APPLIED ISSUES

Changes in macroinvertebrate and fish assemblages in a medium-sized river following a breach of a low-head dam

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SUMMARY

1. Dam removal has great potential for restoring rivers and streams, yet limited data exist documenting recovery of associated biota within these systems following removals, especially on larger systems. This study examined the effects of a dam breach on benthic macroinvertebrate and fish assemblages in the Fox River, Illinois, U.S.A.

2. Benthic macroinvertebrates and fish were collected above and below the breached dam and three nearby intact dams for 1 year pre- and 3 years post-breach (2 years of additional pre-breach fish data were obtained from previous surveys). We also examined the effects of the breach on associated habitat by measuring average width, depth, flow rate and bed particle size at each site.

3. Physical habitat at the former impoundment (IMP) became comparable to free-flowing sites (FF) within 1 year of the breach (width and depth decreased, flow rate and bed particle size increased). We also found a strong temporal effect on depth and flow rate at all surveyed sites.

4. Following the breach, relative abundance of Ephemeroptera, Plecoptera and Trichoptera (largely due to hydropsychid caddisflies) increased, whereas relative abundance of Ostracoda decreased, in the former IMP to levels comparable to FF sites. High variation in other metrics (e.g. total taxa, diversity) precluded determination of an effect of the breach on these aspects of the assemblage. However, non-metric multidimensional scaling (NMDS) ordinations indicated that overall macroinvertebrate assemblage structure at the former IMP shifted to a characteristically FF assemblage 2 years following the breach.

5. Total fish taxa and a regional fish index of biotic integrity became more similar in the former IMP to FF sites following the breach. However, other fish metrics (e.g. biomass, diversity, density) did not show a strong response to the breach of the dam. Ordinations of abundance data suggested the fish assemblage only slightly shifted to FF characteristics 3 years after the breach.

6. Effects of the breach to the site immediately below the former dam included minor alterations in habitat (decreased flow rate and increased particle size) and short-term changes in several macroinvertebrate metrics (e.g. decreased assemblage diversity and EPT richness for first post-year), but longer-term alterations in several fish metrics (e.g.

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decreased assemblage richness for all three post-years; decreased density for first two post-years). However, NMDS ordinations suggested no change to overall assemblage structure for both macroinvertebrates and fish following the breach at this downstream site.

7. Collectively, our results support the effectiveness of dam removal as a restoration practice for impaired streams and rivers. However, differences in response times of macroinvertebrates and fish coupled with the temporal effect on several habitat variables highlight the need for longer-term studies.

Introduction

A wealth of evidence exists highlighting the predominance, increasing age and deterioration, and ecological impacts of dams on the world's rivers and streams (Dynesius & Nilsson, 1994; American Rivers, Friends of the Earth & Trout Unlimited, 1999; Graf, 1999; Heinz Center, 2002; Poff & Hart, 2002; AASHTO, 2005). This evidence, coupled with the decrease in global aquatic biodiversity (Loh & Wackernagel, 2004) and the high number of impaired rivers and streams (e.g. USEPA, 2000), has resulted in dam removal becoming an increasingly practiced form of restoration (American Rivers, Friends of the Earth & Trout Unlimited, 1999; Bednarek, 2001). Although difficult to obtain an exact count, over 600 cases of removed dams have been documented within the U.S.A. alone (American Rivers, Friends of the Earth & Trout Unlimited, 1999; AASHTO, 2005) with most of these removals occurring during the past 25–30 years. Removal rates are expected to increase in the future as more dams surpass designed life expectancies or become obsolete (USDA, 2000; Bednarek, 2001; Poff & Hart, 2002; Stanley & Doyle, 2003). The majority of removed dams have been small, low-head, run-of-the river dams (Poff & Hart, 2002; AASHTO, 2005), likely due to the high number and limited economic benefits of small dams compared to large dams, and the high frequency of small dams that are old, abandoned or in poor-condition with associated safety issues.

Although there are many documented cases of small dam removals, only a few have had associated ecological studies, limiting our understanding of system recovery following removal (Bednarek, 2001;

Hart *et al.*, 2002; Thomson *et al.*, 2005). Moreover, most of these ecological studies have been conducted on small systems (annual average discharge <5.0 m^3 s^{-1} and drainage area <1000 km^2). In general, within these smaller systems physical habitat, hydrologic and biotic patterns change relatively quickly in former impoundments (IMP) to conditions comparable to free-flowing (FF) or reference areas (Kanehl, Lyons & Nelson, 1997; Bushaw-Newton *et al.*, 2002), whereas downstream reaches are inundated with sediments formerly trapped behind the former dam (Doyle, Stanley & Harbor, 2003), potentially affecting, in the short-term, associated periphyton, benthic macroinvertebrates (especially filter-feeders/clingers) and sensitive fish (e.g. lithophilic spawners) (Thomson *et al.*, 2005). Unfortunately, because of important differences in both habitat and biota between larger (rivers) and smaller systems (i.e. streams) (Huet, 1959; Vannote *et al.*, 1980; Creed, 2006) extrapolation from these studies to larger systems might be not applicable. However, limited data exists documenting the effects of dam removal on larger systems (but see Stanley *et al.*, 2002).

Most dam removal studies also have minimally replicated, short-term data (i.e. <1 year). However, as many taxa may require >1 year to recover (e.g. long-lived macroinvertebrates and fish), some biological patterns may not appear in the first year following removal. Furthermore, our understanding of overall community-level response to dam removals is limited as most dam removal studies have examined only one trophic level of the community. Additional studies are needed that incorporate longer-term (>2 years) data on habitat and multiple levels of the community to fully understand the

ecological effects of dam removal. Moreover, only a small number of studies (e.g. Stanley *et al.*, 2002) have incorporated multiple control and reference sites, which limits our understanding of natural temporal variation (i.e. climatic factors) on the effects of removal/failure.

We examined effects of the breach of the South Batavia Dam on the Fox River, Illinois, U.S.A. We compared patterns in benthic macroinvertebrate and fish assemblages above and below this dam, before and after the breach, to patterns observed in sites above and below three nearby, intact dams (one below and two above the South Batavia dam). We also evaluated changes in associated habitat at each site (measured as average width, depth, flow rate and bed particle size) due to the breach. Our main objectives were to 1) evaluate multiyear (3 years) changes in macroinvertebrate and fish assemblages following the breach at sites immediately below and above the former dam and 2) to assess how natural temporal (yearly) variation affects inferences on the response of these assemblages to a dam breach.

Methods

Study site

The Fox River flows southwest for 298 km from headwaters near Waukesha, Wisconsin to its confluence with the Illinois River at Ottawa, Illinois, draining an area of *c.* 6,888 km^2 with an average daily discharge (1980–2005) at the Dayton, Illinois, gauging station (USGS 5552500) of 62.0 m^3 s^{-1} (range 3.5–1319 m^3 s^{-1}). Land cover within the catchment is dominated by agriculture (66%), followed by urban/residential (18%) and woodlands (9.2%) (Santucci, Gephard & Pescitelli, 2005). A series of 15 run-of-river, low-head dams are present on the Fox River between 9.2 and 159.1 km from the confluence with the Illinois River that were originally built for water supply, navigation and milldams. We selected a series of four of these dams for our study (Fig. 1). Study dams were located at 84.7 km (North Aurora, 114 m wide, 2.7 m high), 88.4 km (South Batavia, 105 m wide, 1.7 m high), 90.6 km (North Batavia, 74 m wide, 3.7 m high) and 94.5 km (Geneva, 134 m
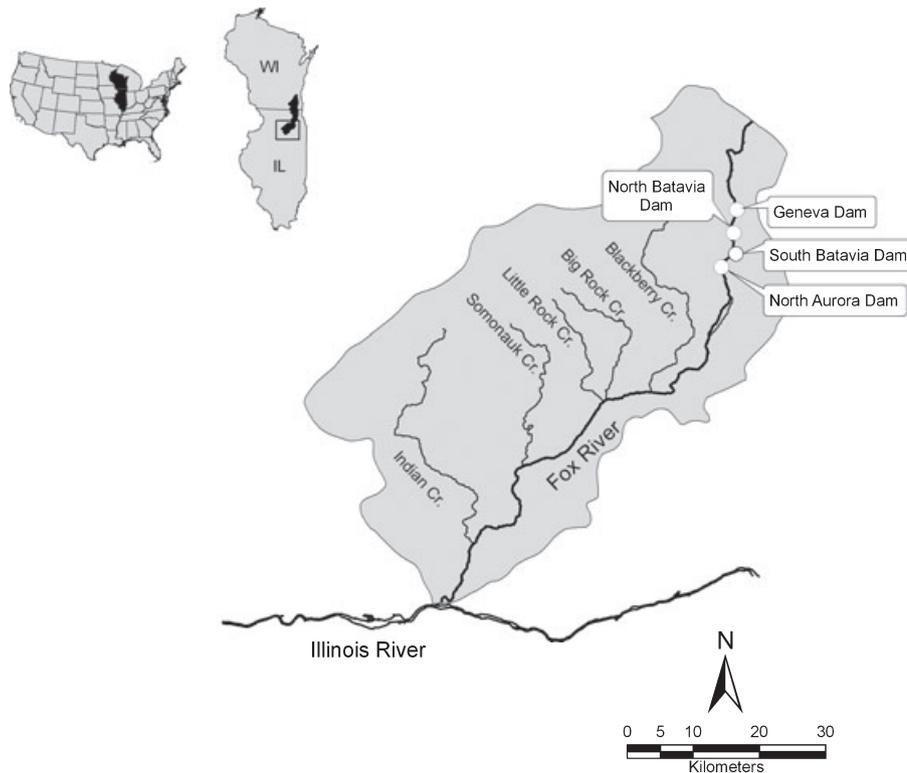

Fig. 1 Map of site location illustrating position of the four dams sampled on the Fox River located in northeast Illinois.

wide, 4.0 m high) from the confluence with the Illinois River. Over this *c.* 10 km reach, the Fox River flows through the cities of Geneva, Batavia and North Aurora, Illinois, U.S.A. As a result, much of the channel over this reach has been modified (e.g. channelization, bank reinforcement and grading) for industrial, commercial, and recreational uses.

General sampling design

Our design was a modified Beyond Before-After-Control-Impact (BACI) design (Underwood, 1991, 1992). We established sampling sites above (IMP) and below (FF areas) three unbreached dams (reference sites) and the South Batavia dam (impacted sites). Two of the unbreached dams (Geneva and North Batavia) were located upstream and one dam (North Aurora) was downstream of the breached South Batavia Dam (Fig. 1). We chose dams upstream and downstream of the breached dam to account for potential longitudinal variation. Preliminary analyses found no effect of the S. Batavia dam breach on width, depth, flow or bed particle size at the North Aurora IMP located 3.7 km below the S. Batavia dam (repeated measures ANOVA, all $P > 0.05$). Analyses also revealed no effect of the breach on bed particle size at the FF site below the North Aurora dam (repeated measures ANOVA, $P > 0.05$); however, a significant time effect ($P < 0.05$) was detected for width, depth and flow at this site but was due to large decreases in these variables at all sites in 2005 because of a severe drought (see Discussion). Further, analyses revealed no effect of the breach on macroinvertebrate or fish assemblages at the IMP and FF sites at North Aurora [see non-metric multidimensional scaling (NMDS) results]. Therefore, we treated the IMP and FF sites at North Aurora as an additional FF and IMP during analyses.

We sampled each site during summer (June/July) from 2002 to 2005. The South Batavia Dam naturally breached (i.e. failed) in the winter of 2002–03, yielding 1 year of pre-breach data for habitat and macroinvertebrates. The breach initially consisted of a *c.* 2–3 m wide opening in the dam (winter of 2002–03) that developed into a *c.* 10 m wide opening prior to our summer 2003 sampling (following high spring flows in 2003), which dramatically altered the geomorphology and hydrology of the former IMP. For fish, we obtained an additional 2 years of pre-breach data

(2000–01) from previous fish surveys on the Fox River that used similar collecting methods (Santucci & Gephard, 2003; Santucci *et al.*, 2005), resulting in 3 pre- and 3 post-breach years data.

Habitat

We measured wetted-width (m), and recorded depth (m), flow rate (m s^{-1}) and particle size (mm) at each site at three equally spaced, cross-channel transects spaced at 80–200 m (scaled to average site width). Due to safety concerns, the first transect at both the downstream FF and upstream IMP sites was *c.* 100 m from each dam. For each transect we measured depth, flow rate and particle size at 5 m increments along each transect. Flow rate was taken at 0.6 times depth in wadeable sections and at 1 m in non-wadeable areas using a Marsh–McBirney (model 2000) flow meter. Particle size was measured using a modified Wolman Pebble count where we measured a random bed particle every 5 m across each transect. Depth and flow rate were averaged within each transect. Particle size was first transformed into corresponding phi scale categories (Cummins, 1962) and then averaged by transect.

Benthic macroinvertebrates

We sampled benthic macroinvertebrates from wadeable areas using both quantitative (Hess) and semi-quantitative (Kick net) techniques. Hess samples (33 cm diameter, 300 μm mesh net) were taken at three locations along shoreline areas of IMP sites and run/riffle habitats of FF sites. Kick nets were taken at three locations in FF and IMP sites (25 × 46 cm, 500 μm mesh, 2–3 min per kick, 7–8 min processing = 30 min per site). In IMP sites, we also collected benthic macroinvertebrates in deep-water, non-wadeable areas using a petite ponar dredge (15 × 15 cm opening) deployed from a boat at three locations (left, right and mid-channel) along two transects (total = six ponars site^{-1}). Ponars were passed through a 500 μm sieve bucket to remove excess clay and silt. Ponars were not taken at the former South Batavia IMP following the breach due to changes in habitat (too shallow, increased bed particle size). All samples were preserved in 4% buffered formalin and returned to the laboratory for processing and identification. Non-insect taxa were identified to

the class⁄order level, whereas insects were identified to family⁄genus. Samples were elutriated using a two sieve series (1 mm and 250 µm) and sorted from organic debris using a dissecting microscope at 10× magnification. Samples with a large number of organisms were sub-sampled. As we were interested in site-scale assemblage changes and because of limitations in each sampling technique (different habitats; some gears qualitative), we combined all samples by site for analysis.

Fish

Fish were sampled at all sites (IMP and FF) with a pulsed-D.C. boat shocker and a D.C. backpack shocker. For boat electroshocking, at all sites each side of the river was shocked for 30 min or *c.* 700 m (total shock time = 60 min, 1400 m per site). Backpack shocking was used to target wadeable habitat (riffles and shoreline areas) for a period of 30 min per site. All fish >100 mm total length were identified, measured and weighed in field and returned to the river; smaller fish were preserved in 10% buffered formalin and taken to the laboratory for identification and measurement. Because of relative strengths of both boat and backpack electrofishing methods (i.e. boat is effective in deeper pools and runs, backpack more effective in shallow riffles and near-shore areas), during analysis we combined fish caught with both methods by site.

Statistical analysis

We tested for habitat differences in the South Batavia Free-Flowing (SBFF) and South Batavia Impoundment (SBIMP) sites from each other and site-average IMP and FF conditions using a repeated measures ANOVA (Proc Mixed, SAS Version 8.2, SAS Institute Inc., Cary, NC, U.S.A.). For this ANOVA, we calculated site-average values for each intact IMP and FF site by averaging the three transect means ($n = 3$). We log-transformed width, depth and flow rate data to satisfy homogeneity of variances requirements. Main effects in models were treatment (SBFF, SBIMP, IMP, FF) and year. The repeated effect was site, nested within treatment, and we included intercept as a random variable. We also tested for a treatment × year interaction, where a significant result would indicate an effect of the breach, and then tested for the simple effects of year on each treatment level (SAS; SLICE

option). Following a significant interaction, we tested for differences among treatments within a year with a Tukey's Studentised Range Test.

We used NMDS ordinations to assess changes in the macroinvertebrate and fish assemblages following the breach. Following removal of rare taxa (<10% of samples) on the ordinations, we root–root transformed the abundance data (Field, Clarke & Warwick, 1982) and performed separate NMDS ordinations (PCORD, version 4.0, slow and thorough autopilot mode, 400 iterations, instability criterion of 0.00001, starting number of axes of 6, 40 real runs and 50 runs with randomized data, McCune & Grace, 2002).

We also examined macroinvertebrate response by focusing on several metrics often used in studies of biotic integrity [total taxa, number taxa in Orders Ephemeroptera, Plecoptera and Trichoptera (EPT richness) and % as EPT]. We also included per cent of the assemblage as Ostracoda (% Ostracoda) due to preliminary analysis indicating these taxa differed between IMP and FF sites. Because we only had 1 year of pre-breach data for macroinvertebrates and our sample size was limited to one (the result of combining gear types), we could not perform any statistical test, therefore, we assessed possible effects of the breach by calculating and plotting 95% confidence limits (CLs) for the IMP and FF sites, and then superimposed values from the SBFF and SBIMP sites onto these CLs. This visual analysis enabled an estimation of whether the SBFF and SBIMP sites fell within or outside the 95% CLs of FF and IMP sites and gave a qualitative assessment of an effect of the breach.

For fish, we also focused our analysis on several often-reported metrics [total biomass, catch per unit effort (CPUE), total taxa, diversity (Shannon *H*′), %benthic invertivores, %sensitive lithophilic spawners]. Additionally, because common carp (*Cyprinus carpio* Linnaeus) was previously reported to be dominant in IMPs on the Fox River (Santucci & Gephard, 2003), we evaluated the effects that the breach had on the percentage of the assemblage as common carp (% as carp). We also calculated a regionally defined fish Index of Biotic Integrity (IBI) for each site (Smogor, 2000). The IBI includes four relative abundance metrics (benthic invertivores, omnivores, sensitive lithophilic spawners, tolerant species) and six natives-only richness metrics [species, sucker species (Catostomidae), Centrarchidae,

intolerants, minnows (Cyprinidae), benthic invertivores]. The 3 years of both pre- and post-breach data enabled us to test for recovery using a 'replicated' BACI analysis. We tested if the difference among treatment levels differed before and after the breach. An effect of the dam breach at the former S. Batavia IMP area would be inferred if a significant effect was found in either the SBFF and SBIMP or SBIMP and IMP comparisons. The effect of the breach on the SBFF site would be inferred if a significant effect was found in either the SBFF and SBIMP or SBFF and FF comparisons. A significant IMP and FF comparison would indicate a natural temporal effect. We also calculated the 95% CLs for IMP and FF and superimposed values for the SBFF and SBIMP to visually highlight and infer patterns.

Results

Habitat

All habitat variables differed between IMP and FF sites as well as among years and except for width all experienced significant treatment × year interactions (all $P < 0.05$, Table 1). In all years, IMP sites were on average wider and deeper with lower flow rates and smaller particle sizes than FF sites (Table 2). Average width did not change over time in either IMP or FF sites (Table 2). Prior to the breach, the SBIMP site was wider than FF sites, however, weak evidence ($P < 0.10$) suggests a decrease in width at this site following the breach, and by 2005 width at SBIMP did not differ from FF (Table 2). The SBFF site was wider than FF prior to the breach, however, following the breach width at this site decreased until it did not differ from FF sites in 2005 (Table 2). Depth did not change over time in IMP sites, however, in FF sites depth was greater in 2003 and 2004 and lower in 2005 (Table 2). Pre-breach, depth at the SBIMP site did not differ from IMP sites; however, following the breach,
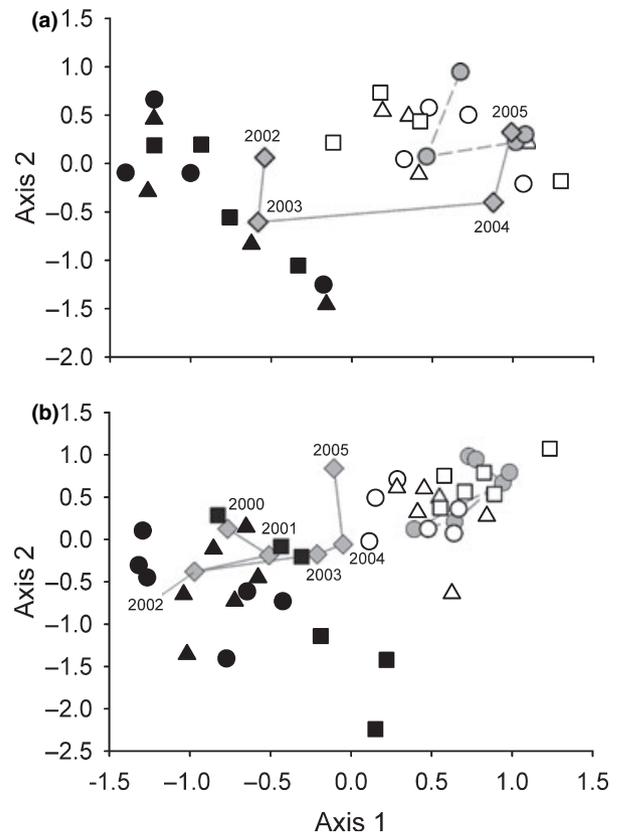

Fig. 2 NMDS ordinations of macroinvertebrate (a) and fish (b) assemblages. Filled symbols indicate impoundments (IMP), open symbols indicate free-flowing (FF) sites; circles = Geneva sites; triangles = North Batavia sites; squares = North Aurora Sites. Shaded diamond with dates = South Batavia impoundment; Shaded circle = South Batavia FF.

depth decreased at this site becoming shallower than IMP sites during 2003–05. Depth at the SBFF site decreased following the breach and did not differ from FF sites in 2002 and 2005. However, depth was shallower at this site from FF sites in 2003 and 2004 (Table 2). Flow rate in both IMP and FF sites changed over time (SLICE results $P < 0.05$, Table 2), being higher in 2003 and 2004, intermediate in 2002 and lowest in 2005 (Table 2), indicating strong temporal

Table 1 Results of repeated measures ANOVA on physical habitat measures; *F*-value (*P*-value)

Effect	NDF	DDF	Width	Depth	Flow rate	Particle size
Treatment	3	8	10.07 (0.0043)	40.76 (<0.0001)	42.73 (<0.0001)	27.63 (0.0001)
Year	3	24	5.63 (0.0046)	43.76 (<0.0001)	20.63 (<0.0001)	4.87 (0.0087)
Treatment × year	9	24	2.08 (0.0738)	5.54 (0.0004)	2.59 (0.0303)	3.13 (0.0123)

Treatment effects were impoundments, free-flowing, S. Batavia impoundment, and S. Batavia free-flowing, years were 2002–05. NDF, numerator degrees of freedom; DDF, denominator degrees of freedom.

Table 2 Mean (±1 SE) of physical habitat variables by year for each treatment in the Fox River, Illinois

River variable	Treatment	Year				Year Effect (P-value)
		2002	2003	2004	2005	
Width (m)	IMP	176.4 (11.4)[a]	178.6 (11.1)[a]	177.1 (11.4)[a]	169.3 (12.7)[a]	0.8835
	FF	70.2 (12.0)[b]	76.3 (12.8)[b]	71.6 (13.5)[b]	66.4 (15.8)[b]	0.2006
	SBIMP	151.3 (15.6)[a]	138.1 (10.9)[a]	125.7 (18.4)[ab]	135.0 (10.4)[ab]	*0.0944*
	SBFF	143.5 (9.4)[a]	140.7 (13.9)[a]	139.0 (9.9)[a]	110.7 (25.3)[ab]	**0.0010**
Depth (m)	IMP	1.51 (0.06)[a]	1.54 (0.02)[a]	1.54 (0.04)[a]	1.27 (0.09)[a]	0.1848
	FF	0.56 (0.10)[b]	0.63 (0.08)[b]	0.64 (0.02)[b]	0.38 (0.08)[b]	**<0.0001**
	SBIMP	1.00 (0.05)[a]	0.62 (0.02)[b]	0.57 (0.05)[b]	0.36 (0.06)[b]	**<0.0001**
	SBFF	0.39 (0.04)[b]	0.39 (0.04)[c]	0.35 (0.04)[c]	0.20 (0.02)[b]	**<0.0001**
Flow rate (m s^{-1})	IMP	0.05 (0.02)[b]	0.07 (0.02)[b]	0.08 (0.01)[b]	0.03 (0.01)[b]	**0.0012**
	FF	0.40 (0.12)[a]	0.56 (0.07)[a]	0.57 (0.02)[a]	0.23 (0.04)[a]	**0.0027**
	SBIMP	0.13 (0.02)[ab]	0.36 (0.03)[a]	0.40 (0.06)[a]	0.19 (0.01)[a]	**0.0002**
	SBFF	0.48 (0.05)[a]	0.46 (0.04)[a]	0.39 (0.08)[a]	0.21 (0.03)[a]	**0.0097**
Particle size (Phi-scale)	IMP	3.4 (0.4)[a]	3.4 (0.4)[a]	3.8 (0.3)[a]	4.0 (0.5)[a]	0.8469
	FF	−4.7 (0.7)[c]	−4.0 (0.6)[b]	−4.6 (0.6)[b]	−3.6 (1.1)[b]	0.4902
	SBIMP	1.3 (0.3)[ab]	−0.3 (2.0)[ab]	−3.2 (1.0)[b]	−2.1 (1.0)[b]	**0.0002**
	SBFF	1.2 (1.0)[b]	−1.4 (0.4)[ab]	−3.5 (0.4)[b]	−2.0 (0.9)[b]	*0.0579*

The breach occurred in late 2002. Values for each variable within a column with same superscript letter are not statistically different ($P > 0.05$, Tukey's). Values in bold indicate a significant time effect at $P < 0.05$ for that treatment (SLICE option), values in italics indicate a significant time effect at $P < 0.10$.
IMP, impoundment ($n = 3$); FF, free-flowing ($n = 3$); SBIMP, S. Batavia Impoundment ($n = 3$); SBFF, S. Batavia free-flowing ($n = 3$).

variation. Pre-breach flow rate at the SBIMP site did not differ from IMP sites, however following the breach, flow rate increased at this site becoming different from IMP sites from 2003 to 2005 (Table 2). Flow rate at the SBFF site decreased following the breach but never differed from FF sites (Table 2). Average particle size did not change over time in IMP or FF sites (Table 2) and irrespective of year, was smallest in IMP sites (phi scale > 3.0, fine sand) and largest in FF sites (phi scale −3.6 to −4.7, mainly pebble). Bed particle size at the SBIMP site increased from coarse sand (phi scale = 1.3) to gravel (phi scale = −2.1) following the breach becoming different from IMP sites in 2004 and 2005 (Table 2). Weak evidence ($P < 0.10$) suggests particle size at the SBFF site slightly increased following the breach but was always categorized as gravel (phi scale = −1.2 to −3.5) and never differed from FF sites.

Biota

Non-metric multidimensional scaling ordinations distinguished between IMP and FF sites for both the macroinvertebrate and fish assemblages and indicated a shift for both assemblages following the breach at the SBIMP site (Fig. 2). A two-dimensional ordination explained 92.8% of the variation (axis 1 = 66.4, axis 2 = 26.4) in the macroinvertebrate assemblage (final stress = 10.39, instability = 0.00000, 45 iterations). For macroinvertebrates, during pre- (2002) and 1-year post-breach (2003) the SBIMP site fell near the IMP sites, however in subsequent years fell near the FF sites (Fig. 2a), indicating a shift of the assemblage by the second post-removal year. A two-dimension ordination explained 90.7% of the variation (axis 1 = 48.2, axis 2 = 42.5) in the fish assemblage (final stress = 13.65, instability = 0.00001, 87 iterations). Similar to the macroinvertebrates, prior to the breach and in the first post-breach year fish assemblages in the SBIMP site resembled IMP sites, however, during the second and third post-breach year fell closer to but not within FF sites (Fig. 2b). Ordination results also suggest no change in both the macroinvertebrate and fish assemblages at the SBFF site following the breach (Fig. 2a,b).

Individual macroinvertebrate metrics showed varied responses to dams and to the breach. The number of total taxa and EPT taxa showed highly variable and broadly overlapping CLs for both IMP and FF sites, which made assessment of the breach difficult (Fig. 3a). Total taxa decreased following the breach at the SBIMP site and these lower values occurred
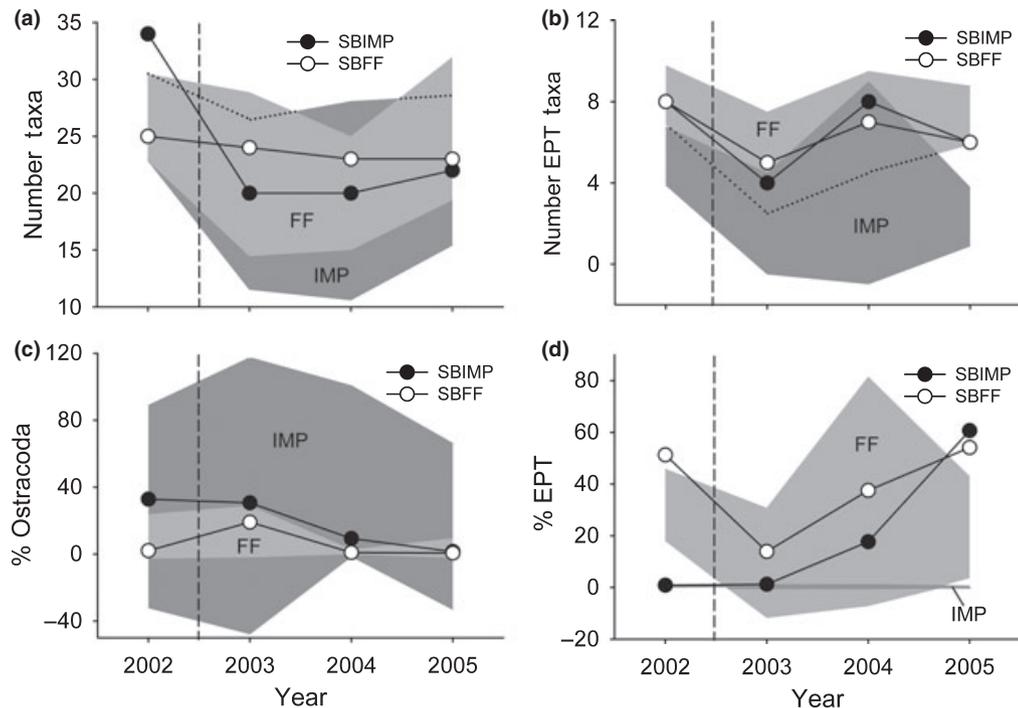
Fig. 3 Macroinvertebrate metrics by year of the South Batavia free-flowing (SBFF) and impoundment (SBIMP) sites superimposed on 95% confidence limits of the three free-flowing sites (FF, light grey) and three impoundments (IMP, dark grey). Vertical dashed lines indicate date of the breach of the South Batavia dam.

Table 3 Pre- and post-breach mean differences (±1 SE) of fish variables by pairwise comparison

Comparison	Period	Biomass (no carp)	CPUE	Total taxa	H'	%Benthic invertivores	%Catch as carp	%Lithophilic spawners	IBI score
FF–IMP	Pre	60.5 (10.2)	688.8 (99.3)	13.3 (0.3)	0.44 (0.13)	**17.5 (2.9)**	−17.9 (7.8)	19.0 (0.6)	**27.1 (1.3)**
	Post	83.2 (7.6)	667.1 (242.9)	11.3 (1.8)	0.58 (0.15)	**4.3 (0.4)**	−21.5 (10.7)	16.2 (3.4)	**20.8 (1.6)**
SBIMP–IMP	Pre	−3.6 (1.5)	33.5 (21.4)	**1.0 (0.7)**	−0.05 (0.08)	−0.3 (1.0)	*11.0 (5.9)*	−2.1 (0.4)	**2.3 (0.3)**
	Post	**14.2 (0.4)**	144.1 (102.9)	**5.0 (0.2)**	**0.41 (0.10)**	0.3 (0.5)	−12.0 (8.2)	1.7 (2.9)	**6.3 (1.2)**
SBFF–SBIMP	Pre	69.2 (24.0)	1175.4 (144.5)	**14.0 (1.7)**	0.28 (0.31)	*17.3 (5.0)*	−33.3 (8.0)	31.3 (6.0)	**28.0 (1.7)**
	Post	129.4 (51.6)	729.0 (550.8)	**6.3 (0.9)**	0.03 (0.02)	*4.7 (2.0)*	−10.9 (3.0)	19.6 (2.1)	**18.0 (0.6)**
SBFF–FF	Pre	5.0 (30.0)	520.2 (71.1)	1.7 (1.7)	−0.21 (0.10)	−0.6 (3.5)	−4.4 (0.7)	10.1 (5.2)	3.2 (1.3)
	Post	60.4 (43.7)	206.0 (688.9)	0.0 (2.7)	−0.14 (0.18)	0.7 (1.0)	−1.5 (5.4)	5.0 (5.5)	3.6 (1.7)

For example, 'pre' under 'FF–IMP' represents the average difference in pre-breach data between free-flowing and impoundments. Values in bold indicate a significant difference from the BACI analysis in pre- and post-breach differences within a comparison at $P < 0.05$, values in italics signify a significant difference at $P < 0.10$.
FF, free-flowing; IMP, impoundment; SBIMP, S. Batavia Impoundment; SBFF, S. Batavia Free-Flowing; CPUE, catch per unit effort; H', Shannon diversity; pre, pre-dam breach; post, post-dam breach.

over the remainder of the study and, except for 2002, fell inside IMP CLs, whereas total taxa at the SBFF site varied little among years and remained within FF CLs (Fig. 3a). A slight decrease in EPT taxa occurred in 2003 at both SBIMP and SBFF sites; however, values returned to pre-breach levels at both sites in 2004, and regardless of year at both sites remained above or near the upper CL of IMP sites (Fig. 3b). Per cent Ostracoda

exhibited high variation in IMP sites but not FF sites (Fig. 3c), whereas %EPT (mainly hydropsychid cad-disflies) showed high variation in FF sites but not IMP sites (Fig. 3d). Per cent Ostracoda at the SBIMP site fell outside the CLs for FF sites during 2002–04, however, in 2005 fell within FF sites CLs and was closer to the SBFF values (Fig. 3c). At the SBFF site, %Ostracoda increased to near the upper CL of FF sites
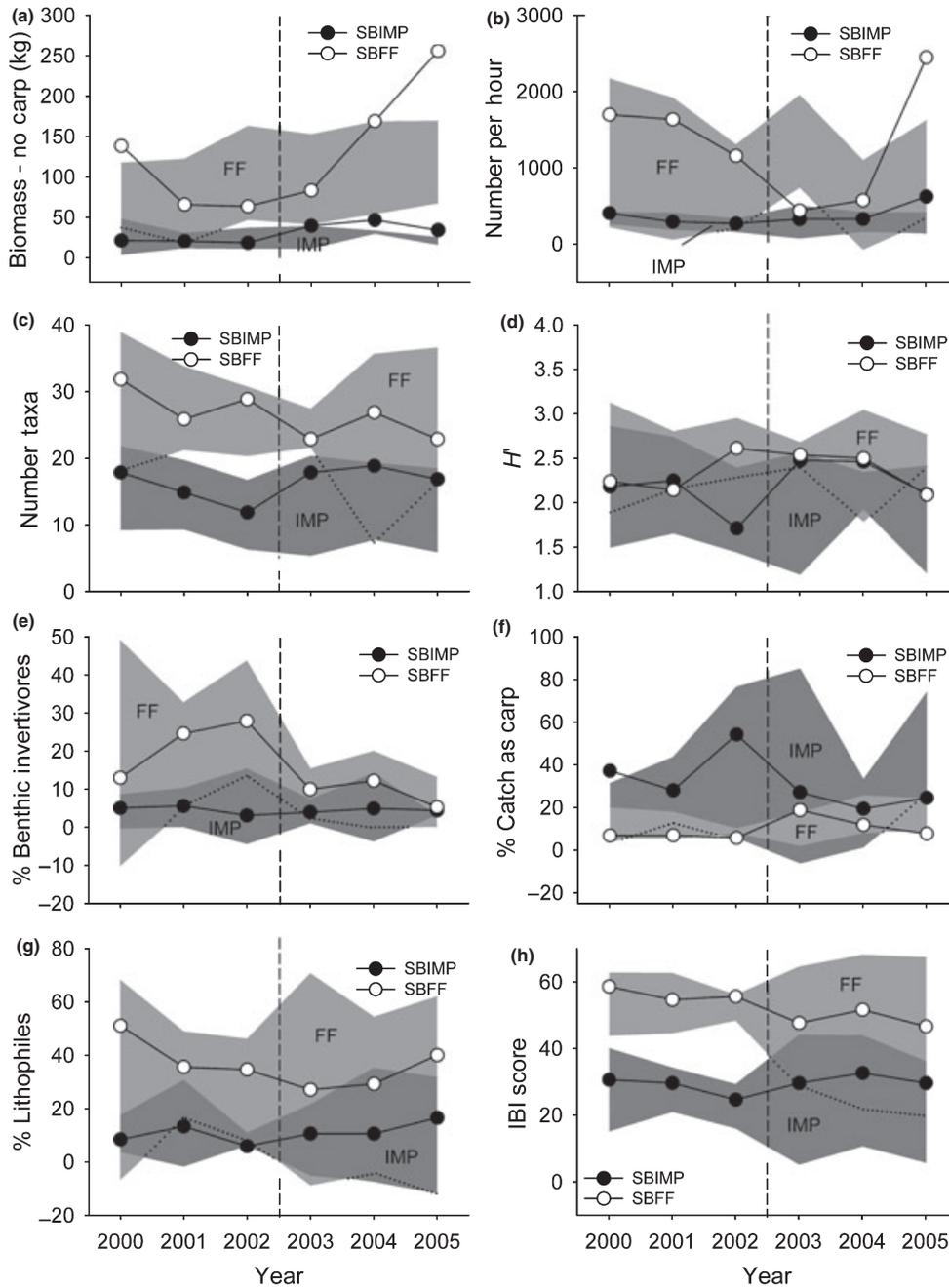
Fig. 4 Fish metrics by year examined at the South Batavia free-flowing (SBFF) and impoundment (SBIMP) sites on the Fox River. Values are superimposed on 95% confidence limits of the three free-flowing sites (FF, light grey) and three impoundments (IMP, dark grey). Vertical dashed lines indicate date of the breach of the South Batavia dam.

during 2003, however, regardless of year %Ostracoda at this site always remained within FF sites CLs (Fig. 3c). Per cent EPT made up <2% of the assemblage in the SBIMP site prior to 2003, however, they became a dominant group during 2004 (17.7%) and 2005 (60.7%, Fig. 3d), falling outside the IMP sites 95% CLs but inside FF sites CLs during 2004 and 2005. Per cent

EPT slightly decreased at the SBFF site in 2003 possibly indicating an effect of the breach, but irrespective of year remained within FF sites CLs (Fig. 3d).

Fish assemblages metrics also differed between FF and IMP sites with all metrics (except % of catch as carp) generally being higher in FF sites than IMP sites (Table 3, Fig. 4). BACI analyses indicated that only

%benthic invertivores and the IBI differed between pre- and post-samples indicating a possible temporal effect on these two metrics (Table 3). Strong evidence ($P < 0.05$) suggests an effect of the breach on biomass, total taxa, diversity and the IBI as all these metrics differed more post-breach than pre-breach between the SBIMP site and IMP sites. Weak evidence ($P < 0.10$) suggests % catch as carp differed more post-breach than pre-breach between the SBIMP site and IMP sites (Table 3). Strong evidence also suggests that the difference between the SBIMP and SBFF sites became less following the breach for total taxa and the IBI, and weak evidence suggests %benthic invertivores and % catch as carp became more similar post-versus pre-breach between these two sites. Results of the BACI analyses also suggested no effect of the breach on any metric in the SBFF site as no pre- and post-breach difference significantly differed (Table 3). These findings are supported by examination of IMP and FF sites CLs, which indicate that biomass, total taxa, % catch as carp, and the IBI moved closer to FF sites CLs and the SBFF line following the breach than before the breach (Fig. 4). Examination of the IMP and FF sites CLs for CPUE also indicates a possible increase in CPUE at the SBIMP site in 2005 (Fig. 4b).

Discussion

Ecologically, the response of the Fox River to dam failure followed predictions based on findings from smaller systems. Following breach of the South Batavia dam, habitat (as depth, flow rate and bed particle size) in the former IMP was comparable to FF sites within 1 year, whereas a shift in the macroinvertebrate assemblage to FF characteristics occurred 2 years post-breach and the fish assemblage only partially shifted to FF characteristics at 3-year post-breach. All these observed patterns agree with findings reported for smaller sized systems (Kanehl *et al.*, 1997; Bushaw-Newton *et al.*, 2002), indicating that, ecologically, larger systems may respond to dam removals/failures in a similar way to those experienced by small systems (e.g. Stanley *et al.*, 2002). Ecological effects to the site immediately below the breached dam also followed expectations postulated from smaller systems (Thomson *et al.*, 2005), being limited to short-term alterations in several habitat, macroinvertebrate and fish metrics. However, several fish metrics (e.g. richness, %benthic invertivores, IBI)

at this site did not follow predictions and remained depressed 3 years after the breach.

We observed no effects at the sites located 3.7 km below the breached dam. The lack of effects on the downstream-most sites may have resulted from the presence of the North Aurora dam, which was 1.6 times higher than the S. Batavia dam (2.7 versus 1.65 m, respectively) and had an associated IMP with 2.4 times more surface area (28.6 ha) than the former South Batavia IMP (11.9 ha). The close proximity (3.7 km) of this larger dam and associated IMP (1.8 km long) may have buffered the effects of the breach on the downstream-most reaches.

Effects of dam breach on habitat

Our finding that wetted width decreased at the former IMP and FF site immediately below the dam agrees with results from several other studies (Bushaw-Newton *et al.*, 2002; Stanley *et al.*, 2002). Following dam removal, rivers incise sediment of former IMPs, which results in a narrowing of local river width (Pizzuto, 2002; Cantelli, Paola & Parker, 2004). Depth decreased but flow and bed particle size increased at the former IMP following the breach, patterns that all agree with previous studies (Bushaw-Newton *et al.*, 2002; Stanley *et al.*, 2002). Interestingly, depth and flow rate exhibited a marked decrease in 2005 at the unaffected FF and IMP sites, indicating a strong influence of natural temporal variation on these variables. Annual precipitation in 2005 (59.3 cm) was the fourth lowest recorded since 1901 (ISWS, 2006), which resulted in extremely low discharge levels during our sampling period (range 6.8–16.5 m^3 s^{-1} compared to the >100 years average of 46.1 m^3 s^{-1}) (USGS, 2006). These 'low-flow' conditions resulted in shallower depths and slower velocities within the river and highlight the need for long-term data sets to account for natural variation in dam removal studies.

Bed particle size at both the former IMP and the FF site immediately below the failed dam increased following the breach. Increased particle size has often been reported to occur in former IMPs following dam removal (Bushaw-Newton *et al.*, 2002; Stanley *et al.*, 2002) and is due to the downstream transport of formerly entrained sediment. However, the increase in particle size at the FF site immediately below the dam is counter to many reports of a short-term decrease in particle size at downstream reaches

following removal (Stanley *et al.*, 2002; Thomson *et al.*, 2005). This observation may be due, in part, to the deposition of large cobble that was removed by scour from within and directly below the breach, shifting the mean particle size upward despite some deposition of finer particles at this site. An additional mechanism that may account for the lack of such a short-term decrease in substrate size may be sediment transport. Average flow rate at the site below the former dam was exceptionally high (0.48 m s^{-1}), suggesting released sediment was transported downstream beyond our sampling locations. Stanley *et al.* (2002) report that following a flood event released sediments were transported 3.5 km downstream until they became trapped in another IMP.

Effects of breach on biota

Following the breach and subsequent changes in habitat, associated biota in the former IMP shifted from a lentic- to lotic-type community, a shift that often has been reported in other systems (Bushaw-Newton *et al.*, 2002; Stanley *et al.*, 2002; Pollard & Reed, 2004). Results of the NMDS suggested this shift occurred by the second year following the breach in the macroinvertebrate assemblage. Such a quick shift is probably attributable to relatively high turnover rates and rapid recolonization ability of benthic macroinvertebrates. The fish assemblage did not fully shift to a lotic assemblage by the end of the study (year 3) suggesting these organisms may take long periods of time to respond to dam modifications.

No change in the macroinvertebrate assemblage at the former IMP was detected using richness measures, such as total taxa or number EPT taxa, suggesting such measures may not be appropriate for assessing changes following dam removal. A likely reason for the lack of change in richness measures is that most taxa were present in both IMP and FF sites. However, relative proportions of certain taxa dramatically differed between IMP and FF sites. The shift in the macroinvertebrate assemblage largely resulted from an increased percentage of net-spinning, filter feeding hydropsychid caddisflies and concomitant decrease in the percentage of Ostracoda. Such increase in hydropsychid caddisflies following dam removals has been reported in other systems (Stanley *et al.*, 2002) and indicates that following the dam removal/failures, the increased flow and particle size in former IMPs

probably favour filter feeding taxa that cling to substrate (e.g. hydropsychid caddisflies). However, the increased flow rate and particle size was probably unfavourable to organisms that utilize the sediment-water interface as refuge (e.g. Ostracoda). The NMDS ordination further supports the shift of the macroinvertebrate assemblage by the second post-breach year indicating that although no response in richness measures or diversity was observed, the assemblage as a whole changed.

A short-term effect was detected on the macroinvertebrate assemblage (as decreased EPT richness, %EPT and %Hydropsychidae and increased %Ostracoda) at the FF site immediately below the failed dam. However, all metrics became comparable to pre-breach reference conditions at the second post-year. Short-term alterations to benthic communities at sites below dams has occurred following removals in other studies, and is attributed to associated short-term alterations in habitat conditions (Thomson *et al.*, 2005). However, we did not detect any strong effect of the breach on associated habitat that would account for the short-term impact on invertebrates. In fact, particle size, which is often attributed as the mechanism behind such impacts on lower sites, actually increased at the site immediately below the failed dam. One possible explanation for the initial effect on macroinvertebrate metrics despite little apparent change in quantitative habitat measures is that some changes to the habitat were not detected by the set of variables that we measured. For example, although particle size demonstrated a slight increase, we observed large shifts in gravel/cobble bars below the former dam (S. Butler, pers. obs.), indicating highly unstable conditions. This instability may have limited invertebrate colonization, especially in the short-term. Another possible mechanism explaining the short-term effect on the macroinvertebrates at the lower site may be the timing of the breach. The dam breached in late 2002 and may have altered habitat conditions sufficiently to depress macroinvertebrates. Even though the altered habitat may have changed, the macroinvertebrate assemblage may not have fully recovered by our sampling in July 2003, in effect exhibiting a short-term lag effect to the breach, reinforcing the need for multiple year studies.

Fish responses indicated that a complete shift to a lotic assemblage had not yet occurred by 3 years following the breach. This lack of shift was most

evident in the NMDS ordination, where the fish assemblage at the SBIMP site fell between the IMP and FF sites at the second and third year after the breach, indicating only a partial shift. This partial shift is further supported by the IBI results, which suggest integrity of the former IMP is improving; however, had not yet reached comparable FF levels. Kanehl *et al.* (1997) previously reported relatively long periods (5 years) for improvements in integrity of former IMPs. In our study, the lack of strong improvements in overall integrity was largely due to limited improvements in three fish metrics (%lithophiles, %benthic invertivores and number taxa). We failed to detect a change in sensitive lithophilic spawning fish; however, at the third year post-breach a slight increase in this metric occurred at the former IMP, indicating these taxa may require somewhat more than 3 years to respond to dam removal. Although the benthic macroinvertebrate assemblage had shifted to a lotic assemblage by 2-year post-breach, relative abundance of invertivorous fish also exhibited no response to the breach. This lack of an effect was probably due to the high variability of this metric in FF sites. Total fish taxa slightly increased at the former IMP following the breach, however by the third year after the breach total taxa remained below CLs for both FF and SBFF sites. Although our analysis revealed no habitat variable that could account for a lack of response in these fish metrics following the breach, alternative habitat metrics unaccounted for in our sampling design may be limiting fish response. For example, suitable flow refugia (boulders, large woody habitat) and spawning habitat (firm gravel, vegetation) were noticeably lacking in the former IMP following the breach. Taken together, these results suggest changes in fish assemblages following dam failure/removal are likely to take longer than 3 years, and may require additional restoration practices (e.g. addition of in-stream cover, habitat heterogeneity).

The decrease we observed in relative abundance of common carp in the former IMP supports previous reports (Kanehl *et al.*, 1997; Bushaw-Newton *et al.*, 2002). Common carp is a non-native species that has significant impacts on associated ecosystem function (Cline, East & Threlkeld, 1994; Lougheed, Crosbie & Chow-Fraser, 1998; Parkos, Santucci & Wahl, 2003). Lentic areas above dams have been suggested as being ideal habitats for facultative riverine species such as common carp, which then invade connected lotic habitats (Rodriguez-Ruiz & Granado-Lorencio, 1992;

Santucci & Gephard, 2003). Dam removal may be one component of a successful management plan to assuage the strong ecological effects of such non-native fishes.

Similar to macroinvertebrates, the breach affected the fish assemblage at the FF site below the dam, a pattern observed in smaller systems (Bushaw-Newton *et al.*, 2002). Most obviously, density (as CPUE) drastically decreased at this site following the breach. The breach also resulted in a slight reduction in fish species richness, relative abundance of lithophilic spawners and IBI scores, as well as an increased relative catch of carp, all of which remained depressed at 3-year post-breach. Associated habitat variables at this site did not change drastically following the breach. However, as with macroinvertebrates, changes to habitat that we were unable to detect using the variables measured (e.g. bed instability) may have had an effect on the fish assemblage. For example, abundance of woody habitat was noticeably lower at this site compared to other FF sites, probably resulting in lower cover/preferred habitat for many fish (Angermeier & Karr, 1984). Moreover, the increased instability following the breach at this site may have reduced suitable sites for lithophilic spawners (Ritchie, 1972; Ryan, 1991).

Even within the highly fragmented Fox River, failure of a dam that was in close proximity to both upstream and downstream dams resulted in changes to both habitat and biotic assemblages in a former IMP to nearby FF conditions. Our findings, taken together with results from other studies (e.g. Stanley *et al.*, 2002; Thomson *et al.*, 2005) have consistently documented the effects of dam removals on former impounded areas. However, although realizing the effects of dam removals on local conditions (e.g. IMPs) have provided insight into the benefits of dam removal, to fully understand the restorative potential of removing dams, future research must move beyond focusing solely on site-scale responses. Measures that incorporate system-scale responses will better highlight the response of entire systems to dam removal. Measurements such as water residence time from headwaters to downstream confluences may provide system-scale estimates on the effects of dam removal on hydrologic conditions, whereas measures of migratory fish and invertebrate assemblages requiring such fish for dispersal (e.g. mussels) may highlight recovery of system-wide biology. When used in conjunction with sufficient temporal and site replication such

studies will greatly increase our understanding of how dam removals affect lotic systems.

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