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USING AN IBI TO ASSESS EFFECTIVENESS OF MITIGATION MEASURES TO REPLACE LOSS OF A WETLAND-STREAM ECOSYSTEM

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Abstract: Approximately 7.3 hectares of wetlands, composed of six separate cells, were created to mitigate the loss of a 6-hectare, beaver-influenced, wetland-stream complex destroyed by the construction of a multi-purpose impoundment in the Cedar Run watershed in Fauquier County, Virginia, USA. The mitigation action physically replaced the lost wetlands and was judged successful in meeting planned objectives and regulatory requirements (which did not include standards for biota). A pre-project fish survey conducted in 1974 in the wetland-stream complex and three nearby streams provided a baseline condition from which to assess project impacts on fish, as determined from yearly surveys in the cells and the stream reach immediately upstream. In addition, fish communities were sampled at 157 stream locations within the northern Virginia Piedmont from 1997 to 1999 to establish a regional Index of Biotic Integrity (IBI) based on fish assemblages. A modification of that IBI was developed to assess the effectiveness of the mitigation based on 22 stream segments that were heavily influenced by beaver. Pre- and post-project conditions were assessed by gauging them against the wetland-stream complexes using this IBI. The IBI score for the mitigation area dropped from the pre-project 34 to 18 the first year after construction and ranged from 18 to 28 over the ten-year post-project monitoring period. A reduction in the number of native species was observed, and there was a dramatic shift in composition and relative abundance within key species groups. In general, the mitigation benefited species favoring lentic environments over those preferring lotic environments and had negative effects on trophic and habitat specialists and less tolerant species. Scores for the mitigation cells were lower than scores for the original wetlands for the following IBI metrics: number of darter species, number of minnow species, percent of the assemblage comprised of the single most dominant species, percent of tolerant individuals, percent of benthic invertivores, and percent of specialist carnivores minus tolerants. Upstream reach IBI scores also diminished over the same 10-year period, although more gradually. The IBI showed that, despite meeting all regulatory requirements, the mitigation failed to replace the original fish community in the wetland-stream complex and adversely impacted additional stream habitat. Using tools such as an IBI to monitor biological condition can help planners effectively mitigate unavoidable project impacts and avoid the unintended loss of important natural resources caused by compensatory mitigation actions.

Key Words: beaver pond, fish assemblage, index of biotic integrity, inundation effects, mitigation, wetland assessment, wetland condition, wetland impairment, Virginia

INTRODUCTION

Despite the current emphasis on wetland conservation (Clean Water Act 1972, White House Office of Environmental Policy 1993), wetlands continue to be lost in the United States due to development, agriculture, impoundment construction, mining, and other causes (USDA SCS 1992, Dahl et al. 1998). Because many wetland functions are valued by society, compensatory mitigation is often employed to replace wetland function lost due to human impact. For over two decades, there have been numerous federal and state requirements for the mitigation of adverse impacts to wetlands (Clean Water Act 1972, Krulitz 1979) that result from public and private projects, as well as programs that promote the voluntary restoration of wetlands on private lands, such as USDA's Wetlands Reserve Program. Unfortunately, many wetland restoration and creation efforts have failed to produce wetland functions or the biological conditions typical of healthy wetlands (Kusler and Kentula 1991, Zedler and Langis 1991). In fact, the few studies that have monitored the results of compensatory mitigation efforts have demonstrated that there is considerable variation in the success of mitigation and that there is substantial room for improvement (Kusler and Kentula 1991, Leibowitz et al. 1992). For example, a 1994 study of freshwater wetland restoration and creation sites in the state of Washington concluded that 65% of the wetlands examined demonstrated poor ecological function (U.S. EPA and U.S. FWS 1994). Furthermore, the National Academy of Sciences report on the effectiveness of compensatory mitigation concluded that ''the goal of no net loss of wetlands is not being met for wetland functions by the mitigation program, despite progress in the last 20 years'' (National Research Council 2001). Mitigation wetlands that are physically created or restored but fail to support diverse floral and faunal communities are not ecologically sound, and neither the environment nor the public realize the full potential benefit from such mitigation.

Wetland ecosystems are spatially and temporally complex, often composed of wetland types to which a variety of plants and animals are adapted. Assessing the condition of these systems can be difficult, as no one technique can easily account for all wetland components or their complex interactions. Accurate assessment of a wetland's condition requires a method that integrates ecological responses to environmental stressors through the examination of patterns and processes from individual to ecosystem levels (Karr et al. 1986). Without such assessment methods, it is difficult to understand the combined impacts of human activities or develop appropriate mitigation alternatives.

The mitigation action evaluated in this study was taken in response to wetland losses associated with the construction of a multi-purpose reservoir in 1992. The impoundment resulted in the destruction of about 6 hectares of a beaver-influenced, wetland-stream complex. To offset this loss and to meet Clean Water Act mitigation requirements, approximately 7.3 hectares of wetlands were created immediately upstream from the impoundment by constructing a series of wetland cells on the stream. One of the permit conditions for this action required the mitigation wetlands to be monitored and evaluated for three to five years. From 1993 to 2001, we monitored the fish community in each of the mitigation cells and in the reach of Cedar Run immediately upstream. Pre-project (1974) fish population data are available for the wetland-stream complex and several other stream segments of Cedar Run, as reported in the Final Plan of and Environmental Impact Statement of the Cedar Run Watershed Project (USDA SCS 1975).

A number of methods have been developed to evaluate the effects of water-resource development activities on wetlands. Most methods focus on measuring physical features (e.g., topography, depth of water, number and size of trees) related to specific wetland functions (e.g., storage of surface water, removal of pollutants, and provision of physical habitat) (Adamus 1983, Ammann et al. 1986, Brinson 1993), but evaluation of the biological attributes of wetlands may be equally important. For example, Zedler (in Jordan 1998) points out the current emphasis on engineering in wetland mitigation projects and the strong need to pay more attention to biology, not only in the design of the mitigation but in evaluating project success or failure. Danielson (1998) stated that, in most cases, the most direct and effective way to assess the ''health'' or biological condition of wetlands is first to measure directly the condition of their biological communities and then, to augment those measurements by assessing physical and chemical condition of the wetland and its watershed.

One widely used technique to assess the biological condition of streams is the index of biotic integrity (IBI) (Karr et al. 1986), which relies on metrics depicting characteristics of faunal assemblages such as fish. The IBI has not only been used to assess conditions of streams and their watersheds in general (Fausch et al. 1990, Roth et al. 1996, Wang et al. 1997) but also to assess the ecological impacts of specific human disturbances (Berkman et al. 1986, Leonard and Orth 1986, Hughes and Gammon 1987, Steedman 1988). Various versions of the IBI are currently used in nearly all of North America (Davis et al. 1996). Due to its wide use and acceptance for stream fish, its ability to assess ecosystem recovery (Hughes et al. 1990), and because the fish fauna was emphasized in the original Cedar Run environmental assessment, we chose an IBI as the method on which to base our evaluation of the mitigation performed. Specifically, we modified an IBI developed for small streams in the Piedmont of northern Virginia (Teels and Danielson 2001) to characterize wetland-stream complexes similar to the impacted site in order to evaluate the biological condition of the original wetland-

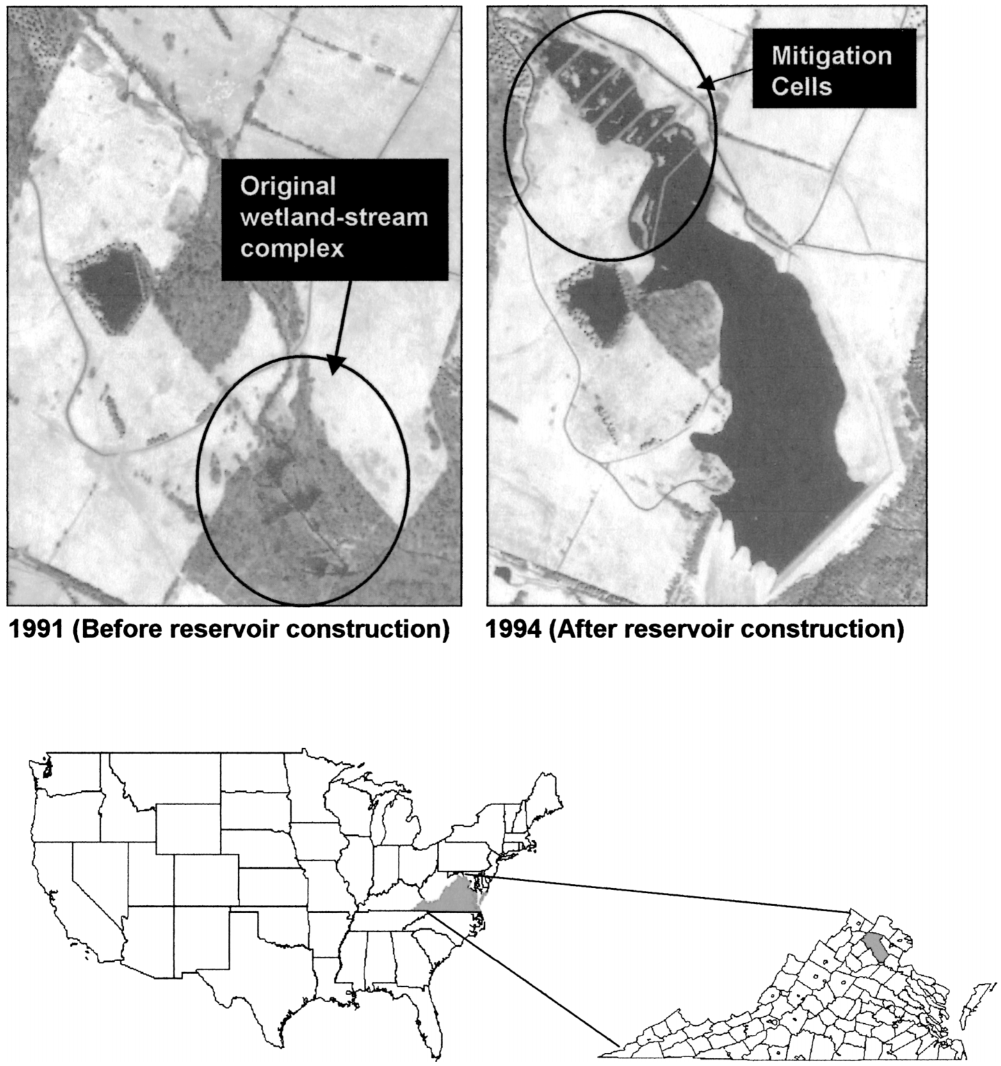

Figure 1. Before (4/3/91) and after (4/2/94) aerial photos of project area.

stream complex and to assess the efficacy of compensatory mitigation performed.

METHODS

Study Site

The site is located approximately 3 km north of Warrenton, Virginia, USA near the headwaters of Cedar Run, a small (9.1 km^2 watershed) tributary of the Occoquan River. Before construction, the affected area consisted of a complex of several small beaver ponds and adjacent saturated wetlands dominated by black willow (*Salix nigra* Marshall) and hazel alder (*Alnus serrulata* Willd.). The area was traversed by a network of interconnected stream channels, forming a mix of lotic and lentic habitats along an approximately 0.5-km reach of Cedar Run. Construction of the reservoir in 1992 entirely inundated the original wetland-stream complex and 0.3 km of free-flowing stream habitat (Figure 1). The project's mitigation was completed that same year by creating six adjacent impoundments (cells) immediately upstream from the water supply structure to mimic the beaver ponds. The cells were formed by constructing a series of rock rip-rap armored embankments across the floodplain to create 7.3 hectares of wetland habitat in six separate shallow-water pools varying in size from 0.5 to 2.4 hectares. Each cell was designed to provide three distinct habitat types: open water, semi permanently-inundated wetlands, and terrestrial islands. Some excavation was performed during cell construction, creating conditions that were generally deeper and less well-vegetated than the original wetland complex. Wetland vegetation was established in the cells by relying on seed banks in the soils excavated from the original wetlands and transferred to the mitigation area. With the construction of the cells, the project's mitigation plan was considered implemented and all regulatory requirements were met.

Fish Sampling

Since seines have been effectively used to sample fish in small, relatively simple streams (Karr et al. 1986), we used hand-held seines to sample fish for this study. Seining was conducted using a 2.4-m (width) by 1.8-m (depth) ''minnow'' seine. All sampling was performed by a 3- to 4-person crew, with the primary investigator present to assist with species identification and to ensure uniform application of the seining technique.

A pre-project fish survey was conducted by the Natural Resources Conservation Service (NRCS—formerly the Soil Conservation Service) in 1974 at the site and at several other stream reaches within the Cedar Run watershed as part of the project's original environmental assessment. While descriptions of the fish sampling methods for the 1974 surveys could not be obtained, sampling locations were recorded precisely. Three of these pre-project watershed data-collection points were selected as comparison sites and were sampled along with the mitigation cells and upstream segment once per year from 1993 to 2002. The mitigation cells were sampled by seining the front and back slopes of each of the embankments and alternating sides of each of the cells. Data from each of the cells were pooled to represent a single yearly sample for all of the cells combined. The upstream segment and comparison sites were sampled using the time-based seining approach described in (Teels and Danielson 2001).

The IBI

The need to test and validate biological responses (metrics) across a gradient of human disturbance is a core assumption of the IBI (Karr and Chu 1997). Although the IBI is widely used, it is not a widely standardized method. Essentially, a unique IBI must be developed for each regional faunal assemblage based on observable biological responses in the area and reference conditions derived from the region's least impaired streams/wetlands. Teels and Danielson (2001) developed an IBI for this region based on a 1997–1999 fish survey that included 157 separate stream reaches on tributaries of the Occoquan River, Goose Creek, and upper Rappahannock River located in the Piedmont physiographic region of northern Virginia. From that 157-site total, 22 stream reaches that were heavily influenced by beaver were used to form the reference conditions for the pre-construction wetland-stream complex and mitigation cells of this study.

A corresponding Human Disturbance Index (HDI) developed by Teels and Danielson (2001) for the region was used to test and validate metric performance. The HDI was based on land-use variables and on-site assessment of stream characteristics using the Stream Visual Assessment Protocol (USDA, NRCS 1998). The HDI was used to evaluate attributes of the fish assemblage to determine which metrics to include in the regional IBI. Metrics validated in the regional IBI were used for this study; however, scoring of these metrics was based on the 22 beaver-influenced stream reaches. The sequence of activities used for developing the regional IBI are described in Teels and Danielson (2001). Biological groupings (guilds) follow (Teels and Danielson 2001) (Table 1) and metrics for the IBI are summarized below.

Number of native species: the total number of species per sample less the species considered to be non-native or probably non-native by Jenkins and Burkhead (1993) for the receiving drainage (Potomac) of the project area.

Number of darter species: the number of species per sample of the genera *Percina* or *Etheostoma*.

Number of minnow species: the number of species of the family *Cyprinidae* per sample.

Percent of the dominant species: the percent of individuals per sample comprised of the single most abundant species.

Number of intolerant species: the number of species per sample considered to be intolerant to the combined effects of human disturbance in the northern Virginia Piedmont.

Percent tolerant individuals: the percent of individuals per sample of species considered to be tolerant to the combined effects of human disturbance in the northern Virginia Piedmont.

Percent omnivorous individuals: the percent of individuals per sample of species that as adults feed across the three food groups of algae, plants, and invertebrates (AHI; Table 1) as designated by Smogor (1996).

Percent benthic invertivores: the percent of individuals per sample that are considered benthic (Ben; Table 1) and as adults feed predominantly on invertebrates (Inv; Table 1).

Percent specialist carnivores minus tolerant pecies: the percent of individuals per sample comprised of species designated as piscivores (Pisc; Table 1) or invertivore/piscivores, excluding individuals from tolerant species.

Percent simple lithophilic spawners minus tolerant species: the percent of individuals per sample comprised of species that scatter their eggs over rock, rubble, or gravel substrates (Lith; Table 1) without nest

Table 1. Biological groupings for fish species collected in the Occoquan Watershed from 1997 to 2001 (Teels and Danielson 2001).

Common Name	Scientific Name	Tol[1]	Non-native	Trophic[2]	Ben[3]	Lith[4]	Late-maturing
Gizzard shad	*Dorosoma cepedianum* (Lesueur)			AHI			
Redfin pickerel	*Esox americanus* Gmelin	I		PIS			
Eastern mudminnow	*Umbra pygmaea* (DeKay)			INV			
Common carp	*Cyprinus carpio* Linnaeus		x	AHI			
Golden shiner	*Notemigonus chrysoleucas* (Mitchill)			AHI			
Rosyside dace	*Clinostomus funduloides* Girard			INV		x	
Fallfish	*Semotilus corporalis* (Mitchill)			IP			
Creek chub	*Semotilus atromaculatus* (Mitchill)	T		IP			
River chub	*Nocomis micropogon* (Cope)			INV			x
Cutlips minnow	*Exoglossum maxillingua* (Lesueur)			INV			
Blacknose dace	*Rhinichthys atratulus* (Hermann)	T		INV		x	
Longnose dace	*Rhinichthys cataractae* (Valenciennes)			INV	x	x	
Eastern silvery minnow	*Hybognathus regius* Girard			AHI			
Common shiner	*Luxilis cornutus* (Mitchill)			INV		x	
Satinfin shiner	*Cyprinella analostana* Girard			INV			
Spotfin shiner	*Cyprinella spiloptera* (Cope)			INV			
Bluntnose minnow	*Pimephales notatus* (Rafinesque)	T	x	AHI			
Fathead minnow	*Pimephales promelas* Rafinesque		x	AHI			
Comely shiner	*Notropis amoenus* (Abbott)			INV		x	
Spottail shiner	*Notropis hudsonius* (Clinton)			INV			
Swallowtail shiner	*Notropis procne* (Cope)			INV		x	
Rosyface shiner	*Notropis rubellus* (Agassiz)	I		INV		x	
White sucker	*Catostomus commersoni* (Lacepede)	T		AHI	x	x	
Creek chubsucker	*Erimyzon oblongus* (Mitchill)			INV	x		
Northern hogsucker	*Hypentelium nigricans* (Lesueur)			INV	x		x
Golden redhorse	*Moxostoma erythrurum* (Rafinesque)		x	INV	x	x	x
Yellow bullhead	*Ameiurus natalis* (Lesueur)			IP			
Brown bullhead	*Ameiurus nebulosus* (Lesueur)			IP			x
Margined madtom	*Noturus insignis* (Richardson)	I		INV	x		x
Banded killifish	*Fundulus diaphanus* (Lesueur)			INV			
Eastern mosquitofish	*Gambulia holbrooki* Girard	T		INV			
Redbreast sunfish	*Lepomis auritus* (Linnaeus)			IP			
Green sunfish	*Lepomis cyanellus* Rafinesque	T	x	IP			
Pumpkinseed	*Lepomis gibbosus* (Linnaeus)			INV			
Bluegill	*Lepomis macrochirus* Rafinesque	T	x	INV			
Redear sunfish	*Lepomis microlophus* (Gunther)		x	INV			
White crappie	*Pomoxis annularis* Rafinesque		x	IP			
Smallmouth bass	*Micropterus dolomieu* (Lacepede)		x	PIS			
Largemouth bass	*Micropterus salmoides* (Lacepede)		x	PIS			
Yellow perch	*Perca flavescens* (Mitchill)			IP			
Shield darter	*Percina peltata* (Stauffer)	I		INV	x	x	
Tesselated darter	*Etheostoma olmstedi* Storer			INV	x		
Fantail darter	*Etheostoma flabellare* (Rafinesque)			INV	x		

[1] Tolerance: T = tolerant to regional human disturbances, I = intolerant to regional human disturbances.
[2] Trophic groups: PIS = piscivore, INV = invertivore, AHI = algivore/herbivore/invertivore, IP = invertivore/piscivore, DAH = detritivore/algivore/herbivore.
[3] Ben = benthic.
[4] Lith = simple lithophil.

preparation or parental care of the eggs, excluding individuals from tolerant species.

Number of late-maturing species: the number of species per sample that normally do not breed before their third year.

Percent anomalies: the percent of individuals per sample with externally visible abnormalities, such as disease, tumors, fin damage, and lesions.

Values for the selected metrics were assigned a score of 5, 3, or 1 depending on whether the data they

Table 2. Metric scoring for the pre-project wetland-stream complex and mitigation cells, based on fish survey data from wetland stream complexes with drainage areas < 17 km².

Metric	Score 1	Score 3	Score 5
Number of native species	<10	10–14	>14
Number of darter species	<2	2	>2
Number of minnow species	<5	5–9	>9
Percent dominant species	>43	35–43	<35
Number of intolerant species	<2	2	>2
Percent tolerant individuals	>57	40–57	<40
Percent omnivorous individuals	>31	16–31	<16
Percent benthic invertivores	<15	15–28	>28
Percent specialist carnivores—tolerants	<8	8–14	>14
Percent simple lithophils—tolerants	<18	18–34	>34
Number of late maturing species	<2	2	>2
Percent anomalies	>4	3–4	<3

represent were comparable to, deviated somewhat from, or deviated greatly from values found for the least-impaired streams/wetlands, respectively (Karr et al. 1986). For the pre-construction wetland-stream complex and mitigation cells, scoring was based on data from the 22 stream reaches heavily influenced by beaver. Metrics were scored by establishing the range in metric values for wetland-stream complexes with similar drainage area (5–17 km²) (Teels and Danielson 2001) and then dividing this set of data into equal thirds (Karr et al. 1986, USDA, NRCS 2003) (Table 2). For the upstream segment, metrics were scored using the trisection technique described by Lyons (1992) based on the reference developed for northern Virginia Piedmont streams (Teels and Danielson 2001). Metric scores were then summed to generate an IBI for each site, with possible scores ranging from 12 to 60.

RESULTS

Nine fish species were collected in the former wetland-stream complex during the 1974 pre-construction survey. Although lentic species were present (e.g., green sunfish), the majority of species were those more adapted to lotic conditions. Blacknose dace was the most abundant species, comprising approximately 34 % of the individuals collected. Minnows (*Cyprinidae*) formed the dominant family, comprising 63.7 % of the individuals. A major shift in species composition occurred after construction of the mitigation cells, with lentic species becoming much more prevalent. The IBI for the site decreased from 34 before construction to between 18 and 28 for the years after construction (Figure 2).

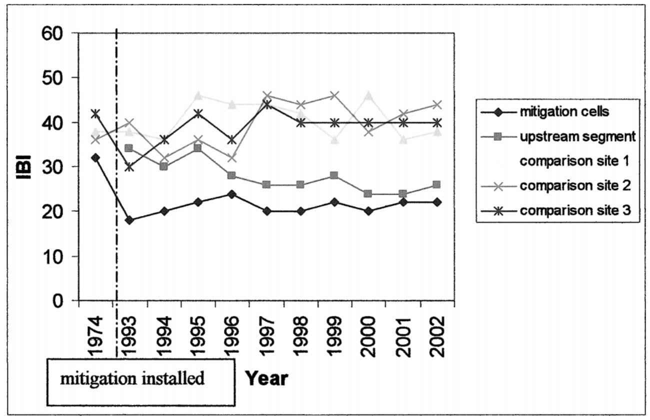

Figure 2. IBI scores for the mitigation cells, upstream segment, and comparison sites (nearby Occoquan Watershed stream segments that were sampled as part of the original environmental assessment and over the 10-year evaluation period of this study).

Mitigation Cells: Comparison of Individual Metrics Before and After Construction

Species Composition and Richness Metrics. Seven native species were inventoried in the wetland-stream complex prior to construction, compared to 2–6 per year in the mitigation cells over the 10-year sampling period after construction (Figure 3). However, there was a dramatic change in species composition. All native species that occurred in the original wetland-stream complex (rosyside dace, fallfish, cutlips minnow, blacknose dace, white sucker, tessellated darter, and fantail darter) were absent in the mitigation cells by the fourth year of post-construction sampling. Blacknose dace and fantail darter were collected in the cells the first year after construction but were not collected thereafter. The presence of rosyside dace in one of the cells in 1995 is probably as a result of unusually high stream flows during the spring of that year, which may have washed a few individuals in from the upstream reach. New species occurring after construction

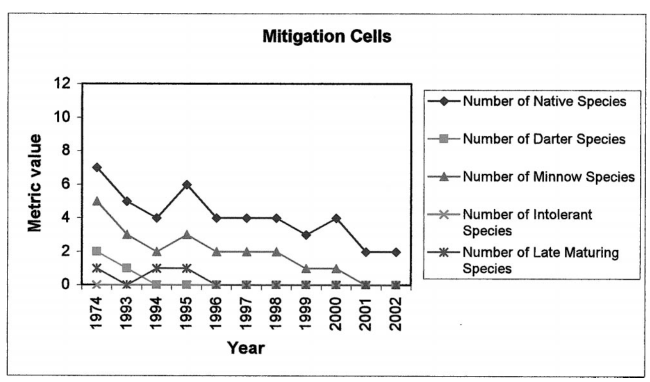

Figure 3. *Number of species* metrics data (number of species per sample) from the mitigation cells.

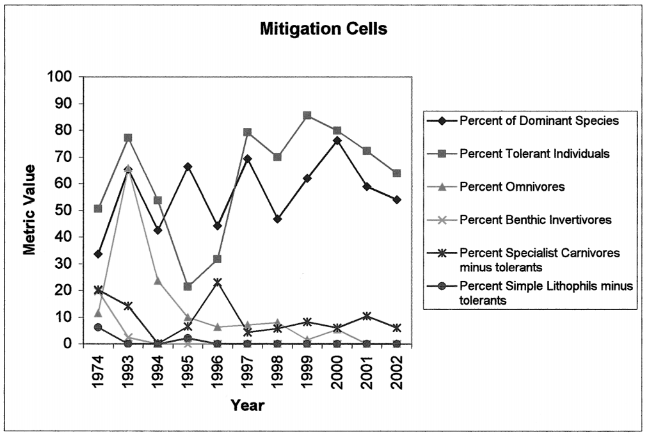

Figure 4. *Proportion metrics* data (metric percentage per sample) from the mitigation cells.

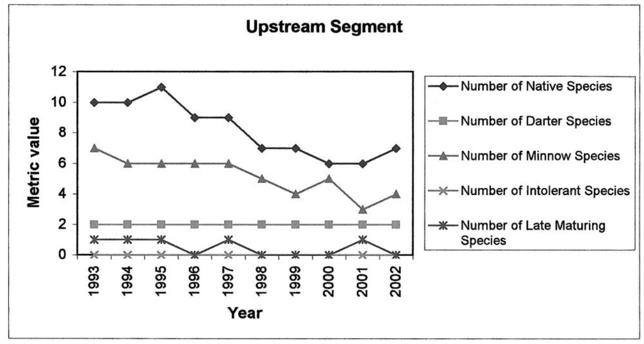

Figure 5. *Number of species* metrics data (number of species per sample) from the upstream segment.

that were absent from the pre-mitigation survey include golden shiner, yellow bullhead, bluegill, pumpkinseed, redbreast sunfish, and largemouth bass. Although the shift in species composition did not impact the score for the *number of native species* metric, it did cause a decrease in scores for *number of darter species* and *number of minnow species* metrics (Figure 3).

Tolerance/Intolerance Metrics. Blacknose dace was the single most dominant species in the 1974 pre-construction survey, comprising 33. 7 % of the individuals collected. Although the dominant species has changed over the years since construction, it has been composed primarily of tolerant species (bluntnose minnow, pumpkinseed, green sunfish, and bluegill). The dominant species comprised a greater percentage of the fish assemblage in the mitigation cells than in the wetland-stream complex before construction (Figure 4). No species designated as intolerant were encountered in either the 1974 survey or in the post-project sampling. Scores decreased for the *percent of the dominant species* and *percent tolerant individuals* metrics but remained the same for *number of intolerant species* (Figure 4).

Trophic Metrics. There was a dramatic increase in the percentage of omnivores immediately following construction of the mitigation cells (11.5 % in the 1974 pre-mitigation survey compared to 65.4 % for the mitigation cells in 1993) (Figure 4). Bluntnose minnow, an omnivore, heavily dominated the assemblage in 1993. However, the emergence of Centrarchids as the dominant taxa produced a substantial reduction in the proportion of omnivorous individuals, resulting in a general increase in the metric's score (Figure 4). Approximately 20 % of the individuals collected during the 1974 pre-construction survey were benthic invertivores. A few benthic invertivores were collected post-construction in 1993, and none were observed thereafter (Figure 4). For all practical purposes, benthic invertivores appear to have been eliminated from the site, and this metric's score has decreased correspondingly (Figure 4). Fallfish was the only specialist carnivore in the 1974 survey, comprising nearly 20 % of that sample (Figure 4). This species was not collected after construction, but two other specialist carnivores, redbreast sunfish and yellow bullhead, were collected in most post-construction years. The score for the percent *specialist carnivores minus tolerants* metric for the mitigation cells varied between 1 and 3 over the sampling period, compared to a score of 5 prior to project construction (Figure 4).

Reproduction/Condition Metrics. Although the values for both percent of *simple lithophils minus tolerants* and *number of late maturing species* metrics differed somewhat before and after construction (Figures 3 and 4), neither metric changed in a meaningful manner. The incidence of anomalies was extremely low at the site both before and after construction resulting in no change in the metric's score over time (Figure 4).

Upstream Segment and Comparison Sites

The reduction in IBI scores over time for the upstream segment was more gradual than in the mitigation cells. The IBI for this reach diminished from 34 in 1995 to between 28 and 24 from 1996 to 2002 (Figure 2). Trends in several metrics are responsible for the IBI decrease, including *number of native species, number of minnow species, percent dominant species,* and *percent tolerant individuals* (Figures 5 and 6). The IBI scores for the comparison sites varied somewhat over the ten-year sampling period; however, no downward trends were observed in the IBI, unlike in the mitigation cells or upstream segment (Figure 2).

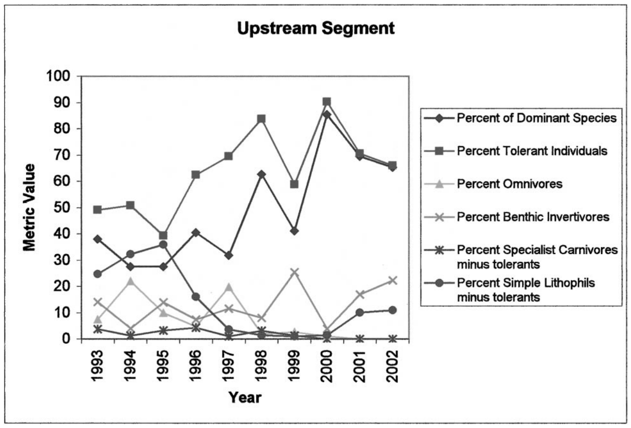

Figure 6. *Proportion metrics* data (metric percentage per sample) for the upstream segment.

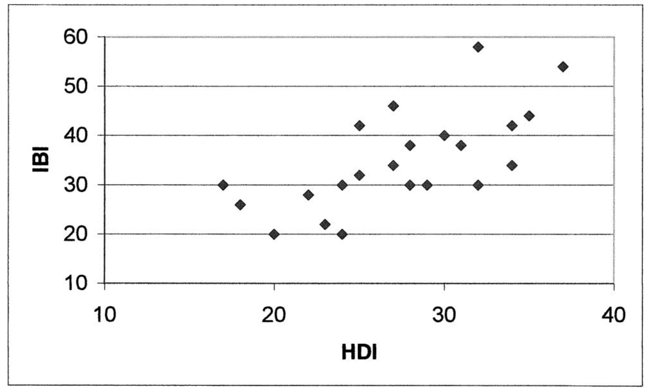

Figure 7. Relationship of the Index of Biotic Integrity to the Human Disturbance Index for the 22 beaver influenced sites that form the reference for the mitigation cells (Pearson's correlation = .68).

DISCUSSION

Although the IBI has been widely used in streams, only within the past decade have biologists begun to evaluate the technique for use in wetlands (EPA 2002a). In an evaluation of IBI for Great Lakes coastal wetlands, Wilcox et al. (2002) found that IBIs developed for wetlands may be less straightforward than the fish or invertebrate IBIs developed for streams. They concluded that after segregation of wetland types by geographic, geomorphic, and hydrologic features, a functional IBI may be possible for wetlands with relatively stable hydrology. However, they describe the problems of developing IBIs for wetlands with highly variable climatic and hydrologic conditions. In such wetlands, they found that the habitat for fish and invertebrates is provided by the complex structural character of plant communities, which can change through time without change in the level of human-induced disturbance. Therefore, they acknowledge that a site-specific, detailed ecological analysis of biological indicators or metrics may be of value in determining the quality or status of wetlands but recommended that IBI scores not be used in wetlands unless the scoring ranges are calibrated for the specific hydrologic history pre-dating any sampling year.

To ensure the appropriateness of the IBI in this study, we took the following precautions in the process of IBI development. First, the original wetland-stream complex and sites that comprise the regional reference set were sampled as stream reaches, not unlike the process used for other fish IBIs that have been developed (Davis et al. 1996, Simon 1999). In addition, the sequence of activities described in detail by Teels and Danielson (2001) were closely followed to ensure that only those metrics that are sensitive to human, and not natural, influences were incorporated into the IBI (Karr and Chu 1997, U.S. EPA 2002b). Teels and Danielson (2001) produced a regional IBI that demonstrated high sensitivity in detecting combined effects of human disturbance across the 157-site regional reference. However, beavers influence the hydrodynamics of streams by altering the flow, circulation, and reach of water and by creating additional physical habitats and plant communities, and thus may have introduced a variable into this study not accounted for by the regional IBI. Therefore, a separate reference based on the 22 stream segments heavily influenced by beaver was created to measure the condition of wetland-stream complexes.

To ensure that the IBI reflected the condition of beaver-influenced reaches, we analyzed the 22 sites that form the wetland-stream complexes reference. Scores for these sites ranged from 20 to 58 (Figure 7). When the IBI is plotted against the HDI, there appears to be little difference between the Pearson's correlation coefficient for all 157 sites in the regional reference ($r = 0.71$; Teels and Danielson 2001) and the beaver-influenced sites ($r = 0.68$), indicating that the IBI is sensitive to the human-disturbance gradient whether or not sites were influenced by beaver.

The IBI score for the pre-construction wetland-stream complex was only mediocre (34) when compared to the other beaver influenced stream segments (Figure 7). Description of the wetland-stream complex from the 1975 environmental impact statement indicated that few land-use or water quality problems occurred at the site or in the upstream watershed. Water quality monitoring from 1993 to 1995 by the Virginia Department of Transportation found extremely low concentrations of nutrients and metals entering the mitigation site (Perry and Fox 1995). The report describing these efforts concluded that the mitigation area had little opportunity to improve water quality under current conditions (Perry and Fox 1995). Although water quality does not seem to be a serious problem at the site, other human disturbances are apparent. For example, construction of Warrenton Reservoir in 1967, located less than one kilometer downstream from the project site, inundated stream habitat and established a formidable impediment to fish movement. The adverse effects of barriers on fish movement have been widely reported (Avery 1978, Winston et al. 1991) and result in a reduction in the number of species, with the effect becoming more severe as the size of the watershed decreases (Begon et al. 1990). The Cedar Run drainage above Warrenton Reservoir is small (approximately 9.1 km^2) and would be susceptible to such species reductions. The construction of the project impoundment further inundated stream habitat and formed another sizable fish barrier. The mitigation measures designed to compensate for loss of the original wetland-stream complex have only exacerbated impacts of the two downstream structures by inundating additional free-flowing stream. The project impoundment and mitigation cells may also have adversely impacted the upstream segment in this study, as indicated by the gradual decline in its IBI since project installation (Figure 2).

Although the mitigation efforts in this study created wetlands that satisfied the project's Clean Water Act requirements, a number of the physical features associated with the original wetland-stream complex have not been replaced, such as saturated wetlands and stream riffles and runs. Neither did the mitigation replace the pre-project fish assemblage with one that represented an equivalent biological condition, based on the IBI. Other studies that have assessed recovery of temperate stream fish communities from large-scale modification of instream and riparian habitat have concluded that recovery time may take over 50 years unless effective habitat mitigation measures are employed (Detenbeck et al. 1992). In this case, the pond-like mitigation cells replaced many features of the original wetland components; however, the flowing water of the original complex and its associated habitats have been lost.

The findings of this study help support the recommendations of the National Academy of Sciences that mitigation should be planned and measured with a broader set of wetland functions than are currently employed and that biological dynamics of mitigated ecosystems should be more thoroughly evaluated (National Research Council 2001). Had they been available at the onset of this study, more recent versions of the IBI developed specifically for wetlands (U.S. EPA 2002a) may have proven useful to more broadly assess the effects of this mitigation (e.g., looking at components of the biota beyond fish). Future mitigation of adverse impacts to wetland complexes should use integrated assessment techniques, such as the IBI or the individual metrics, to ensure replacement of all affected ecosystem components and to ensure that compensatory mitigation actions do not result in additional adverse impacts to other valuable aquatic resources.

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