

The Utility of a Fish Index of Biotic Integrity (IBI) for Assessing Stream Health in
the Oil and French Creek Watersheds of Northwestern Pennsylvania



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Title: The Utility of a Fish Index of Biotic Integrity (IBI) for Assessing Stream Health in the Oil and French Creek Watersheds of Northwestern Pennsylvania

Biomonitoring is often used to assess the net effects of multiple stressors on running water ecosystems. Fish “indices of biotic integrity” (IBIs) have been adapted for use in the French Creek watershed of northwestern Pennsylvania; however, these IBIs have not been evaluated in neighboring watersheds. The purpose of this study was to use the French Creek IBIs to assess the effects of land use on tributaries in the Oil Creek watershed of northwestern Pennsylvania. During the late 1800s, Oil Creek was impacted by intense deforestation and stream habitat degradation associated with oil exploration, whereas French Creek was mainly impacted by agriculture. I predicted that IBI scores would be lower in 2nd to 4th order tributaries of Oil Creek than in tributaries of French Creek of similar size and with similar forest cover because of these differences in “legacy effects” associated with historical patterns of land use. Results show that for a given percent of watershed forest cover, fish species richness is lower in Oil Creek than in French Creek tributaries. This suggests that metric cutoff values will need to be modified to increase the regional utility of the IBI. The results of this study also addressed other between-watershed differences that affect the construction of a regional IBI, including differences in the pool of available species, high forest cover, and stream gradient. Few studies have compared biotic integrity between watersheds with different legacy effects associated with historical differences in land use, a consideration that should be addressed when investigating biotic integrity on a regional level.

INTRODUCTION

Humans rely on aquatic ecosystem services for drinking, fishing, washing, flushing, shipping, irrigating, generating electricity, and for their recreational, spiritual, and aesthetic values. When aquatic ecosystems become degraded from their natural condition, they lose their ability to furnish these services (Karr and Chu 1999). Thus, maintaining the biological components of stream ecosystems (populations, species, genes) and the processes that support them (mutation, selection, fish migration, biogeochemical cycles, and the water cycle) is essential for the supply of goods and services that healthy waters provide (Karr and Chu 1999).

Biological integrity (biotic integrity) is defined as the ability of an ecosystem to “support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region” (Karr 1991). Although initially effective in reducing point-source pollution, legislation focused on regulating chemical effluents (such as the 1972 Clean Water Act, 1972 National Pollution Discharge Elimination System, and 1987 Water Quality Act) has been unsuccessful in meeting water quality objectives in the face of multiple impacts of non-point source pollution. Although chemical and physical approaches to protecting water quality are easy to monitor and legally defensible, they fail to measure more complex impacts on stream ecosystems, such as degradation due to non-point source pollution, habitat destruction, hydrological alterations, and changes in biological integrity. Developing and instituting biological criteria to supplement the narrow focus of point-source based regulations will help to maintain the biotic integrity of waterways (Karr 1991).

Biological Monitoring

In the past ~30 years, biological monitoring (biomonitoring) has emerged to play an essential role in stream watershed management and continues to become more prevalent due to a growing dissatisfaction with the effectiveness of existing water resource programs (Karr 1991; Karr and Chu 1999). Biomonitoring provides information about the biological condition of a sampling site compared to that of a reference site and sheds light on the impacts of human activities on the health of a stream ecosystem (Karr 1991; Karr and Chu 1999; Stoddard et al. 2006). With a scientific grounding, biomonitoring can be used to guide stream conservation and restoration efforts.

Hawkins (2006) reviews three biomonitoring methods currently used in the United States: 1) the biotic index, 1) the multimetric index [e.g. the index of biotic integrity (IBI)], and 3) observed/expected (O/E) assessments. Biotic indices (e.g. Hilsenhoff Index) characterize the biotic integrity of a site based on the pollution tolerance of specific taxa. Multimetric indices incorporate the pollution tolerance of taxa with other biotic metrics and compare degraded sites to reference sites, representing the least disturbed conditions in a given watershed. O/E assessments are used to measure taxonomic completeness on large scales. A commonality that all biomonitoring methods share is that, although they are able to detect degradation, none of these methods can isolate the specific source or sources of the degradation (Hawkins 2006). I used a multimetric biomonitoring approach in this study, because: 1) it is a local comparison of streams in northwestern Pennsylvania, 2) not enough data are available for an O/E assessment, and 3) previous studies in the French Creek

watershed (Allegheny River watershed) have used multimetric indices (IBIs) for over a decade, providing additional data to analyze.

Stream biomonitoring programs focus on fish, macroinvertebrates, and algae, with each protocol having distinct advantages and disadvantages. For example, although fish live longer than macroinvertebrates and algae, therefore better indicating long-term ecological health, they do not always detect non-point source disturbances (Lenat and Crawford 1994; Barbour et al. 1999). Compared to macroinvertebrate sampling, sampling fish is relatively cheap, requires minimal taxonomic expertise, and does not necessitate specimen mortality (Barbour et al. 1999). However, sampling fish requires more than one person, and the mobility of fish and seasonal changes in species distributions complicate sampling protocols (Gorman and Karr 1978; Paller 1995a; Paller 1995b). Fish occupy a range of trophic levels, and piscivorous fish acting as apex predators can reflect impacts on organisms occupying lower trophic levels. Because fish are highly dependent on in-stream habitat, however, fish biomonitoring protocols must be fine-tuned for local use (Kerans and Karr 1984; Schlosser 1990). Finally, much of federal, state, and local legislation focuses on fish populations, and fish are “charismatic megafauna” that are more appreciated by non-scientists than algae and macroinvertebrates (Berkman et al. 1986). Although North America has the greatest temperate freshwater biodiversity of any continent, roughly 39% of its described fish species are imperiled (i.e. vulnerable, threatened, or endangered) (Jelks et al. 2008). Because a diverse fish assemblage exists in northwestern Pennsylvania, fish are a useful bioindicator for local biomonitoring studies.

The Index of Biotic Integrity

The index of biotic integrity (IBI) was developed by James Karr in 1981 to serve as a broadly based, ecologically sound tool to evaluate biological conditions of stream ecosystems in the Midwestern United States (Karr and Chu 2000). By incorporating a broad range of indicators to characterize a condition and realizing that good health is not a simple function of those metrics alone, IBIs function very similarly to assessments of human health (Karr 1991). IBIs rely on the supposition that stream biota are the ultimate integrator of human influence in a watershed and seek to define the attributes of biological communities that become systematically altered by human activities (Karr and Chu 2000; Stoddard et al. 2006). The ideal IBI combines an array of indicators and is responsive to all of the various anthropogenic and natural stressors placed upon an aquatic ecosystem, while having limited sensitivity to natural sources of physical and biological environmental variation (Karr 1991). A key element of developing IBIs is an understanding of the characteristics of a biotic community under reference conditions.

The performance of ecological assessments (e.g. IBIs) depends on how well an aquatic ecosystem is characterized (Hawkins et al. 2010). The reference site approach to biomonitoring evaluates the condition of a biotic community at minimally or least disturbed sites (Stoddard et al. 2006). Reference condition refers to the “naturalness” of a biotic community and implies the absence of significant anthropogenic disturbance (Stoddard et al. 2006). Legacy effects of historical land use can confound biotic integrity at a reference site. The “Reference Condition for Biological Integrity” (RCBI) provides a concrete benchmark to evaluate the physical, chemical, and biological integrity of a waterway, consistent with the Clean Water Act (Stoddard et al. 2006).

The adaptability of an IBI to a broad range of geographical and environmental conditions makes it a useful tool in supporting the goals of the Clean Water Act and evaluating the biotic integrity of watersheds for conservation purposes (Karr 2006; Karr and Chu 2000). Because biomonitoring emphasizes humans' relationship with water, IBIs can be used to influence conservation and restoration efforts and guide the employment of best management practices in impaired watersheds (Karr 2006; Karr and Chu 2000). Quantifying ecologically significant changes in stream biota is crucial to effective ecosystem management (Dolédéc and Statzner 2010). IBI data applied in regulatory contexts can be used to prevent or address ecosystem degradation, preserving a higher water resource quality (Karr 2006). The current IBI used in the French Creek watershed is the FC Index 4 (Rieck 2009). FC Index 4 is a modification of the Goetz 2007 and Roy 2 1998 IBIs. The Roy 2 1998 IBI was based on Karr (1981) and Yoder's modification of it for Ohio streams (Ohio EPA 1996).

Land Use Impacts Biotic Integrity

Streams are the products of their surrounding landscapes and are therefore sensitive to the many different stressors that affect the stream's riparian zone, upland drainage, or in-stream channel (Fausch et al. 2010). Physical (damming, channelization, habitat alterations), hydrological (water withdrawals, canals, changes in flow regime), chemical (point- and non-point source pollution), and biological (invasive species, loss of biodiversity), as well as land use impacts (forest, agriculture, urban), have single and cumulative effects on stream ecosystems and their biotic integrity (Allan and Castillo 2008; Ormerod et al. 2010). Studies focused on the synergistic effect of land use have shown spatial (reach- and landscape-scale) and temporal differences on

the biotic integrity of streams (Allan et al. 1997; Lammert and Allan 1999; Allan 2004). Allan (2004) showed a correlation of biotic integrity and watershed land use, and a correlation of biotic integrity and the intactness of a stream's riparian zone. Stauffer et al. (2000) found that, while watershed land use influenced the biotic integrity of stream ecosystems, riparian cover had a stronger influence on fish populations. Roy et al. (2006) found that the impact of natural riparian cover on stream fish in urban watersheds is contingent on the sediment regime. Lee et al. (2001) investigated the relation between stream fish community composition and riparian intactness in an agricultural watershed, and found that streams with wooded upstream riparian zones had higher species richness and higher IBI scores. IBI studies have shown that watersheds influenced by agricultural land use have lower IBI scores than the least-impacted (reference) watersheds (Lyons 2006; Wang et al 1997). However, the influence of watershed disturbance (e.g. agriculture) may have a threshold of effect on biotic integrity (Wang et al. 1997). Previous biomonitoring studies in the French Creek watershed have used watershed and riparian land use as a stressor gradient to test the responsiveness of locally modified IBIs (Roy 1998, Goetz 2007, and Rieck 2009). This study addressed the relationship of watershed land use, riparian intactness, and biotic integrity, and examined land use at various scales to test the responsiveness of an IBI.

Land Use in the French and Oil Creek Watersheds

The French Creek watershed (3,200 km²) of northwestern Pennsylvania is recognized as one of the most ecologically significant watercourses in the northeastern United States, hosting 25 species of freshwater mussels (including 2 federally endangered species) and approximately 100 species of fish (Nature Conservancy

2008). Threatened or endangered taxa, including several madtom and lamprey species, as well as eight of the 15 species of darters, are included in the watershed's diverse fish assemblage (Straffin 2003). Despite runoff from row-crops and pastures, effluent from dairy farms and wastewater treatment plants, urban runoff, and invasive species, biodiversity in the watershed has remained largely intact, perhaps because the drainage never suffered any of the severe pollution bottlenecks encountered elsewhere in the Allegheny River watershed (Whitney and DeCant 2003; Nature Conservancy 2008). Thus, the diversity of fauna and the comparably lower impact of human degradation in French Creek watershed could make it a suitable reference site for testing IBIs in other watersheds of the upper Allegheny River.

The Oil Creek watershed (827 km²) borders the French Creek watershed to the east. The Oil Creek watershed likely once hosted a diversity of aquatic fauna comparable to that of the French Creek watershed, since both watersheds were subjected to similar glacial drainage reversals and associated species introductions, as well as share connectivity to the Ohio River watershed, through the Allegheny River. However, pollution bottlenecks and drastic changes in watershed land-use have altered the biological diversity of Oil Creek's aquatic fauna (Baxter et al. 2005). On August 27, 1859, Edwin Drake struck oil along the banks of what is now known as Oil Creek. By the 1870s, "oil fever" had spread into the surrounding tributaries of the Oil Creek valley (PA Historical and Museum Commission 2009). During the late 1800s, the Oil Creek valley was a moonscape of deforestation and landscape manipulation, and the stream was a slurry of muddy water and oil (Fig. 1 A). Based on the doctrine of "creekology", the idea that oil was found adjacent to streams, oil pioneers drilled along tributaries,

many of which were dammed to facilitate the flooding of Oil Creek for moving oil barges and timber (Oil Creek State Park 2007). Pipelines eventually replaced wagons, barges, and rail cars as an economic means of transporting oil; today, remnants of these pipelines can be found along many of the watershed's tributaries (Fig. 1 B). The year 2009 marked the 150th anniversary of the birth of the oil industry, although commercial drilling in the area is only a trickle of what it used to be (Fig. 1 C) (PA Historical and Museum Commission 2009). Today, most of the Oil Creek watershed has been reforested, and current land use impacts are comparable to or lower than those of the neighboring French Creek watershed (Whitney and DeCant 2003).

Purpose of Study

The purpose of this study was to use the French Creek IBIs developed by Roy (1998), Goetz (2007), and Rieck (2009) to assess the effects of watershed and riparian land use on tributaries in the Oil Creek watershed of northwestern Pennsylvania. Testing the French Creek IBIs in the Oil Creek watershed provided a broader stressor gradient to examine the utility of the indices against and shed light on the effectiveness of these indices in neighboring watersheds. I hypothesized that IBI scores would be lower in 2nd to 4th order tributaries of Oil Creek than in tributaries of French Creek with similar forest cover because of differences in legacy effects. This study also addressed other between-watershed differences that affect the construction of a regional IBI, including differences in β diversity (pool of available species) and stream gradient.



METHODS

Site Selection

Geographic Information Systems (ARCMAP by ERSI computer programs; ArcGIS Desktop 9.3.1) (GIS) were used to identify potential study watersheds (ERRI 1997) in the Oil Creek drainage; twelve watersheds were chosen for this study (Appendix Fig. 2). Because the standardization of field methods and analytic procedures (quality control/quality assurance) is a fundamental prerequisite to the utility of the IBI, sample reaches were chosen to be near bridges, so that data from this study would be comparable to data from previous IBI studies in the French Creek watershed (Karr 1991). A Trimble GeoExplorer 3 handheld GPS unit was used to record the location of each sample reach. GPS data were imported into GIS and used to delineate watershed boundaries and calculate watershed area (Appendix A). The distance from each sample site to the main stem of Oil Creek was measured according to the segment length of USGS (2004) high-resolution flow lines (Appendix A).

Land Cover Analysis

In previous studies, locally modified IBIs were used to examine the correlations of IBI scores and (1) the percentage of natural watershed cover and (2) the percentage of upstream intact riparian zone in each study watershed (Stauffer et al. 2000). In this study, streams were selected to represent a wide range of reach- and landscape-scale land uses, because the synergistic effect of multiple land use stressors at different scales influences biotic integrity (Allan et al. 1997; Lammert and Allan 1999; Allan 2004). The percentage of natural cover, agriculture, developed space, and open water were calculated for each watershed and within a 30m buffer of the streams using

reclassified 2001 National Land Cover Dataset (NLCD) data (USGS 2003) (Table 1) (Appendix B) .

Table 1. NLCD 2001 landcover reclassification scheme.

NLCD 2001 Classification	Reclassification
41 Deciduous Forest	1 Natural Cover
42 Evergreen Forest	
43 Mixed Forest	
52 Shrub/Scrub	
90 Woody Wetlands	
95 Marsh Wetland	
31 Barren Land	2 Agriculture
71 Grassland/Herbaceous	
81 Pasture/Hay	
82 Cultivated Crops	
21 Developed- open space	3 Developed Space
22 Developed- low intensity	
23 Developed- med intensity	
24 Developed- high intensity	
11 Open Water	4 Open Water
No Data	No Data

Slope Analysis

Stream gradient changes longitudinally in a downstream fashion, creating diverse habitats in a hierarchical fashion (Vannote et al. 1980). Because fish community composition is strongly influenced by flow regime and habitat heterogeneity, stream gradient was analyzed in the Oil Creek study watersheds to help elucidate underlying variation in IBI scores (Schlosser 1990). Main stream channels were divided into 1 km segments, and then raster elevation data (NASA et al. 2002) were converted to polygons and overlaid with the channel division layer to identify the stream elevation at each division (Appendix Fig. 3). Segment length and elevation data were used to calculate the following: 1) total slope [change in rise/run from the top of the main

channel to the sampling point (m/km)], 2) slope 1 km upstream of the sample reach (m/km), and 3) the standard error of the mean for the average slope (m/km) (Appendix C). To investigate the influence of stream slope on fish diversity between the Oil Creek and French Creek watersheds, total slope was calculated for 11 watersheds with comparable drainage areas and natural watershed cover in the French Creek watershed (Appendix D).

Baseline Hydrology and Water Chemistry

To determine the comparability of the Oil Creek streams sampled in this study with streams in the French Creek watershed, base flow discharge was measured with a Marsh-McBirney® Model 2000 Portable Flowmeter at each sample reach in November 2009 (Appendix A). Basic water chemistry tests (temperature, pH, total dissolved solids (TDS), dissolved oxygen, nitrate-nitrogen, total phosphorus, turbidity, and alkalinity) were conducted in January 2010 to reveal the presence or absence of allied stressors (Appendix E) (Miltner et al. 2004; American Public Health Association et al. 1994). Allied stressors (e.g. wastewater discharges, accidental spills, intentional dumping, and stream channel dredging and filling) change the hydrologic and water quality impacts of land cover alone, therefore confounding the effects of land use on stream biotic integrity (Miltner et al. 2004). Onsite, dissolved oxygen samples were fixed according to the standard Winkler method, and temperature, pH, and TDS were measured using Oakton® handheld meters (American Public Health Association et al. 1994). Water samples were collected and stored in a cooler to be brought back to the lab, where nitrate-nitrogen, turbidity, and alkalinity were determined using Hach test kits (Hach Company, Loveland, Colorado, USA). A standard Gran titration was used to measure

alkalinity; total phosphorus was measured according to the molybdate blue method (American Public Health Association et al. 1994; Strickland and Parsons 1968).

Reach Scale Riparian Zone and Channel Evaluation

Biomonitoring protocols recommend evaluating the riparian zone and stream channel characteristics at a sampling site to complement the collection of biological data (Peterson 1992; Stauffer et al. 2000; Allan & Castillo 2008). Riparian zone characteristics and stream channel intactness were evaluated at each site in November 2009 using a Riparian, Channel, and Environmental Inventory (RCE) and the Pennsylvania Department of Environmental Protection Water Quality Network Habitat Assessment (WQNHA) (Appendix F). The RCE was modified for local use from Petersen (1992) by Marion Norris (1997) and Creek Connections (2002) and has been used in previous local biomonitoring studies (Roy 1998, Goetz 2007, and Rieck 2009). The WQNHA (PADEP 2008) is used by the Pennsylvania Department of Environmental Protection throughout the region and has the potential benefit of being used for Total Maximum Daily Load (TMDL) designation. Under section 303(d) of the Clean Water Act, impaired waters are to be ranked according to TMDLs, or calculations of the maximum amount of a pollutant that a water body can receive and still safely meet water quality standards.

Electrofishing

Pulsed DC electrofishing (Smith-Root Model 12; 15-120Hz; 200-1000V) was conducted in October and November 2009. A two-pass removal protocol was followed, since single-pass electrofishing can underestimate cyprinid richness (Meador et al. 2003). At each site, 100m sample reaches were chosen so that they contained at least

one riffle-pool-riffle sequence. Anchored blocking seines (1/4" mesh) were placed 50m apart to delimit a quantitative sample, and an additional 50m stretch of stream adjacent to the quantitative sample reach was sampled qualitatively (Fig. 4 A). Electrofishing was conducted by 2- or 3-person crews in an upstream zigzag pattern. Fish were collected with dip nets, placed into buckets, identified to species, counted, and then released (Fig. 4 B). Because mottled sculpin, *Cottus bairdi*, numerically dominate local fish communities, they were assessed by visual in-stream counts during the first pass of the quantitative sample. Occasionally, specimens could not be positively identified in the field and were brought back to the lab and placed in aquaria overnight for identification (Cooper 1983). Trout were recorded by size (<3", 3-6", 7-12", 13-17", and >18") for the entire sample reach. To avoid errors associated with small cyprinid sample sizes at sites where less than 100 cyprinids were caught in the quantitative sample, additional cyprinids caught in the qualitative sample were added to the quantitative cyprinid data, so that all IBI metric calculations evaluated at least 100 cyprinids.



Figure 4. (A) Anchored blocking seines were used to delimit the quantitative sample reach. Easily identifiable specimens (B) were recorded in the field and promptly released.

Data Analysis

Relationships of land use variables (e.g. the percentage of natural watershed land cover and the percentage of intact upstream riparian zone with 30m of each stream), RCE/ WQNHA scores, and IBI values (FC Index 4, Goetz 2007, and Roy 2 1998 IBIs) (Appendix F) were examined using the least-squares method. These relationships were analyzed to evaluate the influence of watershed and reach scale land use on biotic integrity, and to identify the best IBI for use in the Oil Creek and French Creek watersheds. A weighted score, calculated as:

$$\text{Normalized Score} = \frac{(\text{Site Value}) (50)}{(\text{Average of All Sites})}$$

was calculated for the percentage of natural watershed cover, the percentage of natural riparian cover, and RCE scores for each stream. Weighted scores were summed to produce composite land use scores, which were used to test the responsiveness of IBIs. Watershed area, the distance from sampling sites to the main stem of Oil Creek, and slope data were examined to determine their influences on species richness and IBI scores. Correlations of RCE metrics 1-4 (riparian land use) and metrics 5-12 (channel characteristics), slope data, and land cover data were examined to consider the possible influence of legacy effects on biotic integrity.

RESULTS

Site Exclusion from Analysis

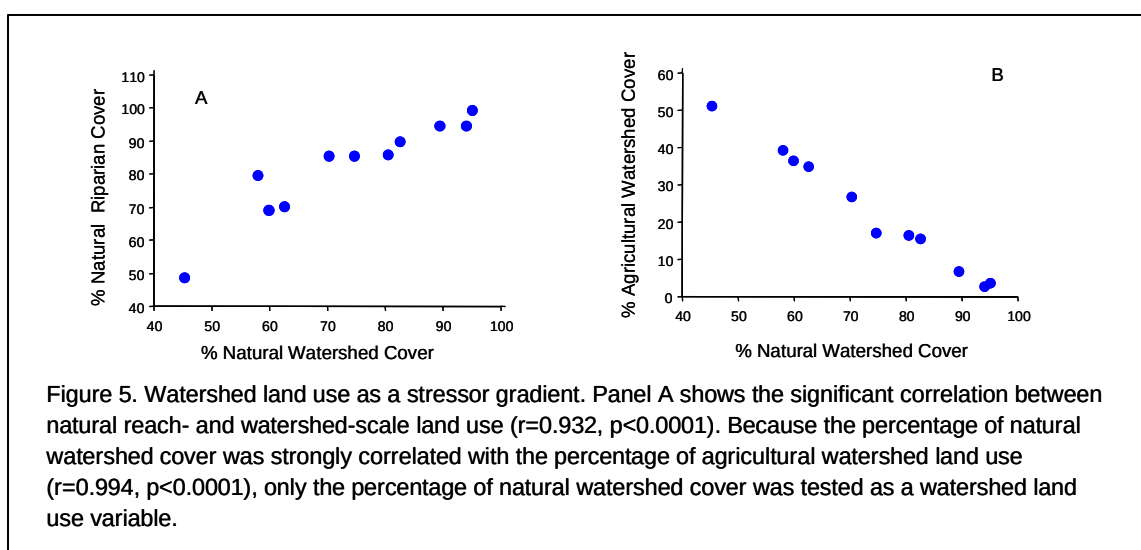
The focus of this study was to determine how non-point source pollution from land use (e.g. farming, timbering, etc.) affects the biotic integrity of streams. One of the 12 sites investigated in this study, Mosey Run, was 1 km downstream from a sewage

treatment plant, a potential source of point-source pollution, referred to as an “allied stressor” in biomonitoring literature (Walsh et al. 2005). Extraordinarily high (compared to other Oil Creek sites) alkalinity and the lowest DO of any site were revealed by chemical analyses, suggested that effluent from the sewage treatment plant confounds the effect of land use on the water quality at this site (Appendix E). Indeed, reclassified 2001 NLCD land use data for the Mosey Run watershed suggested that the site would score much higher on the Index 4 IBI than it actually scored, suggesting that an allied stressor might be confounding the impacts of land use on biotic integrity, as observed in other studies (Van Snik Gray et al. 2005). An analysis of electrofishing data showed an abundance of moderately tolerant centrarchids (sunfish), no sensitive cyprinid (minnow) species, and a domination of pollution-tolerant white suckers (16.9% of fish collected, excluding mottled sculpin) in the Mosey Run sample reach. Thus, to keep the focus of this study on land use impacts, the Mosey Run site was excluded from the analysis.

Land Use as a Stressor Gradient

Across the remaining 11 study watersheds, watershed land use ranged from 45.2% to 95% natural cover; riparian land use ranged from 48.9% to 99.5% natural cover (Appendix B). The percentage of natural watershed cover was strongly correlated with percentage of natural riparian cover ($r = 0.932$; $p < 0.0001$) (Fig. 5). Agricultural land use ranged from 2.7% to 51.4% at the watershed level and ranged from 0.2% to 48.1% in riparian zones (Appendix B). Because the percentage of natural watershed cover was strongly correlated with the percentage of agricultural watershed land use ($r = 0.994$; $p < 0.0001$), only the percentage of natural watershed cover was tested as a watershed land use variable (Fig. 5).

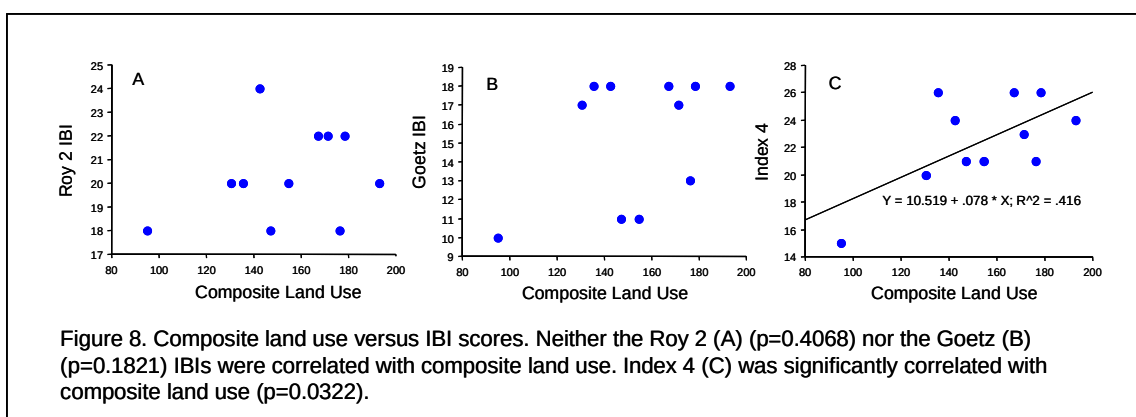
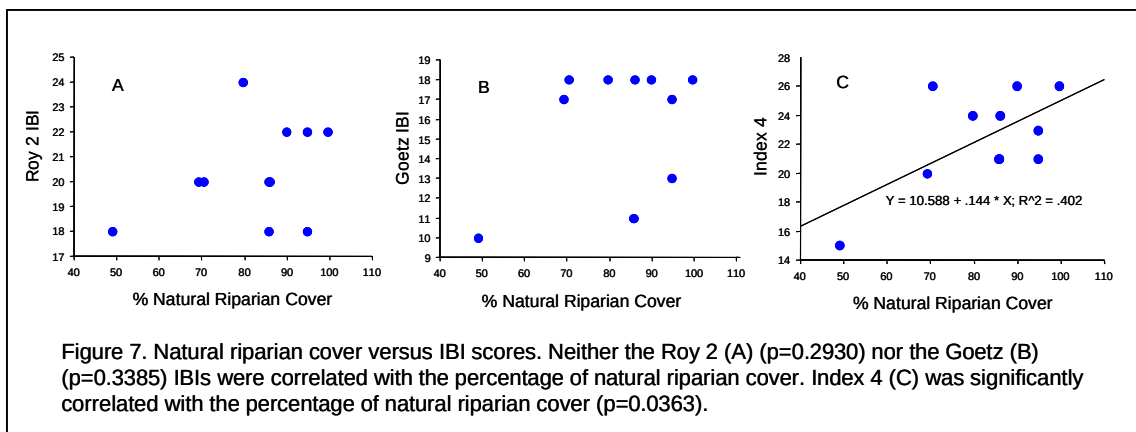
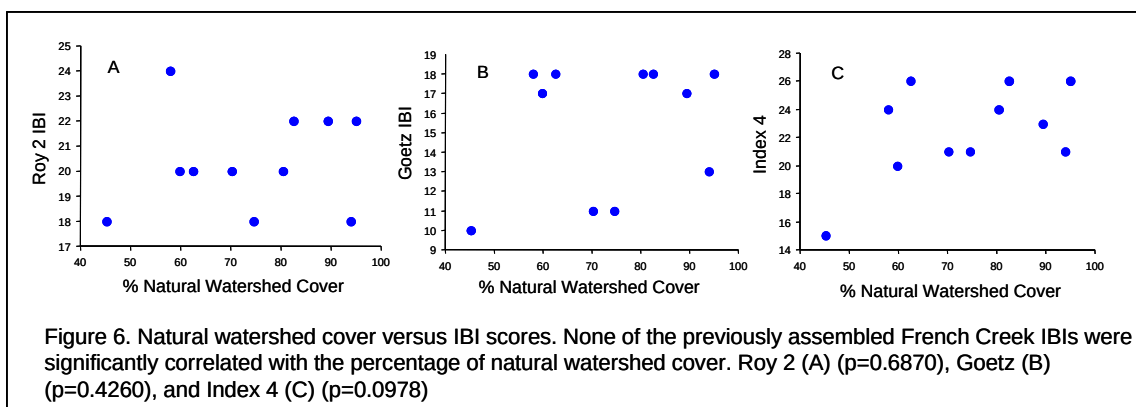
RCE scores ranged from 160 to 259 (highest possible score= 280), and WQHNA scores ranged from 159 to 221 (highest possible score= 240) (Appendix F). RCE and WQHNA scores were significantly correlated ($r= 0.819$, $p=0.0020$). Because habitat evaluations in the French Creek watershed have been done with the RCE and RCE and WQHNA scores are highly correlated, only RCE data were used to evaluate reach scale habitat in this study. RCE metrics 1-4 scores (riparian land use evaluation) were correlated with RCE metrics 5-12 scores (channel evaluations) ($r=0.636$; $p=0.0354$). RCE metrics 1-4 scores (riparian land use evaluation) were correlated with the percentage of natural watershed cover ($r=0.642$; $p=0.0333$) and with the percentage of natural riparian cover ($r=0.801$; $p=0.0030$). RCE metrics 5-12 scores (channel evaluations) were not correlated with natural watershed cover ($r=0.543$; $p=0.0843$), but were correlated with natural riparian cover ($r=0.656$; $p=0.0283$). Composite land use scores ranged from 95.07 to 192.87 and reflected the relative contributions of the percentage of natural watershed cover, the percentage of natural riparian cover, and total RCE scores.



Nitrate-nitrogen concentrations showed a significant negative correlation with the percentage of natural watershed cover ($F_{1,9} = 7.813$; $p = 0.0209$; $R^2 = 0.465$). Significant negative correlations were found between alkalinity levels and the percentage of natural watershed cover ($F_{1,9} = 11.524$; $p = 0.0079$; $R^2 = 0.561$), the percentage of natural riparian cover ($F_{1,9} = 12.098$; $p = 0.0070$; $R^2 = 0.573$), and composite land use ($F_{1,9} = 14.333$; $p = 0.0043$; $R^2 = 0.614$). A significant negative correlation was also found between TDS concentrations and composite land use ($F_{1,9} = 7.281$; $p = 0.0245$; $R^2 = 0.447$). No other significant relationships were found between chemical data and land use.

Fish IBIs and Land Use in the Oil Creek Watershed

None of the previously assembled French Creek IBIs were significantly correlated with the percentage of natural watershed cover (Roy 2 IBI $p=0.6870$, Goetz IBI $p=0.4260$, Index 4 $p=0.0978$) (Fig. 6). Neither the Roy 2 IBI ($p=0.2930$) nor the Goetz IBI ($p=0.3385$) was significantly correlated with the percentage of natural riparian cover. However, Index 4 increased with natural riparian cover ($p=0.0363$) (Fig. 7). Whereas the Roy 2 and Goetz IBIs did not significantly correlate with composite land use scores (Roy 2 $p=0.4068$, Goetz $p=0.1821$), FC Index 4 showed a significant positive correlation with composite land use ($p=0.0322$) (Fig. 8). Within the Oil Creek watershed, composite land use (the sum of the normalized effects of the percentage of natural watershed cover, the percentage of natural riparian cover, and the RCE score) gave the best fit for the Index 4 (Fig. 8c).



Fish IBIs in the French Creek and Oil Creek Watersheds

Electrofishing data collected in previous studies (Roy 1998, Goetz 2007, Rieck 2009) for 39 sites in the French Creek watershed were compared with the electrofishing data collected in this study. There was no statistical difference in species richness ($p=0.6591$) (Table 2) or Index 4 IBI scores ($p=0.3298$) (Table 3) between French Creek sites ($n=39$) and Oil Creek sites ($n=11$). However, when sites were matched according to similar watershed area and land use (Appendix D), species richness was significantly lower ($p=0.0052$) and IBI scores were significantly higher ($p=0.0297$) in Oil Creek watersheds than in French Creek watersheds. Although more species were found at a given site in the French Creek watershed, the same four species [mottled sculpin (*Cottus bairdi*), blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), and central stoneroller (*Campostoma anomalum*)] dominated the total catch in both watersheds (70% for French Creek sites and 79% for Oil Creek sites) (Appendix G). For a given composite land use score, fish richness was lower in the Oil Creek watershed than in the French Creek watershed ($p=0.0079$). There was no statistical difference in stream slope (m/km), calculated as the change in elevation from the top of the main channel to the sample reach divided by the length of the main channel, for the matched Oil Creek and French Creek sites ($p=0.2837$). Neither the distance from sample reaches to the main stem of Oil Creek ($p=0.7353$) nor watershed area ($p=0.9934$) was significantly correlated with species richness. Within the Oil Creek drainage, none of the slope factors calculated in this study had statistically significant correlations with species richness (total slope $p=0.2833$, slope 1km upstream $p=0.4543$, and standard error of the mean slope $p=0.2883$).

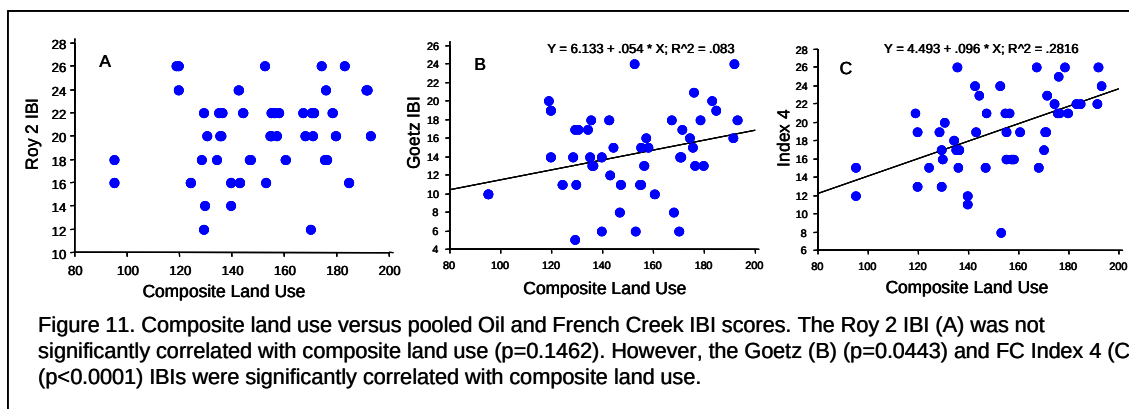
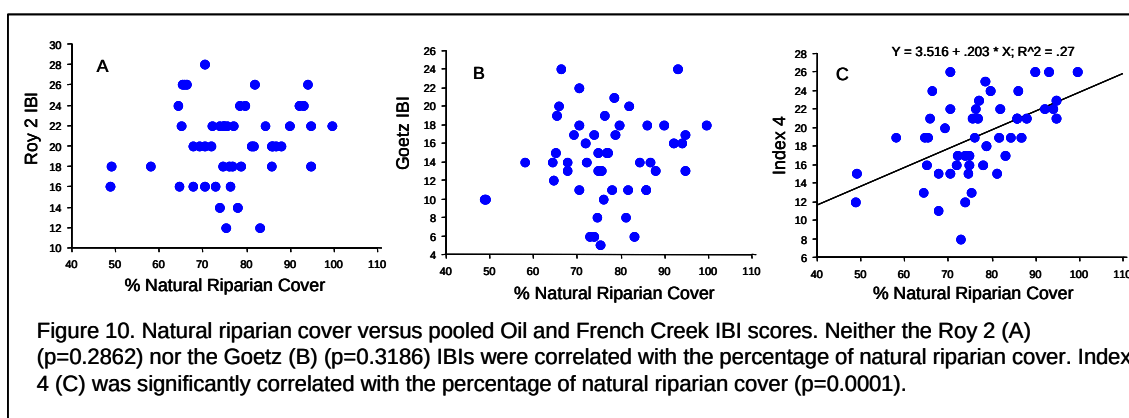
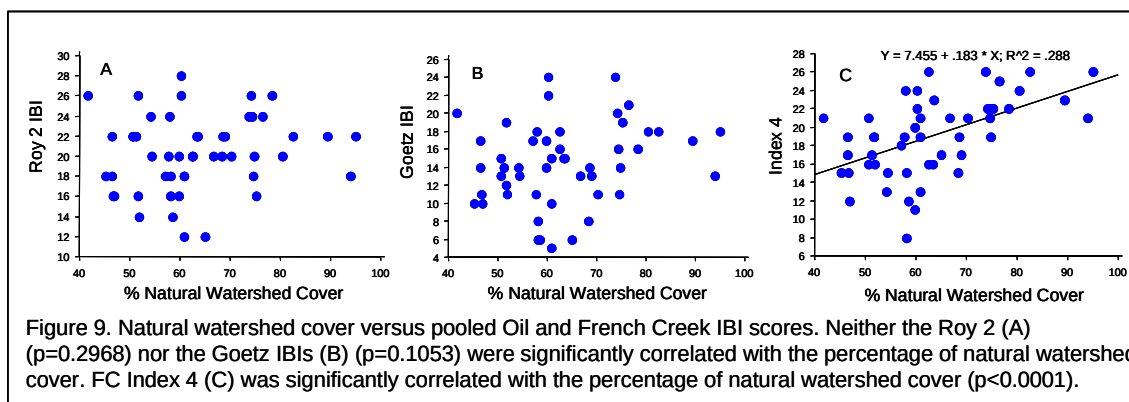
Table 2. ANOVA Table for fish richness between paired Oil Creek and French Creek sites.

	DF	Sum of Squares	Mean Square	F-Value	P-Value	Lambda	Power
Watershed	1	2.804	2.804	.197	.6591	.197	.071
Composite Land Use	1	27.459	27.459	1.932	.1714	1.932	.260
Interaction	1	12.044	12.044	.847	.3622	.847	.140
Residual	45	639.523	14.212				

Table 3. ANOVA Table for Index 4 IBI scores between paired Oil Creek and French Creek sites.

	DF	Sum of Squares	Mean Square	F-Value	P-Value	Lambda	Power
Watershed	1	10.996	10.996	.971	.3298	.971	.154
Composite Land Use	1	115.665	115.665	10.209	.0026	10.209	.896
Interaction	1	25.342	25.342	2.237	.1417	2.237	.294
Residual	45	509.836	11.330				

Index 4 IBI scores from French Creek watershed tributaries (n=39) analyzed in previous studies (Roy 1998, Goetz 2007, and Rieck 2009) were compared to Index 4 IBI data from the Oil Creek study sites (n=11). For the pooled data (n=50), neither the Roy 2 IBI nor the Goetz IBI was significantly correlated with the percentage of natural watershed cover (Roy 2 $p=0.2968$, Goetz $p=0.1053$) (Fig. 9) or the percentage of natural riparian cover (Roy 2 $p=0.2862$, Goetz $p=0.3186$) (Fig. 10). Roy 2 IBI scores were not ($p=0.1462$), but Goetz IBI scores were significantly correlated with composite land use scores ($p=0.0443$) (Fig 11). FC Index 4 performed the best against the pooled French Creek and Oil Creek data, and was significantly correlated with the percentage of natural watershed cover ($p<0.0001$) (Fig 9), the percentage of natural riparian cover ($p=0.0001$) (Fig 10), and with composite land use ($p<0.0001$) (Fig. 11). For the pooled French Creek and Oil Creek data, composite land use gave the best fit for the Index 4 IBI (Fig. 11C).



DISCUSSION

The hypothesis that IBI scores would be lower in 2nd to 4th order tributaries of Oil Creek than in tributaries of French Creek with similar forest cover was not supported. When all sites from both watersheds were pooled, there was no statistically significant relationship between watershed and either fish richness or FC Index 4 scores. However, when sites were paired based on similar size and land use, Oil Creek IBI scores were higher than French Creek IBI scores, and species richness was lower in Oil Creek sites than in French Creek sites. No evidence was found to support the negative influence of legacy impacts on Oil Creek IBI scores. By influencing physical and chemical variables, historical land use can influence the biotic integrity of running waters (Maloney et al. 2009).

The French Creek watershed is acclaimed for its diverse fish fauna, which may be unrivaled by any other similar sized watershed in Pennsylvania (Straffin 2003). The exceptional fish diversity of the watershed can be attributed to a combination of the following: 1) faunal exchanges via glacial drainage reversals at the boundaries between Allegheny River tributaries and those of the St. Lawrence, Susquehanna, and Genesee River drainages, 2) connectivity to the Ohio River fish fauna, which is the most diverse in North America, and 3) a lack of severe historical pollution bottlenecks (Straffin 2003; Trautman 1957; Whitney and Decant 2003). Because of the unique conditions that have established biodiversity in the watershed and because that biodiversity has largely been maintained, using the French Creek watershed to construct a regional IBI for use in Allegheny River tributaries in northwestern Pennsylvania may be unrealistic. Although the FC Index 4 metrics worked in the Oil Creek watershed, IBI metric cutoff

values might need to be modified to reflect watershed differences in the species pool in the two streams. However, as far as I know, there are no compilations of fish species composition for the Oil Creek drainage for comparison with those available for French Creek.

Do Differences in Diversity Confound Interpretations?

Between watersheds, individual Oil Creek sample sites appear to have lower species diversity than those in the French Creek watershed. However, the smaller stream sample size in the Oil Creek watershed (Oil Creek $n=11$; French Creek $n=39$) could be influencing the species richness differences between watersheds. Across all subdrainages, richness in Oil Creek streams was lower than richness in French Creek streams for a given size or a given forest cover. But, in neither watershed is there evidence that watershed size, the percentage of natural watershed cover, the proximity of sample sites to the main stem of the creek, or stream gradient affects fish diversity. Although 70% and 79% of the fish caught at all sample sites in the French Creek and Oil Creek watersheds (respectively) are the same four species and there are roughly the same number of rare species present, at a given site in the French Creek watershed, cyprinid diversity and overall diversity tends to be greater in the latter (Appendix G). There are several non-mutually exclusive hypotheses for the observed differences in between watershed fish richness: 1) differences in watershed size, 2) the main stem of Oil Creek has not recovered from severe historical degradation (land use legacy effect), and the recolonization of fish is incomplete or uneven, 3) high gradient tributaries host fewer species than moderate gradient tributaries, 4) fully forested watersheds have low productivity, which in turn reduces species diversity, and 5) the

high density of piscivorous brown trout in Oil Creek sites reduces diversity. Below, I argue the merits of each hypothesis.

Influence of Watershed Size

In this study, watershed size did not correlate with species richness. To reduce the chances of biogeographical sampling error, the streams sampled in this study were chosen to fit within a range of 10-30 km² (Vondracek et al. 2009). Lyons (2006) developed a fish IBI for intermittent headwater streams (watersheds ranged 1.7-41.5 km²) in Wisconsin, USA, and found that IBI metrics were not related to stream size. It is possible that this range of sizes is too limited to show a relationship between watershed size and species richness. However, watershed size has been shown to correlate with species richness, and expectations for species richness metrics can be a function of ecosystem size (Fausch et al. 1984, Miller et al. 1988). Within a watershed, spatial and temporal differences in disturbance also influence fish community structure (Gorman and Karr 1978, Schlosser 1990). Because sample sites vary in size (drainage area, order, baseflow discharge, and mean width) and longitudinal position within a watershed, an interaction of these and and/or other influences could be determining fish community composition. Fausch et al. (1984) and Karr et al. (1986) showed that species richness can vary with the relative position of a sample reach within a watershed. In my study, the distance of sample sites to the main stem of Oil Creek did not significantly influence species richness.

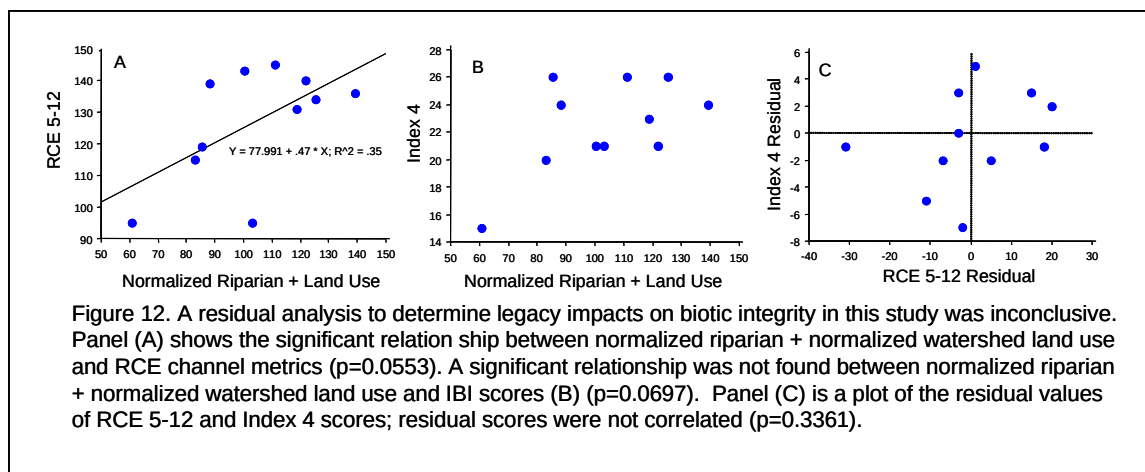
Legacy Effects and Fish Recolonization

Accurately assessing the biotic integrity of streams in minimally impaired watersheds (e.g. high forest cover) requires a better understanding of land use legacies

(Maloney et al. 2009). However, few studies have investigated the influence of legacy effects on present-day biotic integrity (Harding et al. 1998). Historical watershed deforestation can affect stream flow regime, habitat heterogeneity, and woody debris retention (Poff et al. 1997). Alterations of channel morphology and riparian intactness can be influenced by historical watershed deforestation (Zhang et al. 2009). Although the forest cover in many of the watersheds may have regenerated, the angularity of cobble and boulder substrates observed in the forested Oil Creek streams sampled in this study provide some evidence of severe historical watershed disturbance. Because of the observed disparity between stream channel intactness and forest cover, I conducted a residual analysis to attempt to quantify legacy effects of historical land use.

No correlation was found between whether a stream had a low IBI score, given its landscape (natural watershed cover and natural riparian cover), and whether its channel characteristics were low (inferred from RCE metrics 5-12 score), given its landscape ($p=0.4145$) (Fig 12). It is possible that no relationship was found because: 1) there are not strong legacy effects in the Oil Creek watershed, 2) the residual analysis did not adequately quantify the legacy impact of historical deforestation on present-day channel integrity, and/or 3) replication was too low ($n=11$) to adequately test a legacy effect. The most likely explanation for the inability of the residual analysis to detect legacy effects may be that the analysis was not comprehensive enough (e.g. perhaps RCE metrics 5-12 alone do not tell enough about channel condition). Quantifying channel condition in a more rigorous manner (see Peterson et al. 2009) and supplementing land use with this data would create a more robust analysis to test legacy effects. Information on the extent of landscape manipulation associated with the

late 1800s oil boom (Oil Creek State Park 2007; PA Historical and Museum Commission 2009; Whitney and DeCant 2003) highlights the severity of the degradation in the Oil Creek watershed. Zhang et al. (2009) observed changes in macroinvertebrate taxon richness in 11 streams with different historical forestry uses (~40 years). Harding et al. (1998) compared 24 watersheds with different agricultural land-use histories and found that historical watershed land use (~50 years) was a better predictor of macroinvertebrate and fish diversity than was current watershed and riparian land use. Data from Harding et al. (1998) suggest that stream fauna may require decades to recover after short-term natural and human disturbances, even when riparian conditions have returned to a pre-disturbance state.



The ecological characteristics of fish (turbidity tolerance and substrate requirements) can limit their dispersal ability within a watershed (Thornburgh and Gido 2010). Mainstem river assemblages may rely on direct tributaries for refuge from pulsed disturbance, while main stem channels serve as corridors for the connectance of metapopulations (Thornugh and Gido 2010). Because of the movement of fish to and from refugia and source populations in connected rivers, the precision of fish IBI metrics

may be improved by calibrating scoring criteria according to the position of a sample site within a drainage network (Hitt and Angermeier 2008). This study did not find a correlation between species richness and the proximity of sample sites to the main stem of Oil Creek. During the peak of landscape degradation, the fish fauna of the Oil Creek watershed could have found refuge in less-impacted tributaries of the main stem. It is possible that the fish community is still recovering from drastic historical watershed degradation, and that re-colonization from source pools (less-impacted tributaries or the neighboring French Creek watershed) could be ongoing. According to Grossman et al. (1990), the recovery rates of stream taxa are affected by factors such as: “1) persistence of the effects of disturbance, 2) species’ differential abilities to survive disturbance and recovery, 3) presence of refugia, and 4) hydrologic conditions.” Fish communities characterized by species with shorter life-spans, smaller body sizes, earlier sexual maturity, and a potadromous lifecycle (migrating within a freshwater system) can recover more rapidly from severe anthropogenic disturbance than other fish communities (Schlosser 1990).

Stream Gradient

As stream gradient decreases along a longitudinal continuum, habitat heterogeneity (complexity) typically increases as a result of alternating riffles and pools (Vannote et al. 1980). As stream habitat heterogeneity increases, so should species diversity, especially in small streams, where most fish are habitat specialists (Gorman and Karr 1978). Lammert and Allan (1999) found a strong correlation between fish abundance and channel morphology (percentage of pool habitat and percentage of run habitat). Specifically, substrate, depth, and current are three major components that

influence the microhabitat specialization of stream fishes (Gorman and Karr 1978). While investigating the use of an IBI in Wisconsin watersheds ranging 1.7 to 41.5 km², Lyons (2006) found no relation between IBI metrics and stream gradient. Similarly, in my study, species richness was not correlated with stream slope. The presence of deep pool habitats can increase species richness (Schlosser 1982). In many of the Oil Creek streams sampled in this study, pool habitat was scarce, and deep water zones were commonly limited to areas under or downstream of bridges. Not surprisingly, these bridge-created pools reliably produced centrarchid (sunfish) species that were found nowhere else in the sample reach. Similar observations have been made at sample sites in the French Creek watershed, highlighting the importance that new sample sites should include at least one deep-pool habitat, like those typically found at bridges (Barbour et al. 1999).

Primary Productivity

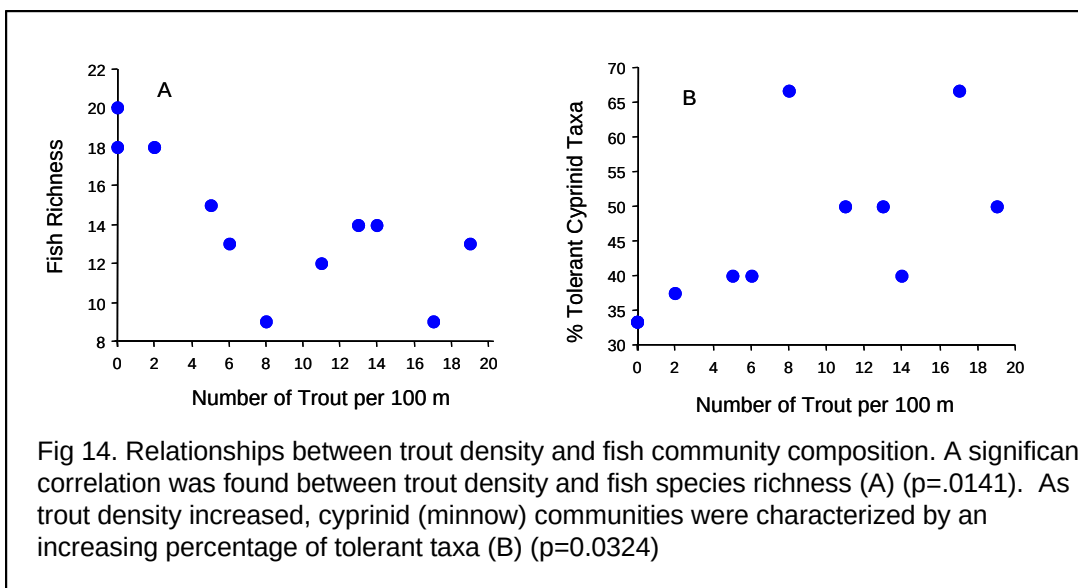
Along with increasing habitat heterogeneity, moderate disturbance also enhances the diversity of stream fish communities (Allan and Castillo 2008). Degraded coldwater streams have been shown to have higher primary productivity and higher fish species richness, relative to non-impaired coldwater streams (Smith et al. 1997). In this study, algivorous fish (central stonerollers, *Camptostoma anomalum*), tended to be found in streams with an open canopy and higher observed primary productivity. Most of the sites with a high percentage of watershed forest cover in this study are also watersheds with steeper slopes (USGS 2003), and lower species richness could therefore be a product of: 1) more uniform in-stream habitat, and/or 2) lower primary productivity.

Trout Piscivory

In the Oil Creek sites, a significant negative correlation was found between the number of trout in a 100 m sample reach and fish species richness ($F_{1,9} = 9.22$; $p = 0.0242$; $R^2 = 0.506$) (Fig 13). Nearly all trout were naturalized brown trout, *Salmo trutta*. Many of these streams had a healthy age structure of mixed size individuals, suggesting that natural recruitment is occurring (Fig 13). Large brown trout (>12") are piscivorous, and it is possible that trout piscivory eliminates rare cyprinid species (regularly encountered in the French Creek study sites but not in the Oil Creek study sites), thus reducing taxon diversity (Jenkins et al. 1999). A significant correlation was also found between the number of trout in a 100 m sample reach and the percentage of cyprinid taxon classified as tolerant species ($F_{1,9} = 6.38$; $p = 0.0324$; $R^2 = 0.415$) (Fig 14). It is plausible that tolerant cyprinids (e.g. blacknose dace (*Rhinichthys atratulus*) and creek chub (*Semolilus atromaculatus*) can reproduce quicker than rare cyprinids, thus maintaining stable populations in the presence of trout (Townsend 1996). The relationship between trout density and species richness could be spurious and might actually be driven by high canopy cover; i.e., such streams are good trout habitat because of cold temperatures, higher dissolved oxygen levels, and low primary productivity (hence low in diversity) because of the canopy cover (Jenkins et al. 1999). Whether the observed negative relationship between trout density and cyprinid (and overall species) diversity is causal or related to some other covariate would require experimental analysis.



Figure 13. The Oil Creek watershed boasts very healthy trout streams. Panel (A): McLaughlin Run. Panels (B) and (C): West Branch Caldwell Creek.



Using Fish IBIs to Estimate the Effects of Land Use on Stream Health

This study demonstrated that a composite land use score (the sum of the normalized effects of reach scale land use, watershed land use, and a stream habitat evaluation) can explain the variation in IBI scores better than land cover or habitat

information alone. Lammert and Allan (1999) found that a combining landscape and habitat variables were better predictors of biotic integrity than landscape or habitat variables alone. McCormick (2001) developed a regional IBI for Mid-Atlantic streams, and found a strong stronger correlation between a measure of aggregate habitat quality and IBI scores than between forest cover, watershed and riparian quality, or channel habitat quality alone. Varying spatial interactions of land use (reach scale and watershed scale) influence stream health. For example, riparian intactness can sometimes rescue land use. Roy et al. (2006) showed that fish communities are influenced by a combination of hydrologic, sediment, and riparian disturbances. Collecting more intensive land use information (e.g. determining sediment loads during peak runoff) and adding more descriptive habitat data (e.g. better quantifying in-stream habitat and channel characteristics) to the composite land use score might help to further explain variation in IBI scores.

When pooled with stream land cover data from previous studies (Goetz 2007, Rieck 2009), the Oil Creek tributaries expand the upper end of the range of natural watershed land cover from 78.3% to 95%. A comparison of land cover (NLCD 2001 data) shows that the overall Oil Creek watershed is 74.8% natural cover and 20.6% agriculture, whereas the French Creek watershed is 56.6% natural cover and 34.8% agriculture. According to Whitney and DeCant (2003), streams in the French Creek watershed are not experiencing the stresses associated with urbanization or converting land for agricultural use. Streams in the Oil Creek watershed might not be experiencing these stressors either. However, disturbances unique to high-forested areas, such as logging road construction, could be adversely impacted water quality (Hassinger 2002).

At some of the most forested Oil Creek sites, gas well drilling and road construction may be contributing to a higher than expected sediment load, given the land cover, and therefore may confound the effects of land use on IBI scores. The narrow scale of land cover investigated in this study (40% to 95% natural watershed cover) might be influencing the limited ability of land use to explain variation in IBI scores. The influence of watershed disturbance (e.g. agriculture) may have a threshold of effect on biotic integrity (Wang et al. 1997). Future investigations of the roles of varying land use disturbance regimes should explore the biotic integrity of streams with less than 40% natural watershed cover.

Improving the Precision of IBIs for Use in Northwestern Pennsylvania

This study confirmed that the FC Index 4 IBI can be reliably used to classify biotic integrity outside of the French Creek watershed in northwestern Pennsylvania. However, the precision of the IBI still needs to be fine tuned against a broader stressor gradient, and IBI metrics need to be re-analyzed to adjust to this broader stressor gradient. Given that some of the most responsive IBI metrics do not necessarily depict a site's overall health, composing an IBI with only responsive metrics may lead to an underestimation of biological integrity (*type I error*) (Vondracek et al. 2009). Because local IBIs have been modified to respond to forest cover, and because the Oil Creek watershed is more heavily forested than the French Creek watershed, Oil Creek IBI scores may have been higher than French Creek IBI scores due to this relationship. Metrics such as the number of lampreys and the number of trout seem to respond better to forest cover, and because these two scores can potentially influence 30% of the total Index 4 IBI score, these metrics may inflate IBI scores in watersheds where other

factors besides forest cover influence biotic integrity. Metrics constructed to respond to known stressors (e.g. forest cover) may or may not adequately detect other stressors, whose effects on biotic integrity are not as thoroughly understood (Vondracek et al. 2009). A potential way to reduce scoring biases of metrics that may have unequal responses to a variety of land use stressors is to use a continuous metric scoring system. The original IBI developed by Karr (1981) and many subsequent IBIs use discrete scoring systems; however, the adoption of a continuous scoring system has been shown to reduce IBI scoring biases (see Dolph et al. 2010). More intensive sampling would also benefit the development of a more precise IBI.

Developing biotic integrity standards from a robust pool of reference sites that accounts for natural regional variation makes evaluations of biotic integrity more defensible (Vondracek et al. 2009). Wang et al. (1997) showed a very strong correlation of land use and IBI scores ($R=0.69$) in an agricultural watershed, while McCormick (2001) did not find a strong correlation between land use and IBI (Mid-Atlantic streams) ($R=0.23-0.29$). Yoder and Rankin (1995) found that 35-40 reference sites per ecoregion was adequate to characterize biotic integrity in Midwestern agricultural streams, although fewer reference sites were adequate in more uniformly disturbed ecoregions. Lyons (2006) conducted electrofishing and habitat surveys four times on 102 small watersheds ($1.7-41.5 \text{ km}^2$) to develop an IBI for Wisconsin headwater streams. Repeated sampling of sites over a number of years helps to better characterize species richness, since the short-term monitoring of fish communities has a severely limited ability to differentiate between the influence of natural and anthropogenic disturbances (Schlosser 1990; Vondracek 2009). The number of fish

and the number of rare taxa collected at a sample site can account for sampling variability in IBI scores, and sampling at various times of the year can account for seasonal differences in fish distribution (Dolph et al. 2010; Thornburgh and Gido 2010). Increasing the number of streams sampled, the number of reaches sampled within each watershed, repeated sampling at sites, and sampling at different times of the year can benefit future IBI studies in northwestern Pennsylvania.

Conclusion

Watershed managers need reliable quantitative measures of biological integrity to guide management decisions. This study confirmed that the FC Index 4 IBI reliably characterizes biotic integrity outside of the French Creek drainage in northwestern Pennsylvania. However, the utility of the index will need to be refined as streams with more intense human impacts (e.g. <40% natural watershed cover) are studied. A composite land use score predicted biotic integrity better than watershed land cover, riparian land cover, or riparian habitat evaluations alone. Expanding the composite land use score to include peak sediment loading and a more rigorous stream channel evaluation could help to strengthen local biotic integrity assessments and account for more variation in IBI scores. Few studies have investigated the legacy impacts of historical land use on present-day biotic integrity. This study attempted to show how historical land use might confound biological integrity expectations based on current land cover and habitat. Future studies should recognize that the legacy effects of historical land use, though difficult to quantify, can underlie differences in biotic integrity between watersheds.

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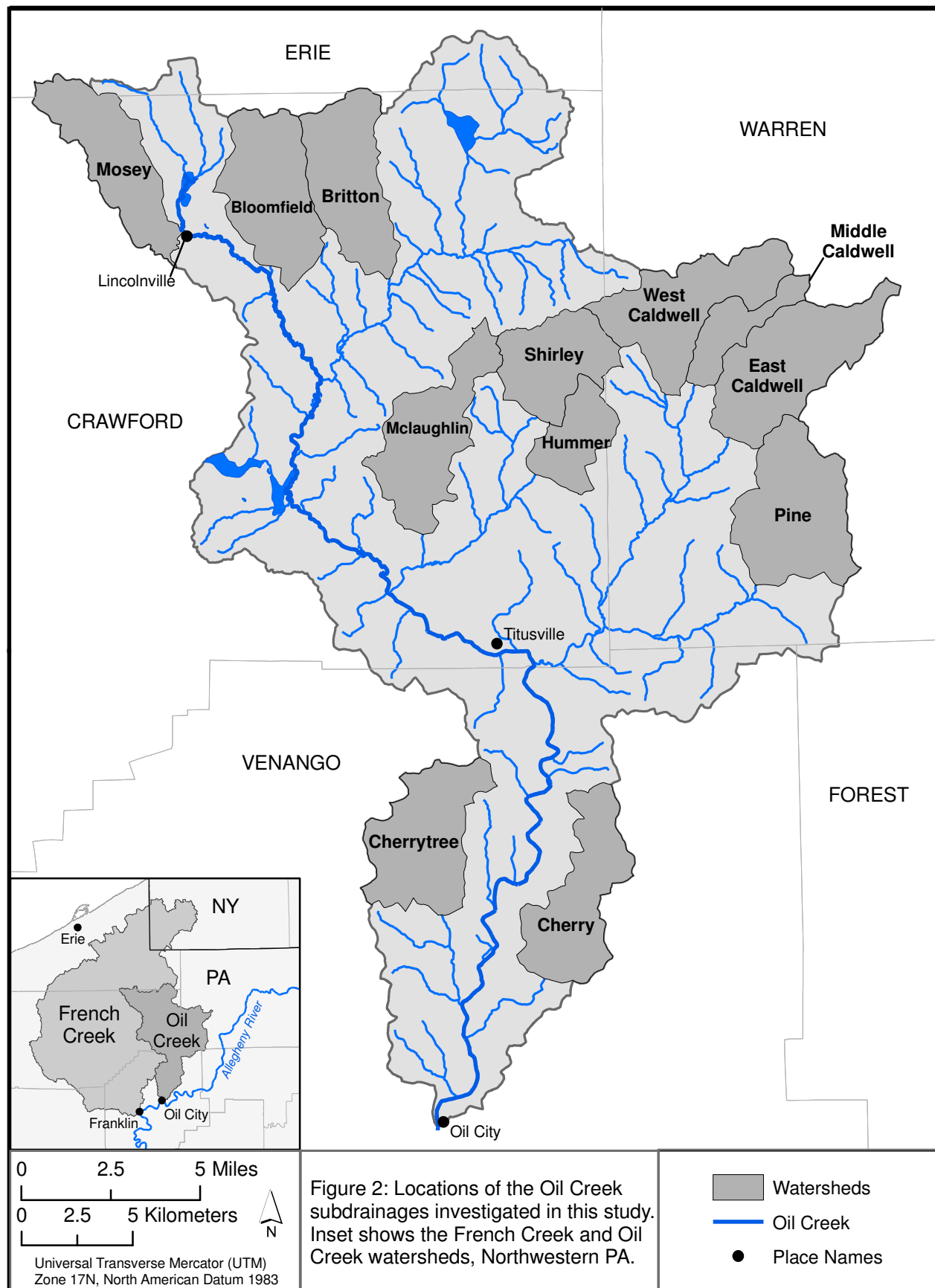
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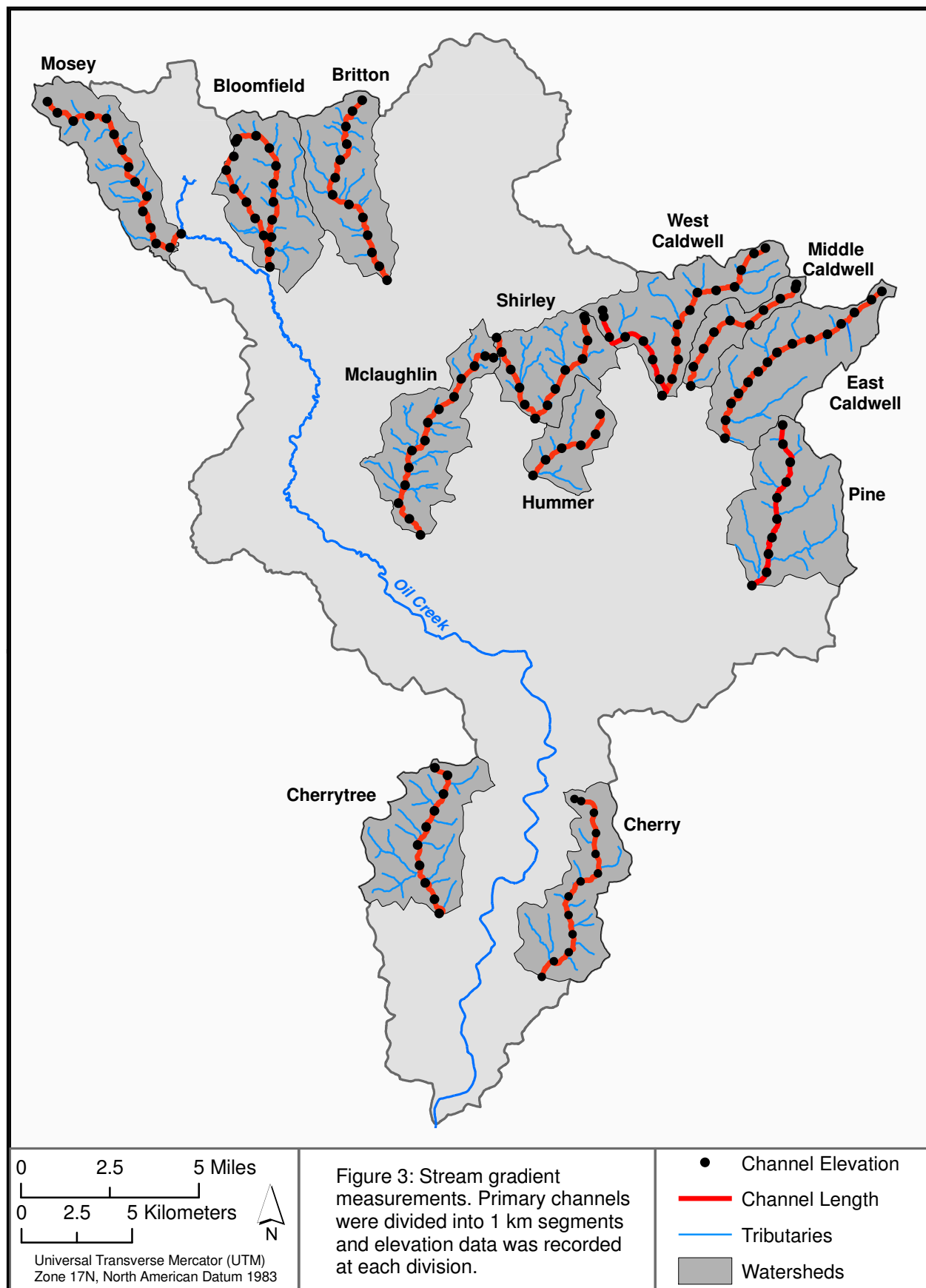
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Appendix A. Indicators of stream size and longitudinal position of sampling sites.

Subdrainage	Stream Name	Drainage Area (km ²)	Stream Order	Discharge (m ³ sec ⁻¹)	Width (m)	Distance to Mainstem (km)	NAD 1983 Latitude & Longitude (DD)
Oil Mainstem	Cherry Tree	25.67	4	0.24	6.25	18.2	-79.705597 41.517329
	Cherry	23.91	3	0.20	6.47	53.9	-79.650372 41.491181
Pine Creek	Pine-Campbell	29.64	3	0.21	7.17	14.5	-79.532095 41.648278
	East Caldwell	29.53	3	0.15	7.18	18.6	-79.548515 41.710446
	Middle Caldwell	9.35	2	0.07	6	20.1	-79.565886 41.731317
	West Caldwell	26.35	3	0.23	7.22	20	-79.583057 41.724848
Thompson Run	Hummer	11.59	3	0.12	8.17	10.8	-79.649644 41.720183
	Shirley	18.11	3	0.28	7.13	14	-79.650988 41.694686
	McLaughlin	24.36	3	0.23	7.49	39.2	-79.714697 41.669904
West Oil Creek	Mosey	21.99	3	0.14	6.15	21.7	-79.839618 41.796367
	Bloomfield	25.38	4	0.12	7.74	23	-79.792789 41.781353
	Britton	23.61	2	0.09	5.59	89	-79.728804 41.776364

Appendix B. Summary of watershed and riparian land use (reclassified 2001 NLCD data).

Stream Name	Watershed Coverage				Riparian Coverage			
	% Natural Cover	% Agriculture	% Urban/Suburban	% Open Water	% Natural Cover	% Agriculture Riparian	% Urban/Suburban	% Open Water Riparian
Cherry Tree	74.50	17.10	8.50	0.00	85.60	4.70	9.70	0.00
Cherry	93.90	2.70	3.40	0.00	94.60	2.10	3.30	0.00
Pine-Campbell	95.00	3.80	1.20	0.00	99.50	0.20	0.30	0.00
East Caldwell	89.40	6.80	3.90	0.00	94.70	2.60	2.70	0.00
Middle Caldwell	80.40	16.70	2.90	0.00	86.00	11.20	2.80	0.00
West Caldwell	82.60	15.50	1.90	0.00	89.90	8.40	1.70	0.00
Hummer	58.00	39.30	2.80	0.00	79.70	18.40	1.90	0.00
Shirley	62.40	34.90	2.60	0.10	70.40	27.10	1.80	0.70
McLaughlin	70.30	26.90	2.90	0.00	85.70	12.20	2.10	0.00
Mosey	65.50	31.80	2.70	0.00	82.60	15.20	2.20	0.00
Bloomfield	59.80	36.60	3.60	0.00	69.10	28.30	2.60	0.00
Britton	45.20	51.40	3.40	0.00	48.90	48.10	2.90	0.00

Appendix C. Slope data.

Stream Name	Total (m/km)	1 km Upstream (m/km)	Standard Error of the Mean
Cherry Tree	12.84	0.007	2.77
Cherry	17.91	0.007	1.37
Pine-Campbell	11.59	0.007	2.45
East Caldwell	8.22	0.009	2.73
Middle Caldwell	12.03	0.021	5.04
West Caldwell	9.99	0.011	2.11
Hummer	12.58	0.011	3.75
Shirley	11.42	0.004	2.78
Mclaughin	13.52	0.01	1.9
Mosey	10.67	0.004	1.16
Bloomfield	8.29	0.002	2.79
Britton	4.29	0.003	2.37

Appendix D. Comparison of Oil Creek and French Creek sites. Streams matched based on basin area and watershed land cover.

Watershed	Subdrainage	Stream Name	Oil Creek - French Creek Match Code	Basin Area (km ²)	% Natural Watershed Cover	Total Slope (m/km)
Oil Creek	Oil Mainstem	Cherry Tree	A	25.67	74.50	12.84
		Cherry	B	23.91	93.90	17.91
	Pine Creek	Pine-Campbell	C	29.64	95.00	11.59
		East Caldwell	D	29.53	89.40	8.22
		Middle Caldwell	E	9.35	80.40	12.03
		West Caldwell	F	26.35	82.60	9.99
	Thompson Run	Hummer	G	11.59	58.00	12.58
		Shirley	H	18.11	62.40	11.42
		Mclaughin	I	24.36	70.30	13.52
	West Oil Creek	Bloomfield	J	25.38	59.80	8.29
		Britton	K	23.61	45.20	4.29
			Mean:	22.5	73.77	11.15

Watershed	Subdrainage	Stream Name	Oil Creek - French Creek Match Code	Basin Area (km ²)	% Natural Watershed Cover	Total Slope (m/km)
French Creek	Sugar Creek	Wolf	A	14.30	78.30	18.56
	Muddy Creek	Mackey	B	14.70	74.80	12.47
	Sugar Creek	Sugar W. Branch	C	20.90	76.50	14.34
	Sugar Creek	Beatty	D	21.40	74.10	6.1
	Sugar Creek	Yellow Jacket	E	6.40	69.00	17.91
	Woodcock Creek	Woodcock	F	18.80	75.30	12.84
	Little Sugar Creek	Little Sugar	G	12.50	59.70	10.12
	Muddy Creek	Muddy	H	19.10	62.40	4.2
	Conneaut Outlet	McMichael Run	I	20.10	63.30	6.9
	French Creek	Gravel	J	19.70	60.90	17.86
	French Creek	Kelly	K	20.40	46.70	11.51
Mean:				17.1182	67.36	12.07

Appendix E. Water chemistry paramaters evaluated in this study.

Stream Name	Temp (C)	TDS (mg/L)	pH	Winkler DO (mg/L)	Hach Nitrate (mg/L)	Alkalinity (mg/L)	Turbidity (JTU)	Total Phosphorus (ug/L)
Cherry Tree	3.00	70.00	8.00	14.60	0.26	17.00	0.00	10.97
Cherry	4.00	80.00	7.80	13.80	0.14	20.00	0.00	16.58
Pine-Campbell	4.00	50.00	7.80	14.40	0.14	15.00	0.00	10.40
East Caldwell	4.00	30.00	6.90	13.00	0.19	12.00	0.00	21.56
Middle Caldwell	3.00	30.00	7.40	13.80	0.49	10.00	0.00	12.58
West Caldwell	4.00	30.00	6.40	14.00	0.42	14.00	0.00	9.21
Hummer	4.00	60.00	7.90	13.60	0.85	22.00	0.00	15.18
Shirley	5.00	70.00	7.80	14.20	0.38	42.00	0.00	19.39
Mclaughin	4.00	50.00	8.00	14.00	0.69	48.00	0.00	12.86
Mosey	4.00	80.00	7.80	12.40	0.28	64.00	0.00	14.40
Bloomfield	4.00	80.00	8.00	13.40	0.16	44.00	0.00	8.87
Britton	3.00	80.00	8.00	13.40	0.81	49.00	0.00	20.99

Appendix F. Riparian and channel habitat evaluation scores and IBI scores for Oil Creek sites.

Stream Name	RCE			WQHNA			IBI		
	Metrics 1-4 (Riparian)	Metrics 5-12 (Channel)	Total Score	Metrics 1-8 (Channel)	Metrics 9-12 (Riparian)	Total Score	Roy 2	Goetz	FC Index4
Cherry Tree	110.00	95.00	205.00	133.00	75.00	208.00	18	11	21
Cherry	113.00	140.00	253.00	130.00	79.00	209.00	18	13	21
Pine-Campbell	112.00	134.00	246.00	138.00	77.00	215.00	22	18	26
East Caldwell	113.00	131.00	244.00	125.00	76.00	201.00	22	17	23
Middle Caldwell	113.00	136.00	249.00	135.00	80.00	215.00	20	18	24
West Caldwell	114.00	145.00	259.00	144.00	77.00	221.00	22	18	26
Hummer	113.00	139.00	252.00	135.00	77.00	212.00	24	18	24
Shirley	113.00	119.00	232.00	123.00	76.00	199.00	20	18	26
Mclaughin	109.00	143.00	252.00	148.00	73.00	221.00	20	11	21
Mosey	85.00	133.00	218.00	132.00	70.00	202.00	16	11	14
Bloomfield	105.00	115.00	220.00	103.00	65.00	168.00	20	17	20
Britton	65.00	95.00	160.00	111.00	48.00	159.00	18	10	15

Appendix G. Comparison of fish species abundance at French (n=39) and Oil Creek sites (n=11).

