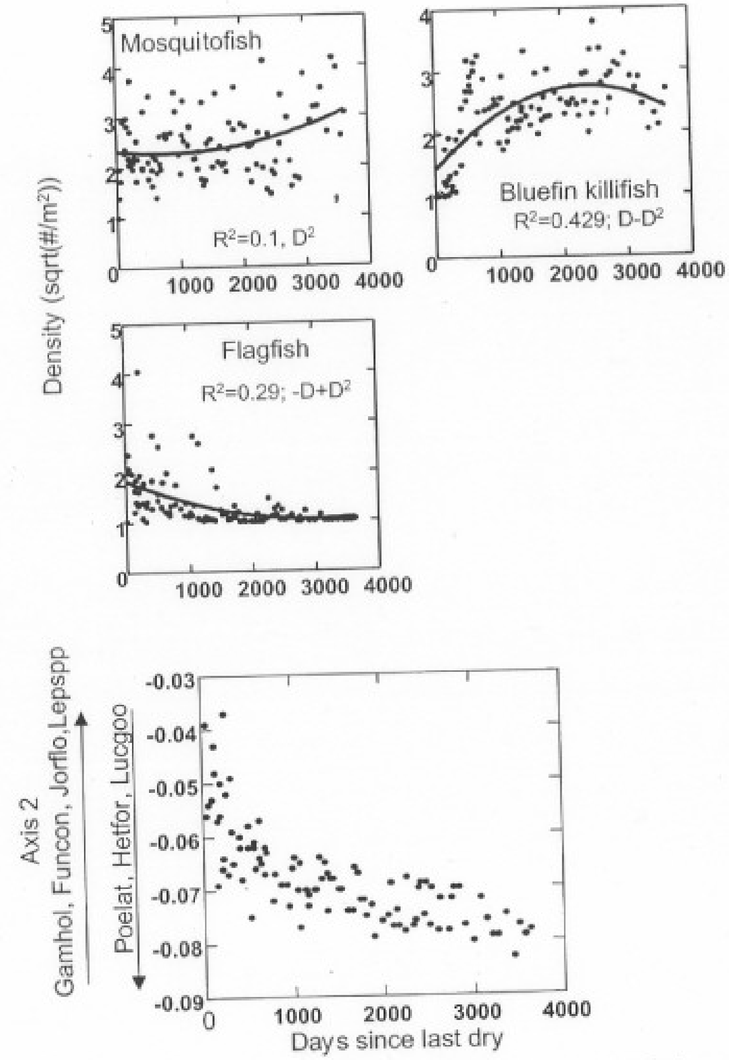

Figure 13-2. Patterns of fish density and relative abundance relative to days since marsh site was last dry for Site 6 between 1977 and 1999. (Top) Square-root transformed density is plotted for three characteristic species: mosquitofish, bluefin killifish, and flagfish. The coefficient of determination and parameters in a regression model are reported on each panel: D = days since last dry. (Bottom) Results from an ordination of relative abundance patterns using nonmetric multidimensional scaling plotted against days since last dry. Gamhol = *Gambusia holbrooki* (mosquitofish); Funcon = *Fundulus confluentus* (marsh killifish); Lepspp = *Lepomis* spp. (sunfish); Poelat = *Poecilia latipinna* (sailfin mollies); Hetfor = *Heterandria formosa* (least killifish); Lucgoo = *Lucania goodei* (bluefin killifish).

last dry-down event and water depth yielded a good fit to those data (fig. 13-3) and the results are presented here to illustrate the approach. Those sites also provide insight into the effect of manipulating hydrology as an ecological restoration tool for the Everglades. Management of northeast Shark River Slough (NESRS), east of the L-67E levee, was modified in

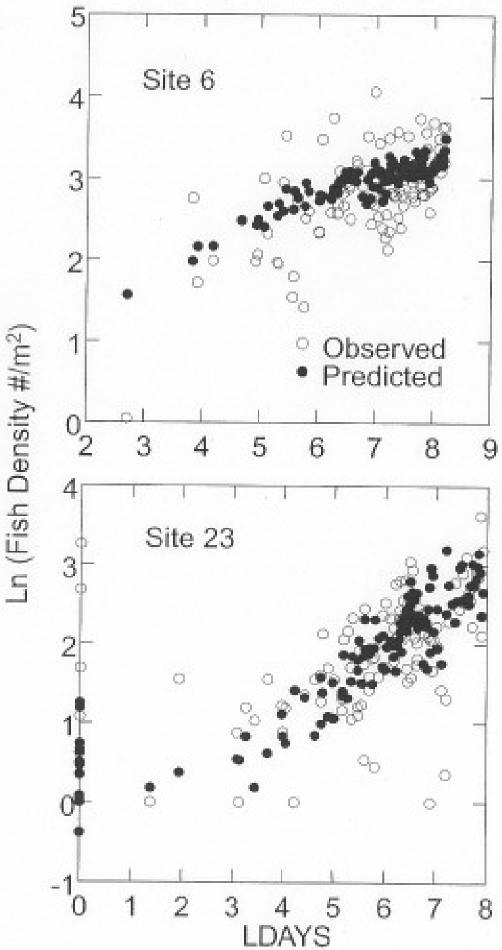

Figure 13-3. Observed and predicted densities of fishes (all species) plotted relative to days since last dry down. The predicted values were derived from a Poisson regression of the model: fish density = (average water depth the month samples were collected) + (natural log transformed number of days since the study site was last dry) + (natural log transformed number of days since the study site was last dry)^2. Site 6 R^2 = 58.4 %; Site 23 R^2 = 78.2%.

1985 under a federal program to lengthen the hydroperiod to better reflect historical conditions there. Management at the SRS site was not modified at that time, and we considered it to be a reference area for this "experiment" in management.

We obtained NSM-derived estimates of water depth for each study site under "natural system" conditions from 1971 to 1995 and used our statistical model to estimate fish density over that time period. We then compared the NSM-predicted density to the density of fishes observed from monitoring. The results indicate that the SRS site was much more similar to NSM-predicted conditions from 1971 to 1995 than was the NESRS site (fig. 13-4). From 1981 to 1986, the SRS site actually had more fishes than expected by the natural system model (fig. 13-4), because managers kept the site from drying out in 1981 when NSM predicted it should have dried. In contrast, the northeast Shark River Slough had fewer fishes than predicted by the natural system model throughout this period. Unfortunately, much of the perceived recovery of this study area occurred after the end of the NSM data record (1995), so further assessment of the success of the management experiment will have to wait for additional NSM calculations. By 1999, the density of fishes observed in the northeast Shark River Slough was in the range of that predicted previously for NSM conditions (approximately twenty fish per square meter), but this corresponded to a series of very wet years. We do not know if the natural system model will predict an even higher density of fishes than in previous years. The density of fishes in the northeast Shark River Slough in 1999 was not different from the reference site in SRS, providing some support for the hypothesis that restoration goals were met by this experimental water delivery.

This exercise illustrates several goals for further improvement in setting ecoregional assessment targets. First, we need to complete calibration of the hydrologically based statistical model to the observed data from our other eighteen study sites. Next, those equations must be incorporated into a landscape-based model that produces output in maps as is used in assessment models like the Across Trophic Level System Simulation (ATLSS; DeAngelis et al. 1998) and the Everglades Landscape Model (ELM; Fitz et al. 1996). Finally, we need access to NSM output that is updated routinely. Our efforts illustrate the importance of uninterrupted gathering of monitoring data to improve the statistical models and provide tests of status and trends in the ecosystem. Gaps in the long-term data greatly hamper this effort. For example, data collection was stopped by budget constraints for most of 1988 in the Shark River Slough, and those data available suggest an unusually high density of fishes (fig. 13-4). Finally, we need to continue to improve the assessment models, most

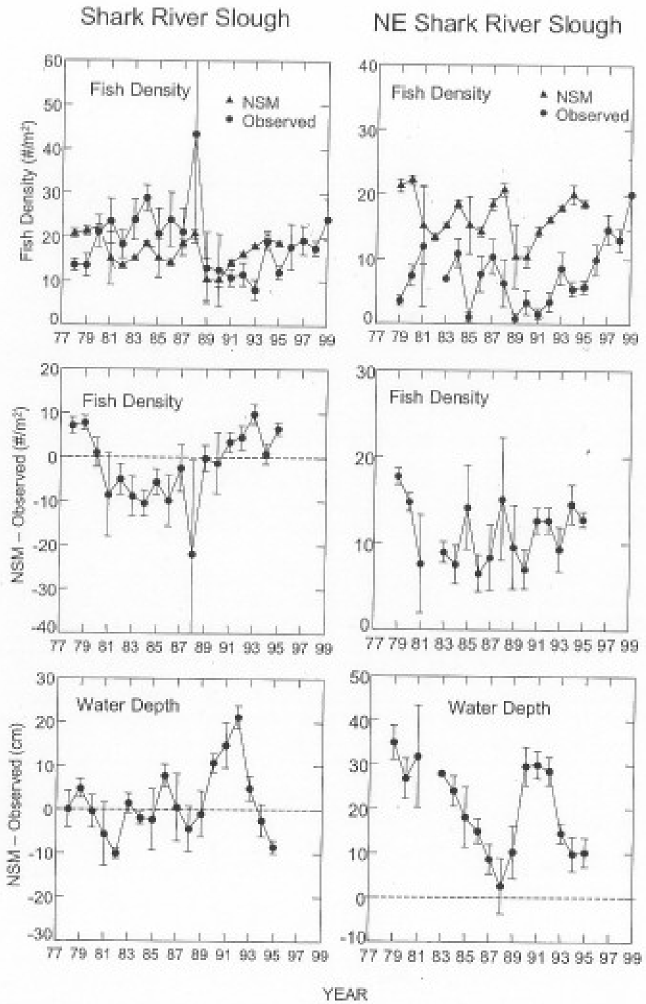

Figure 13-4. Results from analysis of fish density in Shark River Slough and northeast (NE) Shark River Slough. Top Panel: Plot of observed fish density from 1977 and 1999, and density predicted from the natural system model (NSM). NSM data were limited to the years from 1978 to 1995. Middle Panel: The difference between NSM-predicted fish density and observed fish density from 1978 to 1995. Positive values indicate fewer fishes were recorded than predicted under NSM conditions and negative values indicate more fishes than expected under NSM conditions. Bottom Panel: The difference between NSM predicted water depth (centimeters) and observed water depth in central Shark River Slough from 1978 to 1995. Positive values indicate shallower water was recorded than predicted under NSM conditions and negative values indicate deeper water than expected under NSM conditions.

clearly by incorporating information on the nutrient status of areas in the Everglades (see next section). It is expected that changing hydrology will change the landscape pattern of system productivity in subtle, but potentially important, ways that should be incorporated into natural-system scenarios. Current work with the ELM model is addressing this effort, and future development of fish-assessment tools will benefit from this work.

Monitoring Large-Fish Communities

Beginning in 1997, we used airboat electrofishing to monitor the distribution and species composition of large fishes (more than 8 centimeters standard length) in freshwater marshes of the Everglades. From October 1997 to December 1999, we sampled large fishes four times per year at sites within three regions of the Everglades: (1) Taylor Slough and (2) Shark River Slough in Everglades National Park, and (3) Water Conservation Area 3A (WCA-3A) (fig. 13-1). These three regions fall along a gradient of hydroperiod (the proportion of the year when marshes are flooded) with the shortest hydroperiod marshes found in Taylor Slough and the longest hydroperiod marshes in WCA-3A (Trexler et al. 2001). While total catch-per-unit-effort (CPUE) varied significantly through time, among regions, and among sites within regions, regional variation was the greatest. CPUE was consistently greater in both WCA-3A and Shark River Slough compared to Taylor Slough. Large-bodied predators, such as Florida gar (*Lepisosteus platyrhincus*) and largemouth bass (*Micropterus salmoides*), dominated the large-fish community of WCA-3A, whereas the Shark River Slough community was composed primarily of smaller species, such as lake chubsuckers (*Erimyzon sucetta*) and spotted sunfish (*Lepomis punctatus*). Composition of the Taylor Slough large-fish community was highly variable through time. As seen with fishes collected by throw trap, CPUE of large fishes was positively correlated to hydroperiod and soil total phosphorus. Phosphorus is the limiting element for plant growth in the Everglades (Davis 1994), and its availability in soil is positively correlated with hydroperiod.

A simulation model suggested that encounter rates between prey fishes and a large predatory fish is positively related to both predator density and hydroperiod. The potential influence of large-predatory fishes is greatest in the longest hydroperiod and least in short hydroperiod marshes. Interestingly, the density of forage species, estimated by throw trapping at the same study sites, was greatest in Shark River Slough where the large-fish community was dominated by the omnivorous lake chubsucker (diet in SRS approximately 33 percent detritus; 65 percent invertebrates; Loftus

2000). These data illustrate that fish communities vary regionally within habitat types such as ridge and slough. Spatially explicit performance measures, rather than simply habitat-based ones, are needed to incorporate regional variation of this type.

Monitoring Nonnative Fishes

Fish monitoring has provided fortuitous information on the invasion and status of nonnative fishes in the Everglades (Trexler et al. 2000). The monitoring data indicate that introduced fishes are present in the ridge-and-slough community at very low numbers. The long-term nature of the monitoring data has illustrated a boom-bust pattern of invasion by several species entering Shark River Slough. For example, the dry years of 1989 and 1990 corresponded to the range expansion and boom in abundance of at least one nonnative fish, the pike killifish (*Belonesox belizanus*). Monitoring in 2001 and beyond should be focused on identifying similar patterns following this dry period, should they occur. Nonnative species, especially the Mayan cichlid (*Cichlasoma uropthalmus*), are known to be more abundant in habitats surrounding the Everglades ridge and slough. Long-term monitoring in the mangrove zone indicated great fluctuation in the abundance of that species, possibly related to infrequent freeze events (Trexler et al. 2000). Fish monitoring in short-hydroperiod habitats suggests that hydrological management may affect the success of nonnative species there through its influence on their demography and dispersal ecology.

Monitoring Contaminants

The REMAP study revealed "hot spots" of mercury contamination in WCA-3A and Everglades National Park that have led to the development of models of mercury deposition and bioaccumulation (Stober et al. 1998). Also, the temporal data indicate that the concentration of total mercury in Everglades mosquitofish declined by as much as 24 percent from the 1995–1996 sampling events to those in 1999. Similar results have been obtained by monitoring of largemouth bass by the biologists of the Florida Fish and Wildlife Conservation Commission (Lange et al. 2000), a trend that may have reversed in the following drier years of 2000 and 2001. The long hydroperiods of the unusually wet years 1997–1999 have been implicated in this decline, a hypothesis with implications for management that can be tested from future monitoring results. Fish mercury contamination has complex origins in food web ecology that is mediated by both community ecology (food-chain length) and mercury biogeochemistry (availability of methyl mercury to the food web) (Stober et al. 1998). Thus, while mercury should be monitored because of its ecosystem and human health effects, further work is needed before restoration targets can be generated for this parameter.

Assumptions for Natural-System Simulations

Setting monitoring targets by any technique requires that assumptions be made about ecosystem function in general, and specifically in the interpretation of data on which the targets are based. For example, determining whether the Comprehensive Everglades Restoration Plan fulfills the goals of restoring Everglades ridge-and-slough communities requires that targets be identified against which contemporary communities can be compared as restoration progresses. We have illustrated a statistical and simulation approach for establishing NSM targets for fishes, based on ecological studies at a diversity of sites. Those sites represent the spectrum of hydroperiods that we believe also characterized the historical Everglades ridge-and-slough system. Many of those sampling sites may also be considered "reference sites," although they have experienced variable degrees of hydrological alteration. Given the absence of "pristine" Everglades habitats, we have identified nine assumptions inherent in producing and interpreting our natural-system scenario.

1. *Wading bird predation did not regulate numbers of fishes in the historical Everglades.* There is strong evidence that the number of wading birds nesting in southern Florida has decreased significantly in the past century. In the contemporary ecosystem, there is no evidence that wading birds regulate the number of fishes at the landscape scale. The predatory effects of wading birds appear to be very local and limited in time to periods of dry-down when fishes become available. These events are uncommon and generally lead to situations in which most fishes would have perished had they not been consumed. In the unmanaged ecosystem, such events would have been confined to the edges of the system in most years because systemwide dry-downs occurred less frequently than at present.
2. *Recent patterns of wading bird nesting have not grossly altered landscape patterns of fish productivity.* Frederick and Powell (1994) indicated that wading birds could transport and concentrate phosphorus across the ecosystem. Contemporary wading bird populations may be consuming 4.9 tons (dry mass) less prey than historical populations (Frederick and Powell 1994). Thus nutrient transport could have been substantial under historical conditions when bird rookeries of the southern Everglades included thousands of nests. This probably created local

conditions of high primary and secondary production in the vicinity of rookeries. Historically, rookeries were concentrated in the mangrove zone of the extreme southern Everglades, but nesting birds have moved north into areas now in the Water Conservation Areas. Thus, this landscape pattern of avian-supported productivity has changed. At present, these effects are limited to the immediate vicinity of rookeries where they may be important, but they probably have little effect on system-level production.

3. *The Everglades is a smaller ecosystem than it was historically, and the remaining edge habitats have been diminished in area and quality.* Marl prairies and rockland habitats have been reduced in area and in their ability to support aquatic species. Transverse glades draining east to Biscayne Bay have been developed and channelized. Also, the Everglades agricultural area south of Lake Okeechobee was formerly a large area of the historical Everglades ecosystem. The implications of the loss of these habitats, some of which were never well described in their pre-drainage condition, are unknown but are unlikely to have been beneficial.
4. *Drainage has altered soil formation and led to loss of soil in much of the Everglades.* The loss of deep peats and changes to shallow peat and marl substrates has had unknown implications for aquatic community dynamics.
5. *Mercury contamination, which has reached high levels in Everglades fishes, has had little ecological effect on these communities.* While there are no historical data to determine if this is a new phenomenon, studies have demonstrated links to anthropogenic environmental changes (Stober et al. 1998). Mercury contamination may arise from recent mercury deposition and/or biogeochemical processes that methylate elemental mercury naturally present in Everglades sediments. The health effects of chronic low-level mercury exposure to fishes, and implications for their population dynamics, have not been explored for any fish that inhabits the Everglades. Other heavy metals, such as selenium, arsenic, and cadmium are also potential contaminants that may affect fish health and can be transported throughout the ecosystem by the system of canals.
6. *Our results can be generalized across the ridge-and-slough habitat, although our fish data have mainly been collected from one habitat (spikerush-dominated wet-prairie or sloughs).* Wet prairies are one of the two predominant habitats of the central Everglades (Gunderson 1994). Limiting our efforts to this habitat could bias our conclusions:

   First, there is some evidence, largely anecdotal, that historical Everglades wet-prairie sloughs were deeper than those studied today, with abrupt edges. Related to this is the proposal that the relative spatial coverage of sawgrass-dominated ridges versus wet-prairie sloughs was different in the historical ecosystem as a function of more-rapid velocity of downstream flow (Chris McVoy, personal communication).

   Second, applying data from wet-prairie samples may overestimate fish density in the total ridge-and-slough habitat. Data from sampling sawgrass-dominated ridges adjacent to wet-prairie sloughs indicate slightly fewer fishes on the ridges, which can be so densely vegetated that little interstitial aquatic habitat remains. The ridges also dry before the adjacent sloughs (Jordan et al. 1997b; Trexler et al. 2001).
7. *The presence of levees and water-control gates, their effect on hydrology, and their proximity to our study sites do not greatly influence our results.* Preliminary sampling data indicate that the electrofishing CPUE of large fishes diminishes with distance from canals for approximately 1 kilometer. Also, preliminary results from radio tracking Florida gar in WCA-3A suggest that large fishes move several kilometers from canals into the marsh to feed. All of our monitoring sites are at least 3 kilometers from canals, and generally much farther. Canals may function as large, albeit artificial, dry-season refuges. Implications of the large area of such deep-water refuges that were absent under historical conditions are not well understood (but see Loftus 2000; Trexler et al. 2000). Of course, the function of artificial refuges must be contrasted with the historical ecosystem, which held much more water in its central flow-way and seldom dried compared to that habitat today.
8. *Nonnative species, including several species of fishes, have not altered the community dynamics as revealed by our contemporary monitoring and experimental studies.* At least for the ridge-and-slough communities of the southern Everglades, there is reason to accept this assumption (Trexler et al. 2000). New threats from introduced fishes arise routinely in the region (e.g., the recently introduced Asian swamp eel [*Monopterus* sp., probably multiple species] has a life history unique among current invaders, with unpredictable implications [Collins et al. 2002]).
9. *The hydrological Natural System Model provides adequate estimates of historical water depths that can serve as a basis for simulating historical communities.* There are few hydrological data for validation of historical water depths predicted by the NSM. The model's predictions have received extensive evaluation by hydrologists and biologists, who agree with them in general (a process of best professional judgment), but who are often critical of its predictions in local areas. Continued

development of the hydrological Natural System Model is a linchpin to further modeling of biological restoration scenarios.

Model Evaluation, Uncertainty, and Prospects for Monitoring Targets

The advent of societal pressure for science-based management has led to use of simulation models to predict the outcome of ecological management (Sarewitz and Pielke 1999). However, a burgeoning literature emphasizes that it is at best challenging, if not impossible, to verify or validate natural-system or predictive models (e.g., Oreskes et al. 1994, but see Rykiel 1996). In place of validation, it may more appropriate to discuss model evaluation under a diversity of scenarios (Oreskes 1998) leading to delineation of the uncertainties that emerge (Snowling and Kramer 2001). It may be possible to identify and minimize uncertainties (Rotmans and van Asselt 2001), even when predictions per se are impossible.

Despite the reality that we can never validate our NSM predictions, we believe that alternative approaches for goal setting are even more problematic and subject to whims of politics, policy, and expediency. Of course, policy makers and managers appropriately set the final direction of any public works project such as the Comprehensive Everglades Restoration Plan, but our NSM goals provide guidance for benchmarks that seek to minimize subjectivity as a starting point for management action.

The restoration enterprise must include a sustained effort to improve natural-system models and scenarios as new technology and information become available. A key future step, in addition to improving the natural-system model, is to delineate and perhaps quantify uncertainties in the output. Philosophers of science point out that uncertainty cannot be eliminated, and recommend that managers include it explicitly in the decision process (Reynolds and Ford 1999; Rotmans and van Asselt 2001; Reynolds and Reeves, chap. 7). Techniques permitting explicit estimation of uncertainty need to be incorporated into the NSM process to target priority areas for improvement, and minimize the potential of negative impacts of unknowns.

Ecological systems are not static. While the chemical properties of the limiting nutrient, phosphorus, may have been relatively constant throughout the brief history of the Everglades, the same cannot be said for the reactions that determine the abundance of organisms that inhabit it. New species enter ecosystems and others go extinct (naturally or with human assistance), with potential to dramatically change the regulation of biological populations and communities. Species themselves are not stable over evolutionary time, and examples can be cited in which characteristics

influencing population dynamics have changed over ecological time (e.g., Reznick et al. 1997; Grether et al. 2001). We have attempted to delineate the assumptions necessary to develop an Everglades natural-system model based on modern ecological interactions, but uncertainty will remain an inevitable partner in this exercise. Because of this, the use of reference sites must remain an important complement to NSM predictions, even when their comparability is imperfect. For example, the Everglades is an oligotrophic, hydrologically pulsed ecosystem (Turner et al. 1999) with few counterparts. Wetlands in Mexico's Yucatan peninsula and nearby Belize may provide reference ecosystems for examination of key ecological processes that they may have in common. However, in a more practical vein, we have illustrated an example where locations within the restoration served as references for local subprojects within the larger effort. This must be done with circumspection, however, because it precludes documentation of landscape-scale effects and other forms of ecological "surprise."

The Everglades restoration initiative may be relatively well endowed with monitoring data that will assist in assessment of ecosystem alterations as the project proceeds. However, even for this project, many areas are poorly documented and require uncomfortable extrapolation beyond existing data to build NSM scenarios. This hodgepodge of monitoring at the outset of a regional initiative is to be expected (see Hemstrom, chap. 11). Further, we have identified nine challenging assumptions that must be considered for the application of fishery data in assessment of ecosystem restoration. The challenge for the Comprehensive Everglades Restoration Plan, and all ecoregional initiatives, is to maximize the use of existing historical and contemporary data in assessment programs that address both landscape-scale (ecoregion-wide) restoration and short-term local project needs.

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CHAPTER 14

Monitoring Biodiversity for Ecoregional Initiatives

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The conservation of biodiversity has become an important issue receiving national and international attention (Noss 1991; Noss and Cooperrider 1994; Wilson 1992). Biodiversity has been defined in various ways (Harrod et al. 1996), leading to confusion amongst the general public about what biodiversity conservation means (Brussard et al. 1992). For example, Volland (1980) and Gast et al. (1991) define biodiversity as "the variety, distribution, and structure of plant and animal communities, including all vegetative stages, arranged in space over time that support self-sustaining populations of all natural and desirable naturalized plants and wild animals." We use biodiversity to describe the variety of life forms, the ecological roles they perform, and the genetic diversity they contain (Wilcox 1984).

Biodiversity hierarchy theory suggests that what happens at the higher levels of ecological organization, such as the landscape or ecosystem level, influences the lower levels, such as the species or genetic level (Allen and Star 1982; Noss 1990). Biodiversity hierarchy is composed of the genetic, species-population, community-ecosystem, and landscape or regional levels (Grumbine 1992; Harrod et al. 1996). Although these levels are convenient for the sake of discussion, they are not completely isolated or