

# Trajectories of vegetation-based indicators used to assess wetland restoration progress

JEFFREY W. MATTHEWS,<sup>1</sup> GREG SPYREAS, AND ANTON G. ENDRESS

*Illinois Natural History Survey, Champaign, Illinois 61820 USA, and  
Department of Natural Resources and Environmental Sciences, University of Illinois, Urbana, Illinois 61801 USA*

**Abstract.** Temporal trends in attributes of restored ecosystems have been described conceptually as restoration trajectories. Measures describing the maturity or ecological integrity of a restoration site are often assumed to follow monotonically increasing trajectories over time and to eventually reach an asymptote representative of a reference ecosystem. This assumption of simple, predictable restoration trajectories underpins federal and state policies in the United States that mandate wetland restoration as compensation for wetlands damaged during development. We evaluated the validity of this assumption by tracking changes in 11 indicators of floristic integrity, often used to determine legal compliance, in 29 mitigation wetlands. Each indicator was expressed as a percentile relative to the distribution of that indicator among >100 naturally occurring reference wetlands. Nonlinear regression was used to fit two alternative restoration trajectories to data from each site: an asymptotic (negative exponential) increase in the indicator over time and a peaked (double exponential) relationship. Depending on the particular indicator, between 48% and 76% of sites displayed trends that were at least moderately well described ( $R^2 > 0.5$ ) by one of the two models. Floristic indicators based on species richness, including native richness, number of native genera, and the floristic quality index, rapidly increased to asymptotes exceeding levels in a majority of reference wetlands. In contrast, indicators based on species composition, including mean coefficient of conservatism and relative importance of perennial species, increased very slowly. Thus, some indicators of restoration progress followed increasing trajectories and achieved or surpassed levels equivalent to high-quality reference sites within five years, whereas others appeared destined to either not reach equivalency or to take much longer than mitigation wetlands are typically monitored. Finally, some indicators of restoration progress, such as relative importance of native species, often increased over the first five to 10 years and then declined, which would result in a misleading assessment of progress if based on typical time scales of monitoring. Therefore, the assumption of simple, rapid, and predictable restoration trajectories that underlies wetland mitigation policy is unrealistic.

**Key words:** bioindicators; Carex; emergent wetland; floodplain forest; floristic quality; Illinois, USA; invasive species; reference sites; species richness; succession; wetland mitigation.

## INTRODUCTION

A major goal of restoration ecology is to identify commonalities in the temporal dynamics of restored ecosystems in order to make restoration practice more predictable. Restoration trajectories, which plot measures of ecosystem attributes over time, are often used to monitor restoration progress. These trajectories are often assumed to follow smooth increases over time, followed by an eventual arrival at a stable level that is characteristic of a natural reference ecosystem (Bradshaw 1984, Kentula et al. 1992, Aronson and Le Floch 1996). Although some conceptual models include alternative possible endpoints and exceptions to a deterministic increase (e.g., Hobbs and Norton 1996), both restoration practitioners and theorists often assume

that restoration activities guide a site along a simple trajectory of self-repair or self-design by facilitating deterministic succession (e.g., Mitsch et al. 1998, Shuwen et al. 2001, Weinstein et al. 2001). This notion is manifested in the U.S. federal policy of compensatory wetland mitigation, which permits losses of natural wetlands under the assumption that equivalent, replacement wetlands can be constructed over a short time frame. In contrast, a more complex view of restoration trajectories would acknowledge uncertainty in restoration planning and monitoring.

The assumption of predictable temporal trajectories in restored wetlands has been criticized based on empirical evidence (Race 1985, Zedler 1996, Zedler and Callaway 1999, 2000). For example, studies of restored saline and brackish marshes indicate that not all structural and functional properties of restored wetlands follow expected, simple trajectories (Simenstad and Thom 1996, Streever 2000, Whigham et al. 2002), and even when

Manuscript received 21 July 2008; revised 12 January 2009; accepted 2 April 2009. Corresponding Editor: J. C. Callaway.

<sup>1</sup> E-mail: matthews@inhs.uiuc.edu

they do, some properties take much longer than typical monitoring periods to reach equivalency with reference sites (Craft et al. 2002, 2003, Morgan and Short 2002, Edwards and Proffitt 2003). Furthermore, measures of restoration progress sometimes unexpectedly decline after several years of increase. For example, large-scale diebacks of desired vegetation have been reported in constructed estuarine marshes after six to seven years of increasing abundance (Simenstad and Thom 1996, Dawe et al. 2000).

In addition to the unpredictability inherent in restoration trajectories, there is uncertainty regarding how best to monitor restoration progress. Measurable indicators of restoration progress, usually based on vascular plant communities, are often monitored in compensatory mitigation wetlands for three to five years after site construction (National Research Council 2001). Most of these indicators were originally developed and tested in order to evaluate the ecological integrity of existing natural wetlands. For example, percentage of native plant species or native species richness, percentage of or cover of perennial species or hydrophytic vegetation, and richness of *Carex* species have been shown to vary predictably along gradients of human disturbance among wetlands (Mack et al. 2000, DeKeyser et al. 2003, Ervin et al. 2006, Miller et al. 2006, Reiss 2006, Lougheed et al. 2007). These indicators are now frequently applied in monitoring compensatory mitigation progress (Matthews and Endress 2008), but aside from species richness, the pace and direction of restoration trajectories for these indicators have received little study.

One increasingly popular group of indicators for rating ecological integrity and restoration progress is floristic quality assessment (FQA), which is based on two metrics, the floristic quality index (FQI) and the mean coefficient of conservatism ( $\bar{C}$ ) (Swink and Wilhelm 1994). These metrics rely on the concept of plant species conservatism, the notion that species can be ranked based on their relative fidelity to undegraded natural areas. State and federal agencies have used these indicators to quantify the integrity of wetlands that will be impacted by development, to set mitigation ratios (the ratio of wetland area damaged to the area to be created or restored), and to define the performance standards used to judge mitigation progress (Herman et al. 1997, Streever 1999, Matthews and Endress 2008). A few studies have compared FQA values in restored vs. reference communities (Fennessy et al. 2004, Balcombe et al. 2005b, Taft et al. 2006). Surprisingly however, despite their widespread use in restoration monitoring, few published studies have reported changes in FQI and  $\bar{C}$  over time in restorations (but see Spieles et al. 2006, Nedland et al. 2007).

Once appropriate indicators have been chosen to measure restoration progress, indicator values are ideally evaluated relative to natural reference sites (Kentula et al. 1992). In practice, however, compensa-

tory mitigation monitoring rarely involves direct comparisons to reference wetlands. Instead, it usually involves determining whether a few, seemingly arbitrarily set, indicator target levels are reached (Matthews and Endress 2008). This may be attributable, at least in part, to the difficulty of defining an appropriate reference system based on present-day communities or presumed pre-European-settlement conditions (Brown 1994, Cairns and Heckman 1996, Hobbs and Norton 1996, White and Walker 1997). Furthermore, variability among potential reference sites is high even among sites that are relatively free from modern human influence, meaning that the choice of a single reference site will bias the outcome of any assessment (Pickett and Parker 1994, Morgan and Short 2002). Instead of defining a single reference state based on past or present natural communities, the reference concept should be expanded to encompass regional, among-site variation (Brinson and Rheinhardt 1996, White and Walker 1997). Ideally, a restored site or a population of restored sites would be compared to a population of reference sites (Kentula et al. 1992, Kentula 2000). This approach, by defining multiple targets, recognizes that multiple restoration trajectories are possible (Simenstad et al. 2006).

Two general approaches for incorporating information from multiple reference sites have been used. First, restored and reference sites can be displayed together in multivariate (most commonly multispecies) space (e.g., Holl and Cairns 1994, Wilkins et al. 2003, Kirkman et al. 2004). Second, a univariate indicator in the restored site can be compared directly to statistics (such as the mean or minimum) that describe the distribution of that indicator in a population of reference sites (e.g., Simenstad and Thom 1996, Karr and Chu 1999, Rheinhardt et al. 1999, Gutrich et al. 2009). The present study employs the latter approach and presents a novel method for tracking changes in indicators in a large number of restorations over time. Specific objectives were to: (1) compare commonly used, vegetation-based indicators in restored freshwater wetlands to the distribution of these indicators in reference sites, thereby evaluating both the utility of these indicators and the progress of the mitigation wetlands, (2) determine whether a simple, increasing trajectory model was an appropriate description of changes in floristic integrity over time or a peaked model was a better description of temporal dynamics in restored wetlands, and (3) determine whether the typical three- to five-year monitoring time frame is sufficient to describe major trends in restoration trajectories.

## METHODS

### *Study sites and vegetation sampling*

Twenty-nine compensatory mitigation wetlands (14 forested and 15 herbaceous) were included in this study (Appendix A). All were constructed by the Illinois Department of Transportation (IDOT) between 1991 and 2002 to mitigate damage incurred to natural

wetlands during road construction projects. Sites were located throughout Illinois and ranged in size from 0.1 to 7.1 ha. For three to five years following construction, the sites were monitored annually, in late summer. A comprehensive vascular plant species list was compiled during a thorough search of the entire site. In addition, in 16 of the wetlands the herbaceous-layer vegetation was quantitatively sampled using 16–159 square quadrats (1 m<sup>2</sup> or 0.25 m<sup>2</sup>) per site, placed systematically along 3–17 transects per site and encompassing the full range of habitat conditions. Plant species observed in each quadrat were recorded, with the exception of woody plants over 1 m tall, and were assigned a cover class of <1%, 1–5%, 6–25%, 26–50%, 51–75%, 76–95%, or 96–100% (Daubenmire 1959). Cover class data were used to calculate relative frequency, relative cover, and importance value (IV, the sum of relative frequency and relative cover, divided by two) for each species. Although number and size of quadrats varied among sites, sampling was consistent within sites from year to year.

To expand the time window of the analyses, 25 of the 29 sites were revisited from late May through June in 2006. At this time the revisited sites varied in age from 5 to 14 years, and vegetation was resurveyed using methods similar to the original surveys but standardized across sites. Vegetation was sampled as above using 10 0.25-m<sup>2</sup> quadrats placed at equal distances along each of four parallel transects for a total of 40 quadrats per site. In addition, we performed a timed search (20 min plus 7 min per hectare) at each site to compile complete plant species lists. Although the number of quadrats differed from the previous years of sampling, quadrat data were used only in the calculation of three indicators that were not likely biased by differences in sampling intensity: IV of perennials, IV of native species, and IV of hydrophytic species. All other metrics were based on comprehensive species lists.

Four restored wetlands included in this study had been surveyed in a total of four years, 12 had been surveyed in five years, and 13 had been surveyed in six years. We treated each site as an independent test of the trajectory model and compared each site individually to two independent populations of reference sites. The rationale for using two different reference sets was to test the robustness of our results to differences in reference information. The first reference set included 553 naturally occurring wetlands delineated by the Illinois Natural History Survey (INHS) for IDOT from 1992 to 2005 during the months of June through September. The IDOT reference sites encompassed the range of sizes among restored wetlands, varying from 0.1 to 20.8 ha. During on-site field visits, INHS personnel determined each site to be a jurisdictional wetland (U.S. Army Corps of Engineers 1987) and compiled a comprehensive plant species list for the wetland. These reference sites occurred within IDOT project areas proposed for road construction and are

therefore representative of the sites that were compensated for by the mitigation wetlands included in this study. Although reference sites varied widely in ecological integrity, this variation was appropriate for comparative purposes because the goal of wetland mitigation policy is the replacement of destroyed wetlands. However, IDOT reference sites lacked quantitative vegetation sampling of the herbaceous layer. Therefore, we selected a second set of reference sites that had quantitative vegetation data for comparison.

The second reference set included naturally occurring wetlands from the Illinois Critical Trends Assessment Program (CTAP; Illinois Department of Natural Resources 2001). A total of 200 CTAP reference wetlands were sampled from 1998 to 2004 by INHS botanists. These included 181 randomly selected forested ( $n = 45$ ) and herbaceous wetlands ( $n = 136$ ) distributed throughout Illinois and an additional 19 sites that were chosen specifically because they represented the most pristine examples of herbaceous and forested wetlands in the state. Forested sites were sampled from mid-May through June, and herbaceous sites were sampled in July. In forested wetlands all vascular plants, except woody species taller than 1 m, within 30 0.25-m<sup>2</sup> quadrats were assigned cover class values. Quadrats were distributed along three 50-m transects (10 quadrats per transect) that radiated out from a randomly selected center point in randomly selected, nonoverlapping directions. Additionally, because several of our vegetation-based indicators utilize comprehensive species lists, a 0.1-ha plot surrounding one transect was searched for additional species. In herbaceous wetlands vascular plants were sampled in 20 0.25-m<sup>2</sup> quadrats located along a single transect and a larger 0.205-ha plot encompassing the quadrats was surveyed for additional species. Detailed CTAP methods are described elsewhere (Carroll et al. 2002). Groundwater-fed fens and seeps and wetlands fringing lakes were excluded from both reference sets because these types of sites were not represented among the restored wetlands in this study.

Wetlands in each set (restored, IDOT reference, and CTAP reference) were categorized as forested or herbaceous wetlands (marshes and wet meadows) and as northern (>41° latitude), central (39° to 41° latitude) or southern (<39° latitude) (Table 1). We combined central and northern forested wetlands due to the small number of northern forested reference sites. We compared each restored wetland to the full subset of reference sites of the same type and region as the restored wetland.

### *Analyses*

Indicators of plant community integrity were calculated for each reference and restored site. Indicators based on the FQA utilized coefficients of conservatism (*C*) assigned to each species in Illinois (Taft et al. 1997). Coefficients of conservatism are subjective ratings of species' relative fidelity to undegraded natural commu-

TABLE 1. Sample sizes for restored and reference wetlands for each wetland region and type.

Wetland region and type	Restored	CTAP reference	IDOT reference
Northern herbaceous	8	50	165
Central herbaceous	2	51	57
Southern herbaceous	5	39	94
Northern/central forest	9	32	97
Southern forest	5	22	146

Notes: Twenty-nine compensatory mitigation wetlands (14 forested and 15 herbaceous) in Illinois, USA, were included in this study (Appendix A). All were constructed by the Illinois Department of Transportation (IDOT). One reference set included naturally occurring wetlands from the Illinois Critical Trends Assessment Program (CTAP). The other reference set included naturally occurring wetlands delineated by the Illinois Natural History Survey (INHS) for IDOT.

nities and range from 0 (weedy species) to 10 (conservative species, intolerant of habitat degradation). Indicators included: (1) native species richness, (2) proportion of all species at the site that were native, (3) mean coefficient of conservatism ( $\bar{C}$ ), (4) FQI, (5) conservative ( $C \geq 5$ ) species richness, (6) *Carex* species richness, (7) number of native genera, (8) proportion of all species at the site that were perennial, (9) summed IV of hydrophytic species, (10) summed IV of native species, and (11) summed IV of perennials. Native status of species and species life spans (perennial vs. annual/biennial) were based on regional floras (Swink and Wilhelm 1994, Mohlenbrock 2002). Mean  $\bar{C}$  was calculated as the mean of coefficients of conservatism for all plant species at a site and FQI was computed as

$$\text{FQI} = \bar{C} \times \sqrt{S} \quad (1)$$

where  $S$  is the total number of plant species at the site (Swink and Wilhelm 1994). All nonnative species were assigned a  $C$  of zero. Hydrophytic species were defined as facultative, facultative wetland, and obligate wetland species (Reed 1988). Indicators based on importance value were not calculated for IDOT reference wetlands, which lacked quantitative vegetation sampling.

For indicators based on whole-site species lists, we examined the relationship between indicators and wetland type (restored vs. IDOT reference wetlands) using analysis of covariance (ANCOVA) with log-transformed site area as a covariate. Native richness and number of native genera were log-transformed, conservative richness and *Carex* richness were square-root transformed, and proportion of natives and proportion of perennials were arcsine square-root transformed prior to ANCOVA.

Each indicator value in each restored wetland in each year was expressed as a percentile relative to the distribution of the values of that indicator among its reference sites (Fig. 1). The percentile score for an indicator could range from 0 (0th percentile), if the indicator in the restored site was lower than the indicator in every reference site, to 1 (100th percentile),

if the indicator in the restored site was higher than the indicator in every reference site. Some indicators in restored sites might increase beyond the values observed in reference wetlands, obscuring temporal trends beyond the 100th percentile. However, because the goal of compensatory mitigation is to replace and not necessarily to improve upon or exceed natural wetlands, our use of percentiles was suitable. Transformation to percentiles simultaneously allowed for standardization among restored wetlands and allowed for each restored wetland to be scored relative to its unique set of reference sites (i.e., sites of the same wetland type and region). Because reference sites were sampled in different years, our reference distributions capture variability in reference condition in both time and space.

The percentile score for each indicator in each site was plotted against site age to characterize the restoration trajectory (Fig. 1). Nonlinear least squares regression, using a Gauss-Newton algorithm in SYSTAT 11, was used to fit restoration trajectories to the observed data (Engelman 2005). We considered two alternative models. The first assumed that the value of an indicator ( $y$ ) increased to an asymptote, a trend that would be well described by the negative exponential function:

$$y = a(1 - e^{-bt}) \quad (2)$$

where  $t$  is site age in years,  $a$  represents the asymptotic maximum, and  $b$  is a slope parameter (Fig. 2A). A negative exponential shape has been proposed as a likely restoration trajectory in wetlands (Kentula et al. 1992), describes some previously observed trends in coastal wetland restorations (Zedler and Callaway 1999, Morgan and Short 2002, Craft et al. 2003), and reflects the assumption underpinning mitigation policies that wetland restoration is an orderly, predictable process. Alternatively, an indicator could initially increase to a peak and then decline (Fig. 2B). Such a trajectory would be more appropriately described by a double exponential function:

$$y = a(e^{-ct} - e^{-bt}). \quad (3)$$

Note that Eq. 3 reduces to Eq. 2 if the additional slope parameter  $c$  equals zero. A peaked function often describes trends in species richness over the course of succession (Anderson 2007), and we expect it to describe trends in initially successful restorations that subsequently undergo setbacks such as exotic species invasion. Appendix B includes additional details on trajectory models.

For each fitted model,  $R^2$  was calculated as the squared correlation between the observed and predicted values (Engelman 2005). We considered any model with an  $R^2 > 0.5$  to be a moderately well-described restoration trajectory, a model with an  $R^2 > 0.7$  to be a well-described trajectory, and a model with an  $R^2 > 0.9$  to be very well described. Additionally, if  $R^2$  for the double exponential model (Eq. 3) was greater than 0.5, and if it



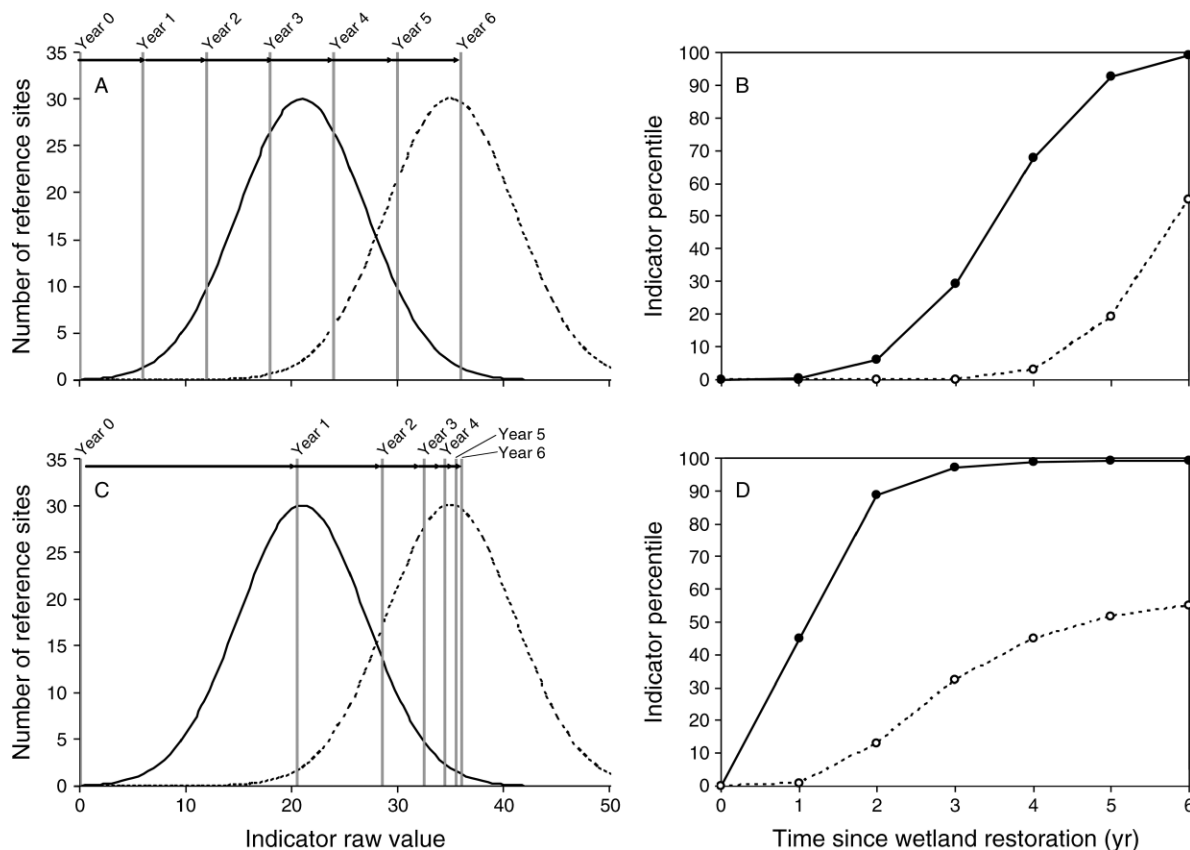


FIG. 1. (A) Frequency distribution of the values of an ecological indicator in a set of reference sites (solid curve). Indicator values for each of the first six years in a hypothetical restoration site are represented as vertical gray lines. Ideally, indicator values in a restored site increase over time, both in raw value and relative to the distribution of the same indicator in the reference sites. In the first example, we assume a linear increase in the indicator over time in the restored site. Restoration progress in each year can be expressed as a percentile relative to the reference distribution by determining the percentage of reference sites with indicator values lower than the value in the restored site (corresponding to the percentage of reference sites to the left of the vertical line). If the restored site is compared to a different set of reference sites (dashed curve) that are of higher average integrity than the first set, the percentile values change. (B) Temporal trajectory of the indicator in the restored site expressed as a percentile relative to the first set of reference sites (solid line) and the second, higher-integrity set of reference sites (dashed line). (C) A second hypothetical restoration site can be compared to the same reference sites. In this example, the raw value of the indicator increases asymptotically over time, which changes the shapes of the trajectories (D) of the indicator relative to the reference sites.

exceeded the  $R^2$  for the corresponding negative exponential model by at least 0.1, we considered the restoration trajectory to be best described by the double exponential function. We used our observations from a large number of independent sites to draw inferences; if consistent trajectories were observed across wetlands, then we inferred the existence of general patterns.

## RESULTS

The IDOT reference wetlands encompassed the range of sizes of the restored wetlands, but on average, the restored wetlands (mean = 2.55 ha) were larger than the reference sites (mean = 1.15 ha). All indicators except proportion of perennial species increased significantly with wetland area ( $\alpha = 0.05$ ; Table 2). However, even after accounting for the effect of site area using ANCOVA, restored wetlands in the final year of site monitoring had greater richness of native, *Carex*, and

conservative species, higher FQI, and a greater number of native genera than reference sites. In contrast, reference sites had greater proportion of natives and proportion of perennials (Table 2). We note that because sampling methods were not identical for restored and reference wetlands, direct comparison of some indicators between restored and reference sites should be interpreted with caution (but see Appendix C). However, these results suggest that the difference in indicator values between restored and reference sites was not solely attributable to differences in wetland area.

Asymptotic (negative exponential) increases and peaked (double exponential) functions were good descriptions of the trends in floristic quality indicators over time in most sites (representative examples are shown in Fig. 2 and Appendix D). Depending on the particular metric and the reference set, between 48% and 76% of sites displayed trends that were at least

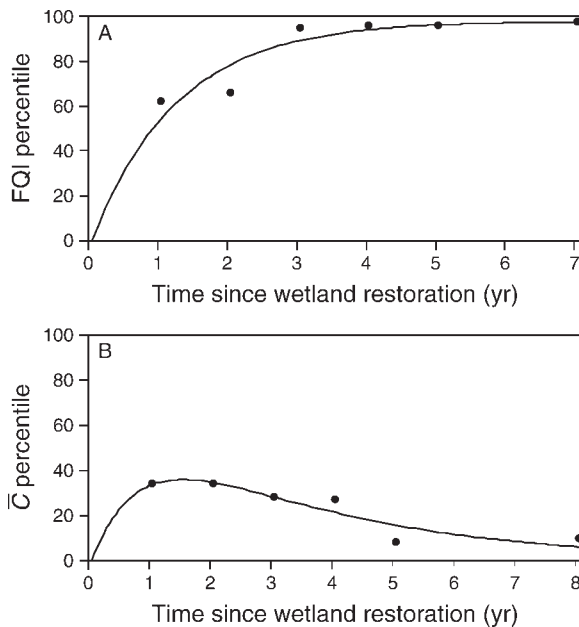


FIG. 2. Representative examples of observed restoration trajectories. (A) Asymptotic trajectory of the floristic quality index (FQI) over time in a restored marsh in Stephenson County, Illinois, USA, relative to the Critical Trends Assessment Program (CTAP) reference data set. Equation:  $y = 0.98(1 - e^{-0.81t})$ ;  $R^2 = 0.82$ . (B) Peaked trajectory of mean coefficient of conservatism ( $\bar{C}$ ) in a restored floodplain forest in Henry County, Illinois, relative to the Illinois Department of Transportation (IDOT) reference data set. Equation:  $y = 0.76(e^{-0.32t} - e^{-1.21t})$ ;  $R^2 = 0.85$ . Coefficients of conservatism ( $C$ ) are ratings of species' relative fidelity to undegraded natural communities and range from 0 to 10. Mean  $C$  was calculated as the mean of  $C$  values for all plant species at a site. The floristic quality index (FQI) was calculated as the product of mean  $C$  and the square root of species richness.

moderately well described ( $R^2 > 0.5$ ) by one of these trajectory models (Table 3). Asymptotic increases were good descriptions of trends in native species richness and other related indicators (number of native genera, FQI, and *Carex* richness). Values of  $R^2$  for these indicators

averaged from 0.63 to 0.71. The trajectory of proportion of perennial species was also well described by an asymptotic increase, but the predicted values of parameter  $b$  were generally much lower, indicating a much slower increase over time (Appendix E). Trajectories of conservative richness,  $\bar{C}$ , and perennial IV were also well described by asymptotic increases, but in 14–21% of sites, they were better described by a peaked function with an initial increase followed by an eventual decline. Trajectories of the remaining indicators (proportion of native species, native IV, and hydrophyte IV) were better described by the peaked function than by the simpler asymptotic increase. Adding the extra parameter,  $c$ , increased the amount of variation explained by models for these three indicators, increasing  $R^2$  by an average of 0.14–0.25 (Table 3).

In most cases in which a peaked function best described the trajectory of an indicator, the indicator reached its predicted peak within five years after site construction. Percentage of native species and importance values of native, perennial, and hydrophytic species peaked, on average, two years after site construction. Conservative richness, on average, peaked between the second and third years. In contrast, where  $\bar{C}$  followed a peaked trajectory, the peak tended to occur later, usually after the fourth or fifth year.

All analyses of trajectories were performed individually for each restored wetland. Therefore, to visually depict generalized trajectories for each indicator, we plotted the percentiles, averaged across sites within site age classes, for each indicator in each year (Fig. 3). Similarly, we plotted generalized trajectories for the raw indicators (Appendix F). Averaged across sites within age classes, native and conservative species richness (Fig. 3A, I), FQI (Fig. 3G), and number of native genera (Fig. 3D) very rapidly increased to levels exceeding those in a majority of natural reference wetlands (i.e., they exceeded the 50th percentile). *Carex* richness (Fig. 3B),  $\bar{C}$  (Fig. 3E), and proportion and importance of perennials (Fig. 3F, K) increased less rapidly (see also

TABLE 2. Summary of analyses of covariance for the effect of wetland type (restored vs. Illinois Department of Transportation [IDOT] reference sites) on floristic integrity indicators.

Indicator	Restored	Reference	Wetland type		Area	
			$F_{1,579}$	$P$	$F_{1,579}$	$P$
Native richness	71.8 (21.0)	26.3 (14.8)	73.57	<0.001	65.01	<0.001
Proportion of native species	0.81 (0.09)	0.85 (0.11)	7.66	0.006	9.75	0.002
$\bar{C}$	2.21 (0.46)	2.28 (0.78)	2.03	0.155	17.61	<0.001
FQI	20.7 (5.8)	12.5 (6.0)	30.56	<0.001	63.60	<0.001
Conservative richness	12.0 (7.9)	4.4 (5.0)	36.66	<0.001	45.54	<0.001
<i>Carex</i> richness	5.6 (4.1)	2.0 (2.2)	36.78	<0.001	19.65	<0.001
Native genera	55.0 (13.3)	22.6 (11.8)	65.81	<0.001	60.32	<0.001
Proportion of perennial species	0.79 (0.08)	0.83 (0.16)	4.97	0.026	2.08	0.150

Notes: Values for restored and reference sites are expressed as means with SD in parentheses. Log-transformed site area was included as a covariate. Coefficients of conservatism ( $C$ ) are ratings of species' relative fidelity to undegraded natural communities and range from 0 to 10. Conservative richness was defined as the number of species with  $C \geq 5$ . Mean  $C$  ( $\bar{C}$ ) was calculated as the mean of  $C$  values for all plant species at a site. The floristic quality index (FQI) was calculated as the product of mean  $C$  and the square root of species richness.

TABLE 3. Summary of results from nonlinear regressions of indicators of floristic integrity, expressed as percentiles, on restored wetland age.

Indicator and reference set	<i>n</i>	Mean $R^2$ for negative exponential	Number moderately well described by trajectory†	Number well described by trajectory‡	Number very well described by trajectory§	Mean increase in $R^2$ for double exponential model	Number best described by double exponential¶
Native richness							
CTAP	29	0.638	20	20	12	0.017	0
IDOT	29	0.707	18	17	16	0.034	0
Proportion of native species							
CTAP	29	0.378	15	11	1	0.200	8
IDOT	29	0.374	18	7	3	0.253	10
$\bar{C}$							
CTAP	29	0.408	14	5	0	0.107	5
IDOT	29	0.430	14	8	1	0.087	4
FQI							
CTAP	29	0.660	22	12	5	0.055	2
IDOT	29	0.693	21	17	10	0.041	1
Conservative richness							
CTAP	29	0.494	20	9	4	0.151	6
IDOT	29	0.558	18	13	5	0.146	6
<i>Carex</i> richness							
CTAP	29	0.633	19	16	5	0.027	1
IDOT	29	0.668	22	17	4	0.025	1
Native genera							
CTAP	29	0.660	21	20	13	0.027	1
IDOT	29	0.653	17	16	13	0.023	1
Proportion of perennial species							
CTAP	29	0.712	21	19	4	0.056	2
IDOT	29	0.728	21	18	6	0.056	2
Hydrophyte IV							
CTAP	16	0.309	10	7	1	0.193	8
Native IV							
CTAP	16	0.367	8	5	2	0.137	5
Perennial IV							
CTAP	16	0.576	12	9	2	0.062	3

Notes: Coefficients of conservatism ( $C$ ) are ratings of species' relative fidelity to undegraded natural communities and range from 0 to 10. Conservative richness was defined as the number of species with  $C \geq 5$ . Mean  $C$  ( $\bar{C}$ ) was calculated as the mean of  $C$  values for all plant species at a site. The floristic quality index (FQI) was calculated as the product of mean  $C$  and the square root of species richness. Importance value is the sum of relative frequency and relative cover, divided by 2. Other abbreviations are: CTAP, Critical Trends Assessment Program; IDOT, Illinois Department of Transportation;  $C$ , coefficient of conservatism.

†  $R^2 > 0.5$  for negative exponential or double exponential models, or indicator was at 100th percentile for entire period of monitoring.

‡  $R^2 > 0.7$  for negative exponential or double exponential models, or indicator was at 100th percentile for entire period of monitoring.

§  $R^2 > 0.9$  for negative exponential or double exponential models, or indicator was at 100th percentile for entire period of monitoring.

¶  $R^2 > 0.5$  for double exponential model and  $R^2$  for double exponential model exceeded  $R^2$  for negative exponential model by at least 0.1.

Appendix E). Averaged across sites, no discernable trajectories were evident for proportion of native species (Fig. 3C), hydrophyte IV (Fig. 3H), or native IV (Fig. 3J). However, for these indicators, a greater number of individual sites followed peaked rather than asymptotic trajectories (Table 3), indicating that a majority of restored sites were trending away from high-quality reference conditions.

The choice of reference data impacted assessments of restoration performance. Restored wetlands, especially restored forested wetlands, had lower average percentile

scores when compared to CTAP vs. IDOT reference wetlands (Fig. 3), indicating that the IDOT reference data set was a lower standard for comparison than the CTAP reference data set. Although the shape of the restoration trajectory was largely unaltered, changing the reference data set led to a different interpretation of restoration progress relative to the reference condition (see also Fig. 1). Relative to their respective set of reference wetlands, restored herbaceous wetlands tended to have indicator values higher than those of restored forested wetlands, presumably because forests require a

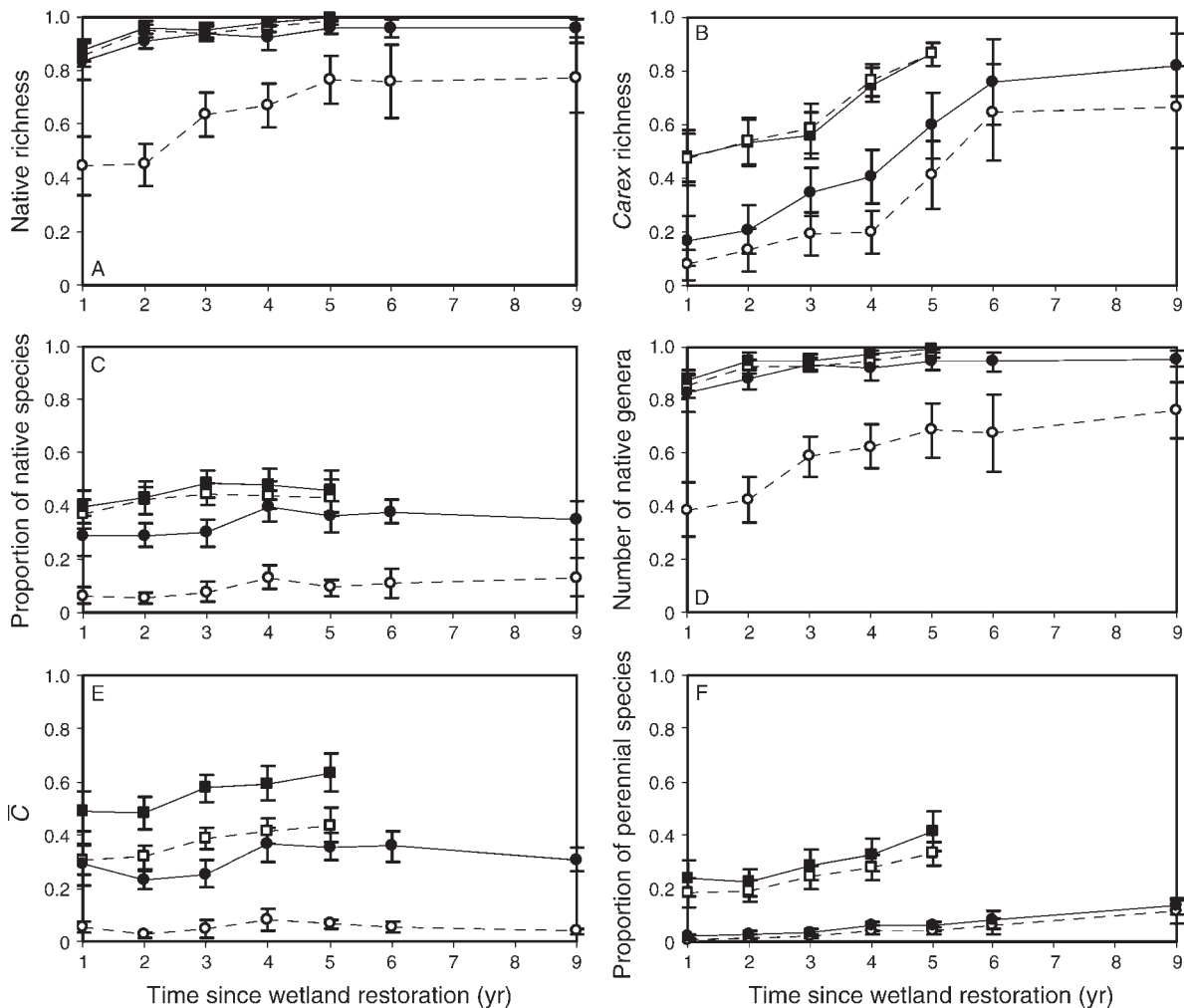


FIG. 3. Pooled trajectories for indicators in restored wetlands relative to reference wetlands. Data points are percentile scores (mean  $\pm$  SE) across restoration sites within site age classes. Solid lines and solid symbols represent percentile scores relative to the Illinois Department of Transportation (IDOT) reference data set, and dashed lines and open symbols represent percentile scores relative to the Critical Trends Assessment Program (CTAP) reference data set. Squares represent values in herbaceous wetlands, and circles represent values in forested wetlands. Data are not shown for age classes where  $n < 6$  (e.g., herbaceous wetlands  $> 5$  yr and forests  $> 9$  yr), and indicators based on importance value (IV) are not shown for forests due to low sample sizes. Sample sizes for herbaceous wetlands were 12, 15, 15, 13, and 11 for sites of ages 1, 2, 3, 4, and 5 yr, respectively, except for indicators based on IV, for which sample sizes were 7, 11, 12, 10, and 10. Sample sizes for forested wetlands were 9, 14, 14, 13, 10, 6, and 7 for sites of ages 1, 2, 3, 4, 5, 6, and 9 yr, respectively. Other abbreviations are: C, coefficient of conservatism; FQI, floristic quality index.

longer time than herbaceous wetlands to approach reference conditions. Indicators followed surprisingly similar trajectories in restored forested and herbaceous wetlands despite the fact that they had different reference sites, though slopes were often lower in forests (Fig. 3, Appendix E).

#### DISCUSSION

##### Mitigation progress relative to reference wetlands

An objective of our study was to compare vegetation-based indicators in restored and reference wetlands. Previous comparisons of mitigation wetlands to reference wetlands based on various ecological indicators have often revealed low levels of restoration

progress (National Research Council 2001). In contrast, many vegetation-based indicators from restored wetlands in the present study compared favorably to reference wetlands. Plant species richness was high in restored freshwater wetlands within one or two years after site construction, and many restorations rapidly achieved plant species richness equivalent to or exceeding the richness of the highest quality natural wetlands. On average, restored herbaceous wetlands achieved levels of richness of native species and genera, FQI, and richness of *Carex* and conservative species that were greater than levels in 85% of sampled reference wetlands within five years. Restored forested wetlands were generally less species-rich compared to



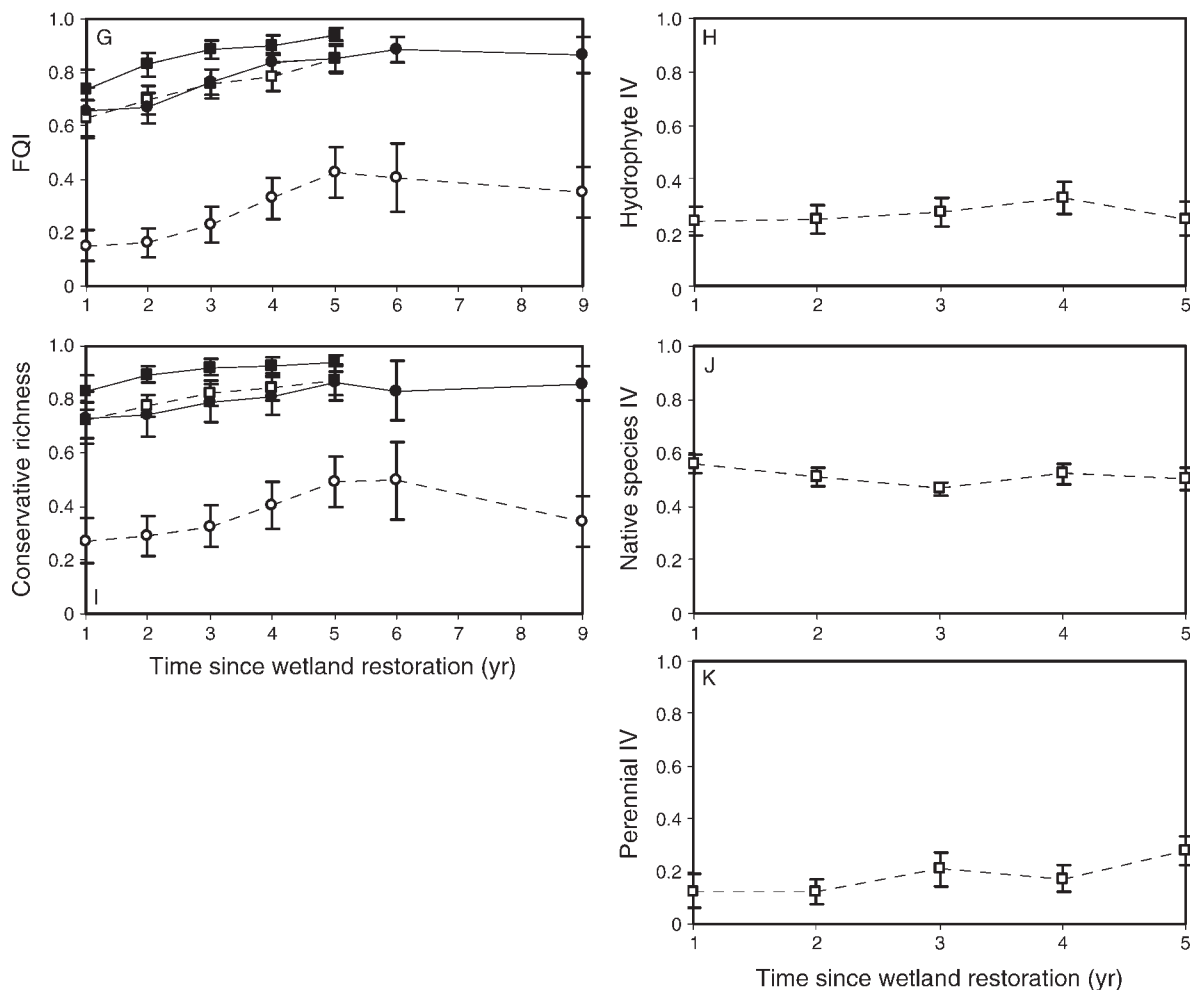


FIG. 3. Continued.

reference forests. Nevertheless, restored forested wetlands, on average, achieved levels of richness of native species and genera, FQI, and richness of *Carex* and conservative species that were greater than in at least 40% of sampled reference forests within five years. The greater richness and FQI observed in restored sites compared to reference sites was not attributable to area alone, as restored wetlands had greater richness and FQI even after accounting for wetland area using ANCOVA. Other indicators of plant community integrity (e.g.,  $\bar{C}$ , proportion of perennial species, and proportion of native species), however, remained at low levels relative to reference sites, indicating that species composition was not equivalent between restored and reference wetlands.

Species richness and diversity are often higher in recently restored freshwater wetlands than in natural reference wetlands (Jarman et al. 1991, Confer and Niering 1992, Kentula et al. 1992, Brown 1999, Magee et al. 1999, Wissinger et al. 2001, Balcombe et al. 2005a, b, Spieles et al. 2006, Gutrich et al. 2009). In contrast, and consistent with trends in the present study,

indicators based on species composition such as proportion of native species, floristic quality, and richness or cover of various plant guilds less frequently reach levels equivalent to those in reference wetlands (Brown 1999, Seabloom and van der Valk 2003, DeBerry and Perry 2004, Fennessy et al. 2004, Brooks et al. 2005, Spieles et al. 2006, Gutrich et al. 2009). Several studies have compared restored wetlands of different ages at a single point in time (i.e., space-for-time substitution studies), and these have reported higher species diversity, floristic quality, or native species presence in older restorations (Reinartz and Warne 1993, Noon 1996, Mushet et al. 2002, Balcombe et al. 2005b), though species richness is sometimes higher in younger restorations (Noon 1996, Campbell et al. 2002). Surprisingly few published studies have documented changes within sites over time in restored freshwater wetland plant communities. Consistent with the present study, existing longitudinal studies have often reported increasing species richness or indicators of floristic quality over the first 10–12 years of site development (Brown 1999, Moore et al. 1999, Mulhouse

and Galatowitsch 2003, Spieles et al. 2006, Nedland et al. 2007, Gutrich et al. 2009).

*Utility of vegetation-based indicators  
for monitoring restored wetlands*

Plant communities may be useful as indicators of restoration progress if they respond predictably to site conditions and age. The value of vascular plants as bioindicators has been discussed by several authors (e.g., Mack et al. 2000, Cronk and Fennessy 2001, U.S. Environmental Protection Agency 2002, Johnston et al. 2008), and vegetation-based indicators, especially FQI and multimetric indices that incorporate FQI, are now widely used in wetland assessment. Coefficients of conservatism have been developed for several U.S. and Canadian regions, and recent studies have evaluated the utility of FQI and  $\bar{C}$  in naturally occurring wetlands. For example, FQI has been shown to be consistently negatively correlated with anthropogenic stress in and around wetlands (Lopez and Fennessy 2002, DeKeyser et al. 2003, Cohen et al. 2004, Bourdaghs et al. 2006, Ervin et al. 2006, Miller and Wardrop 2006, Nichols et al. 2006, Reiss 2006).

An objective of our study was to examine several of these commonly used, vegetation-based indicators to determine which were most useful in evaluating restoration progress. Despite their success in distinguishing between high- and low-quality natural wetlands in other studies, metrics that incorporate plant species richness, such as FQI, can be very high in recently restored wetlands. As a result, they may be insensitive indicators of restoration progress, unable to differentiate successful from unsuccessful restorations. This has significant implications for wetland permitting programs in the United States, which increasingly rely on FQI to determine legal compliance of compensatory wetland mitigation projects (Streever 1999, Matthews and Endress 2008). Species diversity often increases in response to disturbance (Odum 1985, Sheil and Burslem 2003), including disturbance associated with initial restoration activities. Succession studies indicate that diversity often increases to a maximum a few years after disturbance and then declines (Anderson 2007). Indicators, such as FQI, that incorporate species richness may follow a similar pattern (Francis et al. 2000), although we found little evidence of an eventual decline in native species richness or FQI over the time frame of the present study.

Although indicators based on species richness often overestimated wetland performance, indicators based on species composition or dominance were more effective. Trajectories of indicators such as percentage or cover of native species and  $\bar{C}$  were often successful at distinguishing between relatively successful and less successful restored wetlands (Appendix D). Species-specific data, such as species conservatism or life form, have been shown to provide a more informative assessment of restoration progress than species richness

alone (Taft et al. 2006), and some authors have suggested using modified forms of FQI or  $\bar{C}$  instead of FQI, in order to weight species richness less heavily in assessments of floristic integrity (Rooney and Rodgers 2002, Cohen et al. 2004, Miller and Wardrop 2006). Indicators such as  $\bar{C}$  that are based on species composition alone, unlike those that incorporate species richness, are less sensitive to sample area and differences in sample date (Francis et al. 2000, Matthews 2003, Matthews et al. 2005, Bourdaghs et al. 2006). Other metrics such as percentage of perennials and *Carex* richness, which increased slowly relative to reference conditions, may be valuable indicators of successional status in wetland restorations.

*Temporal trajectories of indicators*

Another objective of our study was to describe the temporal trajectories of various vegetation-based indicators in restored wetlands. We demonstrate that some indicators follow simple, increasing, and largely predictable trajectories and achieve levels equivalent to high-integrity reference sites within a short time frame. However, for other indicators, especially those based on species composition, equivalency with high-integrity reference wetlands may never be achieved or will only be achieved over a longer time frame than the typical three to five years of mitigation monitoring. For example, even after 14 years, proportion of perennial species in restored forested wetlands is low relative to reference forests (Appendix D).

Furthermore, some ecosystem attributes may increase over the first few years of site development, and then decline, potentially leading to a faulty assessment of site progress if based on typical short-term monitoring. Restoration practitioners and regulatory agencies should not assume that all attributes of restored communities will follow invariably increasing trajectories toward desired conditions. For example, in the present study, proportion and importance of native species often peaked and then declined. This raises a critical question for defining an appropriate monitoring period: "If sites are to fail, when are they likely to show initial signs of failure?" Peaks in proportion and importance of natives usually occurred within the first three years following site construction, so the typical five-year mitigation monitoring period may be sufficient to determine whether a wetland will develop problems with nonnative species invasion. In contrast, in cases in which  $\bar{C}$  increased and then declined, the peak usually occurred after four or five years, and the rate of decline following the peak was slow (Appendix E), indicating that monitoring beyond five years will be required to determine whether  $\bar{C}$  eventually stabilizes at an acceptable level. Sites in this study ranged in age at the final year of monitoring from four to 14 years; thus it seems likely that if more sites were monitored over longer time periods, even more sites and even more floristic indicators would show similar declines.

There are several reasons why a restored site might not follow a rapid and smooth trajectory toward a desired reference condition. For example, invasion by introduced species can inhibit recovery (Stylinski and Allen 1999, McIver and Starr 2001). The wetlands included in this study often failed to meet legal performance standards requiring that they be dominated by native, non-weedy species, and were frequently dominated by the nonnative, invasive species *Phalaris arundinacea* L. and *Typha angustifolia* L. (Matthews and Endress 2008). Establishment and increasing dominance by these species may be responsible for observed peaked restoration trajectories in some sites (Appendix D). Other factors with the potential to increase the complexity of restoration trajectories include slow succession imposed by propagule limitation (Donath et al. 2003, Galatowitsch 2006, Foster et al. 2007), constraints on the direction or pace of successional trajectories resulting from small restoration areas or altered landscape settings (Cairns and Heckman 1996, MacMahon and Holl 2001, Simenstad et al. 2006), alternative pathways of community assembly resulting from accidents of species arrival order or initial conditions (Lockwood 1997, Young et al. 2001, Trowbridge 2007), and unexpected events such as floods (see Appendix D) and pathogen outbreaks (Simenstad and Thom 1996, Zedler and Callaway 1999, Klötzli and Grootjans 2001).

An alternative to the deterministic trajectory concept views restoration as an attempt to push a site beyond a series of hurdles or thresholds. Restoration is envisioned as a punctuated rather than gradual process, where restoration activities force a site out of the stasis imposed by some limiting constraint to initiate a transition toward a more desirable semi-stable state (Hobbs and Norton 1996, Yates and Hobbs 1997, Lindig-Cisneros et al. 2003, Mayer and Rietkerk 2004, Hobbs 2007). If restoration activities are not sufficient to push the site beyond a "domain of attraction" (Holling 1973), site conditions can revert back to those of the degraded state, resulting in a peaked restoration trajectory similar to several trajectories observed in the present study. Both of these conceptual models may be useful, but the applicability of each model will vary depending on the type of site being restored and the particular ecosystem components considered. Therefore, the choice between applying active management to overcome constraints vs. relying on deterministic succession and passive restoration ultimately depends on restoration objectives, as well as the level of site and landscape degradation and the likelihood that alternative stable states exist in the region and community type being restored (McIver and Starr 2001, Hobbs 2007, Cramer et al. 2008).

*Implications of chosen reference conditions  
for mitigation and restoration monitoring*

It is unrealistic to define a single, targeted reference state (Hobbs and Norton 1996, Whisenant 1999, Choi

2004, Hobbs 2007). The use of multiple reference sites, as in the present study, can overcome several of the difficulties of defining a reference standard for restoration. This approach acknowledges the high variability in structure and function among even pristine sites. Furthermore, it does not rely on difficult-to-support notions of historic wetland condition.

Even when multiple reference sites are used, however, the choice of a population of reference sites influences the interpretation of restoration progress. For example, indicator percentiles of restored wetlands relative to the IDOT references differed from percentiles relative to the CTAP references. The CTAP reference sites were randomly selected and were distributed throughout Illinois, but also included some specifically chosen, high-quality reference wetlands (see *Methods*). In contrast, the IDOT reference sites were more often located in areas already undergoing rapid urban development. These differences may explain the underlying differences in floristic integrity between reference sets. Despite the subjectivity inherent in defining any reference condition, reference sites are still invaluable in setting restoration targets and monitoring restoration sites (Aronson et al. 1995), as long as the reference sites are reasonably similar to the restored site (e.g., of the same community type and region) and are appropriately matched to the restoration goals. The goal of compensatory wetland mitigation is the replacement of destroyed or damaged natural wetlands, many of which are biologically degraded. Therefore, our reference wetlands were chosen to represent typical wetlands affected by development, and as is typical of destroyed wetlands, the reference sites spanned a range of ecological integrity. If the goal of restoration were to restore wetlands to a very high ecological integrity, a different set of reference wetlands, including only pristine examples, should be chosen.

Where measurable performance standards must be established to evaluate legal compliance with mitigation permits, standards would ideally be based on direct comparison to an appropriate regional distribution of reference sites. In the midwestern United States, because existing natural wetlands are heavily degraded by agricultural activities, indicators of vegetation integrity in restored wetlands should, at a minimum, be higher than those in a majority of natural wetlands. For example, based on our analysis, we recommend that mitigation permits specify that restored herbaceous wetlands in Illinois have a  $\bar{C}$  and proportion of native species greater than 50% of similar reference wetlands, and we recommend that the values of these indicators should remain steady or increase each year after site construction. If only relatively pristine wetlands are included in the reference set, then performance standards can be adjusted accordingly. For example, an indicator in a restored site might only be expected to exceed the fifth percentile of a distribution of pristine

wetlands, thus achieving a level comparable to at least a few high-quality reference sites (see Fig. 1).

The small-scale, site-specific focus of many restoration studies and the difficulty of comparing restoration attempts across different situations have limited the development of general models in restoration ecology (Halle and Fattorini 2004). By expressing restoration site performance in each year as a percentile relative to a reference distribution, our approach can overcome some barriers to the development of general models because: (1) it recognizes the natural variability among reference sites and allows a comparison of indicators to the range, not just the mean or median, of reference site values; (2) it recognizes that each restoration has different goals, and it allows each restoration site to be scored relative to its own, appropriately matched set of reference sites; (3) it recognizes that different indicators follow different trajectories, even in the same site, and it standardizes indicator scores to a common scale (percentiles), facilitating comparison among indicators; (4) it allows among-site or among-region comparison of restoration site performance, even when reference sites differ. However, a drawback is that data from a large number of reference sites are required in order to establish a reference distribution against which to compare each restored site.

Vegetation-based indicators in compensatory mitigation wetlands can achieve levels comparable to natural reference wetlands, but judgment of restoration performance depends on both the particular indicator and the reference used for comparison. We recommend the use of indicators based on plant species composition, rather than richness or diversity. We also recommend that restored sites be compared directly to a group of similar reference sites, while emphasizing that legal performance standards must vary depending on the particular reference set used and the specific restoration goals. Finally, the slow and often unpredictable trajectories observed for some vegetation-based indicators suggest that the efficacy of compensatory wetland mitigation is limited by the time that restored wetlands take to reach equivalency with natural wetlands and by the unpredictability of ecosystem dynamics. Therefore, typical three- to five-year monitoring time frames are unsuitable for many ecological indicators.

#### ACKNOWLEDGMENTS

Wetland sites were constructed by the Illinois Department of Transportation (IDOT). Restoration site monitoring and IDOT reference site surveys were performed by the Wetlands Group of the Illinois Natural History Survey (INHS) with funding from IDOT. Critical Trends Assessment Program (CTAP) site surveys were performed by CTAP personnel at the INHS. Funding for 2006 site surveys was provided by the Illinois Department of Transportation and by a National Great Rivers Research and Education Center grant to A. G. Endress and J. W. Matthews. Field assistance was provided by Arun Soni, Patrick Baldwin, Diana Flanagan, and Ariane Peralta. Changwoo Ahn, James Dalling, Jeffrey Brawn, Gregory McIsaac, John Taft, and two anonymous reviewers provided helpful comments.

#### LITERATURE CITED

- Anderson, K. J. 2007. Temporal patterns in rates of community change during succession. *American Naturalist* 169:780–793.
- Aronson, J., S. Dhillon, and E. Le Floch. 1995. On the need to select an ecosystem of reference, however imperfect: a reply to Pickett and Parker. *Restoration Ecology* 3:1–3.
- Aronson, J., and E. Le Floch. 1996. Vital landscape attributes: missing tools for restoration ecology. *Restoration Ecology* 4: 377–387.
- Balcombe, C. K., J. T. Anderson, R. H. Fortney, and W. S. Kordek. 2005a. Vegetation, invertebrate, and wildlife community rankings and habitat analysis of mitigation wetlands in West Virginia. *Wetlands Ecology and Management* 13: 517–530.
- Balcombe, C. K., J. T. Anderson, R. H. Fortney, J. S. Rentch, W. N. Grafton, and W. S. Kordek. 2005b. A comparison of plant communities in mitigation and reference wetlands in the mid-Appalachians. *Wetlands* 25:130–142.
- Bourdaghs, M., C. A. Johnston, and R. R. Regal. 2006. Properties and performance of the Floristic Quality Index in Great Lakes coastal wetlands. *Wetlands* 26:718–735.
- Bradshaw, A. D. 1984. Ecological principles and land reclamation practice. *Landscape Planning* 11:35–48.
- Brinson, M. M., and R. Rheinhardt. 1996. The role of reference wetlands in functional assessment and mitigation. *Ecological Applications* 6:69–76.
- Brooks, R. P., D. H. Wardrop, C. A. Cole, and D. A. Campbell. 2005. Are we purveyors of wetland homogeneity? A model of degradation and restoration to improve wetland mitigation performance. *Ecological Engineering* 24:331–340.
- Brown, J. S. 1994. Restoration ecology: living with the Prime Directive. Pages 355–380 in M. L. Bowles and C. J. Whelan, editors. *Restoration of endangered species: conceptual issues, planning and implementation*. Cambridge University Press, Cambridge, UK.
- Brown, S. C. 1999. Vegetation similarity and avifaunal food value of restored and natural marshes in northern New York. *Restoration Ecology* 7:56–68.
- Cairns, J., Jr., and J. R. Heckman. 1996. Restoration ecology: the state of an emerging field. *Annual Review of Energy and Environment* 21:167–189.
- Campbell, D. A., C. A. Cole, and R. P. Brooks. 2002. A comparison of created and natural wetlands in Pennsylvania, USA. *Wetlands Ecology and Management* 10:41–49.
- Carroll, C., C. Dassler, J. Ellis, G. Spyreas, J. B. Taft, and K. Robertson. 2002. Plant sampling protocols. Pages 11–19 in B. Molano-Flores, editor. *Critical Trends Assessment Program monitoring protocols*. Office of the Chief Technical Report 2002-2. Illinois Natural History Survey, Champaign, Illinois, USA.
- Choi, Y. D. 2004. Theories for ecological restoration in changing environment: toward “futuristic” restoration. *Ecological Research* 19:75–81.
- Cohen, M. J., S. Cartsenn, and C. R. Lane. 2004. Floristic quality indices for biotic assessment of depressional marsh condition in Florida. *Ecological Applications* 14:784–794.
- Confer, S. R., and W. A. Niering. 1992. Comparison of created and natural freshwater emergent wetlands in Connecticut (USA). *Wetlands Ecology and Management* 2:143–156.
- Craft, C., S. Broome, and C. Campbell. 2002. Fifteen years of vegetation and soil development after brackish-water marsh creation. *Restoration Ecology* 10:248–258.
- Craft, C., P. Megonigal, S. Broome, J. Stevenson, R. Freese, J. Cornell, L. Zheng, and J. Sacco. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. *Ecological Applications* 13:1417–1432.
- Cramer, V. A., R. J. Hobbs, and R. J. Standish. 2008. What's new about old fields? Land abandonment and ecosystem assembly. *Trends in Ecology and Evolution* 23:104–112.



- Cronk, J. K., and M. S. Fennessy. 2001. Wetland plants: biology and ecology. Lewis, Boca Raton, Florida, USA.
- Daubenmire, R. 1959. A canopy-coverage method of vegetational analysis. *Northwest Science* 33:43–64.
- Dawe, N. K., G. E. Bradfield, W. S. Boyd, D. E. C. Trethewey, and A. N. Zolbrod. 2000. Marsh creation in a northern Pacific estuary: Is thirteen years of monitoring vegetation dynamics enough? *Conservation Ecology* 4:12.
- DeBerry, D. A., and J. E. Perry. 2004. Primary succession in a created freshwater wetland. *Castanea* 69:185–193.
- DeKeyser, E. S., D. S. Kirby, and M. J. Ell. 2003. An index of plant community integrity: development of the methodology for assessing prairie wetland plant communities. *Ecological Indicators* 3:119–133.
- Donath, T. W., N. Hölzel, and A. Otte. 2003. The impact of site conditions and seed dispersal on restoration success in alluvial meadows. *Applied Vegetation Science* 6:13–22.
- Edwards, K. R., and C. E. Proffitt. 2003. Comparison of wetland structural characteristics between created and natural salt marshes in southwest Louisiana, USA. *Wetlands* 23:344–356.
- Engelman, L. 2005. Nonlinear models. Pages 541–594 in *SYSTAT 11 statistics II*. SYSTAT, Port Richmond, California, USA.
- Ervin, G. N., B. D. Herman, J. T. Bried, and D. C. Holly. 2006. Evaluating non-native species and wetland indicator status components of wetlands floristic assessment. *Wetlands* 26:1114–1129.
- Fennessy, S., J. J. Mack, A. Rokosch, M. Knapp, and M. Micacchion. 2004. Integrated wetland assessment program. Part 5: biogeochemical and hydrological investigations of natural and mitigation wetlands. Technical Report WET/2004-5. Ohio Environmental Protection Agency, Wetland Ecology Group, Division of Surface Water, Columbus, Ohio, USA.
- Foster, B. L., C. A. Murphy, K. R. Keller, T. A. Aschenbach, E. J. Questad, and K. Kindscher. 2007. Restoration of prairie community structure and ecosystem function in an abandoned hayfield: a sowing experiment. *Restoration Ecology* 15:652–661.
- Francis, C. M., M. J. W. Austen, J. M. Bowles, and W. B. Draper. 2000. Assessing floristic quality in southern Ontario woodlands. *Natural Areas Journal* 20:66–77.
- Galatowitsch, S. M. 2006. Restoring prairie pothole wetlands: Does the species pool concept offer decision-making guidance for re-vegetation? *Applied Vegetation Science* 9:261–270.
- Gutrich, J. J., K. J. Taylor, and M. S. Fennessy. 2009. Restoration of vegetation communities of created depressional marshes in Ohio and Colorado (USA): the importance of initial effort for mitigation success. *Ecological Engineering* 35:351–368.
- Halle, S., and M. Fattorini. 2004. Advances in restoration ecology: insights from aquatic and terrestrial ecosystems. Pages 10–33 in V. M. Temperton, R. J. Hobbs, T. Nuttle, and S. Halle, editors. *Assembly rules and restoration ecology: bridging the gap between theory and practice*. Island Press, Washington, D.C., USA.
- Herman, K. D., L. A. Masters, M. R. Penskar, A. A. Reznicek, G. S. Wilhelm, and W. R. Brodowicz. 1997. Floristic quality assessment: development and application in the state of Michigan (USA). *Natural Areas Journal* 17:265–279.
- Hobbs, R. J. 2007. Setting effective and realistic restoration goals: key directions for research. *Restoration Ecology* 15:354–357.
- Hobbs, R. J., and D. A. Norton. 1996. Towards a conceptual framework for restoration ecology. *Restoration Ecology* 4:93–110.
- Holl, K. D., and J. Cairns, Jr. 1994. Vegetational community development on reclaimed coal surface mines in Virginia. *Bulletin of the Torrey Botanical Club* 121:327–337.
- Holling, C. S. 1973. Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4:1–23.
- Illinois Department of Natural Resources. 2001. Critical trends in Illinois ecosystems. Illinois Department of Natural Resources, Office of Realty and Environmental Planning, Office of Scientific Research and Analysis, Springfield, Illinois, USA.
- Jarman, N. M., R. A. Dobberteen, B. Windmiller, and P. R. Lelito. 1991. Evaluation of created freshwater wetlands in Massachusetts. *Restoration and Management Notes* 9:26–29.
- Johnston, C. A., D. M. Ghioca, M. Tulbure, B. L. Bedford, M. Bourdaghs, C. B. Frieswyk, L. Vaccaro, and J. B. Zedler. 2008. Partitioning vegetation response to anthropogenic stress to develop multi-taxa wetland indicators. *Ecological Applications* 18:983–1001.
- Karr, J. R., and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, D.C., USA.
- Kentula, M. E. 2000. Perspectives on setting success criteria for wetland restoration. *Ecological Engineering* 15:199–209.
- Kentula, M. E., R. P. Brooks, S. E. Gwin, C. C. Holland, A. D. Sherman, and J. C. Sifneos. 1992. An approach to improving decision making in wetland restoration and creation. Island Press, Washington, D.C., USA.
- Kirkman, L. K., K. L. Coffey, R. J. Mitchell, and E. B. Moser. 2004. Ground cover recovery patterns and life-history traits: implications for restoration obstacles and opportunities in a species-rich savanna. *Journal of Ecology* 92:409–421.
- Klötzli, F., and A. P. Grootjans. 2001. Restoration of natural and semi-natural wetland systems in central Europe: progress and predictability of developments. *Restoration Ecology* 9:209–219.
- Lindig-Cisneros, R., J. Desmond, K. E. Boyer, and J. B. Zedler. 2003. Wetland restoration thresholds: Can a degradation transition be reversed with increased effort? *Ecological Applications* 13:193–205.
- Lockwood, J. L. 1997. An alternative to succession: assembly rules offer guide to restoration efforts. *Restoration and Management Notes* 15:45–50.
- Lopez, R. D., and M. S. Fennessy. 2002. Testing the floristic quality assessment index as an indicator of wetland condition. *Ecological Applications* 12:487–497.
- Lougheed, V. L., C. A. Parker, and R. J. Stevenson. 2007. Using non-linear responses of multiple taxonomic groups to establish criteria indicative of wetland biological condition. *Wetlands* 97:96–109.
- Mack, J. J., M. Micacchion, L. D. Augusta, and G. R. Sablak. 2000. Vegetation indices of biotic integrity (VIBI) for wetlands and calibration of the Ohio Rapid Assessment Method for Wetlands v. 5.0. Ohio Environmental Protection Agency, Columbus, Ohio, USA.
- MacMahon, J. A., and K. D. Holl. 2001. Ecological restoration: a key to conservation biology's future. Pages 245–269 in M. E. Soulé and G. H. Orians, editors. *Conservation biology: research priorities for the next decade*. Island Press, Washington, D.C., USA.
- Magee, T. K., T. L. Ernst, M. E. Kentula, and K. A. Dwire. 1999. Floristic composition of freshwater wetlands in an urbanizing environment. *Wetlands* 19:517–534.
- Matthews, J. W. 2003. Assessment of the floristic quality index for use in Illinois, USA, wetlands. *Natural Areas Journal* 23:53–60.
- Matthews, J. W., and A. G. Endress. 2008. Performance criteria, compliance success, and vegetation development in compensatory mitigation wetlands. *Environmental Management* 41:130–141.
- Matthews, J. W., P. A. Tessene, S. M. Wiesbrook, and B. W. Zercher. 2005. Effect of area and isolation on species richness and indices of floristic quality in Illinois, USA wetlands. *Wetlands* 25:607–615.



- Mayer, A. L., and M. Rietkerk. 2004. The dynamic regime concept for ecosystem management and restoration. *BioScience* 54:1013–1020.
- McIver, J., and L. Starr. 2001. Restoration of degraded lands in the interior Columbia River basin: passive vs. active approaches. *Forest Ecology and Management* 153:15–28.
- Miller, S. J., and D. H. Wardrop. 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. *Ecological Indicators* 6:313–326.
- Miller, S. J., D. H. Wardrop, W. M. Mahaney, and R. P. Brooks. 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. *Ecological Indicators* 6:290–312.
- Mitsch, W. J., X. Wu, R. W. Nairn, P. E. Weihe, N. Wang, R. Deal, and C. E. Boucher. 1998. Creating and restoring wetlands. *BioScience* 48:1019–1030.
- Mohlenbrock, R. H. 2002. Vascular flora of Illinois. Southern Illinois University Press, Carbondale, Illinois, USA.
- Moore, H. H., W. A. Niering, L. J. Marsicano, and M. Dowdell. 1999. Vegetation change in created emergent wetlands (1988–1996) in Connecticut (USA). *Wetlands Ecology and Management* 7:177–191.
- Morgan, P. A., and F. T. Short. 2002. Using functional trajectories to track constructed salt marsh development in the Great Bay Estuary, Maine/New Hampshire, U.S.A. *Restoration Ecology* 10:461–473.
- Mulhouse, J. M., and S. M. Galatowitsch. 2003. Revegetation of prairie pothole wetlands in the mid-continental US: twelve years post-reflooding. *Plant Ecology* 169:143–159.
- Mushet, D. M., N. H. Euliss, and T. L. Shaffer. 2002. Floristic quality assessment of one natural and three restored wetland complexes in North Dakota, USA. *Wetlands* 22:126–138.
- National Research Council. 2001. Compensating for wetland losses under the Clean Water Act. National Academy Press, Washington, D.C., USA.
- Nedland, T. S., A. Wolf, and T. Reed. 2007. A reexamination of restored wetlands in Manitowoc County, Wisconsin. *Wetlands* 27:999–1015.
- Nichols, J., J. Perry, and D. DeBerry. 2006. Using a floristic quality assessment technique to evaluate plant community integrity of forested wetlands in southeastern Virginia. *Natural Areas Journal* 26:360–369.
- Noon, K. F. 1996. A model of created wetland primary succession. *Landscape and Urban Planning* 34:97–123.
- Odum, E. P. 1985. Trends expected in stressed ecosystems. *BioScience* 35:419–422.
- Pickett, S. T. A., and V. T. Parker. 1994. Avoiding old pitfalls: opportunities in a new discipline. *Restoration Ecology* 2:75–79.
- Race, M. S. 1985. Critique of present wetlands mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. *Environmental Management* 9:71–82.
- Reed, P. B., Jr. 1988. National list of plant species that occur in wetlands: Illinois. NERC-88/18.13. U.S. Department of the Interior, Fish and Wildlife Service, National Wetlands Inventory, Washington, D.C., USA.
- Reinartz, J. A., and E. L. Warne. 1993. Development of vegetation in small created wetlands in southeastern Wisconsin. *Wetlands* 13:153–164.
- Reiss, K. C. 2006. Florida wetland condition index for depressional forested wetlands. *Ecological Indicators* 6:337–352.
- Rheinhardt, R. D., R. C. Rheinhardt, M. M. Brinson, and K. E. Faser, Jr. 1999. Application of reference data for assessing and restoring headwater ecosystems. *Restoration Ecology* 7:241–251.
- Rooney, T. P., and D. A. Rodgers. 2002. The modified floristic quality index. *Natural Areas Journal* 22:340–344.
- Seabloom, E. W., and A. G. van der Valk. 2003. Plant diversity, composition, and invasion of restored and natural prairie pothole wetlands: implications for restoration. *Wetlands* 23:1–12.
- Sheil, D., and D. F. R. P. Burslem. 2003. Disturbing hypotheses in tropical forests. *Trends in Ecology and Evolution* 18:18–26.
- Shuwen, W., Q. Pei, L. Yang, and L. Xi-Ping. 2001. Wetland creation for rare waterfowl conservation: a project designed according to the principles of ecological succession. *Ecological Engineering* 18:115–120.
- Simenstad, C. A., D. Reed, and M. Ford. 2006. When is restoration not? Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. *Ecological Engineering* 26:27–39.
- Simenstad, C. A., and R. M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Tu estuarine wetland. *Ecological Applications* 6:38–56.
- Spieles, D. J., M. Coneybeer, and J. Horn. 2006. Community structure and quality after 10 years in two central Ohio mitigation bank wetlands. *Environmental Management* 38:837–852.
- Streever, W. J. 1999. Examples of performance standards for wetland creation and restoration in Section 404 permits and an approach to developing performance standards. TN WRP WG-RS-3.3. U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi, USA.
- Streever, W. J. 2000. *Spartina alterniflora* marshes on dredged material: a critical review of the ongoing debate over success. *Wetlands Ecology and Management* 8:295–316.
- Stylinski, C. D., and E. B. Allen. 1999. Lack of native species recovery following severe exotic disturbance in southern California shrublands. *Journal of Applied Ecology* 36:544–554.
- Swink, F., and G. Wilhelm. 1994. Plants of the Chicago region. Fourth edition. Morton Arboretum, Lisle, Illinois, USA.
- Taft, J. B., C. Hauser, and K. R. Robertson. 2006. Estimating floristic integrity in tallgrass prairie. *Biological Conservation* 131:42–51.
- Taft, J. B., G. S. Wilhelm, D. M. Ladd, and L. A. Masters. 1997. Floristic quality assessment for vegetation in Illinois, a method for assessing vegetation integrity. *Erigenia* 15:3–95.
- Trowbridge, W. B. 2007. The role of stochasticity and priority effects in floodplain restoration. *Ecological Applications* 17:1312–1324.
- U.S. Army Corps of Engineers. 1987. Corps of Engineers wetlands delineation manual. Technical Report Y-87-1. Environmental Laboratory, U.S. Army Corps of Engineers Waterways Experimental Station, Vicksburg, Mississippi, USA.
- U.S. Environmental Protection Agency. 2002. Methods for evaluating wetland condition: using vegetation to assess environmental conditions in wetlands. EPA-822-R-022-020. U.S. Environmental Protection Agency, Office of Water, Washington, D.C., USA.
- Weinstein, M. P., J. M. Teal, J. H. Balletto, and K. H. Strait. 2001. Restoration principles emerging from one of the world's largest tidal marsh restoration projects. *Wetlands Ecology and Management* 9:387–407.
- Whigham, D., M. Pittek, K. H. Hofmockel, T. Jordan, and A. L. Pepin. 2002. Biomass and nutrient dynamics in restored wetlands on the outer coastal plain of Maryland, USA. *Wetlands* 22:562–574.
- Whisenant, S. G. 1999. Repairing damaged wildlands: a process-oriented, landscape-scale approach. Cambridge University Press, Cambridge, UK.
- White, P. S., and J. L. Walker. 1997. Approximating nature's variation: selecting and using reference information in restoration ecology. *Restoration Ecology* 5:338–349.
- Wilkins, S., D. A. Keith, and P. Adam. 2003. Measuring success: evaluating the restoration of grassy eucalypt

- woodland on the Cumberland Plain, Sydney, Australia. *Restoration Ecology* 11:489–503.
- Wissinger, S. A., S. G. Ingmire, and J. L. Bogo. 2001. Plant and invertebrate communities as indicators of success for wetlands restored for wildlife. Pages 207–236 in R. B. Rader, D. P. Batzer, and S. A. Wissinger, editors. *Bioassessment and management of North American freshwater wetlands*. John Wiley and Sons, New York, New York, USA.
- Yates, C. J., and R. J. Hobbs. 1997. Woodland restoration in the Western Australian wheatbelt: a conceptual framework using a state and transition model. *Restoration Ecology* 5: 28–35.
- Young, T. P., J. M. Chase, and R. T. Huddleston. 2001. Community succession and assembly: comparing, contrasting and combining paradigms in the context of ecological restoration. *Ecological Restoration* 19:5–18.
- Zedler, J. B. 1996. Ecological issues in wetland mitigation: an introduction to the forum. *Ecological Applications* 6:33–37.
- Zedler, J. B., and J. C. Callaway. 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? *Restoration Ecology* 7:69–73.
- Zedler, J. B., and J. C. Callaway. 2000. Evaluating the progress of engineered tidal wetlands. *Ecological Engineering* 15:211–225.

#### APPENDIX A

Detailed information on study sites (*Ecological Archives* A019-087-A1).

#### APPENDIX B

Additional details on trajectory models (*Ecological Archives* A019-087-A2).

#### APPENDIX C

Additional analyses of covariance, addressing sampling differences among sites (*Ecological Archives* A019-087-A3).

#### APPENDIX D

Restoration trajectories in individual wetlands (*Ecological Archives* A019-087-A4).

#### APPENDIX E

Median values of estimated parameters from trajectory models (*Ecological Archives* A019-087-A5).

#### APPENDIX F

Average trajectories of vegetation-based indicators in restored wetlands (*Ecological Archives* A019-087-A6).

# Ecological Archives A019-087-A4

Jeffrey W. Matthews, Greg Spyreas, and Anton G. Endress. 2009. Trajectories of vegetation-based indicators used to assess wetland restoration progress. *Ecological Applications* 19:2093–2107.

## Appendix D. Restoration trajectories in individual wetlands.

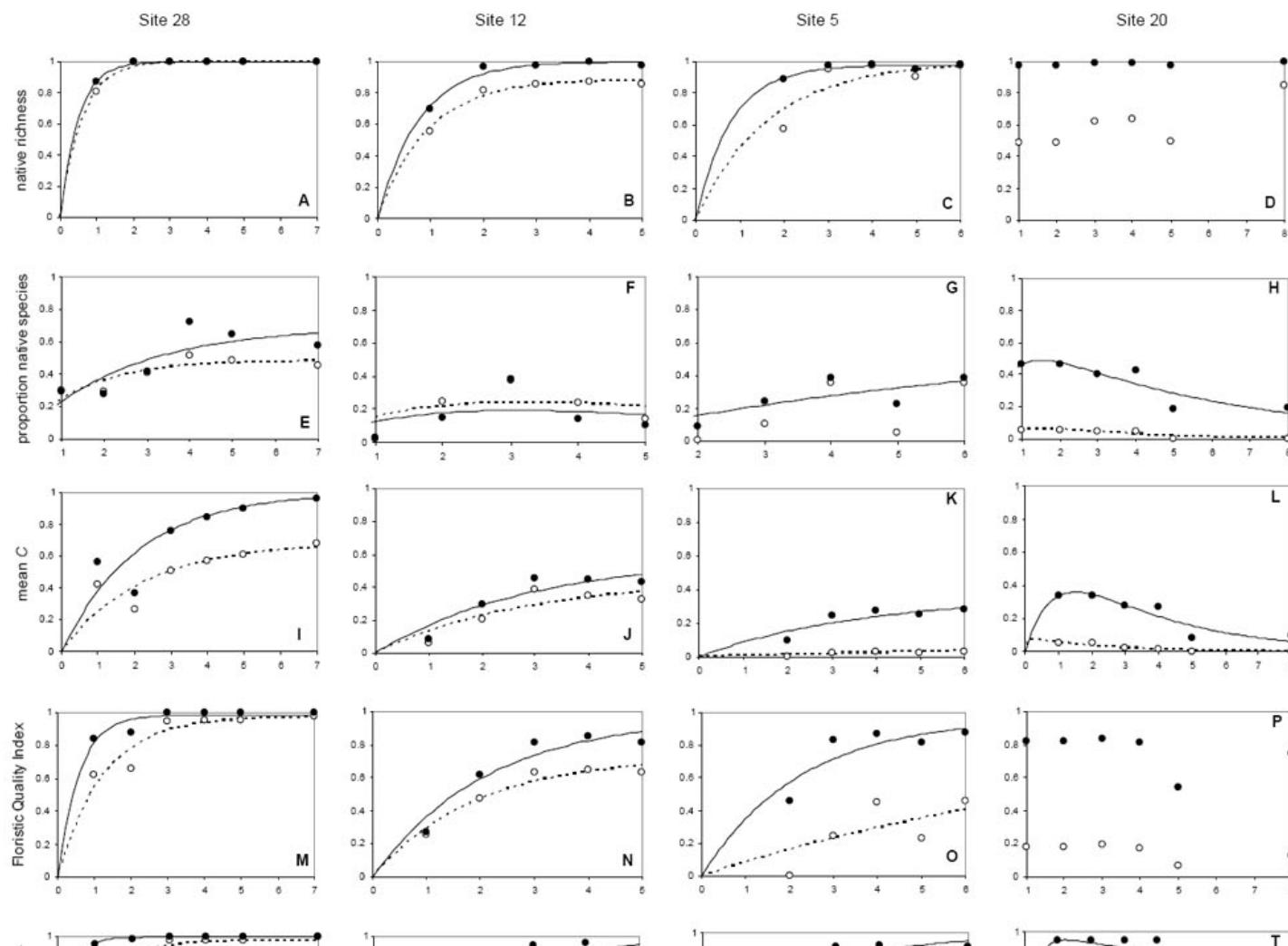
This Appendix provides a more in-depth examination of restoration trajectories from individual wetlands. Restoration trajectories of eight vegetation-based indicators of restoration progress from four sites, including two forested wetlands and two herbaceous wetlands, are shown in Fig. D1. Site 28 is a created marsh in northern Illinois (see [Appendix A](#) for site information). The site was excavated and planted with several marsh and wet prairie plant species. Relative to reference wetlands from northern Illinois, Site 28 achieved high richness of native species (Fig. D1A), *Carex* species (Fig. D1U) and conservative species (Fig. D1Q), as well as a high Floristic Quality Index (FQI; Fig. D1M) and mean Coefficient of Conservatism (mean C, Fig. D1I). Even in this relatively successful restoration, however, proportion native species (Fig. D1E) and total importance value of native species (Fig. D1AC) remained close to the median level in reference wetlands, suggesting that the representation of native relative to exotic plant species may be one the most difficult characteristics of the plant community to restore.

Site 12, in contrast, was less successful. This created marsh was constructed within a highway interchange in central Illinois, and was rapidly invaded by the non-native *Typha angustifolia*, which after five years made up 22% of the plant cover at the site. Nevertheless, indicators based on species richness, including native richness (Fig. D1B), FQI (Fig. D1N) and conservative richness (Fig. D1R), generally increased over time and by the end of five years had achieved levels higher than a majority of reference wetlands. Mean C in this site leveled off below the median level in reference sites (Fig. D1J). Proportion and importance of native species initially increased, but then declined over time (Fig. D1F, D1AD).

Site 5 was a restored floodplain forest on former agricultural land in southern Illinois. As in Site 28, most indicators increased over time, and indicators based on species richness achieved levels exceeding those in a majority of reference, southern Illinois, forested wetlands (Fig. D1C, D1S, D1W). Proportion native species and Mean C, however, remained low relative to reference sites (Fig. D1G, D1K).

Site 20 was also a restored floodplain forest on former agricultural land, but was located in northwestern Illinois. This site was heavily invaded by the aggressive, exotic grass *Phalaris arundinacea*, which comprised 42% of the herbaceous cover after eight years. In contrast to the other sites, most vegetation-based indicators declined over time or showed no discernable trends (Fig. D1). Even in this site, however, native richness (Fig. D1D), conservative richness (Fig. D1T) and FQI (Fig. D1P) were high relative to at least one set of reference wetlands.

Across all four sites, proportion perennial species, where it increased over time, increased only slowly relative to reference sites (Fig. D1Y, D1Z, D1AA, D1AB). This was especially the case for the restored forested wetlands, suggesting that this characteristic of natural forested wetlands recovers very slowly.



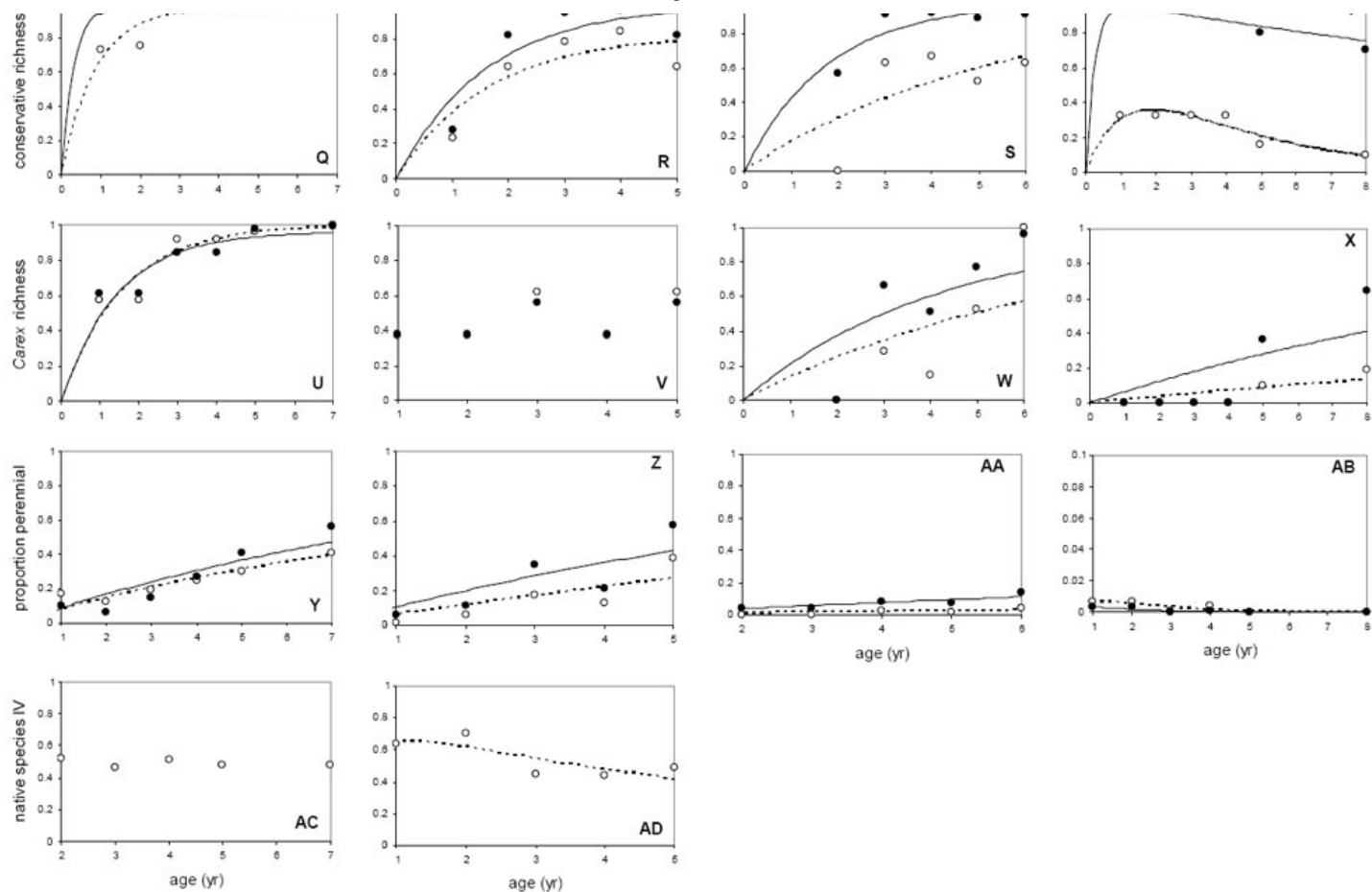


FIG. D1. Restoration trajectories of eight vegetation-based indicators (rows), expressed as percentiles relative to reference site distributions, in four restored wetlands (columns). Filled symbols represent percentiles relative to Illinois Department of Transportation (IDOT) reference wetlands and open symbols represent percentiles relative to Critical Trends Assessment Program (CTAP) reference wetlands. Solid curves are best-fit trajectories relative to IDOT references and dashed curves are best-fit trajectories relative to CTAP references. Trajectory curves are not shown where  $R^2 < 0.5$ . Site numbers refer to [Appendix A](#).

The oldest forested wetland among our sampled wetlands was Site 16, a restored floodplain forest on former agricultural land. Even after 14 years of site development, proportion perennial species remained below the median level in reference forests (Fig. D2D). *Carex* richness, on average, also tended to increase slowly relative to reference wetlands (see Fig. 3), but in many sites, including Site 16 (Fig. D2C), it eventually reached levels higher than in most reference sites. After 14 years, the importance of native species remained low relative to reference forests (Fig. D2B).

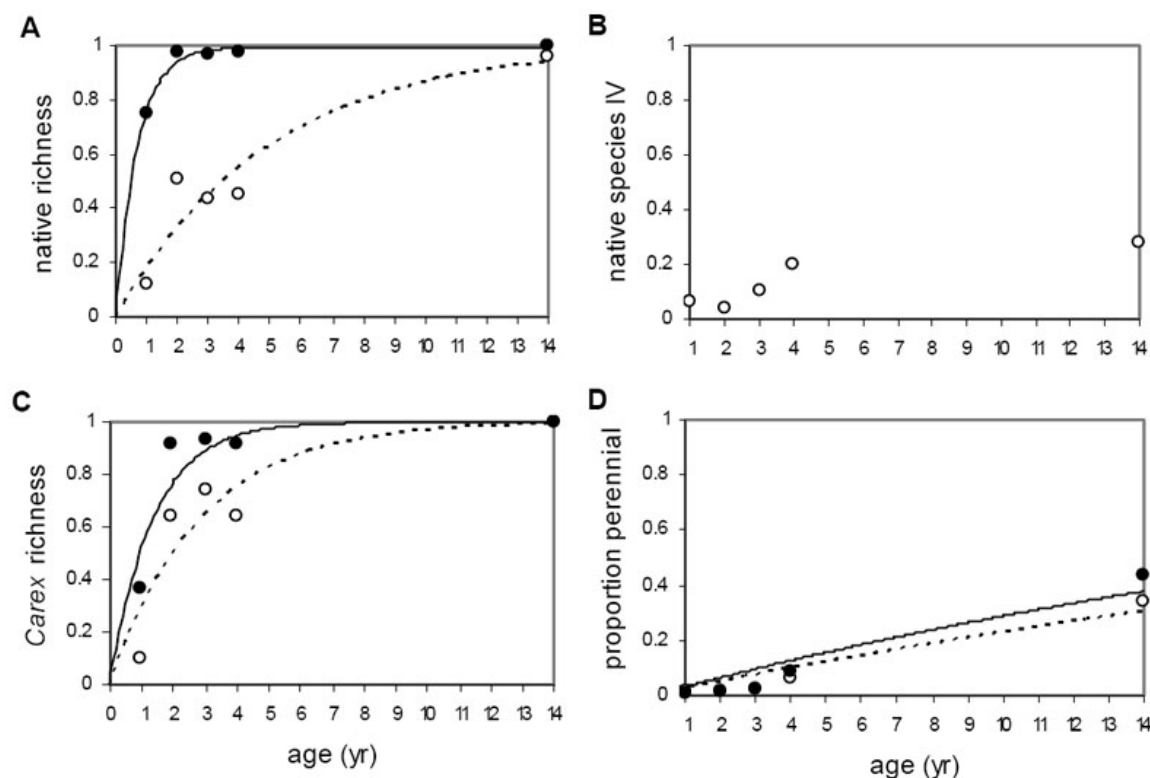


FIG. D2. Restoration trajectories of four vegetation-based indicators, expressed as percentiles relative to reference site distributions, in Site 16, a restored floodplain forest on the LaMoine River in western Illinois. Filled symbols represent percentiles relative to IDOT reference wetlands and open symbols represent percentiles relative to CTAP reference wetlands. Solid curves are best-fit trajectories relative to IDOT references and dashed curves are best-fit trajectories relative to CTAP references.

Indicators in some sites failed to follow trajectories that were adequately described by a negative exponential increase or a peaked double exponential function. For example, in Site 17, a restored floodplain forest that became heavily invaded by *Phalaris arundinacea*, native richness fluctuated widely over 9 years (Fig. D3A). This site was frequently disturbed by high energy flood events, which led to scouring, debris accumulation, sediment deposition, and damage to planted trees. Other indicators in this site increased slowly over time and eventually declined, but trajectories were not well described by a double exponential function (Fig. D3).

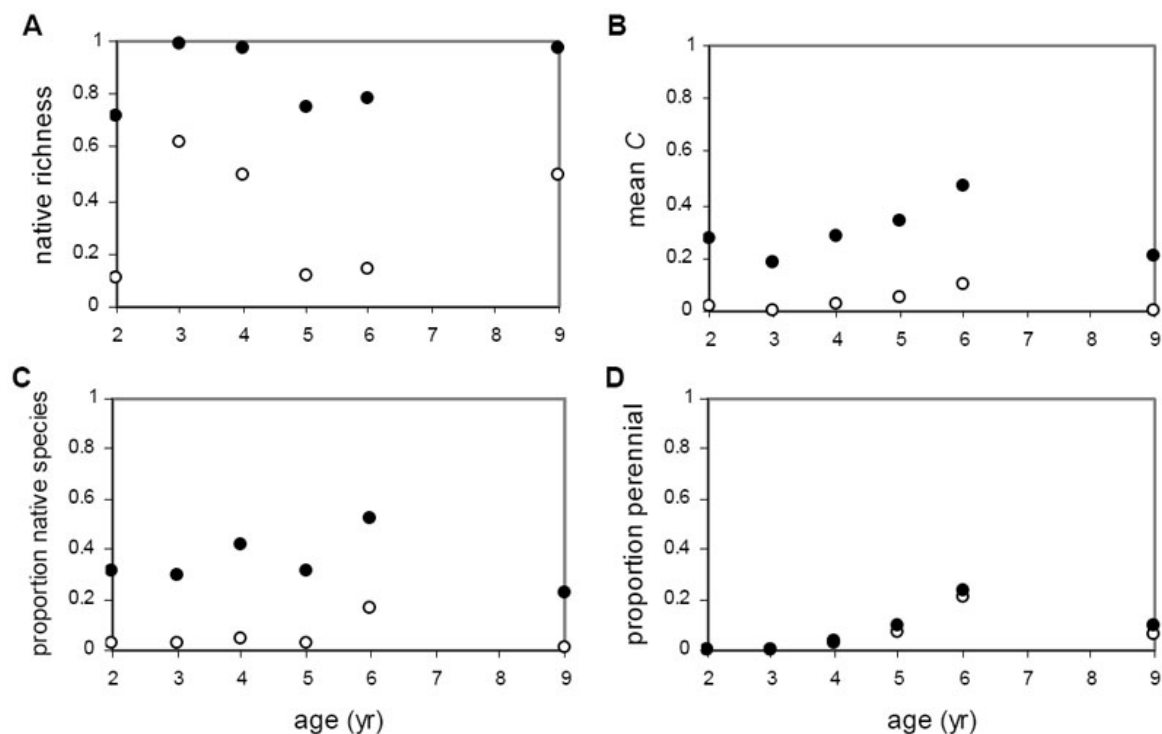




FIG. D3. Restoration trajectories of four vegetation-based indicators, expressed as percentiles relative to reference site distributions, in Site 17, a restored floodplain forest on the Mackinaw River in central Illinois. Filled symbols represent percentiles relative to IDOT reference wetlands and open symbols represent percentiles relative to CTAP reference wetlands. Solid curves are best-fit trajectories relative to IDOT references and dashed curves are best-fit trajectories relative to CTAP references.

Although general patterns emerge when viewing the temporal trajectories of vegetation-based indicators across a large number of wetlands (see Fig. 2, Table 3) individual site trajectories reveal that not all sites follow the general patterns. Some sites follow unique trajectories attributable to external factors such as climatic events. In Site 14, a restored floodplain forest, a flood event breached a nearby levee during the second year after site construction, trapping water on the site for an extended duration and killing most vegetation. Species richness declined, but quickly recovered (Fig. D4A). Although mean *C* (Fig. D4B) and proportion perennials (Fig. D4D) remained low relative to reference wetlands, several water-dispersed *Carex* species were observed on the site in the years following the flood event, and *Carex* richness was higher than all reference wetlands by the fifth year (Fig. D4C).

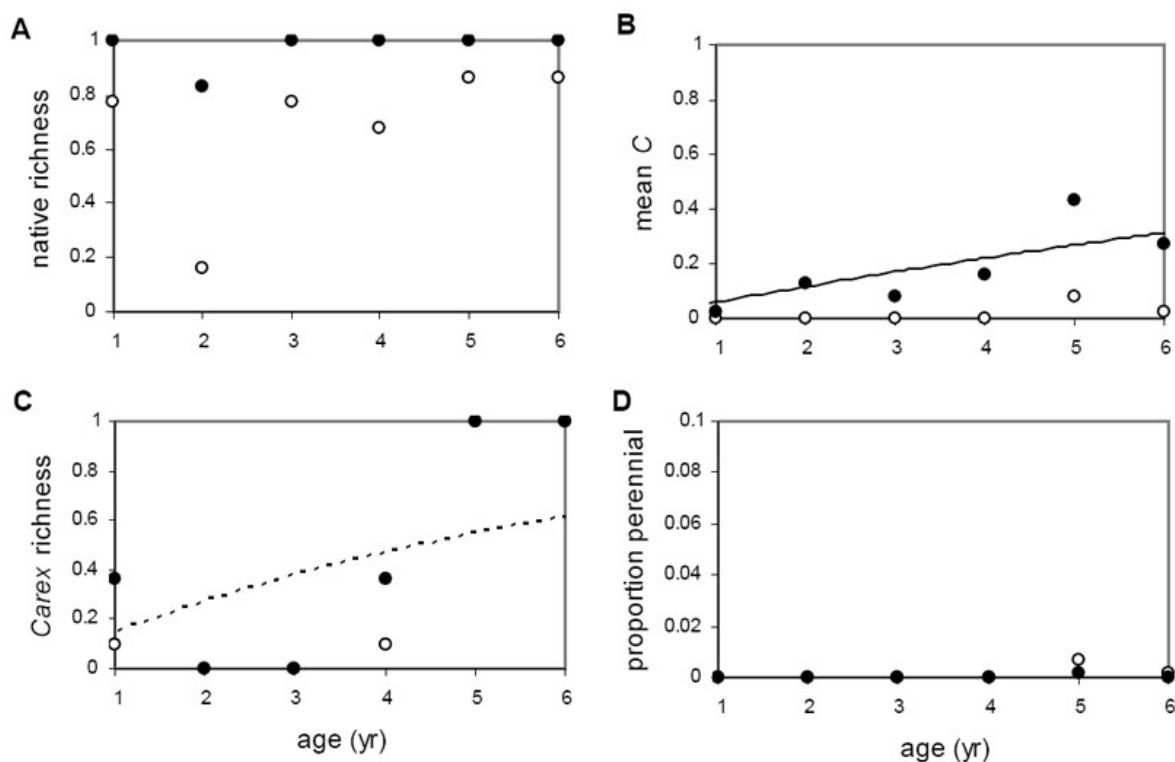


FIG. D4. Restoration trajectories of four vegetation-based indicators, expressed as percentiles relative to reference site distributions, in Site 14, a restored floodplain forest on the Sangamon River floodplain near Springfield, Illinois. Filled symbols represent percentiles relative to IDOT reference wetlands and open symbols represent percentiles relative to CTAP reference wetlands.

[\[Back to A019-087\]](#)

# Performance Criteria, Compliance Success, and Vegetation Development in Compensatory Mitigation Wetlands

Jeffrey W. Matthews · Anton G. Endress

Received: 8 February 2007 / Accepted: 23 May 2007 / Published online: 5 August 2007  
© Springer Science+Business Media, LLC 2007

**Abstract** The US Army Corps of Engineers often requires wetland creation or restoration as compensation for wetlands damaged during development. These wetlands are typically monitored postconstruction to determine the level of compliance with respect to site-specific performance standards. However, defining appropriate goals and measuring success of restorations has proven difficult. We reviewed monitoring information for 76 wetlands constructed between 1992 and 2002 to summarize the performance criteria used to measure progress, assess compliance with those criteria, and, finally, to evaluate the appropriateness of those criteria. Goals were overwhelmingly focused on plant communities. Attributes used to assess the quality of restored plant communities, including percent native species and the Floristic Quality Index, increased over time but were apparently unrelated to the number of species planted. Compliance frequencies varied depending on site goals; sites often failed to comply with criteria related to survival of planted vegetation or requirements that dominant plant species should not be exotic or weedy, whereas criteria related to the establishment of cover by vegetation or by wetland-dependent plants were often met. Judgment of a site's success or failure was largely a function of the goals set for the site. Some performance criteria were too lenient to be of value in distinguishing failed from successful sites, whereas other criteria were unachievable

without more intensive site management. More appropriate goals could be devised for restored wetlands by basing performance standards on past performance of similar restorations, identifying consistent temporal trends in attributes of restored sites, and using natural wetlands as references.

**Keywords** Creation · Restoration · Clean Water Act, Section 404 · Performance standards · Plant community development · Floristic Quality Index

Approximately 52% of the original wetland area in the conterminous United States (Dahl 2000) has been converted to other land uses. The extensive loss of wetlands, in addition to a growing recognition of the value of wetlands to society, led federal and state governments to switch from policies subsidizing wetland conversion to policies with the goal of preventing further loss (Brown and Lant 1999). The major piece of federal legislation regulating wetland conversion is Section 404 of the Clean Water Act, which is enforced by the US Army Corps of Engineers. Permits for regulated activities affecting wetlands often require mitigation in the form of compensation through restoration, creation, or enhancement of wetlands elsewhere. These compensatory mitigation wetlands are typically monitored for a period of 3–5 years to determine whether they meet a set of site-specific performance standards approved by the Army Corps of Engineers (NRC 2001).

The policy of compensatory mitigation assumes that both the structure and function of destroyed wetlands can be predictably recreated, an assumption questioned by several authors (e.g., Niering 1987; Race 1985; Zedler 1996), and that 5 years of monitoring is long enough to assess the progress of compensatory mitigation wetlands (Mitsch and Wilson 1996; Zedler and Callaway 1999).

---

J. W. Matthews (✉)  
Illinois Natural History Survey, 1816 South Oak Street,  
Champaign, IL 61820, USA  
e-mail: matthews@inhs.uiuc.edu

A. G. Endress  
Department of Natural Resources and Environmental Sciences,  
1101 West Peabody Drive, Urbana, IL 61801, USA

Restored and created wetlands have often failed (NRC 2001), and in addition to the progressive degradation of existing natural wetlands, this has led to the continuing loss of both wetland area and wetland ecosystem function at regional scales (Whigham 1999; Zedler and Callaway 1999). Judging whether a restored or created wetland is a failure or a success, however, is a difficult task.

Judgment of success or failure in compensatory mitigation wetlands is ideally based on goals established a priori. A mitigation project is considered a “compliance success” if it meets all goals specified in a permit or agreement among the parties involved (Kentula 2000; Zedler and Callaway 2000). A site can fail to achieve compliance for two reasons: The goals were too stringent to be realistically achievable or the site is not successful as a functioning wetland. On the other hand, even if a site does meet all compliance goals, it is not necessarily a successful ecological restoration because the goals might have been too modest or otherwise inappropriate.

In a regulatory context it is important to establish measurable, realistically achievable performance standards in order to judge a site’s progress with respect to stated project goals. Performance standards, measurable thresholds used to judge compliance or lack thereof, are often included as conditions in mitigation permits (Streever 1999) and determine how a site will be monitored (Ehrenfeld 2000). Performance standards are not uniform among wetland permits (Streever 1999) and are often proposed by a permittee’s consultants with approval from the Army Corps of Engineers (Sudol and Ambrose 2002). Even within similar wetland types, there can be large inconsistencies in actual targets (Breux and Serefidin 1999), suggesting that the performance standards are often set arbitrarily, without reference to similar natural or restored wetlands. Furthermore, performance standards are often unclear, do not set measurable targets, or are poor indicators of site performance or wetland functions (Cole 2002; NRC 2001).

A lack of regional-scale studies of the past performance of compensatory mitigation wetlands has limited the ability of wetland planners to set realistic goals for new sites. Most published assessments of wetland permitting programs have focused very broadly on whether compensatory wetlands were actually created or monitored, and although informative, they necessarily lack details about site performance that might prove useful as models for setting goals. Other studies have focused in more detail on evaluating success in only one or a few sites, but the generalities that can be drawn from such site-specific studies are limited. Furthermore, there has been little discussion of whether the performance standards used to measure site progress are reasonable (but see Breux and Serefidin 1999; Cole 2002).

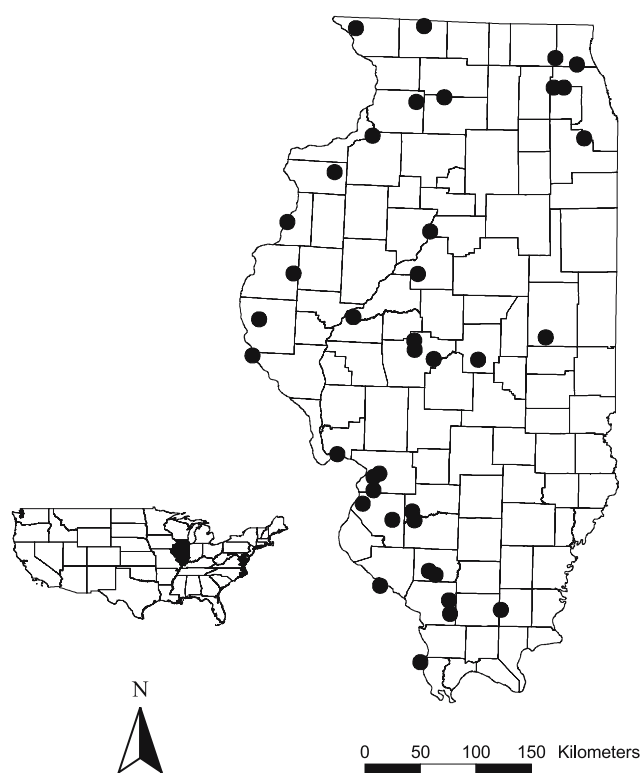
Wetland conversion has been especially extensive in the midwestern United States. In Illinois; an estimated 90% of

original wetland area has been lost (Suloway and Hubbell 1994). Therefore, protection of existing wetlands and effective compensation for unavoidable losses are of great consequence in the state. The Illinois Department of Transportation (IDOT) has created and restored wetlands throughout Illinois to compensate for natural wetlands damaged during road projects. After construction, these sites are monitored annually for a period of up to 5 years. Data derived from the original site-monitoring reports provide a unique opportunity to examine, in detail, the goals and performance levels of a large number of compensatory mitigation wetlands. The specific objectives of this study were to (1) summarize the goals and performance standards set for these sites, (2) determine compliance frequencies for each goal, (3) determine whether performance levels change over time and how they compare to typical performance standards in order to judge whether standards were achievable within the time frame of monitoring, and (4) determine whether planting a greater number of herbaceous species increased performance levels.

## Methods

Seventy-six compensatory mitigation wetlands, constructed between 1991 and 2002 in 38 separate project areas, were monitored annually, in late summer, for 1–5 years by scientists from the Illinois Natural History Survey (INHS). Late summer sampling maximizes the number of identifiable plant species in wetlands (Matthews 2003) but might underestimate richness of early-flowering species (e.g., *Carex* spp.). Sites included both emergent and forested wetlands. Target areas for wetland construction ranged from 0.02 to 10.7 ha (median: 0.6 ha). Sites were located throughout Illinois, from  $\sim 37^{\circ}17'$  to  $42^{\circ}27'$  latitude and  $87^{\circ}53'$  to  $91^{\circ}20'$  longitude (Figure 1). Eight sites were monitored for a single year, 17 were monitored for 2 years, 9 were monitored for 3 years, 18 were monitored for 4 years, and 24 were monitored for 5 years. For analyses focusing on the outcome of restoration, we use only the final year of monitoring data for each site, as it represents the site at its most mature state and is the final year of data upon which regulatory agencies can base decisions about site success. For analyses of trends over time, we use only sites with at least 4 years of data.

At each site, in each year, the area of the site meeting the jurisdictional criteria of a wetland was delineated; a site was determined to be a wetland if it contained dominant hydrophytic vegetation (Reed 1988), hydric soils, and wetland hydrology as described by the US Army Corps of Engineers Wetlands Delineation Manual (USACE 1987). A complete plant species list was compiled annually during a thorough search of the entire site. Additional monitoring was



**Fig. 1** Locations, within Illinois, of compensatory mitigation projects included in this study

performed based on site-specific performance standards. For example, trees were counted in restored floodplain forests when performance criteria required a measurement of the number of planted trees surviving by the end of monitoring. Goals and performance standards were specified by the permitting agency or were developed by the permittee's consultants or personnel from the INHS, with approval by the permitting agency. Several of these sites were also monitored by hydrologists from the Illinois State Geological Survey to determine areal extent of wetland hydrology.

Some sites had performance standards that required measurement of herbaceous species cover. In these sites, in addition to annual whole-site surveys that were used to generate species lists, vegetation was quantitatively sampled in square quadrats (1 m<sup>2</sup> or 0.25 m<sup>2</sup>) placed systematically along transects. All vascular plant species in each quadrat were recorded, and each species was assigned a cover class (<1%, 1–5%, 6–25%, 26–50%, 51–75%, 76–95%, or 96–100%), an estimate of the amount of area within the sample quadrat that is covered by that species (Daubenmire 1959). Cover class data were used to calculate frequency (percent of quadrats in which a species is present), relative frequency, average cover per quadrat, relative cover, and importance value (the sum of relative frequency and relative cover, divided by 2) for each sampled species at each site. Species were arranged by

importance value in decreasing order, and importance values were sequentially summed, starting with the most prevalent species, until the total reached 50. Those species included in the summation were considered dominant species. Additionally, any species having >20% of the total vegetation cover was considered dominant (FICWD 1989). Number of quadrats and transects, distance between transects, and distance between quadrats varied among sites, but were consistent within sites from year to year. Although the monitoring protocol varied among sites, the quantitative sampling employed should have yielded a good representation of the relative importance of species within a site. However, because of the sampling differences among sites, we avoided direct among-site comparisons based on quadrat sampling, and we restricted other analyses based on these data to a few variables (relative cover by all hydrophytic species combined and presence or absence of exotic and weedy dominant species) that were unlikely to be affected by the sampling differences.

Some sites had performance standards requiring the calculation of the mean coefficient of conservatism ( $\bar{C}$ ) and Floristic Quality Index (FQI). Swink and Wilhelm (1994) developed these indexes as a means of rapidly assessing natural areas in the region around Chicago, Illinois, and Taft and others (1997) expanded the indexes for use throughout Illinois. Each native plant species was assigned a "coefficient of conservatism" ( $C$ ), a subjective rating of species fidelity to undegraded natural communities, varying from 0 to 10, with higher values assigned to species less tolerant of degradation. The Floristic Quality Index is computed as  $FQI = \bar{C}\sqrt{S}$ , where  $\bar{C}$  is the mean coefficient of conservatism for all native plant species at a site and  $S$  is the total number of native plant species at the site.

The following approach was used to summarize the goals set for the wetlands and determine compliance frequencies: (1) Goals specified in site-monitoring reports were first categorized to determine what types of goals were set; (2) for each category of goals, we determined how success with respect to that goal has been evaluated (i.e., what specific performance criteria were established); (3) for each major category of goals, we determined what proportion of sites achieved the site-specific performance standards. A site was considered successful with respect to a given goal if it met all stated performance criteria related to that goal in its final year of monitoring. Sites were categorized as successful at meeting all goals, partially successful, or completely unsuccessful, and analyses of variance (ANOVA), followed by Scheffe tests, were used to determine if these categories differed in mean wetland indicator status (Reed 1988) or in the mean number of goals originally set for sites.

Compliance success frequencies are of interest from a regulatory perspective. However, actual target levels for a

given goal might vary widely among sites, so that frequencies of success, based solely on yes-or-no judgments of compliance, provide little insight into the overall level of performance across sites. Instead, performance of sites should be compared to a common standard. Therefore, the performance level for each site, in its final year of monitoring, was quantified for each category of goals, and the distribution of performance levels across all sites was compared to a common standard: the “typical” performance requirement. The typical performance requirement for a given goal was taken to be the mode of the distribution of performance requirements. For example, 53 sites had performance requirements establishing that some minimum percentage of the plant species at the site should be the desirable species. However, the definition of “desirable species,” as well as the percent required for success, varied among sites, making among-site comparison difficult. Therefore, we compared the distribution of percent native, nonweedy perennials across all sites (regardless of whether they specified this particular goal) to the most commonly required performance level for that goal (namely 50% native, nonweedy perennials). A similar approach was used to assess performance with respect to other site goals. Comparing the typical performance requirement to the distribution of success levels allowed for a qualitative assessment of whether the typical requirement was appropriate (i.e., not unachievably ambitious nor overly modest). Because sites were monitored annually for periods varying from 1 to 5 years, sites varied in age at their final year of monitoring.

For this study, native weeds were defined as native species that: 1) were listed as economic, noxious or colonizing weeds (Iverson and others 1999), or are typical of habitats that include waste areas, bare or disturbed ground (Mollenbrock 2002); and 2) have coefficient of conservatism values of two or less (Taft and others 1997). Native status of species was based on Taft and others (2002). *Phalaris arundinacea*, *Typha x glauca*, and *Phragmites australis* were considered non-native to this region due to likely hybridization between native and non-native species or genotypes (Galatowitsch and others 1999; Saltonstall 2002).

We employed a distribution-free, randomization procedure, analogous to a repeated measures one-way ANOVA (Edgington 1995), to determine whether performance levels for common goals (planted tree survival; percent of planted herb species persisting; proportion of flora comprised of native, nonweedy, and perennial species; relative cover by hydrophytic species; number of non-native and weedy dominant species; and FQI) change over the first 4 years of monitoring. This analysis was limited to sites with at least 4 consecutive years of monitoring ( $n = 42$ ). Performance levels are expected to vary widely among sites, and we were primarily interested in whether there were consistent time

trends in performance levels within sites over time, rather than in among-site variability. Data were therefore permuted, 10,000 times, among years, within sites, using Resampling Procedures v. 1.3. The observed  $F$ -statistic for the effect of year was then compared to the distribution of  $F$ -statistics generated under a null hypothesis of no effect of year in order to calculate a  $P$ -value. Significance levels were adjusted using Bonferroni's method ( $\alpha' = \alpha/k$ , where  $k$  is the number of tests and  $\alpha$  was set to 0.05).

Simple linear regressions were used to determine whether native species richness, FQI, mean coefficient of conservatism, or proportion of a site's flora comprised of desirable species increased with the number of herbaceous species planted at a site. Data from only the final year of site monitoring were used in these analyses. The number of species planted was  $\log_{10}$ -transformed prior to analyses. Some project areas consisted of multiple wetland sites in close proximity. Therefore, to ensure statistical independence, a single, randomly chosen site from each project area was included in regressions. Data were available on number of planted species for 33 independent sites.

## Results

Goals and performance standards were grouped into 14 categories that were overwhelmingly focused on the plant community (see the Appendix). The most common goal, not unexpectedly, was to create or restore jurisdictional wetland as defined by the US Army Corps of Engineers (USACE 1987). Other commonly stated goals included ensuring adequate survival of planted species, specifications that the dominant species should not be exotic or weedy, specifications that a minimum proportion of the plant species at a site must be native, nonweedy, and/or perennial, and requirements for a minimum cover by vegetation or by hydrophytic plant species. With the exception of one site with a goal related to sediment accumulation rate, criteria were based on structural attributes and therefore did not require measures of dynamic processes.

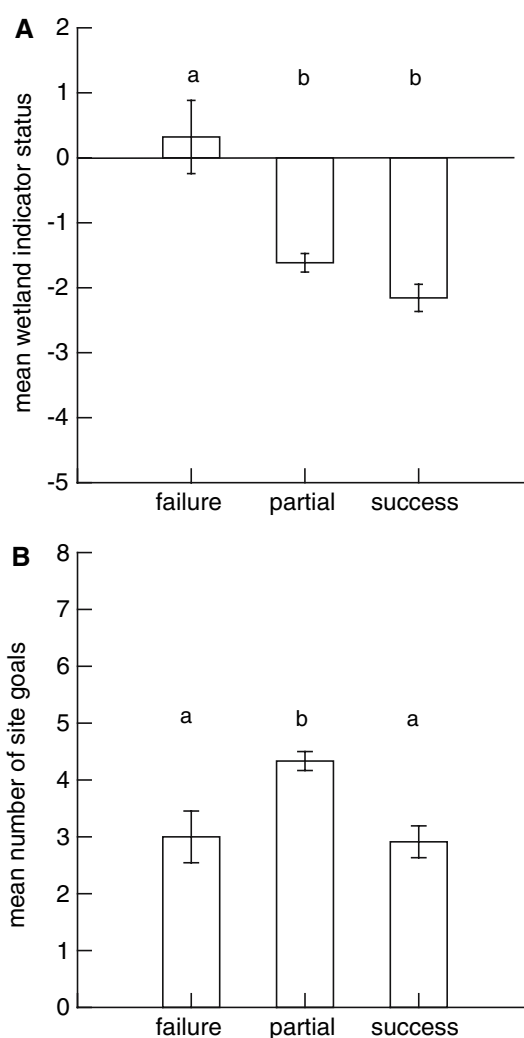
Within broader categories of goals, performance standards and acceptable thresholds for success varied widely for some goals (see the Appendix). For example, acceptable levels of planted tree survival varied from 27% to 100%. Most performance standards established measurable benchmarks for success, but some were subjective and not measurable (e.g., site must have “good survival” of planted trees) and therefore were not useful for determining compliance. Some standards were site-specific and would not be applicable to other sites (e.g., standards specifying particular species that should be dominant), but most were very general and could be applied to any restored or created wetland.



Of a total of 76 sites, 8 sites failed to achieve any project goals in the final year of site monitoring, 45 sites met some, but not all, goals, and 23 sites achieved all goals. Of the sites that were partially successful, seven failed to create jurisdictional wetland and six of these also failed to comply with one or more vegetation-based standards. The remaining 38 partially successful sites did generate jurisdictional wetland but failed to meet some or all vegetation standards. Failed sites had vegetation more characteristic of upland communities (based on mean wetland indicator status) than partially successful or successful sites, reflecting a lack of appropriate wetland hydrology (ANOVA;  $n = 72$ ;  $F_{2,69} = 13.60$ ;  $P < 0.001$ ; Figure 2A). Sites considered partially successful had more goals, on average, than sites considered fully successful (ANOVA;  $n = 76$ ;  $F_{2,73} = 12.87$ ;  $p < 0.001$ ; Figure 2B), suggesting that one reason successful sites were considered “successful” is that they had fewer standards to achieve. It should be noted that the strict assumption of statistical independence among sites has been violated in these ANOVAs because some sites co-occurred within project areas.

Area of jurisdictional wetland created or restored was assessed at the project area level rather than the site level because jurisdictional wetland area was often summed across sites within project areas in the original monitoring reports. The total area proposed for wetland establishment was 113.6 ha in 37 projects, but the area actually established was 81.9 ha in 34 projects, based on an average area of jurisdictional wetland across all years during which a site was monitored. Thus, there was a deficit of 31.7 ha, and three projects failed to produce any wetland due to a lack of wetland hydrology. Two additional projects failed to establish wetland hydrology in a majority of years monitored. Overall, 67% of projects failed to create or restore their minimum required area. On average, projects established 70% of their required area of jurisdictional wetland. This deficit was due to failure to achieve wetland hydrology over the entire area intended for creation or restoration, rather than failure to construct sites. For 22 projects, information was available on the area of natural wetland that was originally impacted, so the mitigation ratio (ratio of area required for compensation to area of original wetland impacted) could be determined. For these projects, the proposed mitigation ratio was 1.55:1, which would result in a net gain of wetland area if all projects were successful. For this subset of projects, the actual, realized mitigation ratio was approximately 1.1:1, indicating a small net gain of wetland area.

For most goals, there were no obvious across-site trends in compliance over time (data not shown). In other words, as sites aged, the proportion of sites considered to be in compliance with permit conditions did not increase or decrease consistently.



**Fig. 2** **A** Mean ( $\pm$  standard error) wetland indicator status of failed, successful, and partially successful compensatory wetlands. **B** Mean ( $\pm$  standard error) number of site goals of failed, successful, and partially successful compensatory wetlands. Different letters above bars indicate significant differences ( $P < 0.05$ ) among groups based on Scheffe test

Compliance success frequencies at the final year of site monitoring varied widely among goals (Table 1). Performance standards related to the survival of planted herbaceous species and planted trees were rarely met by the final year of site monitoring. Similarly, standards specifying a certain vegetation structure were rarely met (e.g., standards requiring some level of interspersed emergent vegetation and open water for waterfowl habitat). A majority of sites with performance standards stating that no non-native or otherwise undesirable species could be dominant at the site failed to meet this standard. However, standards that specified a minimum vegetation cover, or cover by hydrophytic plant species, and standards that established a minimum percent native or nonweedy plant species at a site were frequently met, even in the early years of site

**Table 1** Compliance frequencies for compensatory mitigation wetlands at the final year of site monitoring for major categories of site goals

Goal	No. of sites with goal assessable <sup>a</sup>	No. meeting goal in final year	Success rate
Create jurisdictional wetland <sup>b</sup>	72	58	81%
No undesirable species dominant	53	24	45%
Flora composed primarily of desirable species	53	38	72%
Site must have a minimum of total vegetation cover	31	24	77%
Survival of planted trees	19	4	21%
Goals regarding overall vegetation structure	14	2	14%
Goals regarding buffer vegetation	13	5	38%
Site must exceed a minimum FQI	11	6	55%
Site must have a minimum of cover by hydrophytes	11	10	91%
Survival of planted herbaceous species	9	1	11%
Plant species evenness or richness	1	0	0%
Sediment retention	1	1	100%
Natural regeneration of trees	1	1	100%

<sup>a</sup> For a given goal, the number of sites in which compliance was assessable based on information available in monitoring reports might be less than the number of sites with that goal (see the Appendix)

<sup>b</sup> Considered successful in final year if jurisdictional criteria were met in more than half of the years monitored

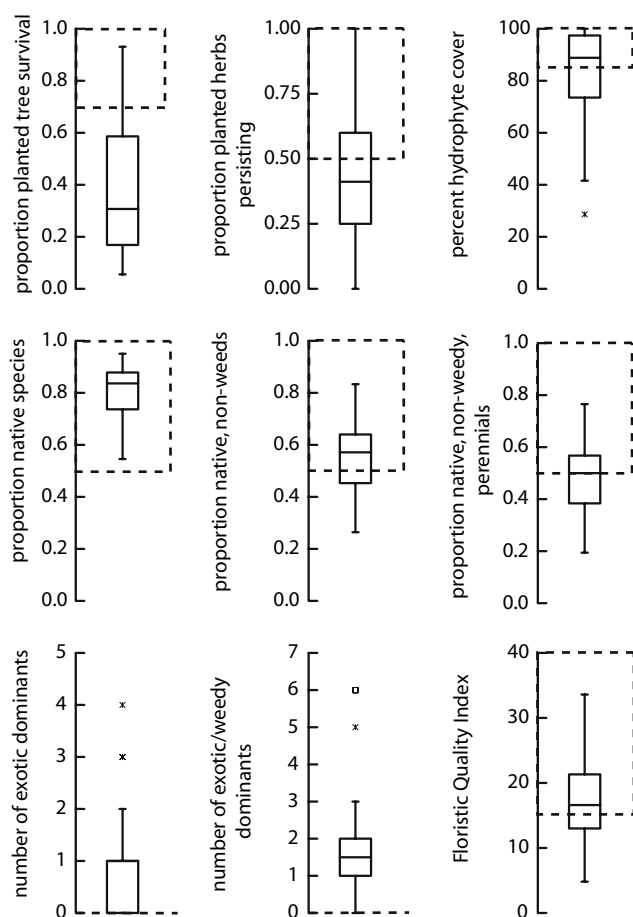
establishment. Compliance frequencies for some other goals could not be evaluated due to a lack of information in site-monitoring reports.

In order to generalize site performance across sites that had different performance standards and goals, measures of performance were compared to typical performance criteria (Figure 3). All sites (given availability of information) were included in these analyses, regardless of site goals and performance criteria. Median percent tree survival, measured as the number of planted trees surviving at the site in the final year of monitoring divided by the number originally planted at the site, was 31%, much lower than the most common criterion for percent tree survival (80%), and this typically required criterion for planted tree survival was met in fewer than 15% of sites. The typical performance criterion of no exotic or weedy dominant species was also met in fewer than 15% of sites. The most frequent exotic dominants were *Phalaris arundinacea* (among the dominants in 19% of sites), *Typha angustifolia* (14% of sites), and *Festuca arundinacea* (8% of sites). Frequent native weedy dominants were *Echinochloa muricata* (11% of sites) and *Eupatorium serotinum* (8% of sites). Typical performance criteria for planted herb persistence, proportion native, nonweedy, or perennial species, and number of exotic dominant species were met in fewer than half of the sites. However, all sites met the typical standard for percent native species, and 67% of sites met this standard for percent native or nonweedy species. The typical performance criterion of at least 75% relative cover by hydrophytic plant species was met in 74% of sites. A majority of sites also met the typical criterion for the FQI.

Randomization tests demonstrated that some, but not all, vegetation-based indicators improved over the first 4 years in sites with at least 4 consecutive years of monitoring (Table 2). Proportion of initially planted trees present in a

given year was calculated as the number of individuals alive in that year divided by the number initially planted. Planted tree mortality was typically high during the first year, and in several cases, additional trees were planted in later years in response. Therefore, the ratio of living planted trees to the number initially planted could increase or decrease over time and potentially could exceed 1 in some years. However, no significant time trend (at  $\alpha = 0.006$  after Bonferroni adjustment) was observed in the number of living planted trees relative to the number initially planted. The proportion of planted herb species persisting at a site tended to increase, although not significantly, over time. Based on information available in site-monitoring reports, this was likely a result of initially seeded species becoming established slowly over time, rather than as a result of subsequent plantings. The FQI and proportion of species that were native, nonweedy, and perennial increased significantly over time, although the increase in mean percent native species over 4 years was small. It should be noted that these variables are interrelated because sites with a high proportion of weedy and annual species will have a low FQI. The percent cover by hydrophytic species also increased over time, although this increase was only marginally significant after a Bonferroni adjustment. There was no significant trend, however, in the number of dominant weedy or exotic species over time.

The number of herbaceous species planted varied among sites from zero to 56 species ( $n = 33$  sites). Native species richness, FQI, mean coefficient of conservatism, and proportion of a site's flora made up by native, nonweedy perennials were not significantly related to log-transformed number of species planted (native species richness:  $r^2 = 0.00$ ,  $\beta = 0.78$ ,  $F_{1,31} = 0.01$ ,  $P = 0.92$ ; FQI:  $r^2 = 0.03$ ,  $\beta = 1.76$ ,  $F_{1,31} = 0.84$ ,  $P = 0.37$ ; mean coefficient of conservatism:  $r^2 = 0.07$ ,  $\beta = 0.23$ ,  $F_{1,31} = 2.36$ ,  $P = 0.14$ ;



**Fig. 3** Box-and-whisker plots illustrating the distribution of wetland site performance levels in comparison to typical thresholds for compliance success. Box-and-whisker plots illustrate the interquartile range (solid box), median (line in box), the range of the distribution (whiskers), and outliers (asterisks and circles). The range of values typically considered successful is represented by the dotted box. Sample sizes were 23 for tree survival, 25 for planted herb persistence, 42 for hydrophyte cover, 73 for proportion desirable species and FQI, and 72 for number of exotic and exotic plus weedy dominants

proportion native, nonweedy perennials:  $r^2 = 0.07$ ,  $\beta = 0.06$ ,  $F_{1,31} = 2.19$ ,  $P = 0.15$ ).

## Discussion

### Performance Standards and Compliance

Goals and performance standards reported here are similar to those reported for compensatory mitigation wetlands elsewhere in the United States. For example, in a review of 300 Section 404 permits, Streever (1999) identified 7 common approaches to performance standards: survival of planted vegetation, standards that are phased in over time, the use of reference sites to set standards, methods that are

based on delineation of jurisdictional wetlands, the use of indexes to condense information, requirements for vegetation cover or plant density, and requirements limiting the occurrence of undesirable species. Six of these approaches were employed in the sites reviewed here. No sites in the present study had performance criteria based on an explicit reference to natural wetlands, however. As in the present study, other studies have reported that the most commonly measured characteristic of compensatory wetlands was vegetation, with performance standards often requiring a certain level of vegetation cover or percent survival of planted vegetation (Breux and Serefidin 1999; Spieles 2005).

Most sites reviewed here can be considered partially successful from a compliance standpoint, meeting some, but not all, project goals. Herbaceous vegetation planting was often unsuccessful in the sites reviewed here. Planted tree survival also failed to meet required goals in a majority of sites. Dominance by exotic and weedy species, most notably *Phalaris arundinacea*, was an additional barrier to compliance. Most sites, however, complied with standards related to percent desirable plant species and, as reported in other studies (Brown and Veneman 2001; Cole and Shafer 2002; Spieles and others 2006), standards for total vegetation cover and cover by hydrophytic plant species.

Very few projects were completely unsuccessful at creating or restoring at least some wetland area, but for a majority of projects, the area of wetland actually constructed was less than the area planned for the project. These results call into question the effectiveness of the Section 404 permitting process with respect to the national goal of no-net-loss of wetland area. However, because this study considered only sites constructed by a single permittee and included sites that ultimately were not accepted as successful by the US Army Corps of Engineers, this study cannot fully address this issue. Previous studies, however, that have reviewed national or state wetland permitting programs have invariably found low success rates for the compensatory mitigation process. Often, required sites are never installed or are never monitored after installation (Brown and Veneman 2001; Hornyak and Halvorsen 2003; NRC 2001; Race and Fonseca 1996; Robb 2002; Sudol and Ambrose 2002). The area of wetland actually created or restored, on a regional or statewide basis, is less than the area required, often leading to a net loss of wetland area at regional scales (Breux and Serefidin 1999; Brown and Veneman 2001; Morgan and Roberts 2003; Kentula and others 1992; NRC 2001; Race 1985; Sifneos and others 1992). When restoration or creation is attempted, compliance with permit conditions is often low (Morgan and Roberts 2003; Sudol and Ambrose 2002; Wilson and Mitsch 1996). A recent review of several studies by the National Research Council (NRC 2001)

**Table 2** Results of repeated measures one-way ANOVA via randomization, testing for an effect of monitoring year on commonly used measures of compensatory wetland performance, over the first 4 years of site monitoring

Measure of success	<i>n</i>	Mean in year 1	Mean in year 4	Observed <i>F</i>	<i>P</i>
No. of trees alive, relative to number initially planted	14	0.54	0.45	1.21	0.33
Proportion of planted herb species persisting	17	0.31	0.41	3.64	0.01
Proportion of flora made up by native species	41	0.79	0.82	6.75	0.0003
Proportion of flora made up by native, nonweedy species	41	0.51	0.56	6.16	0.0005
Proportion of flora made up by native, nonweedy perennials	41	0.41	0.49	9.56	<0.0001
No. of exotic dominants	41	0.85	0.66	0.73	0.54
No. of exotic and weedy dominants	41	2.10	1.71	1.17	0.33
Relative percent cover by hydrophytic plant species	19	82.1	90.7	4.30	0.006
Floristic Quality Index	41	14.7	19.4	27.99	<0.0001

estimated that about half of projects fail to meet their requirements.

### Compliance Versus Success

Compliance with permit conditions is a poor indication of a site's success at replacing the functions of a destroyed natural wetland; therefore, a distinction must be made between compliance success (a regulatory issue) and ecological or functional success (Kentula 2000; Wilson and Mitsch 1996; Zedler and Callaway 2000). Site functional failure has been attributed to a lack of knowledge of wetland ecology among regulators and restoration practitioners (Mitsch and Wilson 1996; Zedler 2000), poor site location relative to the surrounding landscape (Simenstad and others 2006), poor vegetation establishment (Brown and Veneman 2001; Morgan and Roberts 2003; Race 1985), and failure to establish appropriate hydrologic regimes (Brown and Veneman 2001; Galatowitsch and van der Valk 1996; Loucks 1992; Mitsch and Wilson 1996; Morgan and Roberts 2003; Race 1985; Zampella and Laidig 2003; Zedler 2000).

Some of the sites reviewed here were clearly functional failures, largely due to inappropriate hydrology and dominance by exotic species. These functional failures resulted in failure to achieve permit compliance. However, this study demonstrates that a judgment of compliance success or failure is also a function, to a large extent, of the standards chosen to measure site performance. Not unexpectedly, sites with fewer goals were more likely to be considered successful. Furthermore, some standards were apparently either unrealistically stringent or too modest to be of value in assessing site performance. Unrealistically high expectations might reflect overconfidence in restoration technology and our ability to compensate for lost wetland functions, which might ultimately result in an

overreliance on restoration and creation as a form of mitigation. Standards that are too lenient result in the acceptance of poorly performing compensation sites as mitigation for destroyed natural wetlands.

Comparing the site performance from a number of sites to typical goals can help distinguish between situations where goals were not appropriate from situations where sites are not performing. Among sites with planted trees or herbaceous species, survival of planted stock was usually lower than typical standards required. This suggests that if a certain number of planted species or individuals is desired, more individuals, and perhaps a wider variety of species, must be planted with a lowered expectation for establishment.

The typical performance standard for percent native species at a site (>50%) is too low to be of value in distinguishing heavily invaded sites from noninvaded sites. Natural wetlands in this region have, on average, much greater than 50% native species. A set of 551 wetlands sampled throughout Illinois during jurisdictional wetland delineations had an average of 82.4% native species per site (J. Matthews, unpublished data), which is comparable to the average for the compensatory sites in this study (81.3%). Benchmarks for native species presence as well as benchmarks based on indexes like FQI should be based on reference to natural wetlands of similar size, community type, and region as the compensatory wetland so that there is true ecological compensation rather than arbitrary regulatory accomplishment.

Although most sites failed to meet the often-required performance standard specifying that exotic and or weedy species should not be dominant at a site, this standard does not seem inappropriate or overly stringent. However, based on the observed poor performance with respect to this standard, more intensive management is needed to control undesirable species, especially given the high levels of exotic invasion in natural wetlands that act as seed sources

for constructed sites (Spyreas and others 2004; Zedler and Kercher 2004). Some weedy and exotic species are likely to decline in dominance over time (e.g., *Echinochloa muricata* in the present study), and their presence is not necessarily indicative of functional failure in recently restored sites. Other, more invasive, species can form stable communities in restorations, preventing the recruitment of desired, native species (Kulmatiski 2006; Stylinski and Allen 1999). Because of the often inhibitory effect of early invaders on further vegetation change (Connell and Slatyer 1977), restoration practitioners should not assume that invasive species will be replaced by desired native communities via succession (Klötzli and Grootjans 2001; Suding and others 2004). We found no evidence that the number of exotic and weedy dominants declined over 4 years in these sites (see Table 2).

Most sites met the typical performance standard requiring greater than 75% cover by hydrophytic species. It is not clear, however, why a site with more cover by hydrophytic species should be considered more successful. Wetland designers have often erred on the side of greater depth or duration of flooding, creating wetlands that are wetter than the natural wetlands they are meant to replace (NRC 2001). This type of performance standard could have the undesirable effect of encouraging this practice. Given that the primary goal is to create jurisdictional wetland, which requires the establishment of dominant hydrophytic vegetation, wetland hydrology, and hydric soils, this additional performance standard requiring some minimum cover by hydrophytes is redundant and unnecessary as long as the site has an acceptable level of overall vegetation cover.

### Improving Predictability in Wetland Compensation

Ultimately, setting appropriate, scientifically valid performance standards will require improved prediction of restoration outcomes and, thus, an understanding of restoration site development over time. Results of this study suggest that vegetation indicators generally improved over the first 4 years of monitoring. Other studies of restored and created freshwater wetlands have reported increases in plant species richness over time (Campbell and others 2002; Moore and others 1999; Reinartz and Warne 1993), higher FQI in older wetlands (Balcombe and others 2005), and decreases in the presence of exotic species over time (Spieles 2005). However, many studies have reported increases in the abundance or cover of particularly invasive species as restored and created wetlands age (Garde and others 2004; Noon 1996; Moore and others 1999; Reinartz and Warne 1993), suggesting that restored and created wetlands might become biologically simplistic over longer timescales. The timescale of the present study was too

short to determine if increasing dominance of exotic species will eventually lead to a decline in other measures of site performance.

Planting restored wetlands with native species might discourage the establishment of unwanted, highly dominant species and increase the diversity of native species (Armitage and others 2006). Restored wetlands in Wisconsin were found to have lower cover by cattails (*Typha* spp.) and higher diversity of native wetland plants when planted than when unplanted (Reinartz and Warne 1993). Other studies, however, suggest that planting restored wetlands is unnecessary (Kellogg and Brigham 2002; Mitsch and others 1998). Although we found no effect of number of planted species on native species richness, or on other site-scale indicators of floristic quality, planting native species that would not otherwise colonize a site might still increase the site's contribution to local and regional biodiversity (Galatowitsch 2006).

### Recommendations

Two recommendations, not unique to this study but supported by its findings, can be made to improve the establishment of performance criteria. First, the range of performance criteria used to evaluate compliance should be expanded. Performance is most commonly measured by quantifying properties of the vegetation, but such structural properties do not necessarily reflect ecosystem function (Mitsch and Wilson 1996; NRC 2001; Parker 1997). The range of site attributes measured could be expanded to include topography, soils, hydrology, and wildlife, as well as functional attributes of sites (NRC 2001, Ruiz-Jean and Aide 2005; Zedler 1996, Zedler and Callaway 2000). Site-specific goals could be based on wetland functions lost as a result of the permitted activities (NRC 2001; Streever 1999) or based on functions found to be lacking at a watershed scale (Bedford 1999; Brooks and others 2006). Second, more realistic benchmarks for compliance should be set based on reference to the surrounding landscape, natural reference sites, and performance over time in previously restored sites. Permitting agencies will be able to make more informed decisions regarding permit approvals, mitigation site locations, mitigation ratios, performance criteria, and postconstruction monitoring protocols if they can more accurately predict the outcomes of restoration.

**Acknowledgments** Original project monitoring was performed by the Wetlands Group of the Illinois Natural History Survey, under the direction of Allen Plocher, with funding from the Illinois Department of Transportation. Additional hydrologic monitoring was performed at some sites by personnel from the Illinois State Geological Survey. Greg Spyreas, Allen Plocher, Ben O'Neal, and anonymous reviewers provided helpful comments on the manuscript.



## Appendix

### Goals and Performance criteria for 76 Compensatory Mitigation Wetlands Sites in 38 Illinois Project Areas

Goal	Sites	Projects	Goal	Sites	Projects
Create jurisdictional wetland	73	37	Total vegetation cover	31	11
Dominance	56	29	20% vegetation coverage in emergent zone	3	1
No exotic or noxious dominants	3	1	30% vegetation coverage	4	2
No exotic or invasive dominants	4	1	50% vegetation coverage	1	1
No exotic or weedy dominants	1	1	50% vegetation coverage in emergent zone	9	3
None of 3 most dominant exotic	5	3	75% vegetation coverage in zones other than emergent	8	2
None of 3 most dominant exotic or <i>Typha</i> spp.	18	9	75% vegetation coverage in wet prairie zone	1	1
None of 3 most dominant exotic or weedy	11	7	75% vegetation coverage	16	6
None of 3 dominants exotic, weedy, or nonhydrophytic	2	1	Vegetation coverage should be dominant	2	1
None of 3 most dominant in any stratum exotic	1	1	Planted herb persistence or cover	19	8
None of 3 most dominant in any stratum exotic or weedy	4	2	50% of planted herb species persist	14	5
50% of dominants native and nonweedy	5	2	70% of planted herb species should establish	2	1
Dominated by native herbaceous species in ground layer	1	1	50% areal coverage by planted herbs	2	1
Dominated by tall graminoids	1	1	70% of herbaceous cover must be planted species	2	1
Woody species must be dominant	1	1	Good survival of planted herbs	2	1
No woody dominants	2	1	Cover by hydrophytic species	11	7
<i>Sparganium eurycarpum</i> must be a dominant species	1	1	50% of area with hydrophyte cover	5	2
Planted trees should dominate	1	1	75% of area with hydrophyte cover	2	2
Certain planted tree species specified as dominant	1	1	25-80% of area with hydrophyte cover	1	1
Planted tree survival	24	19	Yearly goal for hydrophyte cover (75% by 5 years)	3	3
100% planted tree survival	3	1	Mean native wetness rating of 0 or less	1	1
80% planted tree survival	6	6	Overall vegetation structure/composition	14	6
75% planted tree survival	1	1	Hemimarsch of 50% open, 30% emergent, 20% sedge	8	1
50% planted tree survival	3	3	Establish a floodplain forest	1	1
1500 live trees after 5 years (100% survival)	1	1	Should resemble natural aquatic emergent composition	1	1
60 trees alive after 5 years (92% survival)	1	1	Should resemble natural wet prairie composition	1	1
300 stems/acre (27% survival)	1	1	Less than 40% open water	1	1
100 stems/per acre after 5 years (89% survival)	1	1	Less than 30% open water	1	1
Survival rates specified by seedling type (36% overall)	1	1	High interspersions of vegetation and open water	1	1
Planted trees account for 70% of woody cover	1	1	No planted or volunteer should exceed 40% density	1	1
50% survival of each species	2	1	Floristic quality	11	4
70% of planted species represented by live individuals	1	1	FQI > 20 and mean coefficient of conservatism > 3.5	1	1
50% of planted species represented by live individuals	2	2	FQI > 15	8	2
Good survival of planted trees	2	1	FQI > 7, mean coefficient of conservatism > 2	2	1
Nativeness/nonweediness of flora	53	26	FQI must increase each year	1	1
50% of species native	4	2	Buffer goals	13	4

continued

Goal	Sites	Projects	Goal	Sites	Projects
50% of species native and nonweedy	9	4	Buffer should have 75% vegetation cover	8	1
50% of species native, nonweedy perennials	15	9	No evidence of erosion in the buffer	11	2
50% of species native, nonweedy hydrophytes	2	1	Planted species should establish in the buffer	11	2
70% of species native and nonweedy	1	1	Native perennials persist in buffer	1	1
90% of species native and nonweedy	1	1	Buffer dominated by native, non-weedy perennials	9	2
50% cover by native, nonweedy species	8	1	Good survival of planted trees in buffer	1	1
50% cover by native, nonweedy perennials	8	2	Natural regeneration of trees should occur	1	1
75% cover by native, nonweedy perennials	1	1	Sediments should accumulate at a rate of 0.3–1.1 in/year	1	1
Relative importance of native species increases yearly	1	1	State threatened/endangered birds should be present	8	1
Yearly goal for percent native species (50% after 5 years)	3	3	State EPA water quality standards should be met	8	1

## References

- Armitage AR, Boyer KE, Vance RR, Ambrose RF (2006) Restoring assemblages of salt marsh halophytes in the presence of a rapidly colonizing dominant species. *Wetlands* 26:667–676
- Balcombe CK, Anderson JT, Fortney RH, Kordek WS (2005) Vegetation, invertebrate, and wildlife community rankings and habitat analysis of mitigation wetlands in West Virginia. *Wetlands Ecology and Management* 13:517–530
- Bedford BL (1999) Cumulative effects on wetland landscapes: links to wetland restoration in the United States and southern Canada. *Wetlands* 19:775–788
- Breaux A, Serefidin F (1999) Validity of performance criteria and a tentative model for regulatory use in compensatory wetland mitigation permitting. *Environmental Management* 24:327–336
- Brooks RP, Wardrop DH, Cole CA (2006) Inventorying and monitoring wetland condition and restoration potential on a watershed basis with examples from Spring Creek Watershed, Pennsylvania, USA. *Environmental Management* 38:673–687
- Brown PH, Lant CL (1999) The effect of wetland mitigation banking on the achievement of no-net-loss. *Environmental Management* 23:333–345
- Brown SC, Veneman PLM (2001) Effectiveness of compensatory wetland mitigation in Massachusetts, USA. *Wetlands* 21:508–518
- Campbell DA, Cole CA, Brooks RP (2002) A comparison of created and natural wetlands in Pennsylvania, USA. *Wetlands Ecology and Management* 10:41–49
- Connell JH, Slatyer RO (1977) Mechanisms of succession in natural communities and their role in community stability and organization. *The American Naturalist* 111:1119–1144
- Cole CA (2002) The assessment of herbaceous plant cover in wetlands as an indicator of function. *Ecological Indicators* 2:287–293
- Cole CA, Shafer D (2002) Section 404 wetland mitigation and permit success criteria in Pennsylvania, USA, 1986–1999. *Environmental Management* 30:508–515
- Dahl TE (2000) Status and trends of wetlands in the conterminous United States 1986 to 1997. US Fish and Wildlife Service, Washington, DC
- Daubenmire R (1959) A canopy-coverage method of vegetational analysis. *Northwest Science* 33:43–64
- Edgington ES (1995) Randomization tests, 3rd ed. Marcel Dekker, New York
- Ehrenfeld JG (2000) Defining the limits of restoration: the need for realistic goals. *Restoration Ecology* 8:2–9
- FICWD (Federal Interagency Committee for Wetland Delineation) (1989) Federal manual for identifying and delineating jurisdictional wetlands. Cooperative technical publication. US Army Corps of Engineers, US Environmental Protection Agency, US Fish and Wildlife Service, and USDA Soil Conservation Service, Washington, DC
- Galatowitsch SM (2006) Restoring prairie pothole wetlands: does the species pool concept offer decision-making guidance for revegetation? *Applied Vegetation Science* 9:261–270
- Galatowitsch SM, van der Valk AG (1996) Characteristics of recently restored wetlands in the prairie pothole region. *Wetlands* 16:75–83
- Galatowitsch SM, Anderson NO, Ascher PD (1999) Invasiveness in wetland plants in temperate North America. *Wetlands* 19:733–755
- Garde LM, Nicol JM, Conran JG (2004) Changes in vegetation patterns on the margins of a constructed wetland after 10 years. *Ecological Management and Restoration* 5:111–117
- Hornyak MM, Halvorsen KE (2003) Wetland mitigation compliance in the western Upper Peninsula of Michigan. *Environmental Management* 32:535–540
- Iverson LR, Ketzner D, Karnes J (1999) Illinois plant information network. Database available from <http://www.fs.fed.us/ne/delaware/ilpin.html>. Illinois Natural History Survey and USDA Forest Service
- Kellogg CH, Bridgham SD (2002) Colonization during early succession of restored freshwater marshes. *Canadian Journal of Botany* 80:176–185
- Kentula ME (2000) Perspectives on setting success criteria for wetland restoration. *Ecological Engineering* 15:199–209
- Kentula ME, Sifneos JC, Good JW, Rylko M, Kunz K (1992) Trends and patterns in Section 404 permitting requiring compensatory mitigation in Oregon and Washington, USA. *Environmental Management* 16:109–119
- Klötzli F, Grootjans AP (2001) Restoration of natural and semi-natural wetland systems in central Europe: progress and predictability of developments. *Restoration Ecology* 9:209–219
- Kulmatiski A (2006) Exotic plants establish persistent communities. *Plant Ecology* 187:261–275

- Loucks O (1992) Predictive tools for rehabilitating linkages between land and wetland ecosystems. In Wali MK (ed.), Ecosystem rehabilitation, Volume 2: Ecosystem analysis and synthesis. SPB Academic Publishing, The Hague, The Netherlands. pp 297–308
- Matthews JW (2000) Assessment of the Floristic Quality Index for use in Illinois, USA, wetlands. *Natural Areas Journal* 23:53–60
- Mitsch WJ, Wilson RF (1996) Improving the success of wetland creation and restoration with know-how, time, and self-design. *Ecological Applications* 6:77–83
- Mitsch WJ, Wu X, Nairn RW, Weihe PE, Wang N, Deal R, Boucher CE (1998) Creating and restoring wetlands: a whole-ecosystem experiment in self-design. *BioScience* 48:1019–1030
- Mohlenbrock RH (2002) Vascular flora of Illinois. Southern Illinois University Press, Carbondale
- Moore HH, Niering WA, Marsicano LJ, Dowdell M (1999) Vegetation change in created emergent wetlands (1988–1996) in Connecticut (USA). *Wetlands Ecology and Management* 7:177–191
- Morgan KL, Roberts TH (2003) Characterization of wetland mitigation projects in Tennessee, USA. *Wetlands* 23:65–69
- Niering WA (1987) Vegetation dynamics (succession and climax) in relation to plant community management. *Conservation Biology* 1:287–295
- Noon KF (1996) A model of created wetland primary succession. *Landscape and Urban Planning* 34:97–123
- NRC (National Research Council) (2001) Compensating for wetland losses under the Clean Water Act. National Academy Press, Washington, DC
- Parker VT (1997) The scale of successional models and restoration objectives. *Restoration Ecology* 5:301–306
- Race MS (1985) Critique of present wetlands mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. *Environmental Management* 9:71–82
- Race MS, Fonseca MS (1996) Fixing compensatory mitigation: What will it take? *Ecological Applications* 6:94–101
- Reed PB Jr (1988) National list of plant species that occur in wetlands: Illinois. US Department of the Interior, Fish and Wildlife Service, National Wetlands Inventory, Washington, DC
- Reinartz JA, Warne EL (1993) Development of vegetation in small created wetlands in southeastern Wisconsin. *Wetlands* 13:153–164
- Robb JT (2002) Assessing wetland compensatory mitigation sites to aid in establishing mitigation ratios. *Wetlands* 22:435–440
- Ruiz-Jean MC, Aide TM (2005) Restoration success: How is it being measured? *Restoration Ecology* 13:569–577
- Saltonstall K (2002) Cryptic invasion by a non-native genotype of the common reed, *Phragmites australis*, into North America. *Proceedings of the National Academy of Sciences, USA* 99:2445–2449
- Sifneos JC, Cake EW Jr, Kentula ME (1992) Effects of section 404 permitting on freshwater wetlands in Louisiana, Alabama, and Mississippi. *Wetlands* 12:28–36
- Simenstad CA, Reed D, Ford M (2006) When is restoration not? Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. *Ecological Engineering* 26:27–39
- Spieles DJ (2005) Vegetation development in created, restored, and enhanced mitigation wetland banks of the United States. *Wetlands* 25:51–63
- Spieles DJ, Coneybeer M, Horn J (2006) Community structure and quality after 10 years in two central Ohio mitigation bank wetlands. *Environmental Management* 38:837–852
- Spyreas G, Ellis J, Carroll C, Molano-Flores B (2004) Non-native plant commonness and dominance in the forests, wetlands, and grasslands of Illinois, USA. *Natural Areas Journal* 24:290–299
- Streever WJ (1999) Examples of performance standards for wetland creation and restoration in Section 404 permits and an approach to developing performance standards. WRP Technical Notes Collection TN WRP WG-RS-3.3. US Army Engineer Research and Development Center, Vicksburg, MS
- Stylinski CD, Allen EB (1999) Lack of native species recovery following severe exotic disturbance in southern California shrublands. *Journal of Applied Ecology* 36:544–554
- Suding KN, Gross KL, Houseman GR (2004) Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology and Evolution* 19:46–53
- Sudol MF, Ambrose RF (2002) The US Clean Water Act and habitat replacement: Evaluation of mitigation sites in Orange County, California, USA. *Environmental Management* 30:727–734
- Suloway L, Hubbell M (1994) Wetland resources of Illinois: an analysis and atlas. Illinois Natural History Survey Special Publication 15:1–88
- Swink F, Wilhelm G (1994) Plants of the Chicago region. 4th edition. Indiana Academy of Science, Indianapolis, Indiana, 921 pp
- Taft JB, Wilhelm GS, Ladd DM, Masters LA (1997) Floristic Quality Assessment for vegetation in Illinois, a method for assessing vegetation integrity. *Erigenia* 15:3–95
- USACE (US Army Corps of Engineers) (1987) Corps of Engineers wetlands delineation manual. Technical Report Y-87-1. Environmental Laboratory, US Army Corps of Engineers Waterways Experimental Station, Vicksburg, MS
- Whigham DF (1999) Ecological issues related to wetland preservation, restoration, creation and assessment. *The Science of the Total Environment* 240:31–40
- Wilson RF, Mitsch WJ (1996) Functional assessment of five wetlands constructed to mitigate wetland loss in Ohio, USA. *Wetlands* 16:436–451
- Zampella RA, Laidig KJ (2003) Functional equivalency of natural and excavated coastal plain ponds. *Wetlands* 23:860–876
- Zedler JB (1996) Ecological issues in wetland mitigation: an introduction to the forum. *Ecological Applications* 6:33–37
- Zedler JB (2000) Progress in wetland restoration ecology. *Trends in Ecology and Evolution* 15:402–407
- Zedler JB, Callaway JC (1999) Tracking wetland restoration: do mitigation sites follow desired trajectories? *Restoration Ecology* 7:69–73
- Zedler JB, Callaway JC (2000) Evaluating the progress of engineered tidal wetlands. *Ecological Engineering* 15:211–225
- Zedler JB, Kercher S (2004) Causes and consequences of invasive plants in wetlands: Opportunities, opportunists, and outcomes. *Critical Reviews in Plant Sciences* 23:431–452

# PROFILE

## Wetland Mitigation Banking: A Framework for Crediting and Debiting

**ERIC D. STEIN<sup>1\*</sup>**

**FARI TABATABAI**

Regulatory Branch

U.S. Army Corps of Engineers

Los Angeles District

P.O. Box 532711

Los Angeles, California 90053-2325, USA

**RICHARD F. AMBROSE**

Environmental Science and Engineering Program

University of California, Los Angeles

10833 Le Conte Avenue

Los Angeles, California 90095-1772, USA

**ABSTRACT** / Wetland mitigation banking as a resource management tool has gained popular support for its potential to provide an ecologically effective and economically efficient means to fulfill compensatory mitigation requirements for impacts to aquatic resources. Although this management tool has been actively applied within the past 10 years (C. Short, 1988, Mitigation banking, in *Biological Report* 88(41): 1–103), assessment of credits and determination of a compensation ratio that reflects existing and/or potential functional condition in a mitigation bank has been a formidable

task. This study presents a framework for a systematic approach for determination of credits and debits and subsequently the compensation ratio. A model for riparian systems is developed based on this framework that evaluates credits and debits for spatial and structural diversity, contiguity of habitats, invasive vegetation, hydrology, topographic complexity, characteristics of flood-prone areas, and biogeochemical processes. The goal of developing this crediting and debiting framework is to provide an alternative to the current methods of determining credits and debits in a mitigation bank and assigning mitigation ratios, such as best professional judgement or use of preset ratios. The purpose of this crediting and debiting framework is to develop a method that (1) can be tailored to evaluate ecological condition based on the target resources of a specific mitigation bank, (2) is flexible enough to be used for evaluation of existing or potential ecologic condition at a mitigation bank, (3) is a structured and systematic way to apply data and professional judgment to the decision-making process, (4) has an ecologically defensible basis, (5) has ease of use such that the level of expertise and time required to employ the method is not a deterrent to its application, and (6) provides a semiquantitative measure of the condition of aquatic resources that can be translated to a mitigation ratio.

Urbanization, land development, agriculture, resource extraction, and infrastructure development are often accompanied by impacts to aquatic resources through either direct fill or secondary and cumulative impacts. Discharge of dredged or fill material affecting aquatic resources, such as lakes, rivers, streams, oceans, or wetlands usually falls under the jurisdiction of Section 404 of the Clean Water Act and is regulated by the U.S. Army Corps of Engineers (Corps) regulatory program. Corps' regulations, guidelines, and Memorandum of Agreement (MOA) allow for compensatory mitigation to be performed to offset the unavoidable impacts associated with permitted activities. The 1990 MOA between the Corps and the U.S. Environmental Protection Agency (US EPA) regarding mitigation ex-

presses a clear preference for on-site, in-kind replacement of wetland functions and values. Consequently, compensatory mitigation is often done at or near the project site and consists of either creation of new habitat, restoration or enhancement of degraded habitat, or, in some cases, preservation of intact habitat.

Within the last 10 years wetland mitigation banking has gained popular support as a resource management tool with the potential to provide an ecologically effective and economically efficient alternative to traditional site specific mitigation as a means to fulfill compensatory mitigation requirements (IWR 1992). Mitigation banking is founded on the premise that large, contiguous wetland parcels can have a greater chance of being biologically and hydrologically viable and can accrue more ecologic functions than small, isolated compensatory mitigation sites (Short 1988, Environmental Law Institute 1993). Wetland mitigation banks strive to establish large, contiguous wetland areas that can be used to mitigate for a number of independent impacts. This allows eligible permittees to purchase compensa-

**KEY WORDS:** Wetlands; Mitigation banking; Credits; Debits; Section 404

<sup>1</sup> *Present address:* PCR Services Corporation, One Venture, Suite 150, Irvine, California 92618, USA

\*Author to whom correspondence should be addressed.

tory mitigation functions or credits from another entity that has already produced and banked them, thereby eliminating the need to produce compensatory mitigation areas on site. Mitigation banking can have the added advantage of establishing successful wetland functions in advance of the actual loss of functions associated with a permitted activity (IWR 1992).

Despite the recent rise in popularity and regulatory support for mitigation banks, assessment of credits in a mitigation bank and determination of compensation ratios that reflect existing and/or potential ecologic conditions in a mitigation bank continues to be one of the most problematic yet most essential aspects of mitigation banking (Environmental Law Institute 1993, IWR 1994). The November 1995 Joint Federal Guidance for the Establishment and Use of Mitigation Banks requires that mitigation banks include systems for determining the number of credits needed to compensate the impacts of a given project (i.e., defining the currency of the bank and setting mitigation ratios) (Federal Register 1995). The crediting and debiting methodology is a two-step process where the existing or potential condition of a mitigation bank (credits) and at the impact site (debits) are assessed and translated into a currency such as acreage or habitat units (IWR 1992, Environmental Law Institute 1993). The second step consists of a determination of the number of credits needed to compensate for losses from a project (debits) or the compensation ratio.

Numerous assessment methods have been proposed for the determination of credits and debits in wetland mitigation banks. The majority of wetland mitigation banks to date, however, use best professional judgment or simple indices, such as acreage, to determine the compensation ratio (Tabatabai 1994). The main advantage of simple indices is their lack of complexity and ease of use. These indices can be calculated quickly by project proponents and regulatory staff, often with little or no field work and little expenditure of resources. The disadvantage of simple indices is they ignore the complexities of wetland ecosystems and may not be representative of aquatic resource functions impacted and the existing or potential functions that exist in a mitigation bank (IWR 1994). Using best professional judgment to determine the acreage to compensate for loss of aquatic resources not only is problematic in terms of scientific indefensibility but also poses problems of inconsistency, uncertainty, and irreproducibility. Great caution must be exercised when using best professional judgment or simple indices to protect against wetland losses.

As an alternative to simple indices or best professional judgment, credits and debits can be computed using functional evaluation methods. Numerous tech-

niques developed over the last 20 years attempt to use field indicators as measures of habitat function. These techniques include:

- Biotic indices, such as species density and the Shannon-Weaver index of species diversity. These biotic indices can be multiplied by acreage to yield diversity units.
- Assessments based on species composition or habitat suitability for specific indicator species, such as the Habitat Evaluation Procedure (US FWS 1980), Habitat Evaluation System (Pearsall and others 1986), Biological Evaluation Standardized Technique (Barnett and others 1991), and Index of Biotic Integrity (Karr 1991).
- Surveys of habitat characteristics, such as the Wetland Evaluation Technique (Adamus 1983), Wetland Replacement Evaluation Procedure (Bartoldus and others 1992), and Wetland Evaluation Methodology (WEM) (US ACOE 1988).
- Landscape level assessments using Geographic Information Systems (GIS) and other coarse resolution measures of function in a regional perspective, such as US EPA's Synoptic Approach to Impact Assessment (US EPA 1992).

The most recent and one of the most promising functional assessment techniques is the Hydrogeomorphic Method (HGM) (Smith and others 1995), developed by the U.S. Army Corps of Engineers Waterways Experiment Station (WES). This method uses variables measured in the field to compute functional indices for biotic, hydrologic, and biogeochemical functions. These indices are scaled against locally representative reference sites to account for regional variations in wetland ecosystems. However, development of regional models and reference standards requires considerable time, resources, and technical expertise; to date, few regional reference sets have been developed.

Each evaluation method has strengths and weaknesses, which have been previously discussed by several authors (Margules and Usher 1981, Westman 1985, Lonard and Clairain 1985, Jain and others 1993, Stein 1995). However, because mitigation banks are typically used to compensate for impacts resulting from multiple small projects, methods such as those listed above become cumbersome in terms of personnel resources and inefficient in terms of assessing functions impacted at each site eligible to use the mitigation bank. In addition, regulatory agencies may not have the expertise or resources to apply the functional assessment methods properly; therefore, the designated method may not be used accurately. In his review of functional



assessment methods, Smith (1993) concluded that “no single method reviewed meets the requirements of a quick screening technique to determine a broad spectrum of wetland values and functions.” It is unlikely that any single method could fully satisfy both the quick screening and the comprehensiveness criteria. However, it is our goal to develop a crediting and debiting framework for wetland mitigation banks that will address some of the limitations posed by other crediting strategies while providing a balance between ease of use and defensible measure of ecologic condition. To achieve this goal, a crediting and debiting framework should meet the following criteria: (1) can be tailored to evaluate ecologic condition based on the target resources of a specific mitigation bank, (2) is flexible enough to be used for evaluation of existing or potential ecologic condition at a mitigation bank, (3) is a structured and systematic way to apply data and professional judgment to the decision-making process, (4) has an ecologically defensible basis, (5) has ease of use such that the level of expertise and time required to employ the method are not deterrents to its application, and (6) provides a semiquantitative measure of the condition of aquatic resources that can be translated to a mitigation ratio.

In this paper we present a crediting and debiting framework for wetland mitigation banks that meets the above criteria. The principles of the framework are applied to develop a model for southern California riparian systems. Use of the riparian model is illustrated for Santa Ana River Mitigation Bank (SARMB), located in Riverside County, CA.

### Crediting and Debiting Framework

The crediting and debiting framework is based on assessing changes in structural characteristics at the impact site and the mitigation site. Change is assessed by evaluating conditions before and after alterations to the site. Structural characteristics are used as indicators of ecologic condition of the specific class of aquatic resource.

Credits are determined based on the difference between structural characteristics of the post-restoration condition and pre-restoration (baseline) condition at the bank site. Similarly, debits are assessed by determining the difference between pre-project and post-project structural characteristics at the impact site. Each structural characteristic, or criterion, is evaluated on a linear interval scale and assigned a rating that reflects the relative value of that criterion at a given site. Credits are the sum of net gain of values for all criteria at the bank and debits are the sum of net loss of values for all

Table 1. Crediting and debiting framework. Application of the crediting and debiting framework involves three main steps: evaluation of credits, evaluation of debits, and determination of the mitigation ratio

---

Step 1. Evaluation of credits
Credits = Post-project Rating (or Enhancement Potential Rating) – Pre-project Value (Existing Value)
Step 2. Evaluation of debits
Debits = Pre-project Rating of the Impact Site – Post-project Rating of the Impact Site
Step 3. Determination of the mitigation ratio
Mitigation Ratio = Debits/Credits (or Projected Available Credits)

---

criteria at the impact site. The mitigation ratio is the ratio of debits over the credits. When the mitigation credits must be calculated (or estimated) before the bank is functionally mature, the mitigation ratio can be based on the maximum expected gain at the bank (i.e., the enhancement potential) (Table 1). We will demonstrate the framework using the structural characteristics developed below for the southern California riparian model.

The framework is a systematic approach designed to balance directly measuring hydrologic and physical characteristics of aquatic resources against ease of use. The intent is not to provide an absolute tool for evaluating functional condition, rather to provide an ecologically based framework to organize best professional judgment and apply it in a systematic manner. The framework is intended to apply to the mitigation bank and the typically small impact sites that normally use a mitigation bank. Assumptions associated with this type of crediting and debiting methodology include equal weights assigned to each criteria and a linear increase in values associated with each interval.

This crediting and debiting framework may be applied to various types of ecosystems. Evaluation criteria will vary based on the type of the system being evaluated and should account for the hydrologic, biologic, biogeochemical, and landscape characteristics of the target aquatic system. Below we provide a sample crediting and debiting system developed for southern California riparian wetlands.

### Southern California Riparian Model

Riparian systems in the western United States are typically narrow, linear strips of vegetation along rivers, streams, or lakes and are dependent on perennial or ephemeral surface or subsurface water (Knopf and others 1988, US DOI 1994). Dry climates and porous

soils found in arid regions cause streamside soil moisture to decrease more rapidly with distance from the streambank than in humid regions, resulting in narrower riparian zones (Reichenbacher 1984). However, flooding duration, intensity, and timing are the ultimate determinants of riparian succession. Flooding waters bring nutrient-rich sediments to the flood plain, export organic and inorganic material from the flood plain, scour mature woodlands, and help spread propagules laterally into the flood plain (Strahan 1984, Warner and Hendrix 1985, Dickert and Tuttle 1985, Gosselink and others 1990a). Riparian systems form dynamic mosaics of active channels, terraces, flood plains, and alluvial fans. The composition and distribution of these systems is a product of fluvial processes, which erode material from some areas and deposit it in others during flood events, facilitating channel migration (Gregory and others 1991). This combination of degradation and aggradation results in the formation of bars and terraces with different drainage patterns and elevations. These elevational differences result in the extensive vegetative diversity of riparian systems (Strahan 1984). The viability of terraces and flood plains depends on their proximity to groundwater levels, surface emergent aquifers, or hyporheic zones (porous substrate allowing water to flow immediately beneath the surface of streambeds) (Stanford and Ward 1993). Therefore, in the arid west, the width and distribution of the riparian zone is ultimately determined by the vertical gradient between the benchland and the streambed (Szaro 1990).

Although their areal extent is proportionately less than in other parts of the country, western riparian systems have a proportionately greater significance for some functions because of the arid climates in which they occur (US DOI 1994). In the arid southwestern United States, riparian areas serve as linear or single-point habitat islands on which a multitude of native wildlife species are totally dependent for survival (Warner and Hendrix 1985). The US DOI (1994) estimated that although less than 1% of the western portion of the United States is covered by riparian vegetation, between 51% and 82% of all species in the southwestern United States depend on riparian areas for survival.

#### Evaluation Criteria for Southern California Riparian Systems

Based on the crediting and debiting framework, we developed a model for southern California's riparian systems using the following evaluation criteria: (1)

spatial diversity and coverage of habitats; (2) structural diversity of habitats; (3) contiguity of habitats; (4) percent of invasive vegetation; (5) hydrology; (6) topographic complexity; (7) characteristics of flood-prone area; and (8) biogeochemical processing. These criteria reflect the fact that assessment of riverine systems requires examination of the entire riparian zone and consideration of the interaction between geology, hydrology, and organic and inorganic inputs to the system. In recognition of the fact that functional capacity differs between low-order and high-order streams, for some criteria we have provided different indicators for first- and second-order streams versus higher order streams. Because first- and second-order streams do not typically support the same complexity of habitat as higher order systems, they will typically score lower on the habitat criteria. For the purposes of this method, trees are defined as perennial woody dicots greater than 7.5 cm diameter at breast height (DBH). Saplings are defined as perennial woody dicots less than 7.5 cm DBH.

The first two evaluation criteria address structure, composition, and diversity of the site. Scoring of the first criterion, coverage and spatial diversity of habitats, should consider the site as a whole and evaluates both diversity of habitat types (i.e., interspersed) and species diversity within each patch. Scoring of the structural diversity of habitats criterion should focus only on the structure within the riparian patches on the site (as opposed to the site as a whole). Although this criterion partially captures species diversity, it is to a lesser extent than the spatial diversity criterion. The first two criteria should be scored based on the vegetative composition of the site regardless of whether the vegetation is native or non-native. Effects of non-native species on habitat integrity are addressed by a separate criterion. Evaluation of structure regardless of the geographic origin of the species accounts for the fact that increased biomass (regardless of species type) contributes to a site's ability to retain water and retain nutrients and compounds, thereby increasing some hydrologic and biogeochemical functions. This attribute is also directly accounted for by the biogeochemical processes criterion, which is scored based on abundance of biomass, regardless of whether or not it is native.

#### Coverage and Spatial Diversity of Habitats

Riparian habitats are typically patchy with an interspersed of different habitat types (Faber and Holland 1988). This interspersed allows the activities of animals in dry sites to be more closely coupled to those in wet sites. A mosaic of habitat types provides a richer, more continuous food source for mobile fauna than that of a

homogeneous habitat. For example, Doyle (1990) found a strong correlation between the extent of herbaceous and deciduous shrub cover in riparian habitats and the abundance and diversity of small mammals. Habitat mosaics also allow animals to fulfill several life functions at a single site (e.g., foraging, escape, reproduction) (Warner and Hendrix 1985, Gosselink and others 1990b). Alpha diversity (diversity within a site) has been correlated to the ability of a patch to support a complex food chain and allow interior species with specific habitat requirements to thrive in the face of competition from generalists (Klopatek 1984, Harris 1988). Assessment of changes to the spatial diversity of a project site provides information about impacts to a site's capability to support a variety of different faunal species.

The ratings for the coverage and spatial diversity criterion are assigned based on the following scale:

- 0 = Site permanently converted to land use not able to support native riparian vegetation, such as housing, agriculture, or concrete channel.
- 0.2 = No existing riparian vegetation (e.g., covered with annual grasses and scrub, bare ground). However, site has the potential for revegetation without extensive structural modification.
- 0.4 = Patches of monotypic riparian vegetation covering up to 50% of the site, interspersed among herbaceous species or bare ground.
- 0.6 = Patches of diverse riparian vegetation (e.g., at least three different genera of riparian vegetation present) covering up to 30% of the site, interspersed among grasses, invasive plants, or bare ground; and/or greater than 50% of the site covered with monotypic patch(es) of riparian vegetation, interspersed among herbaceous species or bare ground.
- 0.8 = Diverse riparian vegetation covering between 30% and 75% of the site, e.g., strips or islands of riparian habitat interspersed in open space.
- 1.0 = Diverse riparian vegetation (e.g., at least three different genera of riparian vegetation present) covering between 75% and 100% of the site, interspersed in open space or herbaceous plant communities.

#### Structural Diversity of Habitats

The stratification of vegetation into layers, including shrubs, understory, and canopy, provides a variety of different habitats. This allows a diversity of organisms representing different trophic levels to coexist in a single site, thereby supporting a more complex and resilient food chain (Warner and Hendrix 1985). For

example, diverse ground cover provides habitat for many insects which form the base of the food chain and provide important ecosystem functions, such as pollination. This allows higher-trophic-level organisms to utilize understory and canopy habitat that may be present (Erman 1984). Structural diversity within a site has been correlated with faunal diversity, especially for birds (Gosselink and others 1990b). The presence of a floristic structure consisting of three strata indicates that appropriate soil, moisture, and topographic conditions exist to support a "healthy" riparian system (Warner 1984). Structural diversity of the vegetated portions of the project site is used as a surrogate for general habitat suitability for an assortment of common species.

Because riparian habitats are typically patchy (Faber and Holland 1988), the ratings for this criterion are based on only the vegetated portions of each site:

- 0 = Site permanently converted to land use that will not be able to support native riparian vegetation, such as housing, agriculture, or concrete channel.
- 0.2 = No existing riparian vegetation (e.g., covered with upland grasses and scrub, bare ground). However, site has the potential for revegetation without extensive structural modification.
- 0.4 = Vegetated areas of the site contain sparse, scattered, patchy, or remnant riparian vegetation that is immature and/or lacks structural (vertical) diversity.
- 0.6 = The patches of riparian vegetation on the site contain riparian trees and/or saplings (i.e., perennial dicots), but contain no or poorly developed shrub understory.
- 0.8 = The patches of riparian vegetation on the site contain riparian trees and saplings, plus a well-developed native shrub understory.
- 1.0 = The patches of riparian vegetation on the site are structurally diverse. They contain riparian trees, saplings, and seedlings, as well as developed native shrub understory and herbaceous layer.

#### Contiguity of Habitats

Fragmentation and habitat loss are dominant causes of the decrease in biotic diversity (Harris 1988). The ecological value of disjunct habitat patches can be enhanced if they are connected by strips of protected habitat; these corridors facilitate movement between patches (Diamond 1975, Noss 1987). For animals with a home range exceeding the size of an individual habitat patch, corridors provide a means of moving from one habitat patch to another. Without a system of travel

corridors allowing these animals passage from one refuge to another, they will probably not occur in future landscapes (Harris 1988). Even if partially disturbed, riparian corridors are vital to the successful migration of neotropical birds and other organisms (Croonquist and Brooks 1991). In addition, habitat connectivity helps small populations (such as endangered species) maintain demographic and genetic integrity in the face of the isolation caused by habitat fragmentation (Frankel and Soule 1981). Changes to linear contiguity affect not only corridors but also contribute to overall habitat fragmentation and decreases in patch size. This can be detrimental for resident as well as migrant species (Harris 1988).

The ecological value of riparian habitats also depends on their integration as units within the surrounding landscape (Gosselink and others 1990b). Many organisms have complex life histories in which different stages require distinct habitats within a regional landscape in order to meet their life requirements (Harris 1988). Therefore, continuity between riparian and upland habitats increases utilization by fauna and provides safe passage between riparian oasis and adjacent uplands (Gosselink and others 1990c). Furthermore, the greater the edge area between riparian habitat and developed areas, the greater the potential negative impact from adjacent upland land use (Warner and Hendrix 1985). Additionally, many riparian plants require adjacent uplands as a flood plain for establishment of their propagules during flooding events (Scott and others 1993). These flood plains also provide refuge for fauna during flooding (Gosselink and others 1990c).

The continuity criterion includes two components. Linear continuity refers to riparian habitat upstream and/or downstream of the site. Lateral continuity addresses the quality of upland habitat and reflects the connection of the site to the surrounding nonriparian habitat. The ratings for the contiguity criterion are assigned based on the following scale:

*First and second order streams.*

- 0 = No linear contiguity or transitional upland habitat; completely surrounded by or isolated within an urban setting or converted to an urban/suburban land use.
- 0.2 = No linear contiguity upstream or downstream, but isolated within upland open space habitat.
- 0.4 = Contiguous with comparable habitat on one end of the site (upstream or downstream), but surrounded with urban/suburban or other nonopen

space lands adjacent (lateral to) to the site on at least one side.

- 0.6 = Contiguous with comparable habitat on one end of the site (upstream or downstream) and surrounded by transitional upland habitat which is at least 35 m wide.
- 0.8 = Contiguous with comparable habitat on both ends of the site (upstream and downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
- 1.0 = Contiguous with comparable habitat on both ends of the site (upstream and downstream) and surrounded by transitional upland habitat on both sides which is at least 35 m wide.

*Higher order streams.*

- 0 = No linear contiguity or transitional upland habitat; completely surrounded by or isolated within an urban setting or converted to an urban/suburban land use.
- 0.2 = No linear contiguity upstream or downstream, but isolated within upland open space habitat.
- 0.4 = Contiguous with comparable habitat on one end of the site (upstream or downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
- 0.6 = Contiguous with comparable habitat on one end of the site (upstream or downstream) and surrounded by transitional upland habitat which is at least twice the width of the riparian zone.
- 0.8 = Contiguous with comparable habitat on both ends of the site (upstream and downstream), but surrounded with urban/suburban or other nonopen space lands adjacent (lateral to) to the site on at least one side.
- 1.0 = Contiguous with comparable habitat on both ends of the site (upstream and downstream) and surrounded by transitional upland habitat on both sides which is at least twice the width of the riparian zone.

Percent of Invasive Vegetation

Invasive species often thrive in mesic environments and readily establish following disruption of riparian systems. Many invasive species have few, if any, native pests or diseases and thus grow rapidly. Once established, their proliferation excludes reestablishment of native species following subsequent disturbances, such as floods or fires (Warner and Hendrix 1985). Some invasive vegetation, such as *Arundo donax* and *Tamarix*



spp. provide little to no habitat value for wildlife species (Hanes 1981, Bell 1993). Moreover, *A. donax* and *Tamarix* spp. pose a greater problem for flood control than native vegetation due to the morphological characteristics of the long stalks (*Arundo*) and deep taproots (*Tamarix*), which obstruct flood control channels more than native riparian vegetation. Overall, the replacement of native riparian habitat with *A. donax*, *Tamarix* spp., and other invasive vegetation displaces native fauna, reduces flood conveyance, increases evapotranspirative losses, increases water temperature, and creates fire hazards (Bell 1993). For example, eradication of *A. donax* from the Santa Ana River could reduce annual evapotranspirative water losses by an estimated  $4.6 \times 10^7 \text{ m}^3$ , resulting in an estimated savings of \$12 million annually (Iverson 1993). However, it has been suggested that the increased biomass associated with invasive weed infestation may increase retention times and, therefore, the ability of a site to sequester elements or compounds. The contribution of increased biomass to biogeochemical processes is accounted for in the structural diversity and spatial diversity criteria.

The ratings for the percent of invasive vegetation criterion are assigned based on the following scale:

- 0 = Site is covered by pure stands of invasive vegetation or lacks any riparian vegetation.
- 0.2 = Site is covered by 70–99% invasive vegetation.
- 0.4 = Site is covered by 40–69% invasive vegetation.
- 0.6 = Site is covered by 10–39% invasive vegetation.
- 0.8 = Site is covered by 5–9% invasive vegetation.
- 1.0 = Site is covered by less than 5% invasive vegetation.

#### Hydrology

Hydrology is the most important factor determining the establishment and maintenance of specific wetland functions (Mitsch and Gosselink 1993). Reviews of past mitigation sites reveal that improper hydrology is the most significant problem with many unsuccessful sites (Mitsch and Wilson 1996, Sudol 1996). Riparian systems rely on appropriate and natural hydrology for long-term self-sustainability and viability. This criterion addresses the source of water supporting the wetlands and the exposure of the site to riparian processes, such as scour and overbank flow. The geomorphic structure of the site is addressed by the topographic complexity and flood-prone area criteria. The ratings for the hydrology criterion are assigned based on the following scale:

- 0 = No regular supply of water to the site. Site not associated with any water source, surface drainage, impoundment, or groundwater discharge.

- 0.2 = Water supply to the site is solely from artificial irrigation (e.g., sprinklers, drip irrigation). No natural surface drainage, natural impoundment, groundwater discharge, or other natural hydrologic regime.
- 0.5 = Site is sustained by natural source of water but is not associated with a stream, river, or other concentrated flow conduit. For example, the site is sustained by groundwater or urban runoff. There is no evidence of riparian processes, such as overbank flow or scour or deposition.
- 0.7 = Site is within or adjacent to an impoundment on a natural water course which is subject to fluctuations in flow or hydroperiod.
- 1.0 = Site is within or adjacent to a stream, river, or other concentrated flow conduit that provides the primary source of water to the site. The site contains evidence of riparian processes, such as overbank flow or scour or deposition, or is within the flood-prone area (the channel plus the area defined by a horizontal projection at a height of twice the bankfull thalweg; Rosgen 1994).

#### Micro- and Macrotopographic Complexity

In riparian systems, fluvial processes that erode material from some areas and deposit it in others during flood events form dynamic mosaics of active channels, terraces, flood plains, and alluvial fans with different drainage patterns and elevations (Gregory and others 1991). These elevational differences result in the extensive vegetative diversity of riparian systems (Strahan 1984). Riparian flora depends on connectivity between active channels and flood plains for seed dispersal and germination and on base flow resulting from percolation into flood plain soils for survival during the dry season (Warner and Hendrix 1985, Harris and Gosselink 1990, Faber 1993). The ratings for the topographic complexity criterion are assigned based on the following scale:

#### *First- and second-order streams.*

- 0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
- 0.2 = Flood-prone area is characterized by a homogeneous, flat earthen surface with little to no micro- and macrotopographic features.
- 0.6 = Flood-prone area contains micro- and/or macrotopographic features such as pits, ponds, hummocks, bars, rills, large boulders, but is predominantly homogeneous or flat surface.
- 1.0 = Flood-prone area is characterized by micro- and



macrotopographic complexity, such as pits, ponds, hummocks, rills, large boulders, etc.

*Higher order streams.*

- 0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
- 0.2 = Flood-prone area is characterized by a homogeneous, flat earthen surface with little to no micro- and macrotopographic features.
- 0.5 = Flood-prone area contains micro- and/or macrotopographic features such as meanders, bars, braiding, secondary channels, backwaters, terraces, pits, ponds, hummocks, but is predominantly homogeneous or flat surface.
- 0.8 = Flood plain is predominantly heterogeneous, and is characterized by microtopographic features such as pits, ponds, hummocks, bars. However, there are no macrotopographic features, such as braiding, secondary channels, backwaters.
- 1.0 = Flood-prone area is characterized by micro- and macrotopographic complexity, such as meanders, bars, braiding, secondary channels, backwaters, terraces, pits, ponds, hummocks, etc.

Characteristics of Flood-Prone Area

Riparian systems are defined by the geomorphic structure and fluvial characteristics of the valleys in which they exist (Gregory and others 1991). Development of river flood plains and restriction of channel migrations alters the hydrologic regime of riparian systems and severs the critical link between the aquatic habitat and adjacent upland habitat. Alteration of the flood plain reduces overbank flooding, resulting in less seed dispersal and a reduced ability of riparian vegetation to establish (Harris and Gosselink 1990). Kraemer (1984) reported that loss of riparian flood plain along the Sacramento River led to decreased sediment deposition and energy dissipation, resulting in increased flows and less stable streambeds and banks. Once the flood plain is developed, storms result in more overland flow due to impervious surface, but less percolation (Faber 1993). Furthermore, disconnecting rivers from their flood plains reduces their ability to attenuate flood peaks, limits natural sediment deposition and water quality enhancement, and disrupts downstream successional processes and scour cycles (Warner and Hendrix 1985, Harris and Gosselink 1990, Scott and others 1990). Although specific effects vary, in general channel "improvements" cause downstream flood hydrographs to have higher peaks and also cause peaks to occur earlier (DeVries 1980).

The flood-prone area is defined as the bankfull channel plus the area defined by a horizontal projection at a height of twice the bankfull thalweg (Rosgen 1994). This criterion is based on flood-prone area instead of the flood plain because the former represents the area regularly exposed to overbank flow. Although the margins of the flood plain contribute greatly to the ecological function of the riparian system, these areas are often not subject to Corps jurisdiction (in semi-arid systems) and are therefore not the focus of mitigation efforts. The ratings for characteristics of the flood-prone area criterion are assigned based on the following scale.

*First- and second-order streams.*

- 0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
- 0.2 = Channel has an earthen bottom; however, it is structurally confined (e.g., riprap or concrete sideslopes) such that the flood-prone area is within the confined channel and flow would only overtop the channel during extreme events (i.e., greater than a 50-year-flood event).
- 0.4 = Channel has an earthen bottom and earthen sideslopes; however, it is incised or confined such that the channel would only overtop during extreme flow events (i.e., greater than a 50-year-flood event).
- 0.7 = Channel has an earthen bottom and earthen sideslopes and is mildly incised or confined such that the flood-prone area would be subject to periodic overbank flow (i.e., during a 10-year-flood event).
- 1.0 = Site is a natural channel with little to no evidence of incision or confinement.

*Higher order streams.*

- 0 = All flows, including flood flows, are contained in a concrete-lined channel, culvert, etc.
- 0.2 = Channel has an earthen bottom; however, it is structurally confined (e.g., riprap or concrete sideslopes) such that the flood-prone area is wholly contained within the channel and there is no opportunity for overbank flow, except in extreme events.
- 0.3 = Channel has an earthen bottom and earthen sideslopes; however, it is incised or confined such that the flood-prone area is wholly contained within the channel and there is no opportunity for overbank flow, except in extreme events.

- 0.6 = Site is part of a flood plain, which provides an opportunity for overbank flow during moderate flow events (i.e., during a 2- to 10-year-flood event). However, the flood-prone area is confined by levees, berms, dikes, or other obstructions or barriers such that the area available for overbank flow is less than twice the width of the channel at bankfull conditions.
- 0.8 = Site is part of a flood plain, which provides an opportunity for overbank flow during moderate flow events (i.e., during a 2- to 10-year-flood event). The flood-prone area is confined by levees, berms, dikes, or other obstructions or barriers; however, the area available for overbank flow is equal to or greater than twice the width of the channel at bankfull conditions.
- 1.0 = Site is part of an unconfined natural floodplain at least twice the width of the channel at bankfull conditions and there is evidence of overbank flow.

#### Biogeochemical Processes

The location of riparian areas along streams along with the relatively low topography, natural ponding, and ground surface roughness of riparian zones allows them to act as sinks for sediment and nutrient runoff from adjacent uplands and as sources for conversion of detritus to consumable organic matter (Childers and Gosselink 1990, Scott and others 1990). Rising water overtops streambanks, slowing the flow velocity, allowing water and suspended material to access the adjacent flood plain and riparian zones (Gosselink and others 1990a, Scott and others 1990). Microbial action in the root zone removes toxics, nitrogen, and other nutrients from the runoff; thereby improving water quality and helping to reduce the impacts of nonpoint source pollution (Schaefer and Brown 1992). Peterjohn and Correll (1984) reported that each ha (2.47 acres) of riparian forest removed 4.1 mg of particulates, 11 kg of particulate organic nitrogen, 0.83 kg of ammonium-nitrogen, 2.7 kg of nitrate-nitrogen, and 3.0 kg of total particulate phosphorus per year. Gregory and others (1991) reported that up to 65% of the nitrogen and phosphorus can be removed from agricultural runoff by riparian vegetation. Heterotrophic microorganisms that thrive in riparian areas are also responsible for converting detritus from leaf litter and other dead organic matter into consumable organic matter. This organic material forms the base for the riparian food chain and can be released downstream as dissolved organic matter (Gregory and others 1991, Schaefer and Brown 1992). Knight and Bottoroff (1984) reported that up to 1000

g/m<sup>2</sup>/year of detritus are produced by aquatic macrophytes in riparian zones, and this provides a food chain base for these ecosystems, promoting their biodiversity.

Biogeochemical processes depend on water flow through the site, availability of surfaces to slow water and provide a platform for microbial activity and chemical reactions and as sources of organic carbon. Water flow and availability of flood plain surfaces are addressed by the criteria discussed above. The ratings for surface roughness and sources of organic carbon are assigned based on the following scales:

#### *First- and second-order streams.*

- 0 = channel is contained in a concrete-lined channel, culvert, etc., with little to no vegetation or detritus.
- 0.2 = Site can support grasses, forbs, or other herbaceous vegetation, or there is debris, leaf litter, or detritus present in the channel.
- 0.4 = Channel supports at least 5% relative cover of herbaceous or other vegetation and there is at least 10% relative cover of debris, leaf litter, or detritus in the channel.
- 0.6 = Site contains between 5% and 20% relative cover of any type of vegetation and between 10% and 25% relative cover with debris, leaf litter, or detritus.
- 0.8 = Site contains greater than 20% relative cover of any type of vegetation or between 25% and 60% relative cover with debris, leaf litter, or detritus.
- 1.0 = Site contains greater than 20% relative cover of any type of vegetation and greater than 60% relative cover with debris, leaf litter, or detritus.

#### *Higher order streams.*

- 0 = channel is contained in a concrete-lined channel, culvert, etc., with little to no vegetation or detritus.
- 0.2 = Site can support grasses, forbs, or other herbaceous vegetation, and there is woody debris, leaf litter, or detritus present in the channel.
- 0.4 = Channel supports at least 25% relative cover of grasses, forbs, herbaceous, or riparian vegetation, and there is at least 10% relative cover of woody debris, leaf litter, or detritus in the channel.
- 0.6 = Site contains between 25% and 50% relative cover of any strata of riparian vegetation and between 10% and 40% relative cover with woody debris, leaf litter, or detritus.
- 0.8 = Site contains between 50% and 75% relative cover of any strata of riparian vegetation and

between 40% and 60% relative cover with woody debris, leaf litter, or detritus.

- 1.0 = Site contains greater than 75% relative cover of any strata of riparian vegetation and greater than 60% relative cover with woody debris, leaf litter, or detritus.

#### Calculation of Credits and Debits

Credits and debits are calculated based on the number of condition units per ha (CU) gained at the mitigation bank and lost at the impact site. The number of CU per ha is calculated by adding the scores for most of the evaluation criteria. We chose to multiply the scores for the three habitat criteria by the score for the percent of invasive vegetation criterion because infestation with invasive vegetation tends to depress all habitat function in riparian systems. Using the score for percent invasive vegetation as a multiplier precludes the need to specify native versus invasive vegetation under the other habitat criteria. The effect of increased biomass associated with invasive plants on hydrologic and biogeochemical processes is accounted for in the biogeochemical processes criterion. Hydrology is widely recognized as the driving force behind wetland and riparian systems. Therefore, the condition units formula weights the hydrologic regime criterion at three times the importance of other criteria. This reflects the fact that appropriate hydrology is fundamental to overall riparian function, and it devalues sites with artificial or inappropriate hydrology. We recognize there may be some overlap between criteria, for example, the density of vegetation at a site contributes to the rating under both the coverage and spatial diversity and the biogeochemistry criteria; however, we believe this is appropriate because certain characteristics of a site contribute to multiple functions (e.g., habitat and biogeochemical functions). The number of condition units/ha is calculated using the following formula:

$$CU = [(ST + SP + CNT) I] + FPA + TC + BR + 3H$$

where ST = Habitat – Structural Diversity; SP = Habitat – Coverage and Spatial Diversity; CNT = Habitat – Contiguity; I = Percent of Invasive Vegetation; FPA = Characteristics of the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry – vegetation roughness and organic carbon; H = Hydrology.

When performing functional assessments for their own sake, for the purposes of impact evaluation or for design or evaluation of mitigation sites, functions should not be combined into overall indices. The practice of combining functions can result in certain functions

being masked, thereby underestimating the overall importance of a wetland to watershed ecology and decreasing the resolution of the functional assessment. However, the intent of this method is not to evaluate wetland functions but to provide a tool to calculate mitigation ratios based on the ecologic condition of impact and restoration sites. To accomplish this goal in the context of a mitigation bank, we must generate a single number or index.

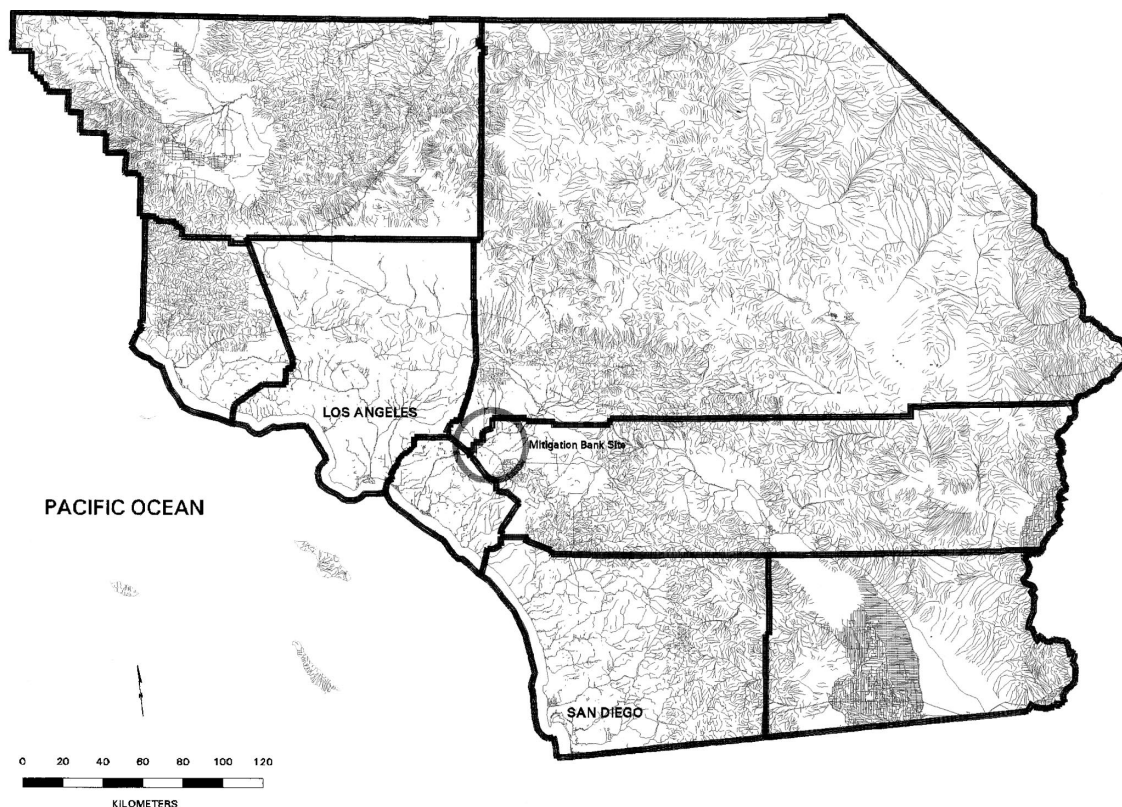
#### Application of the Riparian Model

The Santa Ana River Mitigation Bank (SARMB) provided the earliest application of the southern California's riparian crediting and debiting model. The Santa Ana River, with a watershed of area of 6345 km<sup>2</sup> (2450 miles<sup>2</sup>), is the largest river system in southern California (Hanes 1981). The riparian habitat along the Santa Ana River is southern riparian scrub consisting of *Salix*, *Populus*, and *Baccharis* species. Of the 5667 ha (14,000 acres) of riparian habitat along the Santa Ana River, approximately 2000 ha (5000 acres) are infested with an invasive species commonly known as the giant reed, *A. donax* (Bell 1993). Replacement of the native riparian vegetation with *A. donax* has not only led to loss of suitable habitat for many wildlife species, including the federally listed endangered least Bell's vireo, *Vireo bellii pusillus*, but has also caused problems with water quality and water conservation (Bell 1993, Iverson 1993).

The Santa Ana River mitigation bank is located in the northern portion of Riverside County in the City of Riverside (Figure 1). The goal of the SARMB is to restore a degraded riparian system by reestablishing the native riparian ecological diversity and other riparian functions, such as flood flow alteration, groundwater recharge, improvement of water quality (temperature and organic matter), reduction of fire hazard, and increased recreational use. Credits were established by removal of invasive vegetation and selective planting to encourage natural recruitment of native riparian vegetation.

The size of the initial mitigation bank area was approximately 22.7 ha (56 acres). Since the initial bank establishment, additional areas have been incorporated in the bank, however, the information provided here only reflect the original 22.7 ha. The potential credits were determined using aerial photographs and field surveys. Review of aerial photographs and field surveys revealed three characteristic regions with varying degrees of invasive vegetation infestation within the bank area (Figure 2, Table 2). To determine the available credits, each region of the mitigation bank was rated





**Figure 1.** Location of the Santa Ana River mitigation bank. The area circled shows the location on the Santa Ana River where the mitigation bank is located. Thick lines are county boundaries, fine lines are streams.

separately based on homogeneity of the region or subunit. Post-bank scores were assigned based on the maximum possible score a similar uninfested resource in that region could achieve. The cumulative difference between the baseline rating and the predicted post-bank rating is the total available credits in the mitigation bank.

Region A, which is approximately 12.4 ha (30 acres) (62% of the total area), consists of 100% *Arundo* infestation with no native riparian vegetation and is adjacent to a riparian zone with two species of woody riparian plants and poorly developed understory on one half of the site and a structurally and spatially diverse riparian zone on the remaining half. Because of the high degree of *Arundo* infestation, there is low topographic complexity and diversity of detritus in Region A. Region B is approximately 7.6 ha (26 acres) (38% of the total area), and consists of mixed native riparian vegetation with shrub and herbaceous understory interspersed with 20–40% *Arundo*. Region B is connected to structurally and spatially diverse riparian zone on one half and 100% *Arundo*-infested zone on the remaining half. The topographic complexity and den-

sity of detritus is relatively greater in Region B than in Region A due to the presence of secondary channels and a native riparian vegetation. Region C is not part of the mitigation bank, however, it will be preserved and will function as a buffer between the bank and adjacent land uses. Region C consists of relatively mature riparian species (10–20 years), with a well-developed canopy, diverse understory, and less than 5% *Arundo* present. All three regions possess natural hydrology and are within a flood-prone area greater than twice the width of the active channel at bankful conditions. The scores for pre- and post-bank conditions are shown in Table 3.

The restoration effort began in 1993, when the local community underwent significant threat of fires fueled by *Arundo*. *Arundo* (reaching height of up to 8 m) is a tall grass native to eastern Asia, introduced to southern California in early 1800s for purpose of erosion control. Due to its high rate of growth (5 cm/day), it outcompetes the native riparian vegetation and soon becomes the dominant species in the riparian zone (Bell 1997). The rhizome, which typically reaches depths of 1 m, quickly stabilizes the stream bank and forms terraces severing the riparian zone from fluvial processes typi-

**Figure 2.** Characteristic regions of the Santa Ana River mitigation bank. The map shows the dominant plant community for each subarea of the mitigation bank prior to initiation of any restoration efforts (e.g., baseline conditions). Mapping was based on aerial photography, dated April 1993. Area C was not included in the mitigation bank because it is existing native riparian habitat.

Table 2. Characteristics of regions A & B of the Santa Ana River mitigation bank. Size, percent of total area, and percent infestation with invasive weeds of the two subareas of the Santa Ana River mitigation bank

Region	Size (ha)	% total area	% invasives
A	12.4	62	100
B	7.6	38	20–40

cally occurring within native vegetation-dominated riparian zones. The reduction of overbank flooding limits aggradation and degradation processes consequently limiting native vegetation propagule dispersal. The high growth rate of *Arundo* combined with its high degree of flammability soon redirects the native riparian community to an *Arundo*-infested riparian zone (Bell 1993). The ecological changes that occur within a riparian zone as a result of *Arundo* infestation include

reduction of suitable habitat for native wildlife, highly altered flooding regime, and reduction of biogeochemical processes due to the reduction of surface moisture and presence of noxious chemicals, such as silica, tri-terpines, and sterols (Chanduri and Ghosal 1970, Bell 1997).

Restoration of the Santa Ana River mitigation bank was accomplished using a glyphosate, an EPA-approved herbicide for use in wetlands. The application method varied depending on the extent of *Arundo* infestation and extent of native riparian vegetation present in the treatment areas. The most effective period for applying the herbicide was determined to be during the period when maximum translocation of nutrients to the root is occurring (the period between post-flowering and pre-dormancy) (Bell 1993). Method of application of the herbicide in the SARMB included aerial application (areas >80% *Arundo*), use of all-terrain vehicles (<80% *Arundo*-infested areas easily accessible), and backpack



Table 3. Rating of the two regions in the Santa Ana River mitigation bank. Scores for each criterion for the pre-restoration baseline condition in each subarea. Weighted mean is the average of the criterion score for each subarea adjusted for the subarea's proportion of the total area. Post-project scores reflect the anticipated condition of the site upon maturation of the restoration efforts

Criterion	Region A (62%)	Region B (38%)	Pre-project (weighted mean)	Post-project
ST	0.2	0.6	0.35	1.00
SP	0.2	0.6	0.35	1.00
CNT	0.8	0.8	0.80	0.80
I	0	0.6	0.23	0.80
FPA	0.4	0.4	0.40	0.80
TC	0.2	0.8	0.43	1.0
BR	0.2	0.8	0.43	0.80
H	1.0	1.0	1.0	1.0

Legend: ST = Habitat—Structural Diversity; SP = Habitat—Coverage and Spatial Diversity; CNT = Habitat—Contiguity; I = Invasive Vegetation; FPA = Characteristics and the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry—Vegetation Roughness and Organic Carbon; H = Hydrology.

sprayers (areas difficult to access with vehicles and resprouts). The biomass was cut by hand cutting, chipper, or hydro-ax and removed by hauling to a suitable off-site location or controlled biomass burning. Once the biomass was removed, selective planting was carried out on portions of the bank site to accelerate the natural revegetation process. In the initial 2–3 years treatment or respoutes occurred on a regular basis (2–3 times/year) and continually declined as the native riparian vegetation began to self-recruit and the root mass decomposed. The change in ecological condition of the SARMB became apparent following the third year of treatment as evident by change in characteristics of the flood-prone area, structural diversity of native vegetation, enhancement of topographic complexities (ponds, bars, hummocks, and secondary channels), enhanced biogeochemical processes (due to increase surface moisture and evident by visible microbial activity). The contiguity of the bank area would not be affected as a result of the restoration work, as the site is connected to riparian and upland habitats and no modification is expected to occur in these areas. The SARMB continues to be actively monitored, and it serves as a model for native riparian restoration project throughout California.

Debits at the impact sites are determined by assigning of pre-project and post-project ratings for each criterion. Evaluation of a pre- and post-project at the impact site allows for consideration of any remaining functional characteristics at the impact site following

Table 4. Sample calculation of debits: This table shows a *hypothetical* example of the application of the crediting and debiting framework to an impact site. For this example, the impact would be complete fill of the stream on the project site

Criterion	Pre-project rating	Post-project rating	Net functions lost
ST	0.8	0	0.8
SP	0.8	0	0.8
CNT	0.8	0	0.8
I	0.8	0	0.8
FPA	0.8	0	0.8
TC	1.0	0	1.0
BR	0.8	0	0.8
H	1.0	0	1.0

Legend: ST = Habitat—Structural Diversity; SP = Habitat—Coverage and Spatial Diversity; CNT = Habitat—Contiguity; I = Invasive Vegetation; FPA = Characteristics and the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry—Vegetation Roughness and Organic Carbon; H = Hydrology.

Table 5. Calculation of credits and debits for the Santa Ana River mitigation bank: Sample application of the crediting and debiting framework to determine a mitigation ratio. Credits are determined by using the criteria scores shown in Table 3. Debits are determined by using the hypothetical scores shown in Table 4

#### Step 1. Evaluation of credits

$$CU = [(ST + SP + CNT)I] + FPA + TC + BR + 3H$$

$$\text{Pre-bank } CU = [(0.35 + 0.35 + 0.8)0.23] + 0.4 + 0.61 + 0.43 + 3(1.0) = 4.78$$

$$\text{Post-bank } CU = [(1.0 + 1.0 + 0.80)0.8] + 0.8 + 1.0 + 0.8 + 3(1.0) = 7.84$$

$$\text{Projected Credits Available} = 7.84 - 4.78 = 3.06$$

#### Step 2. Evaluation of debits

$$\text{Debits (Functional Units Lost)} = [(0.8 + 0.8 + 0.8)0.8] + 0.8 + 1.0 + 0.8 + 3(1.0) = 7.52$$

#### Step 3. Determination of mitigation ratio

$$\text{Mitigation Ratio} = 7.52/3.06 = 2.45$$

Legend: ST = Habitat—Structural Diversity; SP = Habitat—Coverage and Spatial Diversity; CNT = Habitat—Contiguity; I = Invasive Vegetation; FPA = Characteristics and the Flood-prone Area; TC = Topographic Complexity; BR = Biogeochemistry—Vegetation Roughness and Organic Carbon; H = Hydrology.

implementation of a project. A hypothetical debiting score scenario to be mitigated at this bank is presented in Table 4, where a project would impact an aquatic resource with a relatively high functional characteristics. The mitigation ratio for the hypothetical debiting scenario for use of the SARMB is calculated to be 2.5:1 (Table 5). This mitigation ratio is based on withdrawal of credits when the credits have reached their expected

Table 6. Credits available during the first 5 years of operation: Discounting of the mitigation ratio (as determined in Table 5) for years 1 through 5. This hypothetical scenario assumes the mitigation site achieves the performance goals by year 5

Year	Total projected credits	Credits available	Mitigation ratio
1	3.06	$3.06 \times 0.2 = 0.61$	$7.52/0.61 = 12.3$
2	3.06	$3.06 \times 0.4 = 1.22$	$7.52/1.22 = 6.2$
3	3.06	$3.06 \times 0.6 = 1.84$	$7.52/1.84 = 4.1$
4	3.06	$3.06 \times 0.8 = 2.45$	$7.52/2.45 = 3.1$
5	3.06	$3.06 \times 1.0 = 3.06$	$7.52/3.06 = 2.5$

restoration potential, which is estimated to take approximately 5 years.

Although mitigation banks should conceptually reach their predicted functional maturity prior to withdrawal of credits, there may be a need to withdraw credits prior to achievement of functional maturity. It should be noted that in these circumstances financial assurance must be secured by the bank sponsor. This crediting and debiting method gives flexibility in withdrawal of credits prior to full functional establishment of an aquatic resource by allowing adjustment of the mitigation ratio to reflect the existing conditions at the mitigation bank. The actual condition of the mitigation bank may be evaluated at a given time interval and the percentage of total expected credits may become available for withdrawal. A simplified example may be a mitigation bank where at the end of the first year credits have 20% of their maximum potential value, 40% of the total potential value in the second year, 60% of potential value in the third year, and 80% of the predicted value at the end of the fourth year (Table 6). Credits would have their full functional maturity at the end of the fifth year. This would allow sale of credits at a partial value to provide funds for the sponsor within the initial establishment period. In addition, if credits have only partial value in the initial 5 years, the mitigation ratios obtained should be high enough to deter the use of this mitigation bank for projects with impacts to riparian habitats with high functional capacity and consequently should encourage avoidance and minimization of impacts to these habitats. By setting a minimum compensation ratio of 1:1 the crediting and debiting methodology prevents loss of acreage.

## Discussion

Despite the existence of numerous methods for assessing functions of aquatic resources, compensation ratios are typically determined based on existing policy

and/or best professional judgment of decision makers. Existing mitigation banks typically use either an acreage-based or case-by-case best professional judgment determination of functional characteristics and compensation ratios (IWR 1994, Tabatabai 1994). As mitigation banking gains support from the regulated public, entrepreneurs, and the resource and regulatory agencies, a greater need arises for use of an appropriate crediting and debiting methodology in any mitigation bank. Use of detailed functional assessment methodologies or site-specific evaluation of function for determination of credits and debits is far superior to rapid approaches, such as the one we present in this paper. However, the constraints posed by their application (e.g., time and resources) makes their use impractical in the mitigation banking context. The goal of this framework is to provide a rapid assessment of credits and debits that does not require extensive field data collection and where the assessment of structural components of an aquatic resource could be used by nonwetland scientists. The proposed framework meets the six objectives necessary for a crediting and debiting system to be useful in a regulatory context.

(1) It can be tailored to evaluate ecologic condition based on the target resources of a specific mitigation bank. The example presented in this paper illustrates application of the proposed framework to a mitigation bank where the goal is to restore riparian habitat. This framework is currently being applied to a bank where the goal is restoration of vernal pools; therefore, specific criteria have been developed that reflect the ecologic conditions of depressional wetlands. For example, one of the evaluation criteria addresses duration of ponding and is scaled as follows:

- 0 = Ponding is transient following storm events and persists for no more than 1 day.
- 0.2 = Site may pond water for several days following storm events; however, ponding seldom persists beyond 10 days. There may be several ponding events during a season.
- 0.4 = Ponding duration is on the order of several weeks. There may be several ponding events during a season.
- 0.6 = Ponding duration is on the order of several months, but less than 6 months. There may be several ponding events during a season.
- 0.8 = On average, site ponds water for more than 6 months.
- 1.0 = Site ponds water year-round.

(2) It is flexible enough to be used for evaluation of existing or potential ecologic condition at a mitigation

bank. Because credits are determined based on the difference between structural characteristics of the post-restoration condition and pre-restoration (baseline) condition at the bank site, the framework can be applied in a predictive manner. A CU score can be calculated based on the expected future condition at the bank and used as the "enhancement potential" for the purpose of determining mitigation ratios. The success of the restoration can be evaluated by comparing the condition of the resources over time to the expected future condition, and remedial measures can be implemented to ensure the target condition is achieved. This crediting and debiting method also provides the flexibility to account for withdrawal of credits prior to full functional establishment of an aquatic resource by allowing adjustment of the mitigation ratio to reflect the condition of the resources at the mitigation bank at time of purchase of credits.

(3) It is a structured and systematic way to apply data and professional judgment to the decision-making process. The intent of the proposed framework is not to provide an absolute tool for evaluating functional condition. However, it does provide an alternative to subjectively applied best professional judgment by establishing a structure to organize information and apply judgment in an objective manner, based on ecological principles. For example, in instances where a mitigation banking agreement contains limits on the quality of riparian habitat that can use the bank for mitigation, this framework can be used to determine whether the "quality" of the resources at the impact site exceeds the stated threshold. Several proposed banks in the Los Angeles District of the Corps involve restoration or enhancement of existing aquatic resources and have stipulations in the banking instruments that only allow impacts to degraded habitats to be mitigated at the bank. In these cases, a proposed project site which receives a pre-project rating of 4 CUs or greater would be precluded from using the bank. This provides an objective way to ensure that the functions gained through the mitigation bank are commensurate with the impacts for which credits were purchased.

(4) It has an ecologically defensible basis. Credits and debits are based on the structural characteristics and landscape setting of the restoration and the impact sites. The evaluation criteria reflect attributes of wetlands shown to be important to their viability and ability to provide a suite of ecologic functions. Many of the criteria presented for riparian systems are similar to one used in established functional evaluation methods, such as HGM and HEP.

(5) It has ease of use such that the level of expertise and time required to employ the method is not a

deterrent to its application. The evaluation criteria have been structured so that they can be applied based on information typically provided in biological resource reports that accompany U.S. Army Corps of Engineers permit applications. Because the criteria are generally descriptive, they can be applied with minimal ambiguity based on information supplied by permit applicants. When being applied in the field, scores can be assigned with a reasonable amount of data collection, yet not so intensive as to be a deterrent to its use. In practice, most sites can be evaluated by review of aerial photography and several hours in the field. This is commensurate with the time and resource constraints of the regulatory program.

(6) It provides a semi-quantitative measure of the condition of aquatic resources that can be translated to a mitigation ratio. Scaling of the evaluation criteria is based on a combination of field data collected during development of a regional HGM assessment model (Lee and others 1997), research on the success of past mitigation projects in southern California (Sudol 1996), and professional judgment of scientists familiar with semi-arid riparian systems. The framework organizes this information in a categorical manner and provides a way to translate information about the conditions of a site to a quantitative mitigation ratio. The framework may also be applicable for determination of out-of-kind compensation ratios. Aquatic resources can be evaluated based on criteria developed for each specific wetland type. The relative conditions can then be translated into a common currency or unit of measure and the compensation ratio assigned using the ratio of debits over credits.

The crediting and debiting framework presented in this work is not designed as a functional assessment methodology; rather, it is intended to be a rapid semi-quantitative measure of structural characteristics of an aquatic resource for the purpose of determining compensatory requirements. The framework, as demonstrated with the southern California riparian model, offers an alternative to use of existing functional assessment methodologies or best professional judgment for determination of credits and debits. We encourage more dialogue and debate among scientists, regulators, and the public on the merits of this approach and the details of its applications, such as choice of indicators, scaling of criteria, and architecture of the CU formula. The ultimate goal should be an objective and systematic way to determine ecologically meaningful mitigation requirements that are commensurate with the impacts and result in a net benefit for the resources protected by the Clean Water Act.

## Acknowledgments

We extend our sincere appreciation to the U.S. Army Corps of Engineers, Los Angeles District, for allowing access to the data used in this study. The comments of Professor L. D. Duke on development of the framework are greatly appreciated. We also thank Aaron Allen, Gary Bell, and Paul Frandsen for valuable discussions and assistance. Permission to publish this paper was granted by the Chief of Engineers, U.S. Army Corps of Engineers.

## Literature Cited

- Adamus, P. R. 1983. A method for wetland functional assessment. FHWA-IP-82-23, US Department of Transportation, Federal Highway Administration, Washington DC.
- Barnett, A. M., T. D. Johnson, and R. Appy. 1991. Evaluation of mitigative value of an artificial reef relative to open coast sand bottom by biological evaluation standardized technique (BEST). Pages 231–237 in M. Nakamura and others (eds.), Recent advances in aquatic habitat technology. Tokyo, Japan.
- Bartoldus, C. C., E. W. Garbisch, and M. L. Kraus. 1992. Wetland replacement evaluation procedure (WREP). Environmental Concerns, Inc., St. Michaels, MD.
- Bell, G. 1993. Biology and growth habits of giant reed. Pages 1–6 in Proceedings of *Arundo donax* Workshop, 19 November 1993, Ontario.
- Bell, G. 1997. Ecology and management of *Arundo donax*, and approaches to riparian habitat restoration in southern California. Pages 103–113 in J. H. Brock, M. Wade, P. Pysek, and D. Green (eds.), Plant invasions: studies from North America and Europe. Bachuys Publishers, Leiden, The Netherlands.
- Chanduri, R. K., and S. Ghosal. 1970. Triterpenes and steroids from the leaves of *Arundo donax*. *Phytochemistry* 9:1895–1896.
- Childers, D. L., and J. G. Gosselink. 1990. Assessment of cumulative impacts to water quality in a forested wetland landscape. *Journal of Environmental Quality* 19:65–70.
- Croonquist, M. J., and R. P. Brooks. 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian wetland areas. *Environmental Management* 15:701–714.
- DeVries, J. J. 1980. Effects of floodplain encroachments on peak flow. The Hydrologic Engineering Center, US Army Corps of Engineers Water Resources Support Center, Davis, CA.
- Diamond, J. M. 1975. The island dilemma: lessons of modern biogeographic studies for the design of nature reserves. *Biological Conservation* 7:129–146.
- Dickert, T. G., and A. E. Tuttle. 1985. Cumulative impact assessment in environmental planning. A coastal watershed example. *Environmental Impact Assessment Review* 5:37–64.
- Doyle, A. T. 1990. Use of riparian and upland habitats by small mammals. *Journal of Mammalogy* 71:14–23.
- Environmental Law Institute. 1993. Wetland mitigation banking. An Environmental Law Institute Report, Washington DC.
- Erman, N. 1984. The use of riparian systems by aquatic insects. In R. E. Warner and K. M. Hendrix (eds.), California riparian systems; ecology, conservation, and productive management. University of California Press, Berkeley, CA.
- Faber, P. M., and R. F. Holland. 1988. Common riparian plants of California. Pickleweed Press, Mill Valley, CA.
- Faber, S. E. 1993. Letting down the levees. *National Wetlands Newsletter* 15 (6):5–7.
- Federal Register. 1995. Federal guidance for the establishment, use and operation of mitigation banks. Volume 60, no. 228:58605–58614.
- Frankel, O. H., and M. E. Soule. 1981. Conservation and evolution. Cambridge University Press, Cambridge.
- Gosselink, J. G., B. A. Touchet, J. Van Beek, and D. Hamilton. 1990a. Bottomland hardwood forest ecosystem hydrology and the influence of human activities: the report of the hydrology workgroup. In J. G. Gosselink and others (eds.), Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems. Lewis Publishers, Chelsea, MI.
- Gosselink, J. G., M. M. Brinson, L. C. Lee, and G. T. Auble. 1990b. Human activities and ecological processes in bottomland hardwood ecosystems: the report of the ecosystem workgroup. In J. G. Gosselink and others (eds.), Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems. Lewis Publishers, Chelsea, MI.
- Gosselink, J. G., G. P. Shaffer, L. C. Lee, D. M. Burdick, D. L. Childers, N. C. Leibowitz, S. C. Hamilton, R. Boumans, D. Cushman, S. Fields, M. Koch, and J. M. Visser. 1990c. Landscape conservation in a forested wetland watershed. *Bioscience* 40(8):588–600.
- Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. *Bioscience* 41(8):540–551.
- Hanes, T. L. 1981. Vegetation of the Santa Ana River. Pages 882–888 In R. E. Warner and K. M. Hendrix (eds.), California riparian systems. University of California Press, Berkeley, CA.
- Harris, L. D. 1988. The nature of cumulative impacts on biotic diversity of wetland vertebrates. *Environmental Management* 12(5):675–693.
- Harris, L. D., and J. G. Gosselink. 1990. Cumulative impacts of bottomland hardwood forest conversion on hydrology, water quality, and terrestrial wildlife. In J. G. Gosselink and others (eds.), Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems. Lewis Publishers, Chelsea, MI.
- Institute for Water Resources. 1992. National wetland mitigation banking study. Wetland mitigation banking concepts. IWR Report 92-WMB-1, Alexandria, VA, 25 pp.
- Institute for Water Resources. 1994. Wetland mitigation banking: resource document. IWR Report 94-WMB-2, Alexandria, VA, 131 pp.



- Iverson, M. 1993. The impact of *Arundo donax* on water resources. Pages 19–26 in *Proceedings of Arundo donax workshop*, 19 November 1993. Ontario.
- Jain, R. K., L. V. Urban, G. S. Stacey, and H. E. Balbach (eds.). 1993. *Environmental assessment*. McGraw-Hill, Inc., New York.
- Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resources management. *Ecological Applications* 1(1): 66–84.
- Klopatek, J. M. 1984. Some thoughts on using a landscape framework to address cumulative impacts on wetland food chain support. *Environmental Management* 12(5):703–711.
- Knight, A. W., and R. L. Bortoroff. 1984. The importance of riparian vegetation to stream ecosystems. Pages 160–167 in R. E. Warner and K. M. Hendrix (eds.), *California riparian systems: ecology, conservation, and productive management*. University of California Press, Berkeley, CA.
- Knopf, F. L., R. R. Johnson, T. Rich, F. B. Samson, and R. C. Szaro. 1988. Conservation of riparian ecosystems in the United States. *Wilson Bulletin* 100(2):272–284.
- Kraemer, T. J. 1984. Sacramento River environment: a management plan. In R. E. Warner and K. M. Hendrix (eds.), *California riparian systems: ecology, conservation, and productive management*. University of California Press, Berkeley, CA.
- Lee, L. C., M. C. Rains, J. A. Mason, and W. J. Kleindl. 1997. Guidebook to hydrogeomorphic functional assessment of riverine waters/wetlands in the Santa Margarita watershed. Seattle, WA.
- Lonard, R. I., and E. J. Clairain. 1985. Identification of methodologies for the assessment of wetland functions and values. In J. A. Kusler and P. Riexinger (eds.), *Proceedings of the National Wetland Assessment Symposium*. Portland, ME.
- Margules, C., and M. B. Usher. 1981. Criteria used in assessing wildlife conservation potential: a review. *Biological Conservation* 21:79–109.
- Mitsch, W. J., and J. G. Gosselink. 1993. *Wetlands*. Van Nostrand Reinhold, New York.
- Mitsch, W. J., and R. F. Wilson. 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. *Ecological Applications* 6(1):77–83.
- Noss, R. F. 1987. Corridors in real landscapes: a reply to Simberloff and Cox. *Conservation Biology* 1(2):159–164.
- Pearsall, S. H., D. Durham, and D. C. Eager. 1986. Evaluation methods in the United States. In M. B. Usher (ed.), *Wildlife conservation evaluation*. Chapman and Hall, New York.
- Peterjohn, W. T., and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of riparian forest. *Ecology* 65(5):1466–1475.
- Reichenbacher, F. W. 1984. Ecology and evolution of southwestern riparian plant communities. *Desert Plants* 6(1):15–22.
- Rosgen, D. L. 1994. A classification of natural rivers. *Catena* 22:169–199.
- Schaefer, J. M., and M. T. Brown. 1992. Designing and protecting river corridors for wildlife. *Rivers* 3(1):14–26.
- Scott, M. L., B. A. Kleiss, W. H. Patrick, and C. A. Segelquist. 1990. The effect of developmental activities on water quality functions of bottomland hardwood ecosystems: the report of the Water Quality Workgroup. In J. G. Gosselink and others (eds.), *Ecological processes and cumulative impacts: illustrated by bottomland hardwood wetland ecosystems*. Lewis Publishers, Chelsea, MI.
- Scott, M. L., M. A. Wondzell, and G. T. Auble. 1993. Hydrograph characteristics relevant to the establishment and growth of western riparian vegetation. In H. J. Morel-Seyteux (ed.), *Proceedings of the thirteenth annual American Geophysical Union Hydrology Days*. Hydrology Days Publications, Atherton, CA.
- Short, C. 1988. Mitigation banking. In US Department of Interior, Fish and Wildlife Services. *Biological Report* 88(41): 1–103.
- Smith, R. D. 1993. A conceptual framework for assessing the functions of wetlands. Technical Report WRP-DE-3, US Army Corps of Engineers Waterways Experimental Station, Vicksburg, MS.
- Smith, R. D., A. Ammann, C. Bartoldus, and M. Brinson. 1995. An approach for assessing wetland functions using hydrogeomorphic classification, reference wetlands, and functional indices. U. S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS, USA. Technical Report WRP-DE-9.
- Stanford, J. A., and J. V. Ward. 1993. An ecosystem perspective of alluvial rivers: connectivity and hyporheic corridor. *Journal of the North American Benthological Society* 12(1):48–60.
- Stein, E. D. 1995. Assessment of the cumulative impacts of section 404 Clean Water Act permitting on the ecology of the Santa Margarita, Ca watershed. Ph.D. diss., University of California, Los Angeles, CA.
- Strahan, J. 1984. Regeneration of riparian forests of the central valley. In R. E. Warner and K. M. Hendrix (eds.), *California riparian systems: ecology, conservation, and productive management*. University of California Press, Berkeley, CA.
- Sudol, M. F. 1996. Success of riparian mitigation as compensation for impacts due to permits issued through section 404 of the Clean Water Act in Orange County, California. Ph.D. diss., University of California, Los Angeles, CA.
- Szaro, R. C. 1990. Southwestern riparian plant communities: site characteristics, tree species distribution, and size-class structures. *Forest Ecology and Management* 33/44:315–334.
- Tabatabai, F. 1994. Wetland mitigation banking: investigation of an innovative approach to off-site compensatory mitigation. Ph.D. diss., University of California, Los Angeles, CA.
- US Army Corps of Engineers (US ACOE). 1988. The Minnesota wetland evaluation methodology for the north central United States. US ACOE Planning Division, Minnesota District, MN.
- US Department of Interior (US DOI). 1994. The impact of federal programs on wetlands. In Volume II, A report to Congress by the secretary of the Interior. Washington, DC.
- US Environmental Protection Agency (US EPA). 1992. A synoptic approach to cumulative impact assessment. A



- proposed methodology. EPA/600/R-92/167, Environmental Research Laboratory, Corvallis, OR.
- US Fish and Wildlife Service (US FWS). 1980. Habitat evaluation procedures, ecological services manual. No. 102-ESM I, Fish and Wildlife Service, Department of Interior, Washington, DC.
- Warner, R. E. 1984. Structural, floristic, and condition inventory of central valley riparian systems. *In* R. E. Warner and K. M. Hendrix (eds.), California riparian systems; ecology, conservation, and productive management. University of California Press, Berkeley, CA.
- Warner, R. E., and K. M. Hendrix. 1985. Riparian resources of the central valley and California desert. California Department of Fish and Game, Sacramento, CA.
- Westman, W. E. 1985. Ecology, impact assessment, and environmental planning. John Wiley and Sons, New York.