water

MDPI

Article

Short-Term Impacts of Remeandering Restoration Efforts on Fish Community Structure in a Fourth-Order Stream

Jennifer M. Clark [1,*] and Justin J. Montemarano [2]

[1] Biology Department, Hiram College, Hiram, OH 44234, USA
[2] Department of Biology, Armstrong State University, Savannah, GA 31419, USA; justin.montemarano@armstrong.edu
* Correspondence: clarkjm@hiram.edu; Tel.: +1-330-569-5315

Received: 1 June 2017; Accepted: 17 July 2017; Published: 20 July 2017

Abstract: Channel reconfiguration is a common but debated method used to restore streams, often causing disturbance and producing subsequent negative impacts on biota. Here, we report results from short-term assessment (i.e., one and three years' post-restoration) of habitat variables (e.g., reach depth, substrate, and canopy cover) and fish community composition and structure (using electrofishing surveys; e.g., proportion of juveniles and tolerant fishes) from a 675 m section of Eagle Creek (Portage County, OH, USA) restored using channel remeandering in August 2013. Mesohabitat analysis was not conducted as part of this study. Sites upstream and downstream of restoration efforts were also monitored. Surveys were completed in 10 separate 50 m stretches: one upstream control site, three new channel sites, two old channel sites, and three downstream sites. Following restoration, fish communities in downstream sites became more similar to new channel sites and diverged from the upstream control site over time, as reflected in increased proportions of juvenile and tolerant fishes. Shifts in fish communities were not explained by habitat variables. Diversity was significantly lower in new channel sites post-restoration than in the upstream control, while downstream sites remained similarly high in diversity compared to the upstream control site over time. Overall, in the short-term, new channel colonizing communities were unable to recover to reflect upstream community composition and structure, and fish communities downstream of restoration were negatively impacted.

Keywords: stream restoration; channel reconfiguration; community composition; diversity; juvenile fishes; tolerant fishes

1. Introduction

To improve stream health and function due to anthropogenic degradation, restoration efforts are now common across the U.S. with more than US $1 billion spent on projects each year [1,2]. The majority of stream restoration projects have one or more goals targeted towards increasing biodiversity, stabilizing the channel, improving riparian habitat, improving water quality, and creating in-stream habitat for biota, and most efforts utilize in-stream hydromorphic changes (e.g., addition of boulders, other sediments, and log jams) and channel hydromorphic alteration/reconfiguration (e.g., raising or lowering the bed to reconnect to the floodplain, creating new meanders, and lateral movement of the channel) along with riparian restoration with riparian plantings to accomplish project goals [3]. Although the practice of stream restoration is widespread, both short-and long-term monitoring to evaluate restoration goals are generally lacking [1,3–5]. However, short-term monitoring efforts have increased in recent years [6] and are largely due to implementation by the U.S. Army Corps of Engineers and U.S. Environmental Protection Agency of the 2008 Compensatory Mitigation for

Losses of Aquatic Resources rule. As part of this legislation, projects under the Clean Water Act section 404 and Department of the Army permits are required to monitor compensatory mitigation efforts (e.g., stream restoration projects) for no less than five years to assess whether or not the performance objectives are met [7]. However, changes in the monitoring timeframe are ultimately determined by the district engineer, and monitoring requirements may be waived, reduced, or increased based on achievement of performance standards [7]. Although the overarching goal of compensatory mitigation is no net loss of function of the system, mitigation requirements by regulatory agencies for streams generally require mitigation based on stream length rather than measurements of stream function [6]. Further, when monitoring occurs, a standard set of criteria are not used, baseline data prior to restoration is limited, and few make comparisons to nearby streams with similar degradation prior to restoration.

Channel reconfiguration/reconstruction is currently the most commonly used approach to restore stream systems [3] but has become a debated approach [5,8] when project goals include ecological improvement because it can cause disturbance to the ecosystem [5,9] and have subsequent negative ecological impacts [10]. Notably, when channel reconfiguration results in loss of canopy cover and increased water temperatures, a shift into an autotrophic stream system can occur [11], resulting in bottom-up trophic cascades, a food web dominated by grazers, and loss of macroinvertebrates and fishes intolerant of warmer water temperatures.

Since fish communities reflect long-term impacts and stressors on streams, they are often used to gauge overall stream health [12] and may be valuable indicators of restoration success [13] when source populations are connected. However, in the past, the majority of studies focused on salmonids or other recreational fishes targeted for recovery following restoration practices, and more attention is now being focused on addressing impacts on fish assemblages as a whole.

Additionally, several, distinct restoration techniques are used with the intention of directly or indirectly improving fish populations and communities, and understanding the overall trends in success or failure of restoration impacts is complex. Recent studies suggest that stream restoration either improves fish communities [13–15] or has little to no effect on the assemblage [13,14,16,17] and is dependent on the restoration method used. However, land use impacts can affect successful recovery of fish communities regardless of the restoration method used. For example, agriculture generally has negative consequences on fish biodiversity [15], while urbanization can have no effect on fishes in some systems [18] or show slight improvement in others [19]. Overall, restoration projects that alter in-stream habitat (e.g., construction of artificial riffles, flow deflectors, large woody debris additions) appear to show the best outcome for fish communities [15]; however, this is not the case for every system [10,16], and effects can be species-specific [16,20–22] and depend on season [20,21]. Further, studies that indicate positive effects of restoration practices on fishes appear to be more common in the literature than those showing negative impacts, suggesting publication bias [23]. In some cases, short-term positive effects show reversal with time [10].

The goal of this study was to assess the impacts of channel remeandering restoration efforts on fish community structure (e.g., age class) and composition (e.g., species abundance and diversity) post-restoration. This study was conducted in Eagle Creek (Portage County, OH, USA), a fourth-order stream impacted by upstream agriculture, with intact source populations for fishes both upstream and downstream of the restored section. Additionally, the diverted site is typical of most stream restoration projects as it is small in scale [1], representing a restored length of approximately 1 km. Since restoration practices in this system diverted the channel from a closed canopy system to full exposure to sunlight and created a relatively homogenous system with regards to channel depth, microhabitat structure (lacking riffles, deep pools, and coarse woody debris), and substrate size, we expected the following short-term impacts to occur: (1) community composition would become less diverse (relative abundance by species, Shannon-Wiener diversity, and species richness) following diversion into the reconfigured channel with substantial impacts on fish communities in downstream areas; (2) juveniles would dominate the newly reconfigured channel; and (3) the abundance of intolerant fishes would decline with an increase in tolerant fishes in both the newly reconfigured channel and in downstream areas. Although the goals of the restoration project in this study are targeted towards channel stability and reconnection with the floodplain, increased biodiversity should be key outcomes of a successful stream restoration project.

2. Materials and Methods

2.1. Field and Survey Site Descriptions

Hiram College's Eagle Creek restoration site (HCECRS) is a 152-acre plot of land located within the Mahoning River watershed at the confluence of Silver Creek and Eagle Creek in Hiram Township (Portage County, OH, USA, 41°28′62″ N, 81°12′09″ W; Figure 1). Land use in the Upper Mahoning River basin is predominantly agriculture (62.7% = cropland and pasture land) along with forested land (13.9%) and urban/residential use (10.2%) [24]. Between 2006 and 2007, this site was heavily logged. It is bordered directly upstream by residential uses and agriculture, causing Eagle Creek to have heavily eroded banks, high turbidity, unstable riffles, and poor canopy cover throughout a substantial portion of the reach. Eagle Creek is a fourth-ordered stream, and sediment ranges from clay and sand to artificially placed boulders. A total of nine sites were surveyed in this study, each consisting of a 50 m stretch of stream. Sites included an upstream control site, two sites within the old, abandoned channel (referred to as the old channel throughout), three sites within the newly, reconfigured channel (referred to as the new channel throughout), and three sites downstream of the reconfigured channel (referred to as downstream sites throughout; Figures 1 and 2). In old channel sites, measurements (fish and habitat variables) were only taken during 2013 (pre-restoration) as it quickly converted into a stagnant oxbow following diversion of water into the new channel.

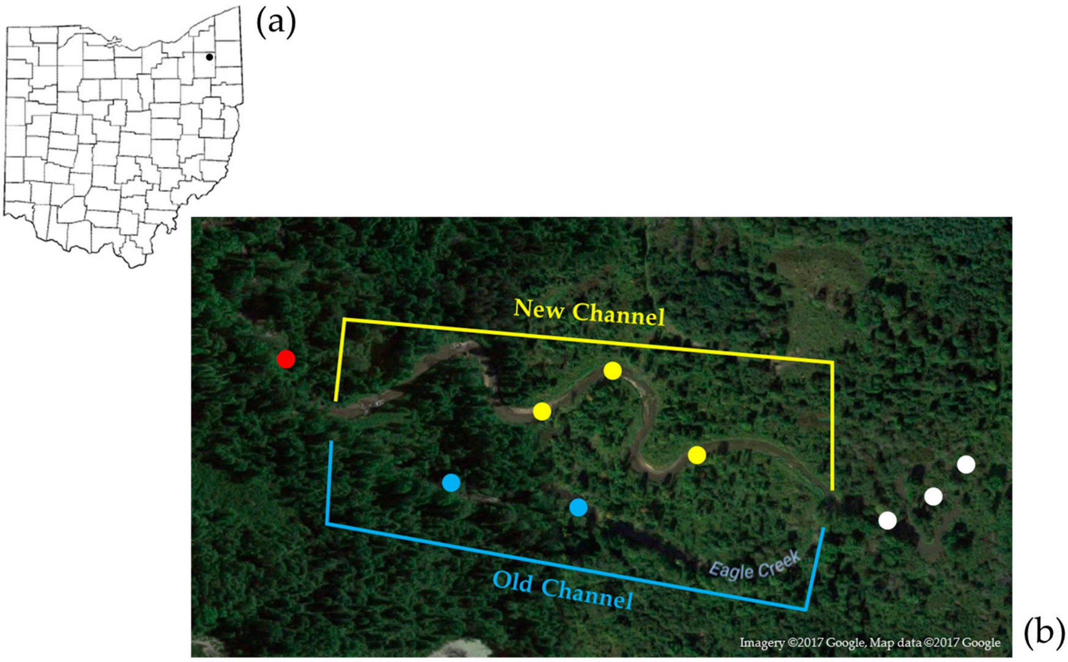

Figure 1. Maps showing (**a**) the location of Hiram College's Eagle Creek restoration site represented by the black circle in northeastern Ohio, USA; and (**b**) an aerial view of the nine sampling sites within Eagle Creek, with New Channel and Old Channel locations bracketed. Site designations are as follows: upstream control site (red circle), old channel sites (blue circles), new channel sites (yellow circles), and downstream of the newly reconfigured channel (white circles).

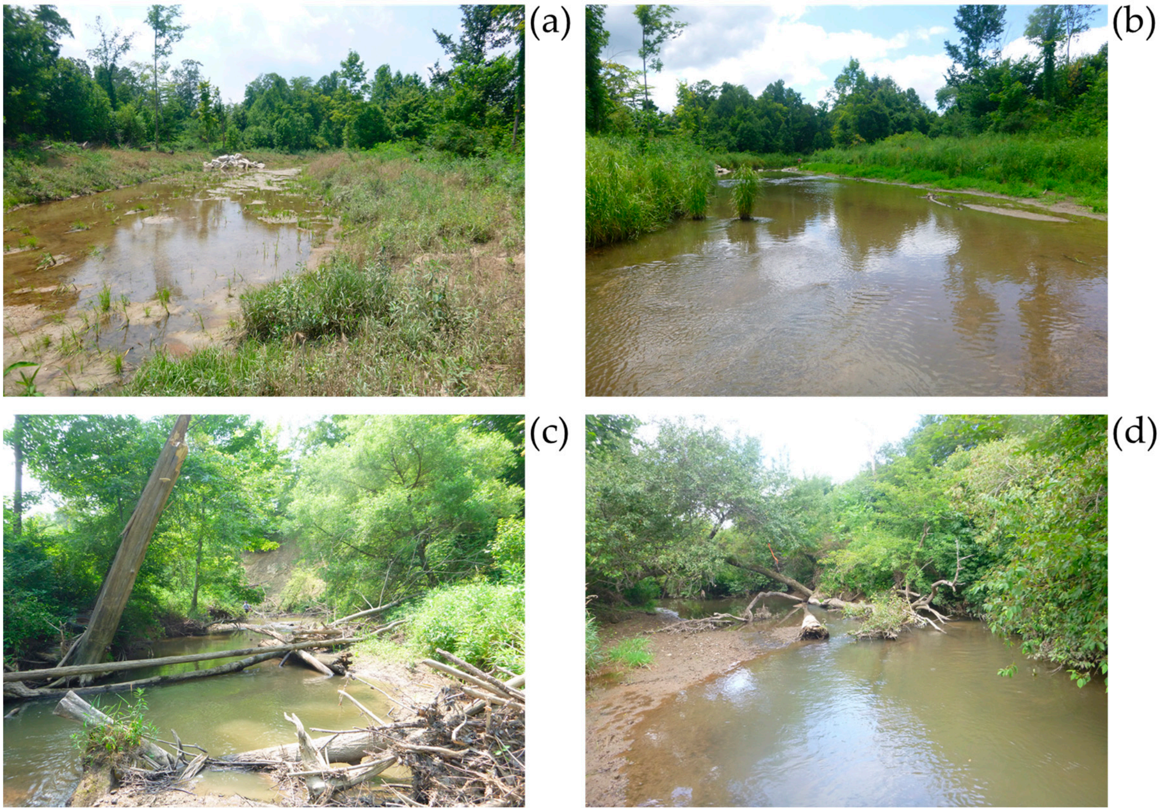

Figure 2. Photographs of sites showing (**a**) the newly excavated channel prior to diversion of water (2013); (**b**) the newly excavated channel following diversion (June 2014); (**c**) the upstream control site; and (**d**) a site downstream of the new channel.

2.2. Restoration Process and Goals

In 2012, approximately 497 linear meters of Eagle Creek was reconfigured and diverted into an excavated meandering channel extending 675 linear meters. The active channel was constructed to be wider than the old channel and upstream and downstream areas to allow movement of the wetted channel over time. Artificial boulders were placed in the new channel to maintain channel grade elevation, dimension, and a thalweg and to provide habitat structure. No other sediments (i.e., gravel or cobble) were placed in the new channel, and natural colonization of particles from upstream has been allowed to fill in with time. Following excavation of the new channel, native trees were planted and native grasses and forbs were seeded along the banks and within the floodplain with the goals of decreasing erosion and sedimentation and to provide future canopy cover to decrease light intensity and in-stream temperatures. Due to severe flooding during the summer of 2012, water was not diverted into the new channel until August 2013, and, subsequently, streamside vegetation had an additional year to help better stabilize the banks. The major goals of this project were to:

(1) re-engage the floodplain and decrease flooding downstream; (2) decrease erosion and siltation within the stream; and (3) to alter the thermal regime by decreasing water temperatures via long-term recolonization of forest. No in-stream habitat improvement for fishes has been done. Restoration design and implementation were completed by a local consulting firm.

2.3. Measurement of Habitat Variables

Average substrate size was measured in each 50 m stretch using Wolman pebble counts [25] (for a full description of methods see [26]) once during each summer. Depth (cm) and current velocity (m/s^2) were measured at 15 random locations within each sampling site using a wading rod and a Hach® flow meter (Model FH950.0, Loveland, CO, USA). All other variables, with the exception of wetted channel width (measured along six transects with a measuring tape), were measured in each 50 m stretch at five cross-sectional transects separated 10 m apart. Dissolved oxygen (mg/L and % saturation), temperature (°C), and pH was measured at the center of each transect using handheld meters (dissolved oxygen and temperature: YSI Professional Series®, Model Pro2030, Yellow Springs Instruments, Yellow Springs, OH, USA, pH: EcoTestr pH2®, Eutech Instruments, Vernon Hills, IL, USA). Percent canopy cover was estimated once during each summer with measurements taken along each transect at the edge of each stream bank (left-and right-side) and directly in the center of the channel (n = 15 per 50 m stretch) using a GRS™ densitometer (Geographic Resource Solutions, Arcata, CA, USA). All habitat variables, with the exception of average substrate size and percent canopy cover, were measured once during June 2013 and four times per year throughout June–July (at two week intervals) in 2014 and 2016.

2.4. Fish Surveys

All fishes were properly handled and collected with recommendations from the American Association for Laboratory Animal Science and approval from Hiram College's Institutional Animal Care and Use Committee (IACUC protocol number 13-05 and 16-03). For field collection, an Ohio Department of Natural Resources collecting permit was obtained (Wild Animal Permit #17-128). Surveys took place prior to restoration during early July in 2013 in the upstream control site, the two sites within the old channel, and the three sites downstream of the reconfigured channel, while sampling occurred post-restoration in the upstream control site, the three sites within the reconfigured channel, and the three sites downstream of the reconfigured channel in early July in both 2014 and 2016. All sampling efforts took place during normal summer baseflow conditions. To survey fishes, two full passes of electrofishing were completed in each 50 m stretch using a backpack Electrofisher (LR-24 Electrofisher, Smith-Root™, Vancouver, Washington, DC, USA) and two dip nets (3 mm mesh) while walking in an upstream zig-zag pattern. Prior to sampling, block nets (2 mm mesh) were stretched across the entire width of the stream both upstream and downstream to prevent movement of fishes into and out of sampling sites. All fishes collected were identified to species and measured for total length (TL, cm) using a standard fish measuring board. At the completion of both passes of electrofishing, all fishes were returned live to the center of the sampling site.

2.5. Data Analysis

Differences in habitat abiotic variables (i.e., depth, substrate grain size, and canopy cover), Shannon-Wiener diversity, and species richness were analyzed using separate ANCOVAs for each response variable with channel type and year included as explanatory variables. Pair-wise comparisons were performed using Tukey's honest significant difference (HSD). All analyses were performed in R 3.3.2 [27].

Constraints on fish community composition were examined using redundancy analysis (RDA) of a Hellinger distance matrix of fish communities at each sampling site and year (for a description of RDA, see [28,29]). Hellinger distance standardizes community composition data by column sum (the abundance of each taxa across all sites), reducing sensitivity of analyses to differences in scale and accounting for absences in taxa across sites [29–31]. The maximal RDA model, which included channel type, year, stream depth, stream width, stream velocity, proportion juvenile fish, site identification, canopy cover, dissolved oxygen, pH, and substrate grain size as constraining variables, was simplified to a parsimonious model using backward and forward stepwise model reduction with the *step* function in R 3.3.2 [27,32], producing a simplified model that included channel type, year, dissolved oxygen, and canopy cover as contributing constraints. Statistical significance of the reduced RDA model, the RDA axes, and constraints were evaluated using permutation ANOVA with 1000 permutations. RDA and associated analyses were performed using the "vegan" package in R 3.3.2 [27,33].

Individual fish were assigned an age class (juvenile or adult) according to records in Trautman, 1981 [34] and to one of the following tolerance categories: intolerant (includes fishes designated moderately intolerant), intermediate (fishes classified as being intermediate in tolerance were omitted from the regression), or tolerant (includes fishes designated as moderately tolerant) from Grabarkiewicz and Davis, 2008 [35]. Binomial logistic regressions were separately fitted to age class and fish tolerance responses, where juveniles and fish classified as tolerant were treated as "successes" in the analyses and year and channel type were used as predictors. An analysis of deviance, goodness-of-fit test was used to test for significance of the independent variables in the logistic regression models (i.e., channel type, year, and the interaction between them). Logistic regressions and associated analyses were performed with function *glm* in R 3.3.2 [27].

3. Results

3.1. Canopy Cover, Depth, and Substrate Type

Regardless of year, canopy cover (ANOVA, $F_{2,312} = 54.55$, $p < 0.0001$), water depth (ANOVA, $F_{2,897} = 88.48$, $p < 0.0001$), and substrate size (ANOVA, $F_{2,2097} = 54.27$, $p < 0.0001$) were higher in both the upstream control site and downstream sites than the new channel for all three parameters (all Tukey's HSD, $p < 0.05$; see Table 1 for means and dispersion). Percent canopy cover was, however, higher in the upstream control site than that in both the downstream sites and new channel sites, whereas water depth was greater in the downstream control sites compared to that in both the upstream control site and new channel sites. Substrate size was statistically similar between the upstream control site and downstream sites (Tukey's HSD > 0.05), and no differences were detected in canopy cover, depth, or substrate size across years ($p > 0.08$).

Table 1. Summary of habitat variables for the upstream control site, sites downstream of channel reconfiguration, and new channel sites. Data is pooled across years and channel type.

Variable	Upstream		Downstream		New Channel	
	Mean ± 1 SE	Range	Mean ± 1 SE	Range	Mean ± 1 SE	Range
Canopy cover (%)	41 ± 7	0–100	26 ± 3	0–100	0 ± 0	0
Depth (cm)	29 ± 2	2–103	33 ± 1	2–105	18 ± 1	2–61
Substrate (mm)	24 ± 2	0.15–361	16 ± 1	0.001–266	11 ± 1	0.001–231

3.2. Effects on Fish Diversity and Community Composition

A total of 22 species were collected across all sites within Eagle Creek over the entire study (Supplementary Materials Table S1). The most common species captured across sites were *Semotilus atromaculatus* creek chub, *Etheostoma flabellare* fantail darter, *Etheostoma nigrum* Johnny darter, *Notropis buccatus* silverjaw minnow, *Rhinichthys obtusus* western blacknose dace, and *Catostomus comersonii* white sucker (Table S1). All common species were captured in the newly reconfigured channel one-year post-restoration with the exception of western blacknose dace, which was collected 3-years post-restoration only (i.e., 2016, Table S1). The channel type significantly affected both Shannon-Wiener diversity (H; ANOVA, $F_{3,13} = 8.66$, $p = 0.0020$) and species richness ($F_{3,13} = 11.55$, $p = 0.0006$), and median H and species richness were higher in the upstream control site and downstream sites than that in both new channel and old channel sites (Figure 3). Median H was also higher in new channel sites than that in old channel sites, but median species richness was similar across new and old channel sites (Figure 3). There was no significant effect of year (H: $F_{1,13} = 1.02$, $p = 0.3321$; species richness: $F_{1,13} = 1.52$, $p = 0.2399$) and no significant interaction effect between channel type and year for both H and species richness (H: $F_{2,13} = 0.26$, $p = 0.7744$; species richness: $F_{1,13} = 3.08$, $p = 0.0801$).

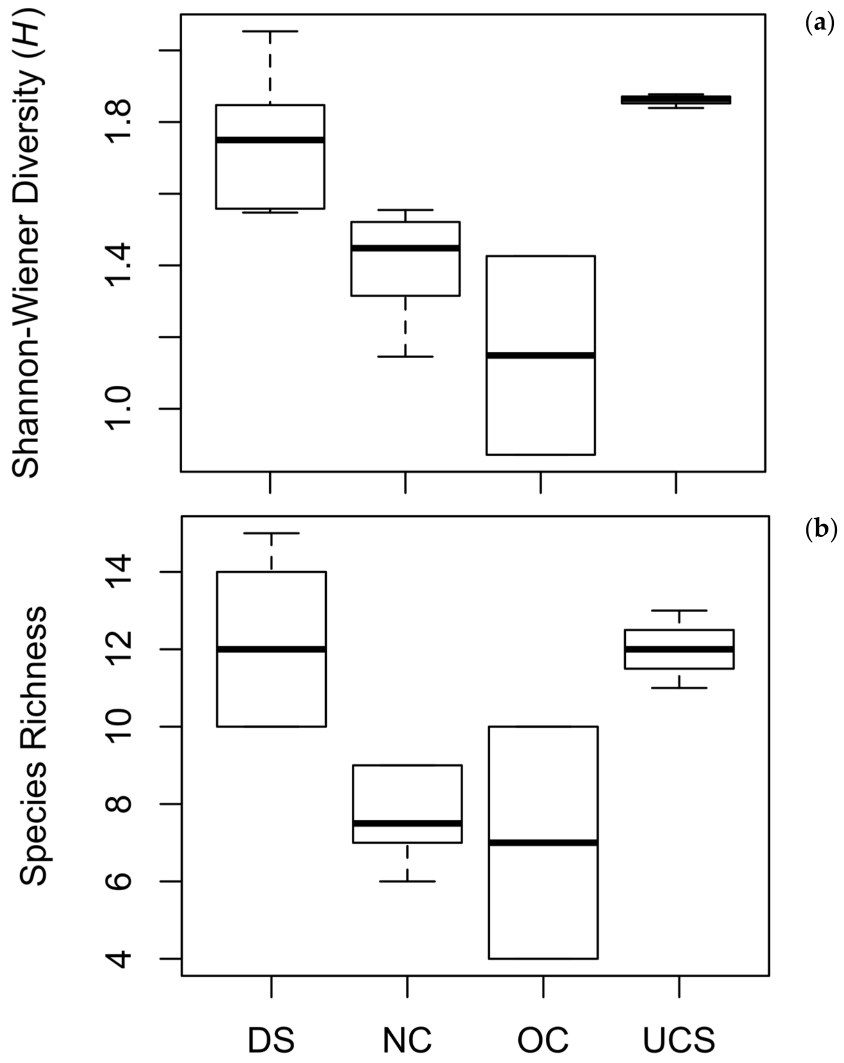

Figure 3. Box plots of (**a**) Shannon-Wiener diversity and (**b**) species richness across the four channel types, where DS are sites downstream of the newly reconfigured channel, NC are new channel sites, OC are old channel sites, and UCS is the upstream control site. Bold lines indicate median, boxes represent interquartile range, and the whiskers are 1.5 times the interquartile range.

RDA axes 1, 2, and 3 were statistically significant and explained 49.7%, 27.6%, and 11.9% of the variance in fish community composition by site, respectively (Table 2). Channel type, year, canopy cover, and dissolved oxygen were maintained in the reduced RDA model, and a permutation ANOVA on the reduced RDA model indicated that channel type and year were statistically significant factors impacting fish communities (Table 2). In contrast, canopy cover and dissolved oxygen (DO) were not statistically significant (Table 2). Year was more strongly correlated with RDA axis 1 than to RDA axis 2 (Table 2, Figure 4).

Old channel site fish community composition differed from all other channel types, and fish communities in new channel sites did not overlap with upstream sites (Figure 4). Prior to restoration in 2013, fish community composition in sites downstream of the newly reconfigured channel were similar to the upstream control site; however, following restoration, fish communities in downstream sites became more similar to the new channel sites and diverged from the upstream site over time. In contrast to the change in fish community composition observed in downstream sites, upstream site fish communities remained relatively stable across time, as indicated by a relatively small shift along RDA axis 1 in upstream site communities.

Table 2. Summary table of redundancy analysis (RDA) results from fish communities at all sampling sites across all years constrained by channel type, year, canopy cover, and dissolved oxygen (DO). Bi-plot scores for continuous constraining variables and centroid scores for channel type (DS = downstream sites, NC = new channel sites, OC = old channel sites, and UCS = the upstream control site) are presented for the first three RDA axes. *p*-values indicated for RDA axes and constraining variables were calculated using permutation ANOVA.

Variable	RDA Axis 1	RDA Axis 2	RDA Axis 3	*p*-Value
Eigenvalue	0.07035	0.3917	0.1692	
% cumulative variance	49.6	27.6	11.9	
p-value	0.001	0.001	0.022	
Variable bi-plot/centroid scores				
Year	0.8197	0.0864	0.3108	0.001
Canopy cover	−0.7342	−0.1443	0.2515	0.151
DO	0.5553	0.3297	0.0008	0.228
Channel type				0.002
DS	−0.0194	0.0874	0.138	
NC	0.328	0.0280	−0.272	
OC	−0.350	−0.858	−0.0656	
UCS	−0.364	0.254	0.171	

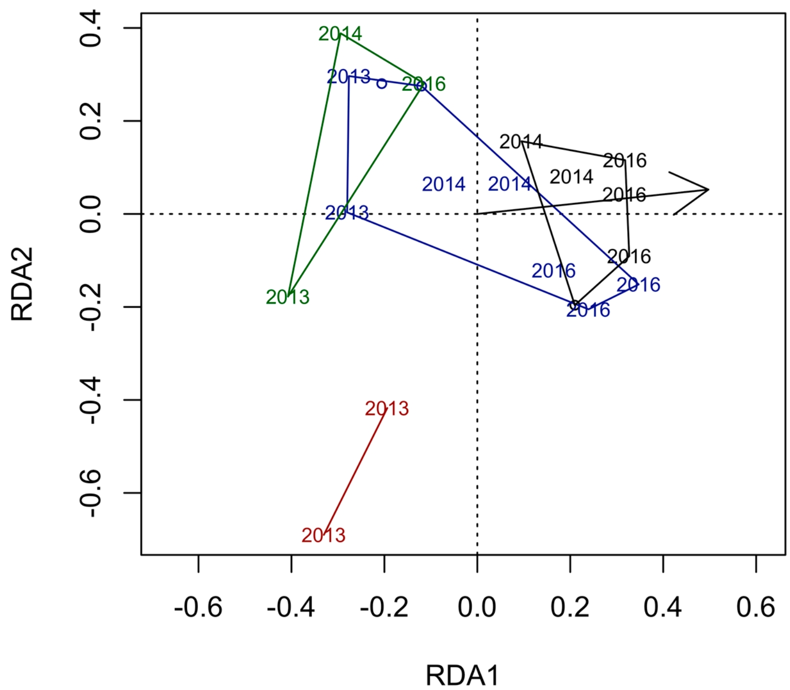

Figure 4. RDA plot displaying axes 1 and 2 with fish community composition of all sampling sites during each year constrained by canopy cover, DO, year, and channel type. Sample sites (labeled with year) are presented in ordination space according to Hellinger's distance and are colored red for old channel, green for upstream control site, blue for downstream sites, and black for new channel sites. Polygons are drawn around the outermost site for each channel type, and a vector is drawn to indicate increasing values for year.

3.3. *Effects on Juvenile and Tolerant Fish*

The main effects of channel type and year, and the interaction between channel type and year each affected the probability of encountering of juvenile fish (Table 3). In the upstream control site, the probability of encountering juvenile fish remained similar across years and was low compared to both old channel and new channel sites (Figure 5). In contrast, the probability of encountering juvenile fish in downstream sites was similar to that of the upstream control site in 2013, but increased in 2014 and 2016, matching the relatively high probability of encountering juvenile fish in the new channel sites (Figure 5).

Table 3. Results of analysis of deviance from logistic regression models fitted to two responses (juvenile fish and tolerant fish) across years and channel types. DF is degrees of freedom.

Term	DF	Deviance	Residual DF	Residual Deviance	*p*-Value
Response: Juvenile fish					
Year	2	66.6	4165	4489	<0.001
Channel Type	3	299	4162	4190	<0.001
Year X Channel Type	3	30.4	4159	4160	<0.001
Response: Tolerant fish					
Year	2	24.2	2085	2287	<0.001
Channel Type	3	128.8	2087	2311	<0.001
Year X Channel Type	3	62.5	2082	2225	<0.001

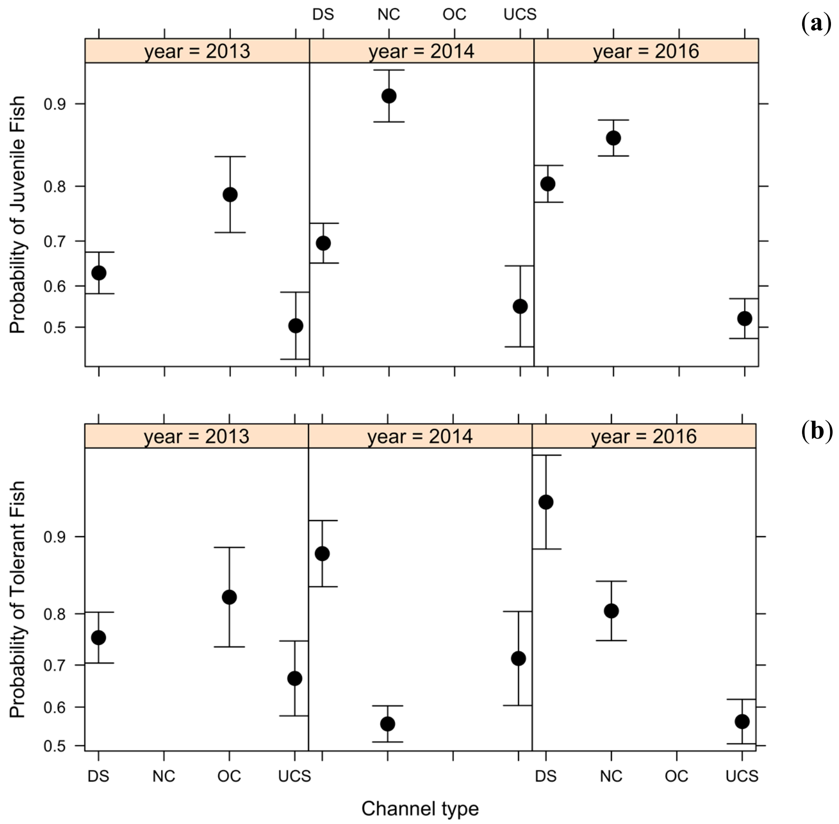

Figure 5. (**a**) Probability of encountering juvenile fish and (**b**) fish classified as tolerant across years and channel types, where DS are sites downstream of the newly reconfigured channel, NC are new channel sites, OC are old channel sites, and UCS is the upstream control site. Error bars represent 95% confidence intervals.

Of the six most common species captured across the study, only one is considered intolerant (fantail darter) while three are considered tolerant fishes (creek chub, western blacknose dace, and white sucker). Similar to the juvenile fish results, the main effects of channel type and year, and the interaction between channel type and year, were all statistically significant factors affecting the probability of encountering tolerant fish (Table 3). Although the probability of encountering tolerant fish was similar across sites in 2013 and remained similar in the upstream control site across all years, probabilities increased in downstream sites with year (Figure 5). In new channel sites, the probability of encountering tolerant fish was lower than both the upstream control and downstream sites in 2014, but increased and was higher than the upstream control site and similar to pre-restoration old channel sites by 2016 (Figure 5).

4. Discussion

Overall, remeandering restoration practices in Eagle Creek had negative impacts on fish communities in the short-term (≤three years). Colonizing communities in the new channel were unable to recover to reflect upstream community composition and structure; further, restoration activity negatively impacted fish communities in sites downstream of restoration over time. Since fish community diversity, proportion of juveniles, and proportion of tolerant fishes remained relatively stable in the upstream control site throughout the study, the impact of restoration appears to be the main driver of observed changes in fish communities downstream of the newly constructed channel. Restoration techniques created a relatively homogenous habitat in the new channel that was shallow, dominated by fine sediments, and completely lacked canopy cover. As of year three (2016), deep pools and logjams, habitat features present and stable throughout other sections of Eagle Creek, had not formed in the new channel, and, although riffle formation was visible in a few locations, riffles were not stable features (shifting or were absent following minor rain events).

Although substantial differences in canopy cover, water depth, and substrate size were observed between the new channel and upstream and downstream sites, these and other habitat variables were not correlated directly with changes in fish community composition. However, other habitat variables not measured in this study, such as presence of coarse woody debris, riffle-pool sequences, and embeddedness, may be important drivers of fish communities. Further, habitat variables may have influenced other biotic interactions (e.g., filamentous algae blooms, predator-prey interactions, competitive interactions, macroinvertebrate assemblages, and food availability). For example, loss of canopy cover is not uncommon with channel modification restoration practices (this study, [11,36,37]) and can lead to increased in-stream temperatures and primary productivity [11,38]; all factors that could impact fish communities. Extensive filamentous algae blooms were observed in this system in the new channel and downstream areas (to a lesser degree) following restoration [39] and may have influenced fish community composition and structure. In other systems, even when channel modification shows improvement of habitat variables, substantial changes in biodiversity generally do not occur [3,5,40].

Fish responses in the new channel reflected community composition that was substantially different from pre-restoration communities throughout all of Eagle Creek, and restoration efforts had negative consequences on fish community diversity. In fact, Shannon-Wiener diversity and species richness were substantially lower in the new channel and did not reach pre- or post-restoration levels found in the upstream control and downstream sites. However, Shannon-Wiener diversity in the new channel was greater than pre-restoration levels in the old channel, while species richness was similar; thus, diversity may be approaching levels found in the old channel. Although community composition and structure did not recover in the new channel, all six of the most common species in this system (creek chub, fantail darter, Johnny darter, silverjaw minnow, western blacknose dace, and white sucker) were able to colonize by year-three post-restoration. Of these six species, all but western blacknose dace colonized one-year post-restoration. Since fishes are generally good dispersers [41] and source populations are present upstream of this study site (unpublished data), rapid colonization by mostly tolerant and intermediate fishes was expected in the short-term. Such rapid colonization (within one-year) is common in other restored reaches [10,42,43] and is generally dependent on the availability [42,43] and proximity [43] of source populations with habitat variables having less of an effect [42].

In other stream systems, channel modification restoration practices appear to have inconsistent impacts on fish communities in the short-term, improving in some streams [13,44], showing no response in others, [10,13,45,46], or reflecting negative shifts in the fish community [10,13]. Even when fish community composition and structure shows rapid improvement (within one year), responses can be short-lived, and patterns of recovery can reverse within the short timeframe (three-five years) [47]. The downstream impacts of channel reconfiguration on fish communities are largely unknown, because studies addressing impacts of restoration generally take a before-after control-impact (BACI) design, which compares restored reaches and control reaches that have not been manipulated (but are considered to have had similar disturbances (e.g., channelization) prior to channel restoration). In this study, we reveal that downstream fish community composition shifted post-restoration, overlapping the composition found in new channel sites. However, neither Shannon-Wiener diversity nor species richness of downstream sites were negatively affected by restoration and remained similar to that of the upstream control site and high in comparison to that of the new channel sites. The shift in community composition evident from ordination, but not in other diversity metrics, is likely due to silverjaw minnows and western blacknose dace becoming more common post-restoration. However, our results contradict those found by Schwartz and Herricks reporting slight positive impacts on fish diversity and community composition downstream of restored riffle-pool structures two years' post-restoration [19].

Observed changes in community structure included increased capture of juvenile fishes post-restoration in both the new channel and downstream sites compared to that of the old channel and upstream control sites. These results suggest that the impacts of restoration either support habitats that promote the recruitment of juveniles or had negative consequences for adult fish colonization in the new channel and persistence in downstream areas. High proportions of juveniles in the new channel are likely explained by shallower water present in this stream section, whereas increases in downstream areas are more likely tied to disturbance from restoration, degrading adult persistence. Since stream depth remained high and deep pools/logjams remained intact in downstream areas, changes in age class structure are likely caused by other factors tied to restoration practices (e.g., sedimentation, degradation of upstream adult source populations in the new channel). Restoration techniques that create shallow margins can increase prevalence of juvenile [48–50] and small-bodied fishes (especially cyprinids) and appear to be important nursery zones [48]. Shallow margins were common in the new channel and colonizing communities were dominated by both juveniles and small-bodied fishes (e.g., cyprinids and darters) post-restoration, and deep-water fishes (e.g., centrarchids) were rarely captured. Increases in the proportion of juvenile fish may also be explained by (1) limitations in movement of juveniles throughout the new channel and downstream areas as young-of year fishes typically have limited swimming capabilities [51] or (2) increases in filamentous algal cover in this system. Anecdotally, filamentous algae became visually abundant post-restoration, while it was not evident pre-restoration, covering large portions of the new channel and some sections (to a lesser degree) of downstream areas with no detectable change in the upstream control site. Although not directly assessed in this study, an increase in algal cover could promote recruitment of juvenile fishes by providing refugia from predators or supporting increased macroinvertebrate abundance (food resource).

Further changes in community structure are reflected in increased capture of tolerant fishes in both the new channel and downstream sites from year one to three post-restoration, suggesting that new channel fish communities have not recovered and are negatively impacting downstream communities. While the proportion of tolerant fishes in new channel sites was lower compared to upstream control and downstream sites one-year post-restoration, by three years' post-restoration, the proportion of tolerant fishes exceeded levels found in both the upstream control and in that of the old channel sites, and the proportion of tolerant fishes continued to climb in downstream sites. Impacts of re-meandering had similar consequences in an Indiana stream, where shifts from communities dominated by intolerant fishes to a greater percentage of tolerant fishes were observed after restoration efforts [10]. Further, in other systems, even when in-stream habitat structure was improved (i.e., creation of riffle-pool sequences), tolerant species persisted post-restoration, and intolerant cyprinids and darters were absent from restored reaches [19]. In our study system, fantail darters (an intolerant fish) remained common in the upstream control site but became rare or absent by year three post-restoration in the downstream sites and were common in one site but rare in the other two. Further, mottled sculpin (another intolerant fish) never colonized the new channel and was absent from downstream areas by year-three, but again persisted in the upstream control site. Mottled sculpin appear to be especially sensitive to disturbance and can take up to six years to recolonize following channel reconfiguration [52].

Although the goals of natural channel design restoration are to achieve bank and channel stability, biodiversity is expected to improve and ecological function is expected to be restored (i.e., the "Field of Dreams" hypothesis, [53,54]). Unfortunately, this restoration technique does not consider restoration of stream chemistry or biological processes [3,55], and the short-term consequences on biota are largely unclear [5]. Meta-analysis of 91 stream restoration projects suggests that techniques utilizing channel reconstruction and improvement of the riparian zone were less effective in improving fish communities than restoration techniques that use in-stream structures to improve habitat [15]; however, the majority of restoration projects analyzed reported short-term effects, and it has been suggested that a minimum of 10 years may be necessary to evaluate the success or failure of stream restoration projects [56]. Regardless of the short-term impacts within a system, long-term monitoring can show improvement in some fish communities (12-years [57], 17 years [52]).

In addition to the importance of restoration technique to fish community recovery, length of the restored section can impact successful recovery [57,58]. Small, reach-scale restoration projects are likely to be unsuccessful when systems are impacted by heavy anthropogenic land use at the watershed-level [5,59,60], and underlying perturbations are likely to outweigh the benefits of restoration efforts if they do not target the major sources of degradation. The watershed surrounding Eagle Creek (this study) is impacted by row-crop agriculture and residential land-use, and a ~8 m tall dam is located approximately 1.5 km downstream, possibly impeding fish movement within this stream system. These watershed-level impacts may ultimately limit or outweigh the future benefits of restoration in this system. Further, bank erosion is common throughout Eagle Creek and sedimentation can limit the recovery of fish communities in restored systems [10]. Since positive responses of the fish community in restored streams may take longer than the timeframe evaluated in this study, long-term monitoring in this system is necessary to evaluate restoration success or failure.

Supplementary Materials: The following are available online at www.mdpi.com/2073-4441/9/7/546/s1, Table S1: Species richness and common, rare, and absent fishes by sampling site.

Acknowledgments: We would like to thank the Paul and Maxine Frohring Foundation and Gerstacker Foundation for funding this project. Additional thanks to all of the helpful hands in the field, especially Zach Nemec, Kailey Cooper, Sara Piccolomini, Maci Nelson, Michael Zielinski, Tricia Bohls, Aaron Acus-Souders, Andrew Runyon, Meredith-Fitschen Brown, Neil Zook, Jane O'Brien, and Matt Sorrick. Special thanks to the James H. Barrow Field Station staff, especially Jim Metzinger, Jim Tolan, and Emliss Ricks, for their support and assistance. Special thanks to the two anonymous reviewers that provided helpful feedback on an earlier version of this manuscript.

Author Contributions: J.M. Clark conceived and designed the study and carried out field surveys; J.J. Montemarano analyzed the data and created figures; both J.M. Clark and J.J. Montemarano contributed to the writing of this manuscript.

Conflicts of Interest: The authors declare no conflict of interest. The founding sponsors had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

References

1. Bernhardt, E.S.; Palmer, M.A.; Allan, J.D.; Alexander, G.; Barnas, K.; Brooks, S.; Carr, J.; Clayton, S.; Dahm, C.; Follstad-Shah, J.; et al. Synthesizing U.S. river restoration efforts. *Science* **2005**, *308*, 636–637. [CrossRef] [PubMed]
2. Jähnig, S.C.; Lorenz, A.W.; Hering, D.; Antons, C.; Sundermann, A.; Jedicke, E.; Haase, P. River restoration success: A question of perception. *Ecol. Appl.* **2011**, *21*, 2007–2015. [CrossRef] [PubMed]
3. Palmer, M.A.; Hondula, K.L.; Koch, B.J. Ecological restoration of streams and rivers: Shifting strategies and shifting goals. *Annu. Rev. Ecol. Evol. Syst.* **2014**, *45*, 247–269. [CrossRef]
4. Brooks, S.S.; Lake, P.S. River restoration in Victoria, Australia: Change is in the wind, and none too soon. *Restor. Ecol.* **2007**, *15*, 584–591. [CrossRef]
5. Tullos, D.D.; Penrose, D.L.; Jennings, G.D. Analysis of functional traits in reconfigured channels: Implications for the bioassessment and disturbance of river restoration. *J. N. Am. Benthol. Soc.* **2009**, *28*, 80–92. [CrossRef]
6. Doyle, M.W.; Shields, F.D. Compensatory mitigation for streams under the Clean Water Act: Reassessing science and redirecting policy. *JAWRA* **2012**, *48*, 494–509. [CrossRef]
7. Department of Defense; Environmental Protection Agency. Compensatory mitigation for losses of aquatic resources (40 CFR Part 230). *Fed. Regist.* **2008**, *73*, 19594–19705.
8. Poff, N.L.; Richter, B.D.; Arthington, A.H.; Bunn, S.E.; Naiman, R.J.; Kendy, E.; Acreman, M.; Apse, C.; Bledsoe, B.P.; Freeman, M.C.; et al. The ecological limits of hydrologic alteration (ELOHA): A new framework for developing regional environmental flow standards. *Freshw. Biol.* **2010**, *55*, 147–170. [CrossRef]
9. Louhi, P.; Mykrä, H.; Paavola, R.; Huusko, A.; Vehanen, T.; Mäki-Petäys, A.; Muotka, T. Twenty years of stream restoration in Finland: Little response by benthic macroinvertebrate communities. *Ecol. Appl.* **2011**, *21*, 1950–1961. [CrossRef] [PubMed]
10. Moerke, A.H.; Lamberti, G.A. Responses in fish community structure to restoration of two Indiana streams. *N. Am. J. Fish. Manag.* **2003**, *23*, 748–759. [CrossRef]
11. Sudduth, E.B.; Hassett, B.A.; Cada, P.; Bernhardt, E.S. Testing the field of dreams hypothesis: Functional responses to urbanization and restoration in stream ecosystems. *Ecol. Appl.* **2011**, *21*, 1972–1988. [CrossRef] [PubMed]
12. Karr, J.R. Assessment of biotic integrity using fish communities. *Fisheries* **1981**, *6*, 21–27. [CrossRef]
13. Haase, P.; Hering, D.; Jähnig, S.C.; Lorenz, A.W.; Sundermann, A. The impact of hydromorphological restoration on river ecological status: A comparison of fish, benthic invertebrates, and macrophytes. *Hydrobiologia* **2013**, *704*, 475–488. [CrossRef]
14. Shields, F.D., Jr.; Knight, S.S.; Cooper, C.M. Can warmwater streams be rehabilitated using watershed-scale standard erosion control measures alone? *Environ. Manag.* **2007**, *40*, 62–79. [CrossRef] [PubMed]
15. Kail, J.; Brabec, K.; Poppe, M.; Januschke, K. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: A meta-analysis. *Ecol. Indic.* **2015**, *58*, 311–321. [CrossRef]
16. Pretty, J.L.; Harrison, S.S.C.; Shepherd, D.J.; Smith, C.; Hildrew, A.G.; Hey, R.D. River rehabilitation and fish populations: Assessing the benefit of instream structures. *J. Appl. Ecol.* **2003**, *40*, 251–265. [CrossRef]
17. Lepori, F.; Palm, D.; Brännäs, E.; Malmqvist, B. Does restoration of structural heterogeneity instreams enhance fish and macroinvertebrate diversity? *Ecol. Appl.* **2005**, *15*, 2060–2071. [CrossRef]
18. Stranko, S.A.; Hilderbrand, R.H.; Palmer, M.A. Comparing the fish and benthic macroinvertebrate diversity of restored urban streams to reference streams. *Restor. Ecol.* **2012**, *20*, 747–755. [CrossRef]
19. Schwartz, J.S.; Herricks, E.E. Evaluation of pool-riffle naturalization structures on habitat complexity and the fish community in an urban Illinois stream. *River Res. Appl.* **2007**, *23*, 451–466. [CrossRef]
20. Cederholm, C.J.; Bilby, R.E.; Bisson, P.A.; Bumstead, T.W.; Fransen, B.R.; Scarlett, W.J.; Ward, J.W. Response of juvenile coho salmon and steelhead to placement of large woody debris in a coastal Washington stream. *N. Am. J. Fish. Manag.* **1997**, *17*, 947–963. [CrossRef]
21. Hrodey, P.J.; Sutton, T.M. Fish community responses to half-log additions in warmwater streams. *N. Am. J. Fish. Manag.* **2008**, *28*, 70–80. [CrossRef]
22. Whiteway, S.L.; Biron, P.M.; Zimmermann, A.; Venter, O.; Grant, J.W.A. Do in-stream restoration structures enhance salmonid abundance? A meta-analysis. *Can. J. Fish. Aquat. Sci.* **2010**, *67*, 831–841. [CrossRef]

23. Stewart, G.B.; Bayliss, H.R.; Showler, D.A.; Sutherland, W.J.; Pullin, A.S. Effectiveness of engineered in-stream structure mitigation measures to increase salmonid abundance: A systematic review. *Ecol. Appl.* **2009**, *19*, 931–941. [CrossRef] [PubMed]
24. Martin, S.C. *Mahoning River Watershed Action Plan*; Youngstown State University: Youngstown, OH, USA, 2004; Available online: http://www.mahoningriver.com/Mahoning%20River%20WAP.pdf (accessed on 31 May 2017).
25. Wolman, M.G. A method of sampling coarse river-bed material. *Trans. Am. Geophys. Union* **1954**, *15*, 951–956. [CrossRef]
26. Clark, J.M.; Kershner, M.W.; Holomuzki, J.R. Grain size and sorting effects on size-dependent responses by lotic crayfish to high flows. *Hydrobiologia* **2008**, *610*, 55–66. [CrossRef]
27. R Development Core Team. *R: A Language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2016; Available online: https://www.R-project.org/ (accessed on 19 July 2017).
28. Bocard, D.; Legendre, P.; Drapeau, P. Partialling out the spatial component of ecological variation. *Ecology* **1992**, *73*, 1045–1055. [CrossRef]
29. Legendre, P.; Legendre, L. *Numerical Ecology*; Elsevier Science: Amsterdam, The Netherlands, 1998; p. 853. ISBN 9780080537870.
30. Anderson, M.J.; Ellingsen, K.E.; McArdle, B.H. Multivariate dispersion as a measure of beta diversity. *Ecol. Lett.* **2006**, *9*, 683–693. [CrossRef] [PubMed]
31. Legendre, P.; Gallagher, E.D. Ecologically meaningful transformations for ordination of species data. *Oecologia* **2001**, *129*, 271–280. [CrossRef] [PubMed]
32. Crawley, M.J. *The R Book*; John Wiley & Sons, Ltd.: West Sussex, UK, 2007; p. 942.
33. Oksanen, J.; Blanchet, F.G.; Friendly, M.; Kindt, R.; Legendre, P.; McGlinn, D.; Minchin, P.R.; O'Hara, R.B.; Simpson, G.L.; Solymos, P.; et al. Vegan: Community Ecology Package. R Package Version 2.4-2. 2017. Available online: https://CRAN.R-project.org/package=vegan/ (accessed on 19 July 2017).
34. Trautman, M.B. *Fishes of Ohio*; Ohio State University Press: Columbus, OH, USA, 1981; ISBN 0-8142-0213-6.
35. Grabarkiewicz, J.D.; Davis, W.S. *An Introduction to Freshwater Fishes as Biological Indicators*; United States United States Environmental Protection Agency: Washington, DC, USA, 2008; pp. 1–79.
36. Ernst, A.G.; Warren, D.R.; Baldigo, B.P. Natural-channel-design restorations that changed geomorphology have little effect on macroinvertebrate communities in headwater streams. *Restor. Ecol.* **2012**, *20*, 532–540. [CrossRef]
37. Violin, C.R.; Cada, P.; Sudduth, E.B.; Hassett, B.A.; Penrose, D.L.; Bernhardt, E.S. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. *Ecol. Appl.* **2011**, *21*, 1932–1949. [CrossRef] [PubMed]
38. Kiffney, P.M.; Richardson, J.S.; Bull, J.P. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. *J. Appl. Ecol.* **2003**, *40*, 1060–1076. [CrossRef]
39. Clark, J.M.; Drinkard, M.K. Short-term impacts of remeandering restoration practices on macroinvertebrate assemblages in a fourth-order stream. Unpublished work, manuscript in preparation. 2017.
40. Ernst, A.G.; Baldigo, B.P.; Mulvihill, C.I. Effects of natural-channel-design restoration on habitat quality in catskill mountain streams, New York. *Trans. Am. Fish. Soc.* **2010**, *139*, 468–482. [CrossRef]
41. Fausch, K.D.; Torgersen, C.E.; Baxter, C.V.; Li, H.W. Landscapes to riverscapes: Bridging the gap between research and conservation of stream fishes. *BioScience* **2002**, *52*, 483–498. [CrossRef]
42. Stoll, S.; Kail, J.; Lorenz, A.W.; Sundermann, A.; Haase, P. The importance of the regional species pool, ecological species traits and local habitat conditions for the colonization of restored river reaches by fish. *PLoS ONE* **2014**, *9*, e84741. [CrossRef] [PubMed]
43. Stoll, S.; Sundermann, A.; Lorenz, A.W.; Kail, J.; Haase, P. Small and impoverished regional species pools constrain colonisation of restored river reaches by fishes. *Freshw. Biol.* **2013**, *58*, 664–674. [CrossRef]
44. Baldigo, B.P.; Warren, D.R.; Ernst, A.G.; Mulvihill, C.I. Response of fish populations to natural channel design restoration in streams of the Catskill Mountains, New York. *N. Am. J. Fish. Manag.* **2008**, *28*, 954–969. [CrossRef]
45. Baldigo, B.P.; Ernst, A.G.; Warren, D.R.; Miller, S.J. Variable responses of fish assemblages, habitat, and stability to natural-channel-design restoration in Catskill Mountain Streams. *Trans. Am. Fish. Soc.* **2010**, *139*, 449–467. [CrossRef]

46. Klein, L.R.; Clayton, S.R.; Alldredge, J.R.; Goodwin, P. Long-term monitoring and evaluation of the Lower Red River Meadow Restoration Project, Idaho, USA. *Restor. Ecol.* **2007**, *15*, 223–239. [CrossRef]
47. Moerke, A.H.; Gerard, K.J.; Latimore, J.A.; Hellenthal, R.A.; Lamberti, G.A. Restoration of an Indiana, USA, stream: Bridging the gap between basic and applied lotic ecology. *J. N. Am. Benthol. Soc.* **2004**, *23*, 647–660. [CrossRef]
48. Jungwirth, M.; Muhar, S.; Schmutz, S. The effects of recreated instream and ecotone structures on the fish fauna of an epipotamal river. *Hydrobiologia* **1995**, *303*, 195–206. [CrossRef]
49. Langler, G.J.; Smith, C. Effects of habitat enhancement on 0-group fishes in a lowland river. *Regul. Rivers Res. Mgmt.* **2001**, *17*, 677–686. [CrossRef]
50. Lorenz, A.W.; Stoll, S.; Sundermann, A.; Haase, P. Do adult and YOY fish benefit from river restoration measures? *Ecol. Eng.* **2013**, *61*, 174–181. [CrossRef]
51. Wolter, C.; Arlinghaus, R. Navigation impacts on freshwater fish assemblages: The ecological relevance of swimming performance. *Rev. Fish. Biol. Fish.* **2003**, *13*, 63–89. [CrossRef]
52. Shirey, P.D.; Brueseke, M.A.; Kenny, J.B.; Lamberti, G.A. Long-term fish community response to a reach-scale stream restoration. *Ecol. Soc.* **2016**, *21*, 11. [CrossRef]
53. Palmer, M.A.; Ambrose, R.F.; Poff, N.L. Ecological theory and community restoration ecology. *Restor. Ecol.* **1997**, *5*, 291–300. [CrossRef]
54. Doll, B.A.; Grabow, G.L.; Hall, K.R.; Halley, J.; Harman, W.A.; Jennings, G.D.; Wise, D.E. *Stream Restoration: A Natural Channel Design Handbook*; North Carolina Stream Restoration Institute: Raleigh, NC, USA, 2003.
55. Kondolf, G.M. River restoration and meanders. *Ecol. Soc.* **2006**, *11*, 42. [CrossRef]
56. Kondolf, G.M. Five elements for effective evaluation of stream restoration. *Restor. Ecol.* **1995**, *3*, 133–136. [CrossRef]
57. Schmutz, S.; Jurajda, P.; Kaufmann, S.; Lorenz, A.W.; Muhar, S.; Paillex, A.; Poppe, M.; Wolter, C. Response of fish assemblages to hydromorphological restoration in central and northern European rivers. *Hydrobiologia* **2016**, *769*, 67–78. [CrossRef]
58. Schmutz, S.; Kremser, H.; Melcher, A.; Jungwirth, M.; Muhar, S.; Waidbacher, H. Ecological effects of rehabilitation measures at the Austrian Danube: A meta-analysis of fish assemblages. *Hydrobiologia* **2014**, *729*, 49–60. [CrossRef]
59. Schiff, R.; Benoit, G.; Macbroom, J. Evaluating stream restoration: A case study from two partially developed 4th order Connecticut, USA streams and evaluation monitoring strategies. *River Res. Appl.* **2011**, *27*, 431–460. [CrossRef]
60. Lorenz, A.W.; Feld, C.K. Upstream river morphology and riparian land use overrule local restoration effects on ecological status assessment. *Hydrobiologia* **2013**, *704*, 489–501. [CrossRef]

© 2017 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license (http://creativecommons.org/licenses/by/4.0/).