INVESTIGATING EROSION AND ECOLOGICAL IMPACTS TO AN URBAN COLDWATER STREAM USING MULTIPLE TECHNIQUES

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DEDICATION

For my family and friends.

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PREFACE

Over the past few months I have realized that the Indian Mill Creek Watershed is a special place. The mix of beautiful rolling pasture, flowering fruit trees, and attractive parks and natural areas are unique to this part of Kent County. The fragility of the creek makes it extra special. Here exists a vulnerable ecosystem with universal interest in restoration, from the farmers who want to see spawning salmon return to their drains, to the school groups who study stream critters and water quality.

We are at a significant point in Indian Mill Creek's history: an era of cooperation between the Cities, township, schools, businesses, and residents to restore the creek. I am confident the creek has a bright future, but it needs your help! The Friends of Indian Mill Creek was formed in 2016. Check out their webpage and participate in events at the link below:

http://www.lgrow.org/watershed/indianmillcreek/about

ABSTRACT

Sediment pollution is a major cause of stream degradation. Our objectives were three-fold. First, we assessed the impacts of environmental stressors on the structure of fish and macroinvertebrate communities across a gradient of agricultural to urban land cover in a coldwater tributary of the Grand River, Michigan, USA called Indian Mill Creek. We found that instream woody debris, streambed substrate, riffle and pool habitat variability, and riparian conditions affected aquatic macroinvertebrates along an agricultural to urban land cover gradient. We also found that variation in fish community assemblage was driven largely by stream flow and temperature regimes and could be impacted by episodic pollution events that have occurred within the lower, urbanized part of the watershed. Second, we identified critical areas for sediment management in the Indian Mill Creek watershed using the Enhanced Generalized Watershed Loading Functions (GWLF-E) model and MapShed plugin for MapWindow GIS to simulate the water budget, field erosion, and streambank erosion in 20 subbasins from 1997-2015. We found that southwest subbasins had the highest rates of runoff because of impervious surfaces and urbanization. Field erosion was greatest in the lower watershed with steep slopes and erodible soils. The proportion of sediment load from streambanks and the lateral bank erosion rate increased in a downstream direction. Third, we evaluated three techniques for quantifying sediment pollution from streambank erosion: erosion pins, total station surveyor, and terrestrial laser scanning, and assessed the spatial distribution of erosion rates in the watershed in relation to the GWLF-E modeling. We found that erosion pins and total station surveying are preferable for vegetated banks, while laser scanning can collect high quality data for barren banks. We also found that streambank erosion rates vary spatially in the watershed, with the lower reaches experiencing net deposition of sediment on the banks, while the upper reaches

experience net erosion that contributes to sediment loading in the creek. We estimate that streambank erosion contributes 28.5% of sediment to the creek's total sediment load. Findings of these studies help watershed managers prioritize restoration programs to reduce sediment loadings and have broad applications for streams degraded by sediment.

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CHAPTER I: INTRODUCTION

Changes in the Earth's land use and climate have impacted aquatic ecosystems. The land draining into streams is being converted to agriculture and urban uses, while precipitation patterns are more intense due to shifting climate (Bartolai et al. 2015). These changes create unstable landscapes that release sediment into streams from streambank erosion and runoff. This sediment is a major cause of water quality impairment worldwide (Narasimhan et al. 2017). Throughout the United States, sediment pollution is the second highest cause of water quality degradation, impairing the quality and habitat of 225,000 km of streams (USEPA 2016). In Michigan alone, sediment pollution has an enormous effect on aquatic life and impairs the quality and habitat of approximately 2,000 miles of streams (USEPA 2016). In both the United States and Michigan, sediment is the greatest pollutant by volume to enter streams (NOAA 1978; Bernard et al. 1996). Sediment is also notorious for carrying attached phosphorus pollution into surface waters (Miller et al. 2014). Sediment pollution can be carried into a stream by runoff and by eroding streambanks, and can smother habitats that would otherwise be used by aquatic organisms (Junk et al. 1989; Allan 2004). It is thus associated with a decrease in abundance and density of aquatic macroinvertebrates and fish (Pennak and Van Gerpen 1947; Chin et al. 2016).

The composition of macroinvertebrates and fish in a stream is reflective of the quality and conditions of a study site (Hilsenhoff 1987; Karr 1991). Our objective in Chapter II was to assess the impacts of environmental stressors on the structure of fish and macroinvertebrate communities across a gradient of agricultural to urban land cover in a coldwater tributary of the Grand River called Indian Mill Creek. We hypothesized that environmental stressors affect the structure and function of biological communities and that these impacts occur along an agricultural to urban gradient. This information will be critical for restoring ecological function of the creek; based on its location, it is a high priority tributary for coldwater fisheries restoration in Michigan's Lower Grand River watershed (LGROW 2011). Increased urbanization and agricultural land conversion is prevalent worldwide; therefore this study also has broad applicability to other watersheds with similar land cover gradients.

Quantifying sediment pollution on a catchment scale is important for resource management but also problematic. It requires an understanding of the pathways sediment enters the water and the complex factors that affect its movement; data at the catchment scale may not be available (Dietrich et al. 1999; Kiesel et al. 2009). To address these problems, sediment transport to streams can be estimated using models that calculate pollutant export coefficients, loading functions, and chemical simulation (Haith and Shoemaker 1987). The purpose of Chapter III is to identify critical areas for sediment pollution management in the Indian Mill Creek watershed of Michigan, USA using a nonpoint source pollution model. To accomplish this, we modeled runoff and sediment loading from 20 subbasins and their matching stream sections from 1997-2015. We aimed to determine if agricultural areas in the upper watershed contribute the most sediment from field erosion and if urban areas in the lower watershed have the highest streambank erosion rates because of increased runoff from impervious surfaces. This information will be used by water quality mangers and local units of government to prioritize restoration programs to reduce sediment loadings and improve stream habitat.

Eroding banks are a natural occurrence in streams, affected by factors such as climate, geology, and topography (Rosgen 1994; Montgomery 1999). However, changes to the landscape surrounding a stream, such as agricultural and urban land use, can increase bank erosion because of powerful flows from reduced infiltration of precipitation and increased runoff (Paul and Meyer 2001; Allan 2004). Bank erosion can also be affected by factors such as cattle access and vegetated riparian corridors (Zaimes et al. 2005). One difficulty with managing sediment pollution is that it is hard to quantify sediment pollution from streambank erosion, which can be the main contributor of sediment pollution in some watersheds (Fox et al. 2016). In Chapter IV, we aimed to evaluate three techniques for quantifying sediment pollution from streambank erosion: erosion pins, total station surveyor, and laser scanning. We hypothesized that these techniques would provide different estimates of streambank erosion due to their ability to resolve spatial change, and that each technique would either be advantageous or disadvantageous to use under different conditions. This better allows watershed managers to address streambank erosion as a source of sediment pollution to be controlled in their watersheds. We also found that streambank erosion rates vary spatially in the watershed, with the lower reaches experiencing net deposition of sediment on the banks, and the middle watershed and agricultural headwaters experiencing net erosion that contributes to sediment loading in the creek.

The scope of this research is the Indian Mill Creek watershed of Michigan, USA (HUC 040500060504). It is a tributary to the Grand River and is 18.5 km long with a 44 km^2 watershed. The watershed is on the Michigan 303(d) list of impaired water bodies, with sediment loading and deposition identified as the cause of impairment (Sigdel 2017). The watershed land cover is predominately urban (43%) and agricultural (39%), with commercial and residential development in the lower watershed, natural and urban lands in the middle watershed, and farmland and orchards in the upper watershed (LGROW 2011). Indian Mill Creek is designated as a coldwater trout stream by the State of Michigan; however, it is currently not supporting its coldwater fishery designated use per Michigan Department of Environmental Quality (MDEQ) standards (Goodwin et al. 2016). Following these chapters are a synthesis, extended review of literature and methods, bibliography, and appendices.

CHAPTER II: IMPACTS OF AN AGRICULTURAL/URBAN LAND COVER GRADIENT IN A COLDWATER STREAM

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Core Ideas

1. Stream habitat, fish, and macroinvertebrates relate to environmental stressors.
2. Environmental stressors occur along an agricultural to urban land cover gradient.
3. This gradient affects aquatic communities in a coldwater stream.
4. Woody debris are associated with high macroinvertebrate community integrity.
5. Fine sediment is associated with degraded macroinvertebrate communities.

Abbreviations

AWRI, Annis Water Resources Institute; BC, Brandywine Creek; CA, Correspondence Analysis; CCA, Canonical Correspondence Analysis; EPT, Ephemeroptera, Plecoptera, and Trichoptera; FBI, Family Biotic Index; GLEAS, Great Lakes Environmental Assessment Section; IMC, Indian Mill Creek; MDEQ, Michigan Department of Environmental Quality; MRPP, Multi-response Permutation Procedure; NMDS, Nonmetric Multidimensional Scaling; P 51, Procedure 51; PPM, Parts per Million; SSC, Suspended Sediment Concentration; U/S, Upstream; WAD, Walker Avenue Ditch.

2.1 ABSTRACT

Throughout the United States and the world, urban areas are often built along large rivers and surrounded by agricultural land cover. Examples are the numerous metropolitan areas along the Grand River in Michigan, USA. Tributaries that flow through these areas often have agricultural headwaters and an urbanized, lower watershed. This land cover gradient can have significant impacts on the chemical, physical, and biological attributes of lotic ecosystems. Our objective was to assess the impacts of environmental stressors on the structure of fish and

macroinvertebrate communities across a gradient of agricultural to urban land cover in a coldwater tributary of the Grand River called Indian Mill Creek. Instream woody debris were lacking and functioned as the strongest driver of EPT (Ephemeroptera, Plecoptera, and Trichoptera) macroinvertebrate abundance and richness, especially in agricultural headwaters. Fine streambed substrate had the strongest relationship with degraded macroinvertebrate communities, with a high abundance of Diptera and surface air breathers, and was most dominant in agricultural headwaters. Habitat variability was often insufficient for trout because of a paucity of pool/riffle areas, and was lowest in the agricultural headwaters. Diminished riparian conditions were prevalent, especially in agricultural areas, and correlated with impacted macroinvertebrate traits. Variation in fish community assemblage was driven largely by stream flow and temperature regimes and could be impacted by episodic pollution events that have occurred within the lower, urbanized watershed. This information will be critical for restoring ecological function of the creek and also has broad applicability to other watersheds with similar land cover gradients.

2.2 INTRODUCTION

Agricultural and urban land cover can have significant impacts on the chemical, physical, and biological attributes of lotic ecosystems. These impacts alter the structure and functions of biological communities (Allan, 2004; Walsh et al., 2005b; Merritt et al., 2006). The US Environmental Protection Agency (USEPA, 2000) listed the top three causes of stream impairments as agriculture, hydromodification in the form of channelization and dams, and urbanization/storm sewers. Impacts to streams from changing land cover can occur along an agricultural to urban gradient (O’Brien and Wehr, 2010).

The distribution and abundance of stream biota often respond to the interaction of multiple environmental stressors (Raleigh et al., 1984; Poff, 1997). A greater diversity of macroinvertebrates, especially those sensitive to stressors, indicates a healthier stream, whereas a greater abundance of pollution tolerant organisms indicates poor stream quality (Hilsenhoff, 1987). The traits and feeding guilds of stream organisms can also reflect environmental stressors (Usseglio-Polatera et al., 2000; Merritt et al., 2006). Relationships between these stressors and the health of biological communities can be difficult to characterize quantitatively because they are often complex, with numerous stressors impacting multiple facets of biological health (Johnson et al., 2007; McNair, 2009; Table 2.1).

Throughout the United States and world, there are many instances where an urban area was built along a large river surrounded by agricultural land cover. Many large cities are located along navigable waterbodies because of historic and/or present day uses of the rivers such as commercial transport, hydroelectric production, and waste disposal. Oftentimes, the soils in large river valleys are very fertile, which also encourages agricultural land use in river floodplains (Gallup et al., 1999). The Grand River, in the southern Lower Peninsula of Michigan, USA, flows through several urban areas including Jackson, Lansing, Portland, and Grand Rapids. Much of the land cover surrounding the metropolitan areas of southern Michigan is agricultural. As a result of this land cover pattern, numerous small, tributary stream watersheds have become characterized by predominantly agricultural land cover in the headwaters and urban land cover in the lower reaches. These waterbodies are surrounded by dense populations and provide numerous recreational opportunities, thereby generating interest in their restoration (Moerke and Lamberti, 2004; Alexander and Allan, 2007; Schwartz and Herricks, 2007). However, agricultural and urban land cover can have multiple impacts on stream systems, and understanding their interactions is necessary for successful restoration efforts (Cooper et al., 2009).

Table 2.1. Common environmental stressors in agricultural and urban streams.

Stressor	Impacts	Sources
Reduced habitat variability	Decreased macroinvertebrate abundance Decreased macroinvertebrate diversity Reduced habitat suitability for fish	Raleigh, 1982; Raleigh et al., 1984; Hogg and Norris, 1991; Hawkins et al., 1993; Paul and Meyer, 2001
Riparian vegetation loss	Increased water temperature Increased bank erosion Increased pollutant loading Decreased terrestrial energy subsidies Decreased organic matter input	Delong and Brusven, 1991; Sweeney, 1993; Maloney and Lamberti, 1995; Weller et al., 1998; Magana, 2001; Nakano and Murakami, 2001; Allan, 2004; Anbumozhi et al., 2005; Palmer, 2008
Increased sediment load	Decreased biological production Fewer macroinvertebrate grazers Reduction in habitat variability Fish populations decline Displaced macroinvertebrates	Pennak and Van Gerpen, 1947; Junk et al., 1989; Allan et al., 1997; Paul and Meyer, 2001; Allan, 2004; Chiu et al., 2016
Fine substrate / sedimentation	Reduced habitat variability Reduced invertebrate diversity Reduced habitat suitability for fish Impeded fish reproduction	Raleigh et al., 1984; Alexander and Hansen, 1986; Culp et al., 1986; Paul and Meyer, 2001
Altered water velocity	Changed macroinvertebrate composition Impeded fish feeding and resting Impeded fish reproduction	Raleigh et al., 1984; Paul and Meyer, 2001; Schoen et al., 2013
Woody debris reduction	Decreased habitat variability Decreased organic matter retention Decreased macroinvertebrate habitat Decreased macroinvertebrate diversity	Ehrman and Lamberti, 1992; Brookshire and Dwire, 2003; Johnson et al., 2003; Cordova et al., 2007; Nakamura et al., 2017
Episodic pollution events	Toxic sediments Fish mortality Fish emigration	Seager and Maltby, 1989; Van Sickle et al., 1996)
Increased stream temperature	Altered temperature regime Altered fluctuation regime Altered fish composition Reduced habitat suitability for fish Altered growth rates of organisms	Crisp and Howson, 1982; Raleigh et al., 1984; Sinokrot and Stefan, 1993; Wehrly et al., 1999

Indian Mill Creek is a coldwater tributary to the Grand River that has predominately agricultural headwaters and flows through the metropolitan area of Grand Rapids, Michigan. This study used multivariate statistics to understand how complex, environmental stressors affect the structure and function of biological communities in the creek. Our objective was to analyze the impacts of environmental stressors (habitat variability, riparian vegetation condition, sediment loading, substrate composition, stream temperature, water velocity, episodic pollution events, and instream woody debris abundance) on the structure of fish and macroinvertebrate communities across a gradient of agricultural and urban land cover. We hypothesized that environmental stressors affect the structure and function of biological communities and that these impacts occur along an agricultural to urban gradient. This information will be critical for restoring ecological function of the creek; based on its location, it is a high priority tributary for coldwater fisheries restoration in Michigan's Lower Grand River watershed (LGROW, 2011). Increased urbanization and agricultural land conversion is prevalent worldwide; therefore this study also has broad applicability to other watersheds with similar land cover gradients.

2.3 STUDY AREA

Indian Mill Creek (HUC 040500060504) is a third-order tributary to the Grand River in Kent County, Michigan, USA. It is 18.5 km long with a 44 km^2 watershed. The watershed is predominately urban (43%) and agricultural (39%), with commercial and residential development in the lower watershed, natural and urban lands in the middle watershed, and farmland and orchards in the upper watershed (Figure 2.1; Figure 2.2; LGROW, 2011). Impervious surfaces cover 12% of the entire watershed, and up to 25% of some lower catchments (AWRI, unpublished data; Sigdel, 2017). Indian Mill Creek is designated as a coldwater trout stream by the State of Michigan; however, it is currently listed as impaired by the Michigan Department of Environmental Quality (MDEQ) due to degraded fish and benthic invertebrate communities (Goodwin et al., 2016).

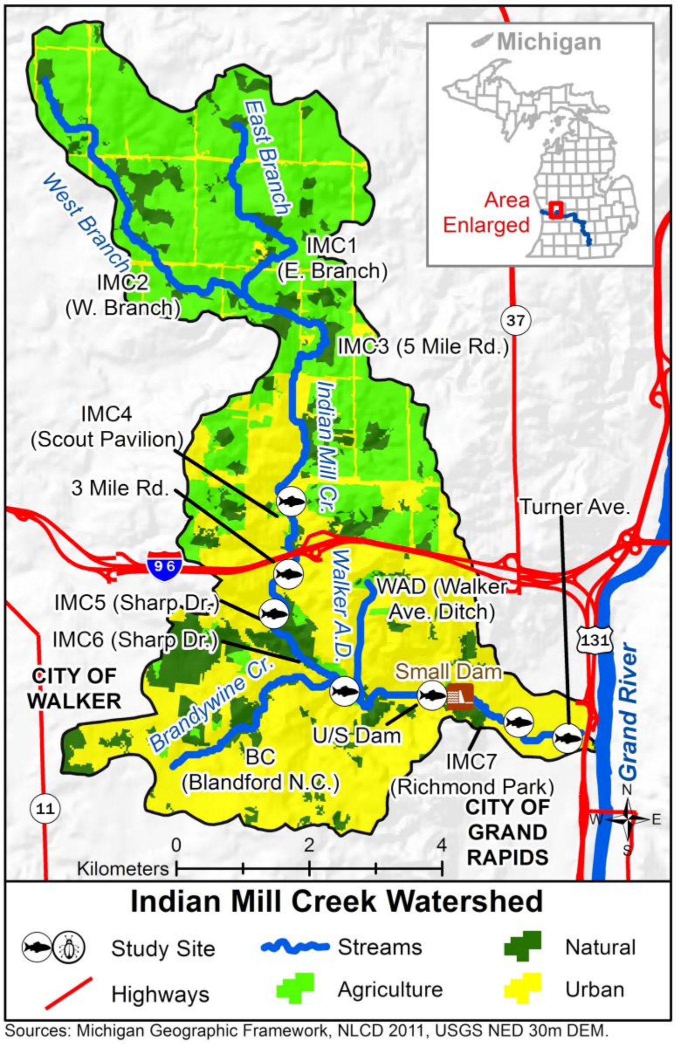

Figure 2.1. Monitoring sites in the Indian Mill Creek watershed (2017).

Figure 2.2. Photographs of the watershed's agricultural to urban land cover gradient in Indian Mill Creek (2017).

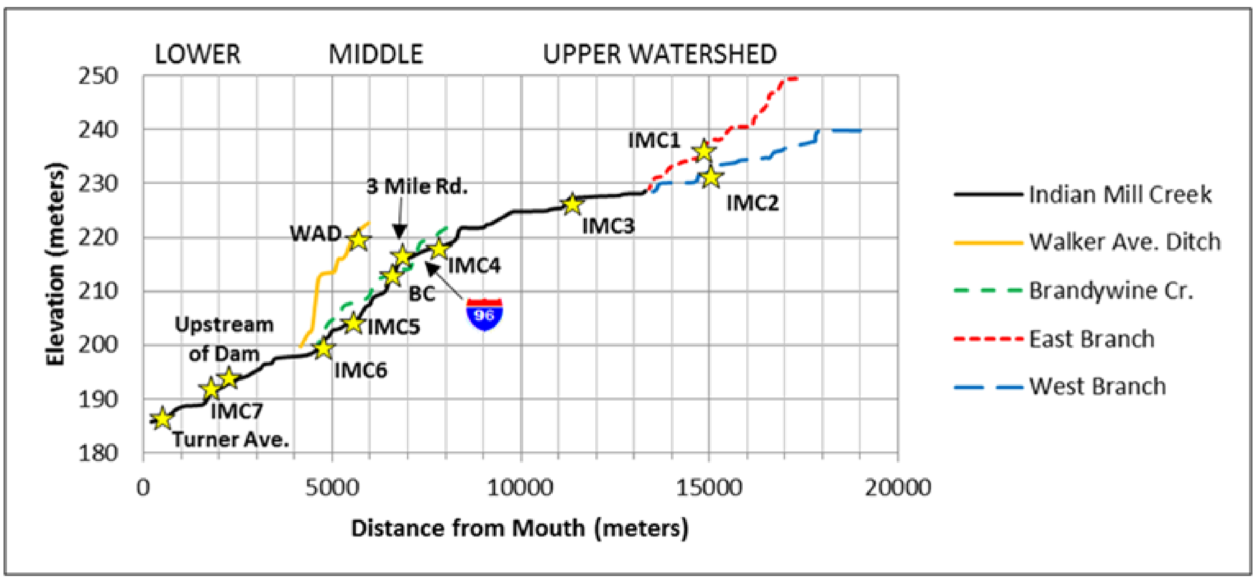

Figure 2.3. Indian Mill Creek and its tributaries following a topographical gradient (elevation data from Gesch et al., 2002). Stars indicate sampling sites.

Geologic features of the Indian Mill Creek watershed were formed by retreating glaciers that deposited hills of medium-textured till in the upper watershed, which contributed cobble and rock to the creek (Farrand and Bell, 1982). Glacial meltwater carved the larger Grand River Valley (Larson and Schaetzl, 2001). Indian Mill Creek descends this valley for five kilometers starting downstream from the present location of Interstate 96 and descending 24 meters in elevation (Fig. 3; Gesch et al., 2002). Overall, the creek descends 65 meters in elevation from its headwaters to its mouth. The lower watershed gently slopes in an outwash of sand and gravel with postglacial alluvium (Farrand and Bell, 1982). One low-head dam (Richmond Dam) is present just upstream of the Indian Mill Creek (IMC) site IMC7 (Figure 2.1). The upper watershed has loamy hydrologic group C and C/D soils with low infiltration in uplands, but sandy A/D and B/D soils along the West Branch and main channel Indian Mill Creek (Soil Survey Staff, 2009). The middle watershed has loamy C and C/D soils in uplands but sandy A and B soils with high infiltration adjacent to the creek and Walker Avenue Ditch (Figure 2.1). The lower watershed has sandy alluvial group A and B soils.

Nine sites in the Indian Mill Creek watershed were monitored for stressors and macroinvertebrates (Figure 2.1). IMC1, IMC2, and IMC3 are in agricultural areas of the upper watershed. IMC4, IMC5, and IMC6 are in the urbanized middle watershed but with wide forested riparian areas. IMC7 is in the urbanized, lower watershed, while the Walker Avenue Ditch (WAD) and Brandywine Creek (BC) sites are on tributaries. Seven sites were monitored for fish, including IMC4, IMC5, IMC6, IMC7, and three additional sites: Turner Avenue, Richmond Dam impoundment, and 3 Mile Road.

Whole sediment toxicity tests of samples from the Richmond Dam impoundment, IMC7, and Turner Street sites, collected in 2017, resulted in reduced 10-day growth of *Chironomus* *dilutus* and reduced 10- and 28-day growth and survival of *Hyalella azteca*, which may be a result of elevated sediment PAH (Polycyclic Aromatic Hydrocarbon) concentrations (MDEQ Surface Water Assessment Section, 2017 unpublished data). Indian Mill Creek has also experienced numerous episodic pollution events in the lower reach. The most noteworthy event occurred in 1998 when ammonia refrigerant from a meat-packing facility spilled into the creek, resulting in a complete fish kill in the ~3 km reach from the discharge point to its confluence with the Grand River (Hanshue, 1998). Within the last decade, the MDEQ has cited 16 facilities either for spill incidents or for directly discharging contaminated wastewater into Indian Mill Creek via illicit drain connections. Illicitly discharged and spilled materials have included oil, sodium hydroxide, and metal plating wastewater effluent (MDEQ MiWaters Explorer, https://miwaters.deq.state.mi.us/nsite/). The latest illicit discharge incident in Indian Mill Creek occurred in the fall of 2017 between sites IMC7 and Turner Street. During that event, industrial, foam adhesive wastewater was being illicitly discharged into Indian Mill Creek at such a volume that the stream was visibly discolored. The source of the wastewater was located and upon interviewing facility operators, MDEQ learned that the company had been intermittently discharging the foam adhesive waste into Indian Mill Creek for about 20 years. The MDEQ has also received several complaints in the last decade about pollution incidents in Indian Mill Creek; however, because of the irregular, ephemeral nature of illicit discharges, the sources were not located. Thus, there are likely other existing, illicit drain connections within industrial facilities that ultimately discharge to Indian Mill Creek. No industrial facility violations have occurred upstream of site IMC3 in the last decade, most likely because the predominant land use above IMC3 is agriculture.

2.4 METHODS

Environmental Stressor Inventory

Nine representative sites were chosen for stressor inventories to reflect the watershed's spatial variation (Figure 2.1). These sites overlapped with all macroinvertebrate and four fish assessment sites. All locations within the watershed were perennially flowing except for the Walker Avenue Ditch site. All technicians were trained together for each inventory and practiced collecting data together until we were confident that data collection was standardized among technicians; this would take a few hours for each inventory. Habitat components were surveyed in June and July 2017. Riffles, pools, and other geomorphic habitats (runs, glides, and cascades) were surveyed using a modified Basinwide Visual Estimation Technique (Dolloff et al., 1993; Tip of the Mitt Watershed Council, 2015). Riparian and bank structure conditions were documented using the Great Lakes Environmental Assessment Section (GLEAS) Procedure 51 habitat survey (MDEQ, 2008). A riparian and bank structure score out of 60 is the sum of these scores for each site and used in the ordination. Scores of 49-60 are excellent, 31-48 are good, 13-30 are fair, and 0-12 are poor. Substrate was examined using the zigzag method of Wolman pebble counts (Bevenger and King, 1995). Median particle size was calculated (Bevenger and King, 1995), as well as the proportion of fine substrate under 2 mm along the intermediate axis. Woody debris was surveyed using methods of Cordova et al. (2007), which counts all wood pieces greater than one meter in length and ten centimeters in diameter.

Suspended sediment concentration was sampled monthly from May through September 2017, with two additional sampling events immediately after storms. Water samples were collected in one-liter polyethylene bottles in the center of the stream at mid depth. Suspended sediment concentration was analyzed by method 2540 D (Greensberg et al., 1992). Stream discharge was measured in transects during these same events at 60% depth with a Marsh-McBirney Flow Mate 2000 velocity meter (Hach Company, Loveland, CO) attached to a top-setting wading rod. Bedload sediment was sampled using a Helley-Smith Sampler (Bunte et al., 2008).

Stream temperature was recorded every 30 minutes in July and August 2017 using automated Tidbit loggers (Onset Computer Corporation, Bourne, MA). Data were checked visually for any temperature spikes or fluctuations that suggested the logger was out of water. We determined that loggers were in the water throughout the entire deployment. Thermal and fluctuation regimes were examined (Wehrly et al., 1999), as well as the number of hours the stream temperature was above the optimal brown trout (*Salmo trutta*) temperature range of 12° to 19° C (Raleigh et al., 1984).

Benthic Macroinvertebrate and Fish Surveys

Stream macroinvertebrates and fish were surveyed in July 2017 with GLEAS Procedure 51. Procedure 51 is used by the MDEQ to evaluate macroinvertebrate communities, fish, and habitat of wadeable streams throughout Michigan (MDEQ, 2008; Riseng et al., 2010). The framework of Procedure 51 surveys is a regionally modified Index of Biotic Integrity (Karr, 1991). It relies on fixed-count subsampling, which is widely used to reduce costs and time for assessing impairments (Barbour and Gerritsen, 1996). It also relies on multi-habitat sampling, which best represents community structure (Haller, 2010). Procedure 51 metrics successfully assess differences in stream communities based on physical stressors (Haller, 2010). All Procedure 51 total scores were negative, therefore absolute values were used for the ordinations. Additional calculated macroinvertebrate metrics included the Family Biotic Index (FBI;

Hilsenhoff, 1988), total taxa richness, and Shannon diversity and Pielou's evenness (Qu et al., 2017). Macroinvertebrate functional feeding groups and habitat traits were assigned to each taxon (Bouchard et al., 2004; Merritt et al., 2006, 2008; Supplementary Data).

Fish were sampled with a backpack electroshocker in July 2017 at seven fish study sites (Fig. 1). Per MDEQ (2008) protocol, sites were sampled with a single pass in a section of stream that was ten times the width of the stream. Fish were identified to species, enumerated, and released back into the stream. Each site was rated using the MDEQ (2008) scoring scheme. The MDEQ considers a site to be "poor" and thus not attaining its fishery designated use if fewer than 50 fish are caught or anomalies are found on greater than two percent of fish at a site. If ≥ 50 fish are collected, the percentage of salmonids relative to total fish number needs to exceed 1% for a stream to meet its coldwater fisheries designated use.

Ordination

Canonical Correspondence Analysis (CCA) was performed using the Vegan package of R 3.3.2 for each analysis (R Core Team, 2016; Oksanen et al., 2017). This method was chosen because it is widely used in aquatic sciences, which often contain both constrained and unconstrained datasets, and zero-inflated data (ter Braak and Verdonschot, 1995). The community data are macroinvertebrate indices or traits composition, and the constraining data are potential stressors. Scaling 2 was used to display data. A Nonmetric Multidimensional Scaling (NMDS) test was performed alongside each CCA to assess robustness of the main results (Oksanen et al., 2017).

NMDS was used to produce a biplot displaying fish communities at the seven survey sites. NMDS was performed on Bray-Curtis similarity matrices calculated from raw species abundances and standardized by maximum abundances for 400 iterations (Faith et al., 1987; McCune et al., 2002). To verify visual interpretations of fish community groupings in the NMDS biplot, a multi-response permutation procedure (MRPP; Mielke, 1984; Zimmerman et al., 1985) was performed. Euclidean distance measures and a natural weighting, recommended by Mielke (1984), was used in the MRPP. Significance was defined as $\alpha = 0.10$ for the MRPP comparisons because of the low sample size.

2.5 RESULTS

Environmental stressors, macroinvertebrate metrics, macroinvertebrate traits, and fish metric results are ordered along an upstream to downstream and agricultural to urban gradient (i.e. IMC1 to IMC7), followed by the two tributary sites (Tables 2.2, 2.3, 2 .4, and 2.5). Site specific data for fish and macroinvertebrate taxa are included in the Supplementary Data (Tables S2.1, S2.2, and S2.3). Two CCAs were performed, each describing eight axes of relationships, as well as three NMDS analyses.

The first two CCA axes (CCA1 and CCA2) of the macroinvertebrate metrics ordination explain 58.6% and 16.6% of the total variation in the data (Figure 2.4). The first CCA axis appeared to represent a gradient of substrate size and macroinvertebrate community integrity. Positive values of CCA1 corresponded to fine substrate associated with increased tolerant taxa, while negative values of CCA1 corresponded to high richness of Ephemeroptera (mayfly) and Trichoptera (caddisfly) taxa as well as increased amounts of woody debris and riparian vegetation. The second CCA axis largely represented a flow gradient. Positive values of CCA2 corresponded to fast flowing habitats, while negative values of CCA2 corresponded to slow flowing habitats. The remaining constrained axes explained 20.5% of the total variation and one residual CA axis explained 4.3%. Additionally, the bedload sediment constrainer was removed from the multivariate analyses because of a weak association with the axes (CCA1=0.13, CCA2=0.17 in macroinvertebrate metrics CCA). Taxa richness was removed for the same reasons (CCA1=0.04, CCA2=0.04). NMDS also showed an association between Ephemeroptera and woody debris, Trichoptera with high velocity and suspended sediment, and Dipterans and air breathers with fine substrate and pools (stress = 0.028).

The first two CCA axes (CCA1 and CCA2) of the macroinvertebrate feeding traits ordination explained 51.1% and 21.3% of the total variation (Figure 2.5). The first CCA axis represented a flow velocity gradient. Negative values of CCA1 corresponded to fast flowing habitats, while positive values of CCA1 corresponded to slow flowing habitats. The second CCA axis appeared to show a substrate size gradient. Negative values of CCA2 corresponded to fine substrate, while positive values of CCA2 corresponded to greater amounts of woody debris and coarse substrate. The remaining constrained axes explained 21.4% of the total variation and one residual axis explained 6.2%. NMDS showed an association between predators and skaters with fine substrate; collector filterers and sprawlers with water velocity and suspended sediment; and climbers and swimmers with pools (stress = 0.050). However, this analysis did not associate burrowers or clingers with woody debris and riffles, as the CCA weakly does.

The NMDS biplot of the fish community showed three distinct groups that appeared to be driven by velocity and temperature regimes (Fig. 6). The MRPP confirmed our visual interpretation of differences among the groups (A = 0.278, p = 0.009). The MRPP comparisons between groups revealed that there was no difference between the fish communities in the fast and slow flow reaches (A = 0.288, p = 0.33). However, communities did appear to differ as a function of temperature regime with marginally significant differences between the fast velocity and warm temperature communities ($A = 0.23$, $p = 0.10$) and between the slow velocity and warm temperature communities ($A = 0.22$, $p = 0.10$).

Table 2.2. Environmental stressor results from the habitat surveys, water quality monitoring, and temperature loggers in Indian Mill Creek (2017).

Site	Riffles (% area)	Pools (% area)	Riparian Scr. (P 51)	% Fine Substrate	Wood Deb. (per 100 m)	Velocity ($m^3 s^{-1}$)	SSC (PPM)	Bedload (kg day^{-1})	Avg. Temp. (C°)	Temp. Fluct. (C°)	Hours >19 C°	Max. Temp. (C°)
IMC1	0	0	23	58	0	0.17	12.8	28	-	-	-	-
IMC2	4	9	12	71	0	0.16	2.6	27	-	-	-	-
IMC3	0	48	27	69	9	0.15	4.4	53	-	-	-	-
IMC4	19	47	42	60	22	0.54	7.8	124	18.3	4.8	547	21.4
IMC5	35	17	31	23	30	0.27	5.4	8	18.2	5.7	549	22.5
IMC6	46	14	43	20	14	0.48	4.9	200	17.7	4.5	295	21.3
IMC7	21	30	31	33	16	0.55	9.1	2426	16.2	5.3	45	21.0
BC	21	50	35	86	29	0.12	3.3	74	-	-	-	-
WAD	0	65	29	90	1	0.05	2.7	11	-	-	-	-

Table 2.3. Macroinvertebrate metrics results calculated using data from the Procedure 51 surveys in Indian Mill Creek (2017).

Site	Mayfly Richness	Caddisfly Richness	EPT Richness	% Mayfly	% Caddisfly	% EPT	% Air Breathers	% Diptera	P 51 Score	FBI	Total Taxa Richness	Shannon Diversity	Pielou's Evenness
IMC1	0	2	2	0	3	3	7	15	-4	6.04	15	1.45	0.54
IMC2	0	0	0	0	0	0	45	5	-8	6.41	12	1.50	0.60
IMC3	1	1	2	1	6	7	8	10	-6	6.16	21	1.88	0.62
IMC4	2	1	3	3	2	5	2	5	-4	7.65	19	1.72	0.59
IMC5	2	2	4	1	10	11	4	4	-2	5.19	19	2.04	0.69
IMC6	1	1	2	1	2	3	3	3	-6	6.13	17	1.56	0.55
IMC7	1	1	2	10	20	30	1	14	-4	6.14	14	1.69	0.64
BC	1	1	2	4	1	4	78	10	-3	5.81	17	1.32	0.47
WAD	1	0	1	1	0	1	68	55	-5	7.61	19	1.80	0.61

Table 2.4. Macroinvertebrate trait results assessed using data from the Procedure 51 surveys in Indian Mill Creek (2017).

Site	Shredders	Predators	Collector Gatherers	Collector Filterers	Scrapers	Herbivores	Swimmers	Burrowers	Clingers	Climbers	Skaters	Sprawlers
IMC1	170	40	32	17	5	0	8	29	22	1	11	193
IMC2	14	144	16	0	138	3	14	15	138	5	138	5
IMC3	150	30	78	25	4	1	28	19	79	4	10	148
IMC4	190	37	129	13	5	0	11	17	120	18	7	201
IMC5	154	32	65	30	3	0	45	8	87	21	10	113
IMC6	129	21	190	9	0	0	18	7	190	9	8	117
IMC7	152	3	69	85	1	0	38	19	109	1	2	141
BC	12	247	38	6	5	0	24	27	12	5	238	2
WAD	3	98	189	1	23	0	214	12	43	39	1	5

Table 2.5. Fish metric results using data from the Procedure 51 surveys in Indian Mill Creek (2017).

Site	Total Individuals	Taxa Richness	Shannon Diversity	Pielou's Evenness	% Salmonids	P 51 Score
IMC4	78	5	1.08	0.67	2.6	-8
3 Mile Rd.	66	9	1.83	0.83	16.7	-4
IMC5	41	10	1.85	0.80	14.6	-5
IMC6	35	8	1.79	0.86	37.1	-2
U/S Dam	25	6	1.20	0.67	20.0	-4
IMC7	26	4	1.27	0.92	69.2	-5
Turner St.	26	9	1.86	0.85	15.4	-1

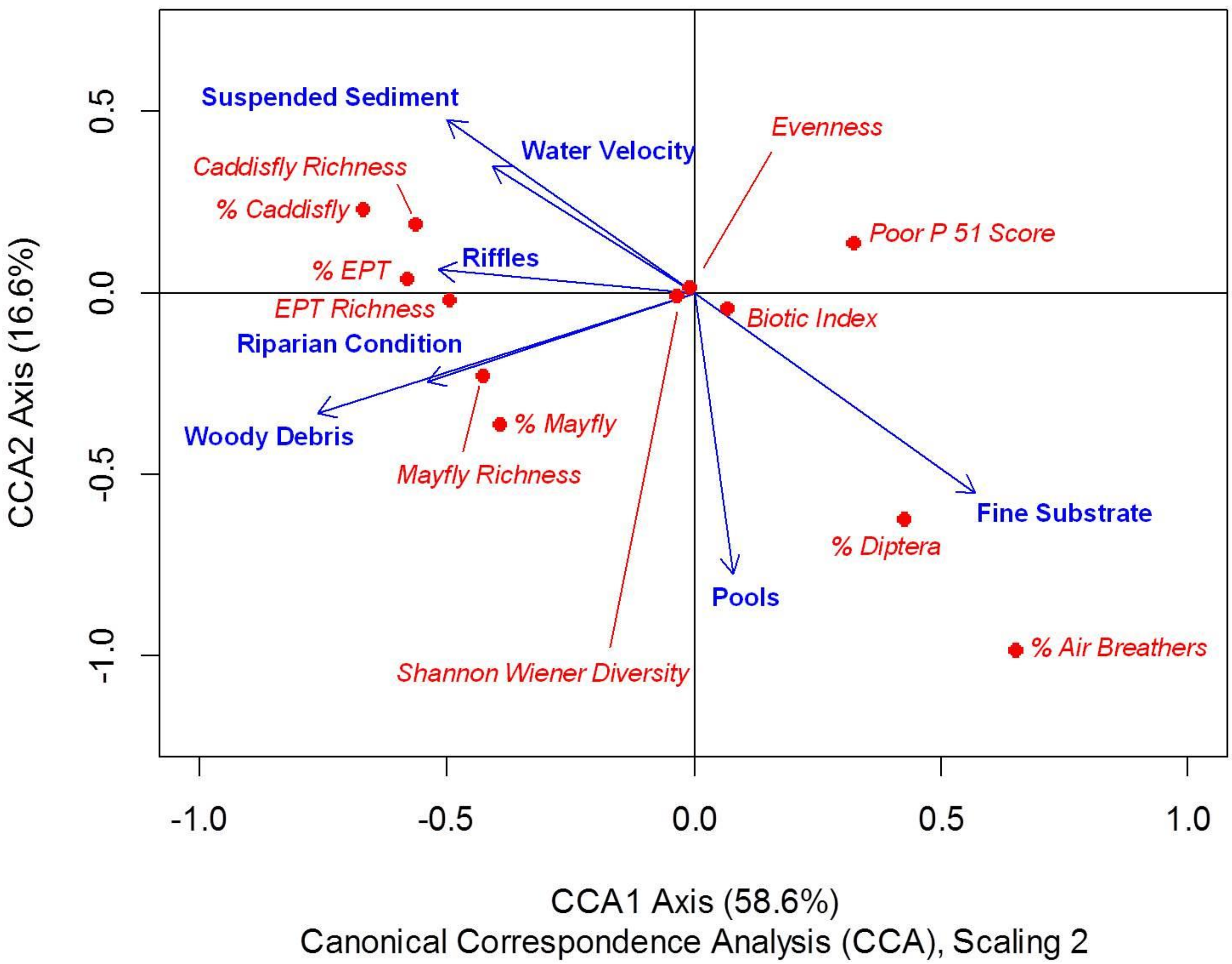

Figure 2.4. CCA Ordination showing the relationships between environmental stressors and macroinvertebrate metrics in Indian Mill Creek (2017).

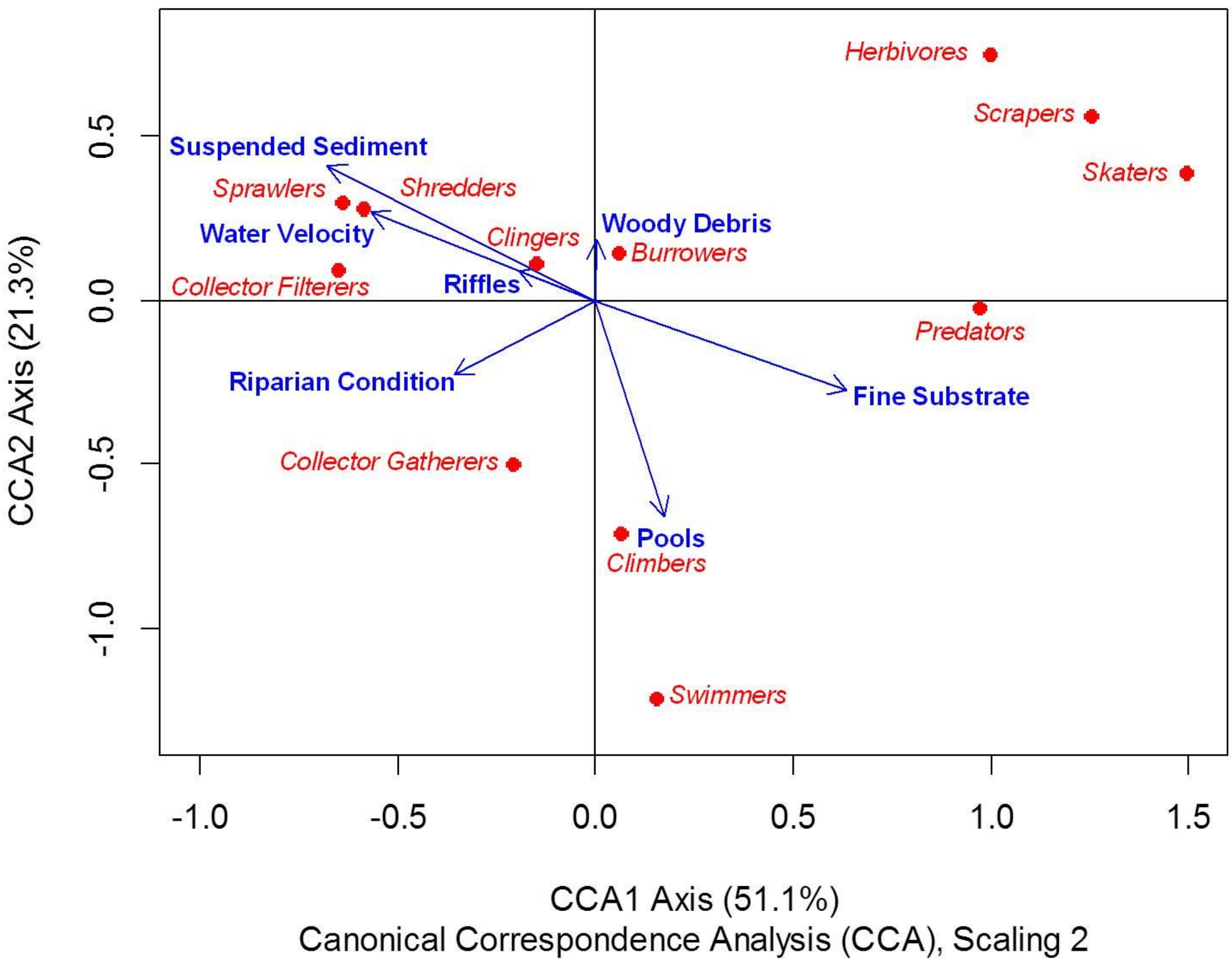

Figure 2.5. CCA Ordination describing the relationships between environmental stressors and macroinvertebrate traits in Indian Mill Creek (2017).

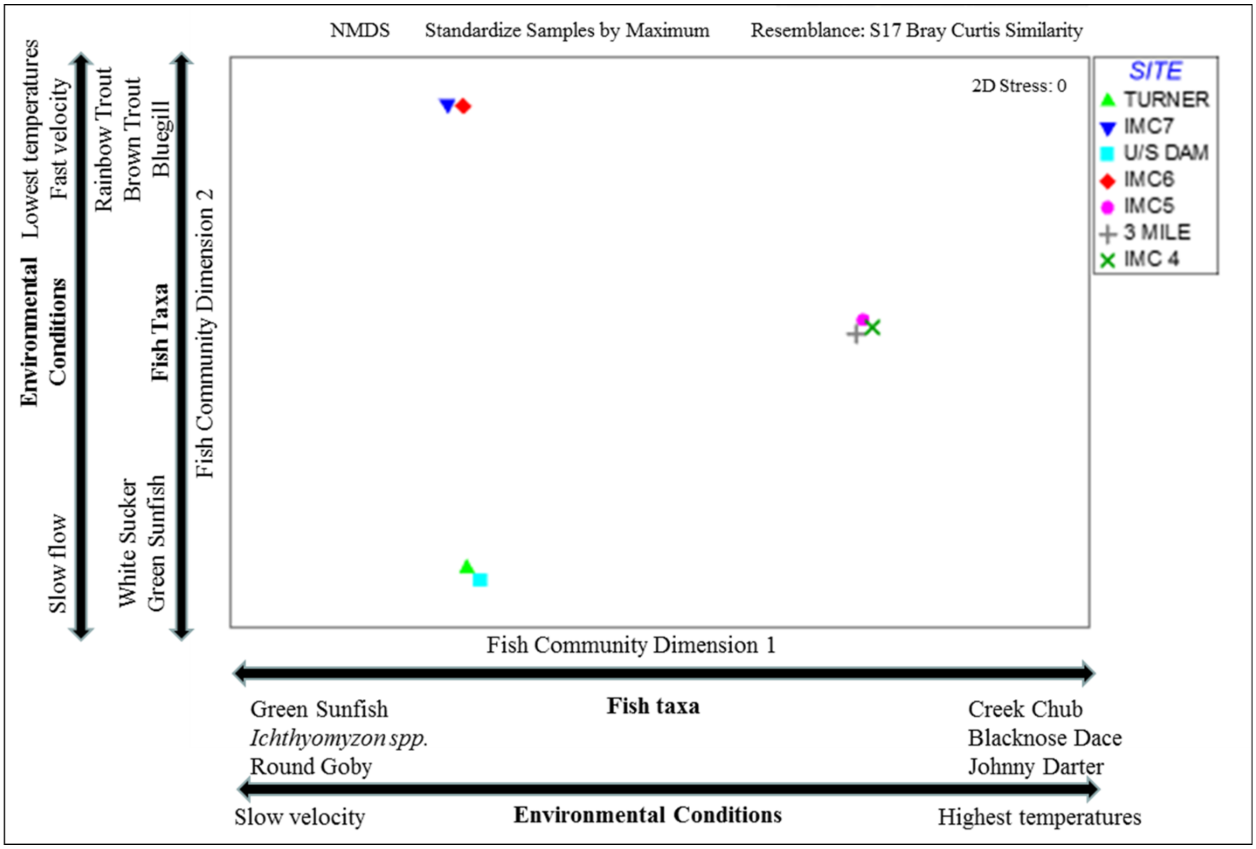

Figure 2.6. NMDS of fish communities in three distinct groups driven largely by stream temperature and flow regime in Indian Mill Creek (2017; A = 0.278, p = 0.009).

2.6 DISCUSSION

A gradient of agricultural to urban landscapes has created a series of environmental stressors that alter biological communities in Indian Mill Creek (Figure 2.7). Environmental stressors include increased sedimentation, loss of habitat variability, woody debris reduction, and riparian zone degradation. Stream temperature regimes also appeared to structure fish communities. Understanding these stressor impacts is important for successful restoration of a waterbody.

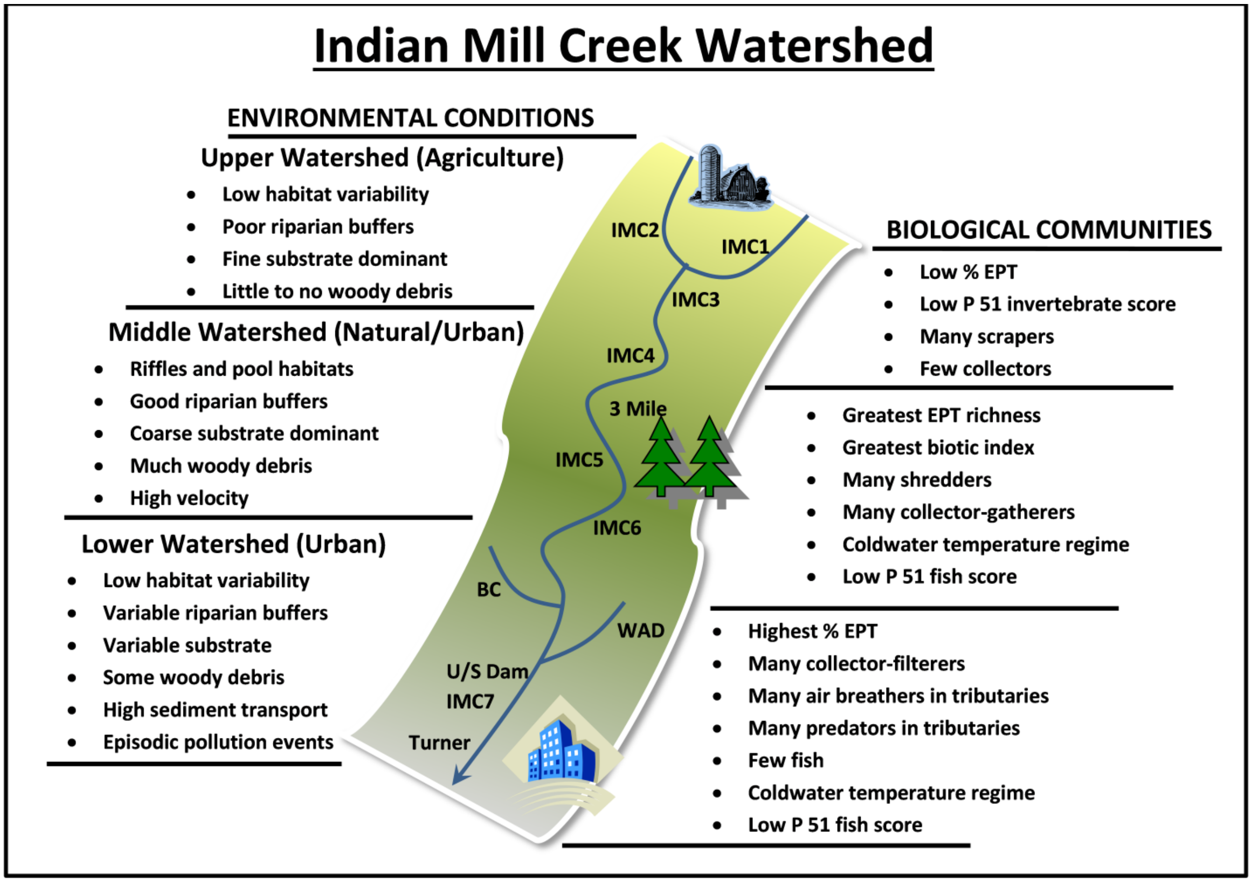

Figure 2.7. Conceptual model of biological communities and their interactions with environmental conditions along a gradient of agricultural to urban land cover in Indian Mill Creek (2017).

Woody Debris

Woody debris abundance was the strongest driver of healthy macroinvertebrate communities in Indian Mill Creek based on the CCA results. It was positively associated with abundance and richness of EPT. However, no sites met the representative condition for Midwest streams of 32.6 pieces per 100 meters (Cordova et al., 2007). Reduction was most evident in the agricultural headwaters where very few wood pieces were found. Deforested riparian zones and lack of mature trees in the headwaters (Figure 2.2) likely limit the amount of large woody debris input to upper Indian Mill Creek. Though still below reference conditions, woody debris was more abundant in the urbanized, lower watershed. The low abundance of woody debris in the agricultural and urban reaches is an expected effect of land cover changes and can influence macroinvertebrate and fish diversity (Allan, 2004). Urban and agricultural streams are typically channelized and cleared of debris to enhance water conveyance from upland sources and to prevent flooding (Booth et al., 1996; Johnson et al., 2003).

Woody debris was most prevalent in the middle watershed with forested riparian areas. This could be explained by recruitment from bank erosion and retention by debris jams (Martin and Benda, 2001). The prevalence of woody debris could also explain the formation of pools at these sites (Evans et al., 1993; Martin and Benda, 2001). The structure and retention of woody debris pieces can shape aquatic communities, especially in streams lacking coarse streambed substrate (Schoen et al., 2013). For example, the presence of woody debris habitat in Michigan agricultural streams increases the number of macroinvertebrate taxa by an average of 55% (Johnson et al., 2003). Woody debris in pools and jams were observed throughout the Indian Mill Creek watershed; however, they were often imbedded in the substrate due to sedimentation.

Sediment and Substrate

Fine sediment in the streambed was the strongest driver of degraded macroinvertebrate communities in Indian Mill Creek and was associated with low abundance and richness of EPT. Fine sediment sources include field erosion, urban stormwater, altered hydrology, and streambank erosion (Paul and Meyer, 2001; Allan, 2004; Kiesel et al., 2009). The distribution of substrate and transport of bedload sediment in the watershed can be explained by a combination of geomorphology and an agricultural to urban land cover gradient. Lane’s Balance (Dust and Wohl, 2012; Pollock et al., 2014) was used to geomorphically explain the substrate composition in the watershed. Lane's balance states that water discharge and channel slope are related to sediment load and representative particle size; it shows whether aggradation or degradation will occur under changing scenarios. The coarsest substrate was in the middle watershed as the creek descends the Grand River Valley. Here, a steep gradient (Figure 2.3) tips Lane's Balance toward increased particle size and erosion of fine particles. The agricultural upper watershed has more gradual slopes, tipping the balance toward finer substrate and sediment deposition. Sediment input from field and streambank erosion could also explain the fine substrate in the agricultural areas (Allan, 2004).

The lower, urban reach had the largest amount of sediment bedload. This section has more gradual slopes because it is in the Grand River floodplain. However, channelization and high flows from runoff and impervious surfaces (Walsh et al., 2005b) counteract the flatter slope, increasing velocity and tipping Lane's Balance toward coarser substrate. Sigdel (2017) found that an increase in stream discharge from impervious surfaces in lower Indian Mill Creek caused banks to erode and moved large amounts of bedload sediment. Excessive bedload buries fish and macroinvertebrate habitat, causes fish populations to decline, and displaces macroinvertebrates (Alexander and Hansen, 1986; Culp et al., 1986; Sigdel, 2017). One of the sources of bedload is streambank erosion, which can be a major contributor of sediment pollution and is a source of sediment to Indian Mill Creek (Fox et al., 2016; Sigdel, 2017). Streambank erosion and sand deposition are evident in the watershed, especially in the Walker Avenue Ditch (Figure 2.8). Streambank erosion can be substantial in Indian Mill Creek, with lateral bank retreat of over 60cm documented in the middle watershed in a single summer (Sigdel, 2017).

Figure 2.8. Habitat in the Walker Avenue Ditch (2017), visually degraded by excessive sand deposition (Sigdel, 2017).

Riffle and Pool Habitats

In Indian Mill Creek, reduced habitat variability occurs along an agricultural to urban gradient. Bedload transport was highest in the urbanized, lower watershed. This is likely reducing habitat variability because excessive bedload sediment has been shown to homogenize the stream channel and create a long run with uniform depth and velocity instead of a riffle/pool series (Alexander and Hansen, 1986). Reduced habitat variability can be unsuitable for fish; for example, brown trout have optimal conditions of 30-50% riffle and 50-70% pool area (Raleigh et al., 1984). This may explain why few fish were found at the middle and lower watershed sites, where pools occupy as low as 14-17% of the habitat area. Agricultural areas in the upper watershed had the least habitat variability, with virtually no riffles and limited pool habitat. Poor macroinvertebrate community integrity in the agricultural, upper watershed could be explained in part by insufficient riffle and/or pool habitat (Raleigh et al., 1984).

Fluvial processes naturally cause variability in stream habitat that structures biological communities (Montgomery, 1999). These processes can be affected by location between the headwaters and mouth, though often not in a uniform way because of the influence of tectonics, lithology, and climate (Statzner and Higler, 1985). The transport of water, sediment, and organic debris often drives changes in channel morphology and habitat characteristics (Montgomery and MacDonald, 2002). However, aquatic habitat variability is often lost in urban and agricultural streams because of channelization, altered hydrology, and deposition of sediment in pools (Paul and Meyer, 2001; Allan, 2004; Lau et al., 2006).

Riparian Condition

Riparian conditions were degraded along a land cover gradient throughout the watershed. The highest level of degradation was in agricultural areas of the upper watershed. Riparian conditions were fair to good in urban areas of the lower and middle watershed, where the sites often had vegetated riparian buffers. Good riparian condition was positively associated with abundance and richness of EPT; thus, degraded conditions could be affecting the integrity of macroinvertebrate communities in the watershed.

Stream restoration can be severely limited if riparian vegetation is lost (Walsh et al., 2005b). Poor riparian conditions contribute to streambank erosion, high water temperatures, increased pollutant loading, and decreased inputs of leaf litter and terrestrial invertebrates that provide energy for aquatic organisms (Delong and Brusven, 1991; Magana, 2001; Nakano and Murakami, 2001; Allan, 2004). Improvement of these conditions is essential for recovery of biological communities and can be done through conservation easements, vegetation buffers, and bank restoration (Natural Resources Conservation Service, 2001; Walsh et al., 2005b).

Macroinvertebrate Traits

In Indian Mill Creek, linkages of environmental stressors with macroinvertebrate feeding groups and traits often occurred along an agricultural to urban gradient. Scrapers and herbivores were most common in the upper watershed, particularly the IMC2 site, and were associated with poor riparian condition. This pattern could be from fertilizer runoff entering the stream and increasing periphyton growth (Compin and Céréghino, 2007), coupled with the absence of shading from riparian vegetation that would otherwise reduce periphyton abundance (Wooster and DeBano, 2006). Scrapers and herbivores were rarely found in the lower, urbanized areas.

Collector gatherers and filterers were found throughout the watershed, as predicted by the River Continuum Concept (Vannote et al., 1980), but were most abundant in the middle and lower reaches. They were associated with high water velocity and good riparian conditions. Collectors were not as common in agricultural headwaters. This could be because of a lower proportion of fine particulate organic matter for headwater streams predicted by the River Continuum Concept (Vannote et al., 1980).

Climbers and swimmers were associated with pools of the two tributary sites. Additionally, predators comprised 80% of the macroinvertebrate community at the Brandywine Creek (BC) site. A balanced stream ecosystem should have only 10-20% predators (Merritt et al., 2006). The top-down control of predators of this site could indicate rapid turnover of prey (Merritt et al., 2006) or large terrestrial prey subsidies from the surrounding riparian area (Nakano and Murakami, 2001).

Shredders and sprawlers were found throughout the watershed but were most associated with high velocity habitats and coarse substrate. These conditions were prevalent in the urbanized middle and lower watershed. Shredders and sprawlers were not commonly found at the agricultural site IMC2 or the tributary sites, which had a high proportion of fine streambed substrate. Improved riparian vegetation condition could be expected to increase the proportion of shredders at IMC2, which had the most degraded condition (Merritt et al., 2006).

Macroinvertebrate traits and feeding groups are used to link biological responses with stressors in streams (Richards et al., 1997; Merritt et al., 2006; Díaz et al., 2008; Menezes et al., 2010). These linkages can occur over many scales, but are often strongest at local levels (Poff, 1997; Richards et al., 1997). Urbanization and agriculture can both alter the composition of macroinvertebrate communities by changing food availability (Compin and Céréghino, 2007). Quantifying the traits and feeding groups helps explain how the macroinvertebrate community will respond to changing environmental conditions (Usseglio-Polatera et al., 2000). For example, reductions in hydrological disturbance in the urban, lower watershed could reduce the proportion of sprawlers and clingers at a site (Townsend et al., 1997). An increase in instream woody debris abundance could benefit clingers, which perch on the structure, and scrapers, who graze its surface (Johnson et al., 2003). Also, improved riparian vegetation conditions could be expected to increase the proportion of shredders in the watershed (Merritt et al., 2006).

Fish and Temperature

Fish survey results revealed low catch numbers at all sites, though the surveys were confined to priority sites in the lower and middle watershed. Fish numbers were particularly low in the lower, urban reaches and increased in an upstream direction. Fish community assemblage appeared to be largely structured by stream temperature and flow. Salmonids, white sucker (*Catostomus commersonii*), and bluegill (*Lepomis macrochirus*) were associated with stable, coldwater reaches, while small Cyprinids and Johnny darter (*Etheostoma nigrum*) were associated with higher temperature reaches. The low-head dam (Figure 2.1) is located between sites IMC6 and IMC7, which had the highest numbers of salmonids. However, the fish community in the small dam impoundment was more similar to the community in the slow-flowing, lower stream reach, near the confluence with the Grand River, suggesting that the dam may be artificially affecting the fish community.

Each site had a poor Procedure 51 fishery score, either due to low values or insufficient catch. The proportion of salmonids suggests that the creek could meet its coldwater fishery designation if abundances improve (MDEQ, 2008). Indian Mill Creek is a coldwater stream with stable to moderate temperature fluxes, conducive to rainbow trout, brown trout, and sculpins (*Cottus* spp.) in the lower watershed (Wehrly et al., 1999; Sigdel, 2017). This can be explained by an increased cold groundwater influx to the creek along an upstream to downstream continuum (Sigdel, 2017). Water temperatures in the middle watershed were above the optimal brown trout limit of 19 °C (Raleigh et al., 1984) for nearly 550 hours throughout July and August, 2017. These elevated temperatures could be a concern for the coldwater fishery and should be further monitored. Riparian vegetation should also be improved because it shades the stream and reduces water temperature (Paul and Meyer, 2001; Allan, 2004). Also, stream channelization can decrease fish populations by reducing habitat variability (Oscoz et al., 2005); this could be affecting the fishery in the lower watershed.

Episodic Pollution Events

The absence/low numbers of small-bodied fishes such as sculpin, darters (*Etheostoma* spp.), and small minnows (Cyprinidae spp.) in the lower reach of Indian Mill Creek was peculiar and may be a result of the episodic pollution events that have occurred within the study area. Sites IMC3 and IMC4, which were the furthest upstream sites, contained the largest number of fish and also had high proportions of small minnow and darter species. Darters (Mundahl and Ingersoll, 1983), small-bodied Cyprinids (Mundahl and Ingersoll, 1989), and sculpin (Breen et al., 2009) tend to have relatively small home ranges in stream systems. Episodic pollution events can have major impacts to fish communities via direct mortality or by causing fish to seek refuge in unpolluted waters (Seager and Maltby, 1989; Van Sickle et al., 1996). Full recovery of small fish populations from these events can take several years (Albanese et al., 2009; Kubach et al., 2011). Slow recovery rates can be further exacerbated when dispersal barriers are present (Albanese et al., 2009); the lower reach of Indian Mill Creek contains a low-head dam that could be acting as a dispersal barrier. Thus, episodic pollution events in the lower, urbanized reach of Indian Mill Creek are one possible explanation for the low numbers of fish, particularly small-bodied, sedentary species. This highlights the need for further toxicity studies in the watershed.

Restoration

An understanding of environmental stressors and their interactions is important for successful restoration of Indian Mill Creek, which is a high priority catchment in Michigan's Lower Grand River Watershed (LGROW, 2011), and other watersheds with similar land use patterns. Additional quantitative tools are available that spatially link the land cover of a catchment with aquatic community integrity and should be employed to aid in restoration planning (Johnson et al., 2007; McNair, 2009). We are currently evaluating modeling with the Enhanced Generalized Watershed Loading Functions (GWLF-E) model (Evans et al., 2003) as it includes estimates of sediment loading from overland flow and streambank erosion (See Chapter III). It is important to note that restoration of stream habitat and riparian conditions in urban streams can be ineffective for recovery of aquatic life if the destructive impacts of intense stormflows aren't addressed (Walsh et al., 2005a). Restoration of aquatic habitat in the urbanized, lower watershed should focus on reducing the amount of sediment-laden runoff through low impact development and best management practices, as described by Southeast Michigan Council of Governments (2008). Suggested practices for the lower, urbanized watershed include a reduction in impervious surface area, bioretention basins, pervious pavement, detention basins, floodplain avoidance, wetland conservation, and vegetated swales. A restoration plan for the agricultural, upper watershed's riparian corridors should be designed and implemented as per Natural Resources Conservation Service (2001) guidance. If storm flows are reduced, then managers should restore woody debris habitat in both urban and agricultural areas; this restoration of woody habitats has been shown to increase the richness of macroinvertebrate taxa and functional groups (Lester et al., 2007). Riffle and pool habitat variability can be restored through dechannelization of the creek in agricultural headwaters and the lowest kilometer that has been artificially straightened by urbanization. A channelized Indiana stream that was experimentally restored by constructing riffle and pool habitats, adding woody debris, and reducing sedimentation saw a recovery of macroinvertebrates and fish within one year and remained high after five years of monitoring (Moerke et al., 2004). However, this study noted that long-term effects could be uncertain if sedimentation is not controlled at a watershed scale.

Therefore, a stand-alone watershed plan should be developed for Indian Mill Creek that summarizes watershed conditions, identifies priority pollutants and critical areas, cooperatively develops goals and objectives, and outlines an action plan with realistic projects to control nonpoint source pollution (Brown et al., 2000). Sediment sources should be spatially analyzed using a watershed model that includes both field and bank erosion (Evans et al., 2003; Kiesel et al., 2009) to identify priority areas for sediment load reduction. Part of watershed plan development should be a road-stream crossings inventory following Great Lakes Road Stream Crossing Inventory Protocol (US Forest Service et al., 2011). This inventory would assess the impacts of crossings on hydrology, sediment transport, and fish passage in the watershed. Poorly-designed crossings have been an impediment to fish passage in other Michigan streams (Briggs and Galarowicz, 2013; Evans et al., 2015). We inventoried one crossing in the upper watershed near the IMC5 site following this protocol and found it to be in poor condition. We recommend culvert replacement and erosion control to remediate poorly-designed crossings. A long-term monitoring program should be developed for stream habitat, water quality, and biological communities; one option is participation in the Michigan Clean Water Corps Volunteer Stream Monitoring Program (MiCorps, 2006). More sites would improve statistical power, while repeated measures over many years would help understand temporal variation in environmental stressors and biological communities. Conservation Practice Standards should be implemented in agricultural areas to control runoff and reduce nonpoint source pollution. These standards, with Natural Resources Conservation Service Conservation Practice Standard guides in parenthesis, include riparian cover (guides 390 and 391), filter strips (393), conservation cover (327), and residue and tillage management (329, 345).

Successful restoration of Indian Mill Creek is dependent on continued involvement of watershed organizations, local governments, researchers, and other stakeholders. Watershed organizations can stimulate community involvement and cooperatively search for funding for restoration projects, such as funding through the MDEQ 319 Program. The Friends of Indian Mill Creek group has been formed to bring stakeholders together and address the issues in the watershed. The local governments of Alpine Township and the cities of Grand Rapids and Walker can develop planning and zoning ordinances for water resources protection with the help of the model stormwater ordinances from Kent County Drain Commissioner's office and the USEPA (https://www.epa.gov/nps). These cooperative efforts among stakeholders are vital to the watershed's restoration and should be continued.

Conclusion

A combination of environmental stressors from both agricultural and urban land cover is affecting the structure and function of aquatic communities in the Indian Mill Creek watershed. Multivariate statistics were used to understand relationships between environmental stressors and aquatic communities. The largest stressors affecting macroinvertebrate communities were increased sedimentation, loss of habitat variability, woody debris reduction, and riparian zone degradation. The main stressors affecting fish communities appeared to be stream temperature and flow, although episodic pollution events in the watershed could also be important. These effects occurred along an agricultural to urban gradient. Understanding how complex environmental stressors affect aquatic communities along this gradient is important for successful restoration of a waterbody. Agricultural and urban land cover changes and their associated impacts to lotic ecosystems are prevalent worldwide; therefore, this study has broad applications.

Supplementary Material

Supplementary material includes tables of fish data, macroinvertebrate data, and macroinvertebrate traits and can be found after the literature cited.

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2.8 SUPPLEMENTARY MATERIAL

Impacts of an Agricultural/Urban Land Cover Gradient in a Coldwater Stream

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Contents

Supplemental Table S2.1. Fish data from Procedure 51 surveys.

Site	Group	Brown trout (Salmo trutta)	Rainbow trout (Oncorhynchus mykiss)	Smallmouth bass (Micropterus dolomieu)	White sucker (Catostomus commersonii)	Green sunfish (Lepomis cyanellus)	Round goby (Neogobius melanostomus)	Silver lamprey (Ichthyomyzon unicuspis)	Bluegill (Lepomis macrochirus)	Chestnut lamprey (Ichthyomyzon castaneus)	Brook stickleback (Culaea inconstans)	Johnny darter (Etheostoma nigrum)	Creek chub (Semotilus atromaculatus)	Pumpkinseed (Lepomis gibbosus)	Blacknose dace (Rhinichthys atratulus)	Central mudminnow (Umbra limi)	Lake chubsucker (Erimyzon sucetta)	Central stoneroller (Campostoma anomalum)
Turner	Slow	2	2	1	8	7	3	1	1	1	0	0	0	0	0	0	0	0
IMC 7	Fast - stable	9	9	0	0	2	0	0	6	0	0	0	0	0	0	0	0	0
Upstream of Dam	Slow	3	2	0	16	0	0	0	0	0	1	2	1	0	0	0	0	0
IMC 6	Fast - stable	1	14	0	0	3	0	0	7	0	0	5	4	2	1	0	0	0
IMC 5	Warm	3	3	0	3	1	0	0	2	0	0	1	15	3	7	1	0	0
3 Mile	Warm	7	4	0	12	2	0	0	0	0	0	7	9	0	23	0	1	1
IMC 4	Warm	2	0	0	12	3	0	0	0	0	0	20	41	0	17	0	0	0

Supplemental Table S2.2. Macroinvertebrate data from Procedure 51 surveys.

	Macroinvertebrates Collected at Each Site								
Macroinvertebrate Taxa	**IMC7**	**BC**	**IMC6**	**IMC5**	**WAD**	**IMC4**	**IMC3**	**IMC2**	**IMC1**
PLATYHELMINTHES (flatworms)									
Turbellaria	22		167	40	2	86	50		
NEMATOMORPHA (roundworms)			1			1	9		
ANNELIDA (segmented worms)									
Hirudinea (leeches)		2			17				
Oligochaeta (worms)									1
ARTHROPODA									
Crustacea									
Amphipoda (scuds)	7	9	14	42			3	10	
Decapoda (crayfish)		2	9	6		4	6	2	3
Isopoda (sowbugs)	141		106	105		186	140	2	165
Arachnoidea									
Hydracarina					3	11	1	1	25
Insecta									
Ephemeroptera (mayflies)									
Baetidae	30	12	2	1	3	10	4		
Heptageniidae				1		3			
Odonata									
Anisoptera (dragonflies)									
Aeshnidae		2	5	15	1	5	4		
Zygoptera (damselflies)									
Calopterygidae	1	2	3	6		9		5	
Coenagrionidae		1	1		38	4			
Hemiptera (true bugs)									
Corixidae							1	3	
Gerridae	1	32	1	9		1	1	40	2
Notonectidae			1		4		1		
Pleidae							4		
Veliidae	1	206	7	1	1	6	9	98	9
Trichoptera (caddisflies)									
Hydropsychidae	61	1	8	28		7	18		6
Limnephilidae				1					
Phryganeidae									1
Coleoptera (beetles)									
Dytiscidae (total)		2			30				4
Haliplidae (adults)					2		1		
Hydrophilidae (total)	1	1		1	17		4	1	4
Dryopidae				2					

Elmidae	1		14	14		19			
Diptera (flies)									
Athericidae			2	1					
Ceratopogonidae					2		1		
Chironomidae	15	25	7	8	9	14	19	15	27
Culicidae					158		1		
Dixidae				1					
Sciomyzidae					2				
Simuliidae	24	4	1	2	1	3	7		11
Tipulidae	4	1			1				1
MOLLUSCA									
Gastropoda (snails)									
Lymnaeidae					2			3	2
Physidae	1	5			21	1	4	135	3
Planorbidae						1			
Pelecypoda (bivalves)									
Sphaeriidae (clams)		1				3			

Supplemental Table S2.3. Macroinvertebrate trait data from Procedure 51 surveys.

Common Name	Scientific Name	Number Collected	FFG	Habitat Trait	Source
Mayflies	**Ephemeroptera**				
Swimming Mayfly	Baetidae	62	Collector gatherers	Swimmers	Merritt et al. 2008
Flat Head Mayfly	Heptageniidae	4	Scrapers	Clingers	Merritt et al. 2008
Caddisflies	**Trichoptera**				
Net-Spinning Caddisfly	Hydropsychidae	129	Collector filterers	Clingers	Merritt et al. 2008
Northern Caddisfly	Limnephilidae	1	Shredders	Sprawlers	Merritt et al. 2008
Giant Casemaker	Phryganeidae	1	Shredders	Climbers	Merritt et al. 2008
Dragon & Damselflies	**Odonata**				
Darner Dragonfly	Aeshnidae	32	Predators	Climbers	Merritt et al. 2008
Broad-Winged Damsel	Calopterygidae	26	Predators	Climbers	Merritt et al. 2008
Narrow-Winged Damsel	Coenagrionidae	44	Predators	Climbers	Merritt et al. 2008
True Bugs	**Hemiptera**				
Water Boatman	Corixidae	4	Herbivores	Swimmers	Merritt et al. 2008
Water Strider	Gerridae	87	Predators	Skaters	Merritt et al. 2008
Backswimmer	Notonectidae	6	Predators	Swimmers	Merritt et al. 2008
Pygmy Backswimmer	Pleidae	4	Predators	Swimmers	Merritt et al. 2008
Small Water Strider	Veliidae	338	Predators	Skaters	Merritt et al. 2008
Beetles	**Coleoptera**				
Predaceous Diving Beetle	Dytiscidae	36	Predators	Swimmers	Merritt et al. 2008
Crawling Water Beetle	Haliplidae	3	Shredders	Swimmers	Merritt et al. 2008
Water Scavenger Beetle	Hydrophilidae	29	Collector gatherers	Swimmers	Merritt et al. 2008
Long-Toed Water Beetle	Dryopidae	2	Scrapers	Clingers	Merritt et al. 2008
Riffle Beetle	Elmidae	48	Collector gatherers	Clingers	Merritt et al. 2008
True Flies	**Diptera**				
Watersnipe Fly	Athericidae	3	Predators	Sprawlers	Merritt et al. 2008
No-See-Ums	Ceratopogonidae	3	Predators	Sprawlers	Merritt et al. 2008
Midge	Chironomidae	139	Collector-gatherers	Burrowers	Merritt et al. 2008

Mosquito	Culicidae	159	Collector gatherers	Swimmers	Merritt et al. 2008
Meniscus Midge	Dixidae	1	Collector gatherers	Swimmers	Merritt et al. 2008
Marsh Fly	Sciomyzidae	2	Predators	Burrowers	Merritt et al. 2008
Black Fly	Simuliidae	53	Collector filterers	Clingers	Merritt et al. 2008
Crane Fly	Tipulidae	7	Shredders	Burrowers	Merritt et al. 2008
Worms	**Platyhelminthes, Nematoda, & Annelida**				
Flatworm	Turbellaria	367	Collector gatherer	Clingers	Bouchard et al. 2004
Roundworm	Nematomorpha	11	Predator	Swimmers	Bouchard et al. 2004
Leech	Hirudinea	19	Predator	Clingers	Bouchard et al. 2004
Earthworm	Oligochaeta	1	Collector gatherer	Burrowers	Bouchard et al. 2004
Crustaceans	**Crustacea**				
Scud	Amphipoda (Gammar.)	85	Shredders	Swimmers	Merritt and Cummins 2006; Bouchard et al. 2004
Crayfish	Decapoda	32	Shredders	Sprawlers	Merritt and Cummins 2006; Bouchard et al. 2004
Sowbug	Isopoda	845	Shredders	Sprawlers	Merritt and Cummins 2006; Bouchard et al. 2004
Arachnids	**Arachnoidea**				
Water Mite	Hydracarina	41	Predator	Sprawlers	Bouchard et al. 2004
Mollusks	**Mollusca**				
Pond Snail	Lymnaeidae	7	Scraper	Clingers	Bouchard et al. 2004
Pouch Snail	Physidae	170	Scraper	Clingers	Bouchard et al. 2004
Ram's Horn Snail	Planorbidae	1	Scraper	Clingers	Bouchard et al. 2004
Fingernail Clams	Sphaeriidae	4	Collector filterer	Burrowers	Bouchard et al. 2004

CHAPTER III: WATERSHED AND STREAMBANK EROSION MODELING IN A MICHIGAN, USA STREAM USING THE GWLF-E MODEL AND MAPSHED GIS PLUGIN

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Research Impact Statement

We used the GWLF-E Model and MapShed plugin for MapWindow GIS for the first time in Michigan to simulate water budget, field erosion, and streambank erosion in an urban coldwater stream's watershed.

3.1 ABSTRACT

Sediment pollution is a major cause of stream degradation. Our purpose was to identify critical areas for sediment management in the Indian Mill Creek watershed, Michigan, USA. We used the Enhanced Generalized Watershed Loading Functions (GWLF-E) model and MapShed plugin for MapWindow GIS to simulate the water budget, field erosion from the landscape, and streambank erosion in 20 subbasins from 1997-2015. Annual water budget results suggest that Indian Mill Creek is primarily groundwater fed, but that a per-subbasin average of 6% to 15% of precipitation becomes runoff. Stream discharge data collected with a flow meter suggest that GWLF-E, although not calibrated to our study catchment, follows the same pattern of increasing discharge toward the outlet of the creek, but may be overestimating discharge in subbasins by a factor of 2.8 to 11.0. Field erosion contributed a per-subbasin average of 0.5 to 2.5 Mg ha^{-1} yr^{-1} of sediment, while streambank erosion accounted for 0.2% to 50.1% of the subbasins' total sediment yields. Average lateral erosion rate of streambanks in subbasins ranged from 0.04 to 7.37 cm yr^{-1}. Southwest subbasins had the highest rates of runoff because of impervious surfaces and urbanization. Field erosion was greatest in subbasins with steep slopes and erodible soils. The proportion of sediment load from streambanks and the lateral erosion rate increased in a downstream direction. We found the GWLF-E model and MapShed plugin understandable and easy to use. However, model simplicity could introduce uncertainty. Findings will help watershed managers prioritize restoration programs to reduce sediment loadings.

Keywords

Rivers/streams; watersheds; erosion; sediment; soils; evapotranspiration; precipitation; runoff; land use/land cover change; urbanization; geospatial analysis; nonpoint source pollution; watershed management)

3.2 INTRODUCTION

Sediment pollution is the second-highest cause of stream degradation in the United States, impairing the health and designated uses of nearly 225,000 km of streams (USEPA, 2016). Sediment pollution can enter a stream through various pathways including bank erosion, runoff from the landscape, and drains (Kiesel *et al.*, 2009). The movement of sediment into streams is becoming more intense because of urban and agricultural land use changes and climate change (Allan, 2004; Bartolai *et al.*, 2015). Stream sediment loads increase with agriculture and urban development in a watershed because fields, ditches, impervious surfaces, and stormwater conveyance systems increase sediment-laden runoff and cause high peak flows that erode the banks (Allan *et al*., 1997; Carpenter *et al*., 1998; Jones *et al*., 2001; Paul and Meyer 2001; Allan 2004). Extreme storms and increases in runoff because of climate change can intensify erosion from landscapes and stream channels, increasing the delivery of sediment into streams (Bartolai *et al.*, 2015). Sediment pollution negatively affects streams by reducing habitat variability, invertebrate diversity, and habitat suitability for fish (Alexander and Hansen, 1986; Culp *et al.*, 1986; Raleigh *et al.*, 1984). It also reduces water clarity, increases water treatment costs, decreases reservoir storage area, and carries phosphorus pollution into streams (Fox *et al.*, 2016a).

Quantifying sediment pollution on a catchment scale is important for resource management but also problematic. It requires an understanding of the pathways sediment enters the water and the complex factors that affect its movement; data at the catchment scale may not be available (Dietrich *et al.*, 1999; Kiesel *et al.*, 2009). To address these problems, sediment transport to streams can be estimated using models that calculate pollutant export coefficients, loading functions, and chemical simulation (Haith and Shoemaker, 1987). Data sources for the models can be readily-available spatial data and/or field-collected information (Haith and Shoemaker, 1987; Kiesel *et al.*, 2009). Managers are often more focused on the distribution of erosion risk throughout a watershed than quantifying soil loss; these measured quantifications can have limitations of cost, representativeness, and reliability that make them unrealistic for assessing spatial distributions of erosion risk over a large area (Lu *et al.*, 2004). When choosing a watershed model for a study, it is important to consider sediment pathways, incorporation of complex factors, data attainability, and the distribution of erosion risk.

The Generalized Watershed Loading Functions (GWLF) model has been used extensively in Pennsylvania, New York, Virginia, and Illinois to model nonpoint source pollution in watersheds and develop sediment and nutrient Total Maximum Daily Loads (TMDLs; Borah *et al.*, 2006; Evans *et al.*, 2003). GWLF is a mid-range process-based model that predicts the transport of water, sediment, and nutrients in a watershed without flow routing (Haith and Shoemaker, 1987; Shoemaker *et al.*, 2005, 1997). GWLF uses readily-available spatial data including land cover, soil characteristics, precipitation patterns, and topography to estimate pollutant loads and hydrological regimes (Haith and Shoemaker, 1987). An advantage of GWLF is that it is easy to use and relies on simpler data inputs than other more-complex watershed models (Markel *et al.*, 2006). Another advantage is that GWLF can be used in watersheds without gauges and with mixed land uses (Borah *et al.*, 2006). The limitation of the model is the degree of uncertainty. Different sources of input data can cause changes in loading outputs that affect pollutant load requirements. For example, using land cover data from the National Land Cover Dataset versus the Digital Ortho-Quarter Quads in the GWLF model can change TMDL reduction estimates from 13% to 74% (Wagner *et al.*, 2007). However, we deem the GWLF model appropriate for this study because we are assessing the spatial distribution of sediment loading in the watershed rather than defining numerical targets.

A major need for research in watershed modeling is the prediction of sediment export from streambank erosion, which can be the primary contributor of alluvial materials to streams (Fox *et al.*, 2016a). This makes streambank erosion a very important component, though often absent, in sediment TMDLs (McMillan *et al.*, 2018). Streambank erosion is difficult to model because of complex environmental factors and drastically varying erodibility characteristics (Evans *et al.*, 2003; Fox *et al.*, 2016b). These include groundwater seeps, channel curvature, and riparian vegetation (Fox *et al.*, 2007; McMillan and Hu, 2017; Purvis and Fox, 2016). This environmental complexity magnifies uncertainty in streambank erosion assessments (Kiesel *et al.*, 2009).

An enhancement to the GWLF model called Enhanced GWLF (GWLF-E) estimates the sediment loads of eroding streambanks at watershed and subbasin levels. It uses readily available spatial data, requires no field data, and has been refined through testing of twenty eight Pennsylvania watersheds and subsequent adjusting (Evans *et al.*, 2003). It can be run using the MapShed plugin of MapWindow GIS and the GWLF-E model (Penn State Institutes of Energy and the Environment, University Park, Pennsylvania, USA) or the Stroud Water Research Center's Model my Watershed website (www.wikiwatershed.org/model, Accessed March 23, 2018). We used the MapShed plugin and GWLF-E model for this study because it incorporates streambank erosion, uses attainable data, and can assess the erosion risk for different subbasins of our watershed. Alternative streambank erosion models that could have been used are summarized in the Discussion.

The purpose of this study is to identify critical areas for sediment pollution management in the Indian Mill Creek watershed of Michigan, USA using a nonpoint source pollution model. To accomplish this, we modeled runoff and sediment loading from 20 subbasins and their matching stream sections from 1997-2015. We aimed to determine if agricultural areas in the upper watershed contribute the most sediment from field erosion and if urban areas in the lower watershed have the highest streambank erosion rates because of increased runoff from impervious surfaces. This information will be used by water quality mangers and local units of government to prioritize restoration programs to reduce sediment loadings and improve stream habitat. To the best of our knowledge, this is the first time the MapShed plugin and GWLF-E model have been used for a watershed study in Michigan.

3.3 METHODS

Study Area

Indian Mill Creek in Kent County, Michigan, USA (HUC 040500060504) is on the Michigan 303(d) list of impaired water bodies, with sediment loading and deposition identified as the cause of impairment (Sigdel, 2017). It is a tributary to the Grand River and is 18.5 km long with a 44 km^2 watershed (Figure 3.1). The creek resides in the Southern Michigan Northern Indiana Till Plains ecoregion, characterized by irregular plains, cropland, pasture, and oak/hickory/beech/maple forests (Omernik, 1987). The watershed land cover is predominately urban (43%) and agricultural (39%), with commercial and residential development in the lower watershed, natural and urban lands in the middle watershed, and farmland and orchards in the upper watershed (Figure 3.2, LGROW 2011). This land cover pattern affects the distribution of erosion risk in the watershed. The National Weather Service classifies the area as a humid continental climate with distinct summers and winters and fairly even distribution of precipitation throughout the year (www.weather.gov). Climate predictions are that the region will have more frequent extreme precipitation events, which can increase erosion rates (Bartolai *et al*., 2015). Indian Mill Creek is designated as a coldwater trout stream by the State of Michigan; however, it is currently not supporting its coldwater fishery designated use per Michigan Department of Environmental Quality (MDEQ) standards (Goodwin *et al.*, 2016).

Geologic features of the Indian Mill Creek watershed were formed by retreating glaciers that deposited hills of medium-textured till in the upper watershed (Farrand and Bell, 1982).

Glacial meltwater carved the larger Grand River Valley, which Indian Mill Creek descends for five kilometers starting downstream from the present location of Interstate 96 and descending 24 meters in elevation (Gesch *et al.*, 2002; Larson and Schaetzl, 2001). The side of the valley in these reaches has steep slopes, from 25% to 50% or greater along its southern edge (Figure 3.3). This topography can affect erosion rates, with higher erosion in areas with steeper slopes (Wischmeier and Smith, 1978). Overall, the creek descends 65 meters in elevation from headwaters to mouth. The lower watershed gently slopes in an outwash of sand and gravel with postglacial alluvium (Farrand and Bell, 1982). It contains alluvial hydrologic group A and B soils however urban land areas have patchy data availability (Figure 3.4; Soil Survey Staff, 2017). In contrast, the upper watershed has loamy hydrologic group C and C/D soils with low infiltration in uplands, but sandy A/D and B/D soils along the West Branch and Indian Mill Creek. The middle watershed is a transition zone and has loamy C and C/D soils in uplands and sandy A and B soils with high infiltration by the main channel and the Walker Avenue Ditch. These soils affect the distribution of runoff and erosion risk in the watershed; high runoff is associated with groups C and D soils, such as those in upper and middle watershed's uplands (U.S. Department of Agriculture, Chapter 7 of Part 630 Hydrology of the National Engineering Handbook), These soils also are associated with higher erodibility in the watershed (Soil Survey Staff, 2017).

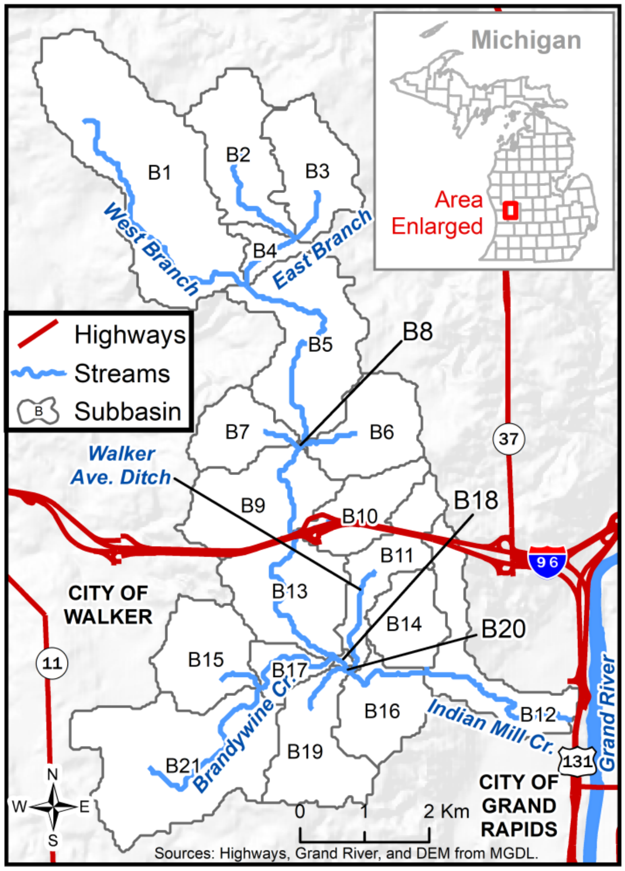


Figure 3.1. Map of the Indian Mill Creek watershed.

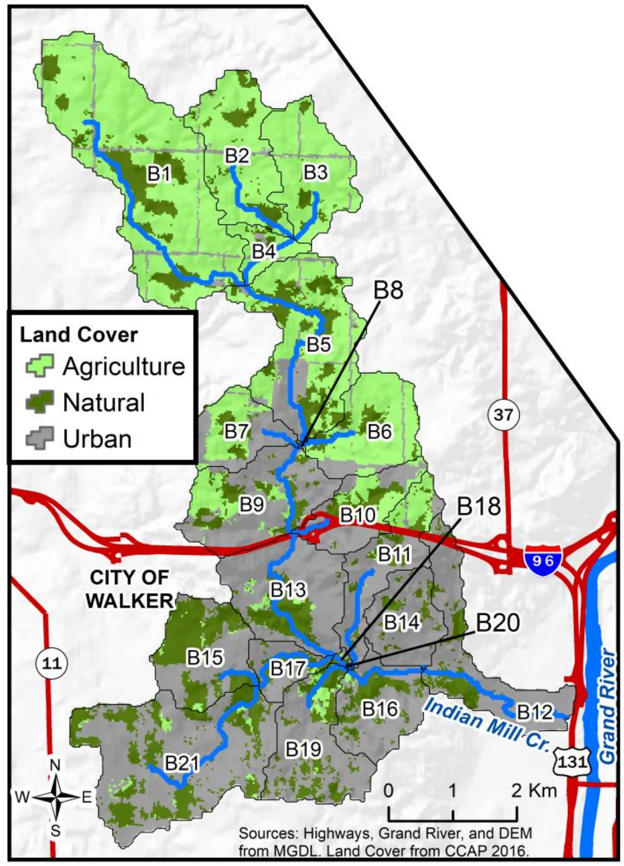


Figure 3.2. Land cover of the Indian Mill Creek watershed.

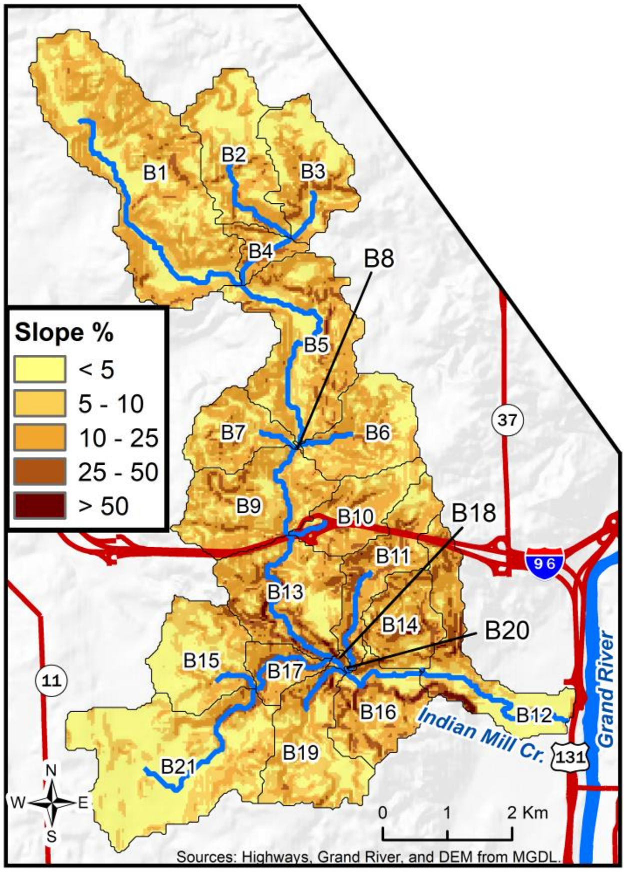


Figure 3.3. Slopes of the Indian Mill Creek watershed.

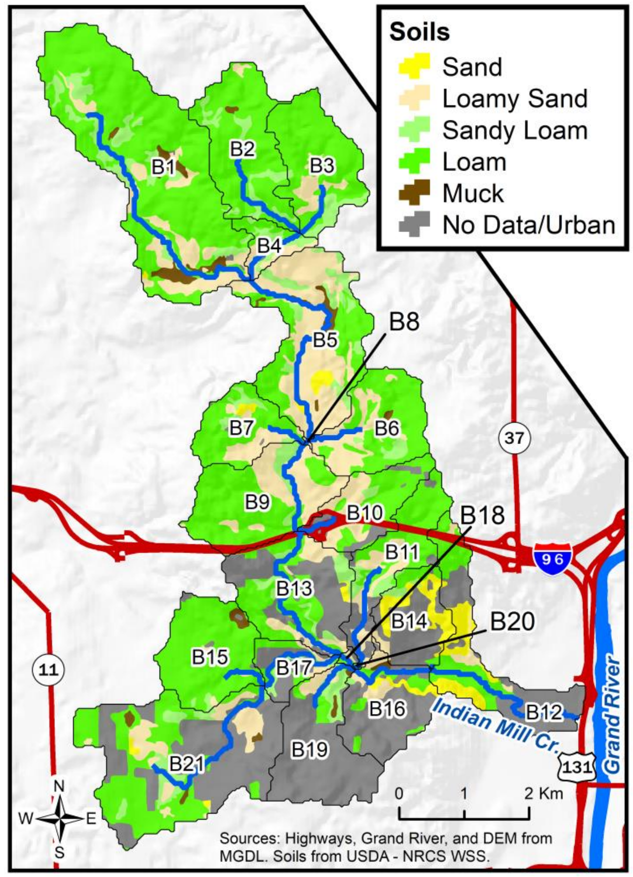


Figure 3.4. Soil types of the Indian Mill Creek watershed.

Modeling

The GWLF-E model was run using Penn State's MapShed model and GWLF-E model with widely available spatial data. The MapShed model uses MapWindow Geographic Information System software to create an input file for the GWLF-E model. GWLF-E was then used to process the input file and simulate watershed hydrology and pollutant loadings from 1997-2015.

Spatial data for the MapShed model were collected from multiple sources. Originally 21 subbasins and streams were delineated using the Watershed Delineation plugin of MapWindow.

Subbasin B20 was removed because it had an area (0.7 ha) and stream length (32 m) that were too small to run GWLF-E and negligible overall. Thus, the model was run on 20 subbasins. Elevation data was derived from a 30 meter digital elevation model from the National Elevation Dataset (Gesch *et al.*, 2002). Land cover data was from the National Oceanographic and Atmospheric Administration Coastal Change Analysis Program (Office for Coastal Management, 2016). Spatial Soil Survey Geographic Database (SSURGO) soils data was downloaded from the Natural Resources Conservation Service's Web Soil Survey and joined with tabular data (Soil Survey Staff, 2017). Soils of the urban land type and gaps in soil data availability in the lower watershed interfered with the model; we thus assumed them to be impervious surfaces assigned a soil erodibility factor of zero, hydrologic group D, and Available Water Capacity of zero. This method inevitably introduces uncertainty into the outputs of affected subbasins; however, we deemed it the best approach because collecting data in the field would require more resources than available for the study. Precipitation and temperature data were downloaded from Michigan Enviro-Weather's Sparta station (https://www.enviroweather.msu.edu. Accessed March 23, 2018). Although the MapShed plugin requests two stations of weather data, we used only one because other nearby Enviro-Weather stations lacked the desired time span of data. We suggest future modelers incorporate additional weather data sources like the NOAA National Climatic Data Center to better account for spatial variability. All spatial data were projected as NAD 1983 Michigan GeoRef meters and reformatted to match the requirements of the MapShed model (Evans and Corradini, 2016). Data layers and alignment were checked for errors before proceeding. Streamflow Volume Adjustment Factors were calculated to account for the contribution of stream flow from upper basins to lower basins (Evans and Corradini, 2016).

A GWLF-E input file was created for each subbasin using the MapShed Tools. This file was created for the years of available weather data (1997-2015) and a growing season of May to September. This input file was then imported into the GWLF-E model for each subbasin. The GWLF-E model was run; the output files included summaries for each basin. Results for runoff, field erosion, and streambank erosion were extracted from these summaries. Lateral erosion rate (LER), although not a direct output of the model, was calculated by dividing the mass of erosion by the GWLF-E's default bulk density (1.5 Mg/m^3), default bank height (1.5 m), and length of stream in the subbasin (m), and then converted to centimeters. The outputs were joined with the subbasins' and streams' GIS data and given quantitative symbologies.

Discharge Estimate Evaluation

The reliability of GWLF-E discharge estimates was evaluated by comparing GWLF-E outputs with manually collected discharge data. Stream discharge was measured in transects during seven monitoring events in 2017 at 60% depth with a Marsh-McBirney Flow Mate 2000 velocity meter (Hach Company, Loveland, CO) attached to a top-setting wading rod. These events were May 30, June 15, June 20, July 13, July 25, August 24, and September 12. June 15 and July 13 were rain events, while the other samples were of baseflow. The GWLF-E model was run for 2017; average daily discharges for those sampling events were extracted from the results and divided by the number of seconds in a day (86,400) to get an estimate of discharge in $m^3 s^{-1}$. This estimate was averaged for each site and compared with the average discharge collected by the flow meter.

3.4 RESULTS

Water budget, field erosion, and streambank erosion outputs from the GWLF-E model were calculated for 20 subbasins (Table 3.1). Annual water budget results suggest that Indian Mill Creek is primarily a groundwater fed stream. Approximately 85 cm of precipitation falls in the watershed annually. Evapotranspiration removes between 16% and 23% of this water depending on the subbasin. The remaining water feeds the creek as either groundwater flow or runoff. Groundwater flow, the main source of water to the creek, contributes a per-subbasin average of 63% to 78% of the stream flow. The other 6% to 15% of the precipitation in subbasins becomes runoff and is quickly exported from the subbasins. Urbanized subbasins in the southern part of the watershed have the highest proportion of water becoming runoff, especially in subbasins of Brandywine Creek (Figure 3.5).

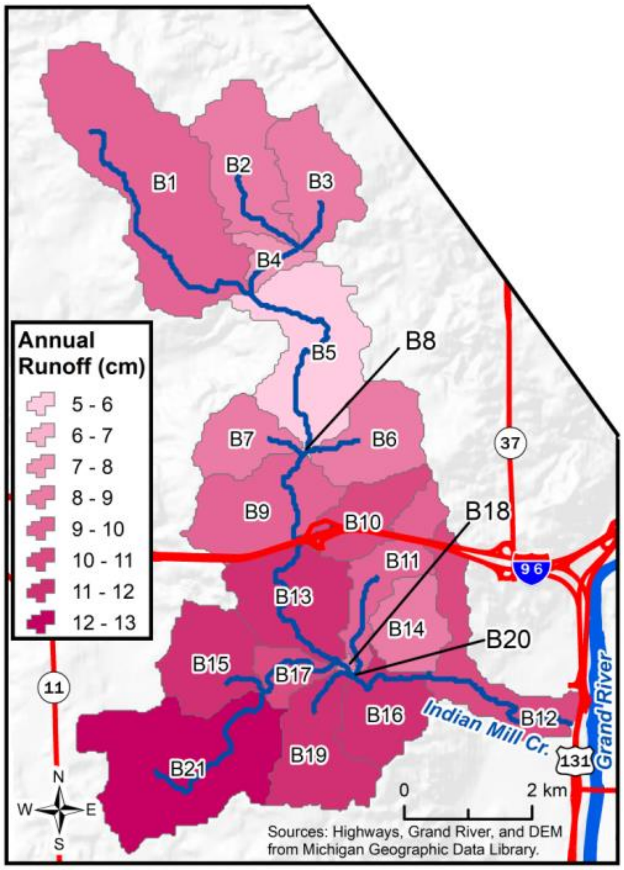
The sediment loading outputs of the GWLF-E model predict that the creek receives a total load of 6,109 Mg/yr of sediment from field and streambank erosion. Field erosion contributes an average by subbasin of 0.2 to 2.5 Mg/ha/yr of sediment to the creek. The greatest rates of field erosion occur in the middle and southern subbasins of the watershed (Figure 3.6). Streambank erosion contributes an average by subbasin of 0.2 to 508.6 Mg/yr of sediment to the creek, accounting for 0.2% to 50.1% of the subbasins’ sediment budgets (Figure 3.7). The lateral erosion rate of streambanks varied by subbasin from 0.04 to 7.37 cm/yr. Both the proportion of sediment load from streambanks and the lateral erosion rate increased in a downstream direction, with less erosion in the headwaters and more erosion in lower reaches (Figure 3.7, Figure 3.8). Total sediment loading varied by subbasin but was greatest in the lowest subbasin BC12 (Figure 3.9).

The evaluation of GWLF-E discharge estimates shows that they follow the same pattern as manually collected estimates of increasing discharge closer to the outlet of Indian Mill Creek (Figure 3.10). However, GWLF-E can overestimate discharge by a factor of up to 11.0 compared with collected discharge estimates in headwater subbasins like B21, and by a factor of 2.8 by the outlet of the creek (B12). Subbasin B11 had patchy data because of its ephemeral nature. This overestimate could be explained by the GWLF-E model not being calibrated to Indian Mill Creek or that it is predicting greater storage of water, leading to higher base flow estimates; implications of this are that the model is appropriate for assessing spatial distribution of erosion risk but could be less effective for numerical targets without calibration.

Table 3.1. Results from the GWLF-E model for 21 subbasins in the Indian Mill Creek Watershed 1997-2015. Subbasins B1 to B8 are in the upper watershed, dominated by agricultural land cover, and B9 to 21 are in the middle to lower watershed, dominated by urban land cover, with spatial reference in Figure 1.

Geography		**Annual Water Budget**				**Annual Field Erosion**		**Annual Streambank Erosion**		
Basin	**Area (ha)**	**Precipitation (cm/yr)**	**Evapotranspiration**	**Groundwater**	**Runoff**	**Mg**	**Mg/ha**	**Mg**	**% of sediment load**	**LER* (cm)**
B1	677	85	21%	69%	11%	924.4	1.4	9.5	1.0%	0.09
B2	247	85	22%	68%	10%	266.0	1.1	2.7	1.0%	0.06
B3	212	85	21%	69%	10%	199.8	0.9	0.9	0.5%	0.04
B4	57	85	20%	71%	9%	38.8	0.7	5.9	13.2%	0.22
B5	374	85	20%	74%	6%	579.5	1.5	21.3	3.6%	0.24
B6	200	85	21%	68%	11%	279.8	1.4	0.8	0.3%	0.04
B7	136	85	22%	68%	10%	176.3	1.3	0.7	0.4%	0.04
B8	1	85	16%	78%	6%	1.8	1.2	1.8	50.1%	0.61
B9	272	85	22%	67%	11%	539.8	2.0	43.6	7.5%	1.15
B10	151	85	21%	66%	12%	125.8	0.8	3.3	2.6%	0.20
B11	178	85	22%	67%	11%	275.0	1.5	6.4	2.3%	0.15
B12	238	85	21%	66%	13%	593.2	2.5	508.6	46.2%	7.37
B13	269	85	21%	65%	14%	505.9	1.9	271.4	34.9%	3.98
B14	145	85	21%	69%	11%	109.5	0.8	0.2	0.2%	0.11
B15	209	85	23%	63%	14%	128.9	0.6	1.5	1.1%	0.08
B16	180	85	21%	67%	13%	151.0	0.8	116.5	43.5%	3.49
B17	74	85	21%	67%	12%	47.7	0.6	10.1	17.5%	0.25
B18	6	85	16%	74%	10%	11.5	1.9	11.0	48.9%	1.53
B19	232	85	20%	67%	14%	39.5	0.2	1.6	4.0%	0.07
B21	518	85	21%	63%	15%	83.7	0.2	13.5	13.9%	0.19

*LER means lateral erosion rate in cm/yr.



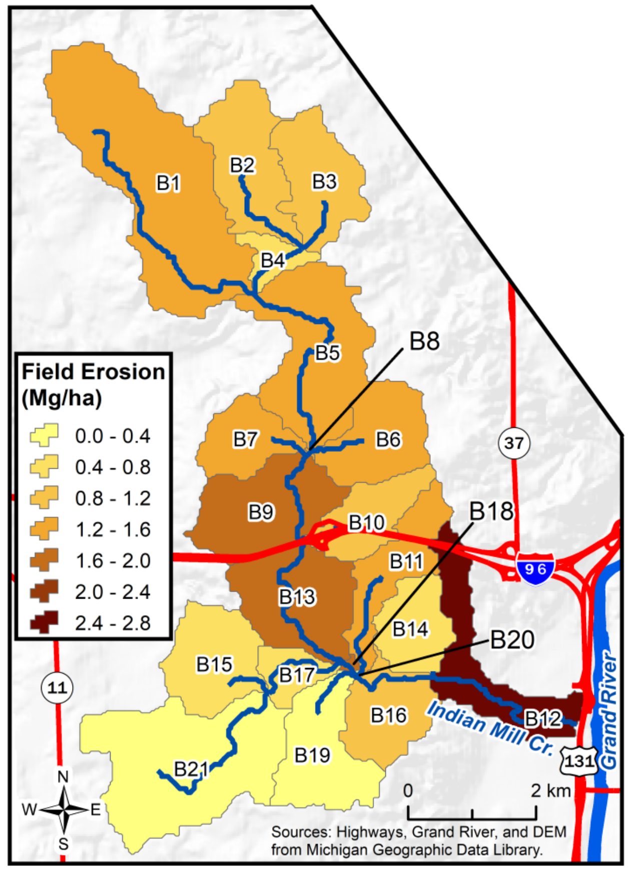


Figure 3.5. Annual runoff results from the GWLF-E model for subbasins in the Indian Mill Creek watershed 1997-2015.

Figure 3.6. Annual field erosion results from the GWLF-E model for subbasins in the Indian Mill Creek watershed 1997-2015.

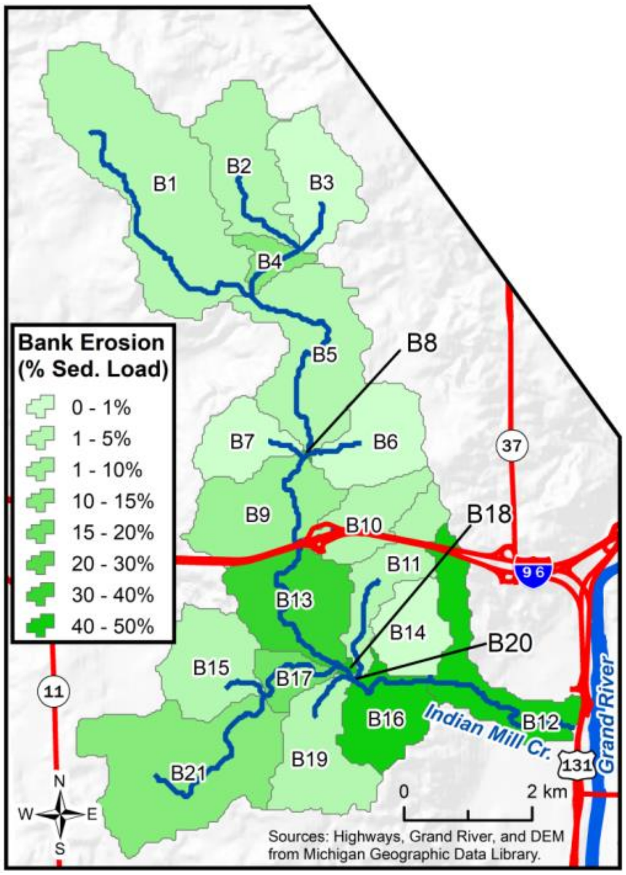


Figure 3.7. Percent of total sediment load from bank erosion from the GWLF-E model for subbasins in the Indian Mill Creek watershed 1997-2015.

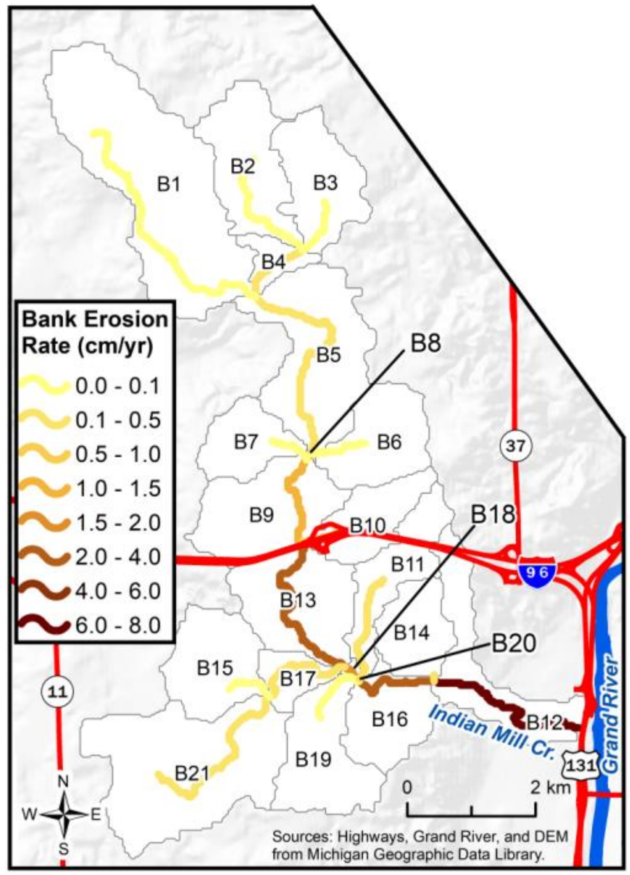


Figure 3.8. Lateral streambank erosion rates from the GWLF-E model for subbasins in the Indian Mill Creek watershed 1997-2015.

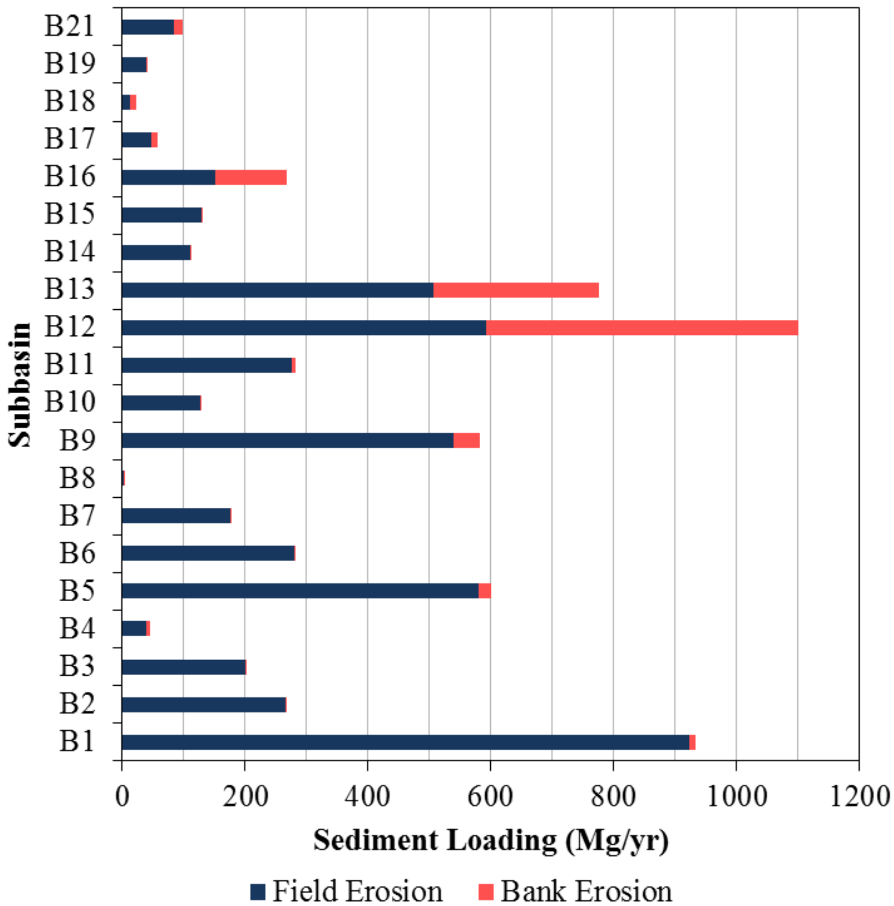
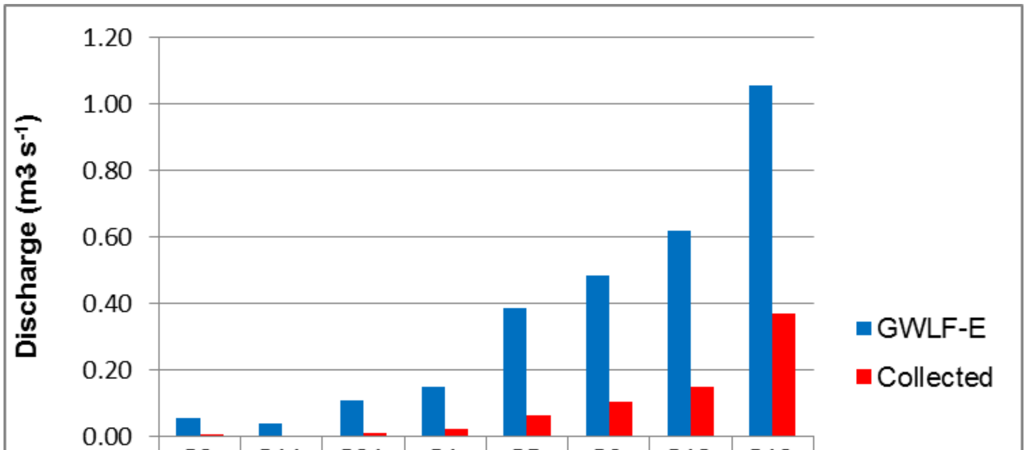


Figure 3.9. Total annual subbasin sediment loading from field and bank erosion from the GWLF-E model in the Indian Mill Creek watershed 1997-2015.



	B2	B11	B21	B1	B5	B9	B13	B12
GWLF-E	0.05	0.04	0.11	0.15	0.39	0.48	0.62	1.05
Collected	0.01	0.00	0.01	0.02	0.06	0.10	0.15	0.37

Figure 3.10. Evaluation of GWLF-E discharge estimation using manually collected discharge data, averaged in eight subbasins over seven monitoring events in 2017. B2, B11, and B21 are tributaries while B1 to B12 progress from headwaters to the outlet of Indian Mill Creek.

3.5 DISCUSSION

The transport of water and sediment through the Indian Mill Creek watershed is affected by a combination of soils, topography, land cover, and climate. Knowledge of these relationships is important for nonpoint source pollution management and predicting impacts from climate change. Relationships can be interrelated and complex; a watershed model can piece together their story and identify critical areas for nonpoint source pollution management. We used the GWLF-E model and MapShed plugin of MapWindow GIS to simulate the water budget, field erosion, and streambank erosion in 20 subbasins of the Indian Mill Creek watershed. To the best of our knowledge, this is the first time these have been used for a watershed study in Michigan.

We created a map with recommendations for each subbasin based on land cover data and proximity to the creek (Figure 3.11). We identify the following subbasins as critical areas for runoff, field erosion, and/or streambank erosion management and discuss management recommendations.

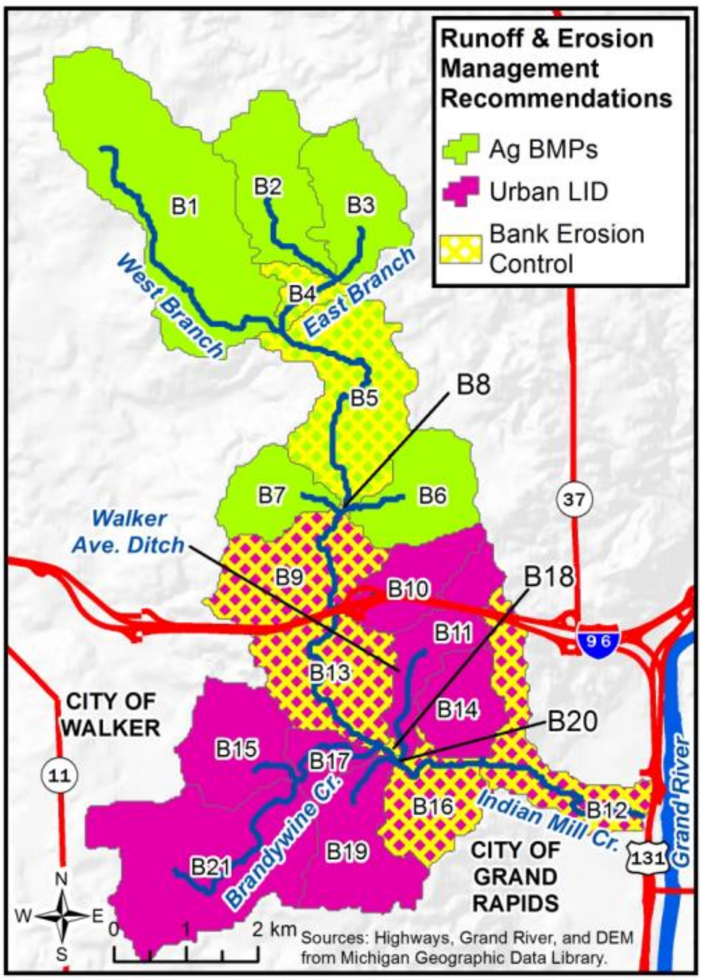


Figure 3.11. Runoff and erosion management recommendations including agricultural best management practices (Ag BMPs), urban low impact development (LID), and streambank erosion control for subbasins in the Indian Mill Creek watershed.

Runoff

The Brandywine Creek area in the southwest portion of the watershed proportionally contributed the greatest amount of runoff to the creek, and the least amount of groundwater to feed base flow. A very low base flow in mid-summer and evidence of powerful floods after rain storms were observed in the adjacent grassy floodplain (Figure 3.12). This flow regime is likely caused by loamy soils of the C and D hydrologic groups with high runoff potential plus the high amount of urban development and impervious surfaces in the subbasin, which cause increased runoff and decreased infiltration of water to the soil (Paul and Meyer, 2001). Subbasin B21, the headwaters of Brandywine Creek, should be a priority for projects that capture runoff and increase infiltration of precipitation into the ground, followed by B19, B15, B13, and B16. This reduction in runoff is vital to restoration of the watershed; restoring stream habitat and riparian conditions in urban streams can be ineffective for recovery of aquatic life if the impacts of intense stormflows are not addressed (Walsh *et al.*, 2005). Low impact development guidelines are available to help plan these projects (Southeast Michigan Council of Governments, 2008). Suggested low impact development projects in these subbasins include a reduction in impervious surface area, bioretention basins, detention basins, pervious pavement, wetland conservation, floodplain avoidance, and vegetated swales.

Figure 3.12. Flood-washed grass along Brandywine Creek in the Indian Mill Creek watershed, October 2017.

Field Erosion

We aimed to determine if agricultural areas in the upper watershed contributed the most sediment from field erosion. The GWLF-E model predicted that subbasins with high sediment loading from field erosion were spread throughout the watershed. Urbanized subbasins along the middle and southern areas of the watershed had the highest predicted per hectare rates of field erosion. This is different from what we expected and could be explained by a combination of steep slopes, erodible soils, and the urban land cover that increases the risk of field erosion in these subbasins (Lu *et al.*, 2004; Paul and Meyer, 2001; Wischmeier and Smith, 1978). However, agricultural subbasins in the upper watershed still contributed considerable sediment to the creek by field erosion and are no less important for nonpoint source pollution management. We identified critical areas for agricultural best management or urban low impact development projects to manage field erosion based on their per hectare contribution of sediment to the creek. Subbasin B12 should be a priority for field erosion management, followed by B9, B18, B13, and B5. Conservation Practice Standards should be implemented in agricultural lands to reduce field erosion. These practices, with Natural Resources Conservation Service guide in parentheses, include conservation cover (327), filter strips (393), residue and tillage management (329, 345), and riparian cover (390, 391). Field erosion results are consistent with Wagner et al. (2007), which modeled rates of 0.4 to 2.8 Mg/ha/yr in four Virginia catchments with the GWLF model. Our results are also similar to Kiesel et al. (2009), which modeled rates of zero to 3.5 Mg/ha/yr in a lowland German catchment using a German revision of the Universal Soil Loss Equation.

Streambank Erosion

We aimed to determine if urban subbasins in the lower watershed had a higher rate of streambank erosion because of increased runoff from impervious surfaces. The rate of sediment loading from streambank erosion modeled with GWLF-E followed a longitudinal pattern in the watershed. Streambanks in headwater subbasins experienced low lateral erosion rates; streambank erosion was thus a small fraction of the overall sediment load from these subbasins. The lateral erosion rate increased in a downstream direction along with the proportion of sediment loading from streambank erosion, predicting that urban areas in the lower watershed would thus have the highest erosion rates. This longitudinal pattern is an effect of the GWLF-E streambank erosion model, which relies on the effect of mean monthly discharge to calculate erosion rate (Evans *et al.*, 2003). Field-collected flow data by the authors from five dry and two storm sampling events in May to September 2017 confirm that there is a trend of increasing discharge from headwaters to mouth of the creek that could be affecting erosion rates (Figure 3.10). Thus, the stream corridor of the lower watershed should be a critical area for streambank erosion control. This corridor has the largest modeled streambank erosion rates because the subbasins have the strongest discharge. Erodible sandy soils in the streambanks observed in the lower watershed could also influence high erosion rates (Figure 3.13). The lowest subbasin B12 should be a priority, followed by B13, B16, B18, and B9. Subbasin B20, though it was too small to model, should be a priority as well because of its lower location along the stream and expected high lateral erosion rate. Bank erosion results are consistent with Kiesel et al. (2009), who measured bank erosion rates of 0.1 to 12.8 cm yr^{-1} in a lowland German catchment; Zaimes et al. (2005), who measured mean bank erosion rates of 0.7 to 5.1 cm yr^{-1} in Iowa, USA streambanks; and Laubel et al. (1999), who measured mean rates of 0.6 to 2.6 cm yr^{-1} in a Danish watershed.

Figure 3.13. Sandy eroding banks observed in subbasin B12 in the lower Indian Mill Creek watershed, April 2017.

Low impact development projects to reduce runoff should be the primary activities to reduce streambank erosion. These projects can minimize the physical disturbances to a stream by the erosive power storm flows, including bank erosion and incision (Walsh *et al.,* 2005). Additionally, a restoration plan for degraded riparian corridors in critical subbasins for streambank erosion control should be developed and implemented. Guidance is available from the Natural Resources Conservation Service's Federal Stream Corridor Restoration Handbook (part 653 of the National Engineering Handbook). This handbook includes approaches for streambank stabilization and stream channel restoration, such as plantings and geotextile systems, based on conditions of the stream corridor.

The GWLF-E didn't identify subbasin B11, the Walker Avenue Ditch, as a priority for streambank erosion. However, we have observed severe erosion occurring in B11 along with intensive sedimentation in the streambed, which appears to be among the worst in the Indian Mill Creek watershed (Figure 3.14). A concurrent study by the authors at nine sites in the Indian Mill Creek watershed will provide measurements of streambank erosion rates over the course of one year, May 2017 to May 2018, for a site in the Walker Avenue ditch and eight other sites in the Indian Mill Creek watershed. The purpose of this concurrent study is to compare four techniques for measuring streambank erosion: erosion pins, total station surveying, terrestrial laser scanning, and photogrammetry. The results will provide an empirical assessment of streambank erosion patterns in the watershed and see if it agrees with the upstream-downstream pattern of the GWLF-E model.

Figure 3.14. Severe incising and bank erosion observed in Walker Avenue Ditch (subbasin B11) of the Indian Mill Creek watershed, April 2017.

Nonpoint Source Pollution Management

A 2016 study (Sigdel, 2017) along with a concurrent study of stream habitat, fish, and aquatic invertebrates in the Indian Mill Creek watershed by the authors identified the hydrologic effects of runoff impacting the structure of fish and macroinvertebrate communities. These studies classified Indian Mill Creek as a coldwater stream by the criteria of Wehrly et al. (1999). The large inputs of cold groundwater from the GWLF-E outputs support this classification. The fish community assemblage of the creek was found to be driven largely by stream flow and temperature regimes, which are directly influenced by runoff during storms. Fine sediment in the streambed was found to be the strongest driver of degraded macroinvertebrate communities and associated with a low abundance and richness of EPT taxa. The cause of this fine sediment was explained by both the geomorphology of the creek (Dust and Wohl, 2012) and the effects of land cover along an agricultural to urban gradient. Sigdel (2017) also reports that an increase in stream discharge from impervious surfaces in the lower watershed increased the rate of streambank erosion and created excessive bed load sediment, which could be reducing the integrity of aquatic communities. Further, is unlikely stream habitat could be successfully restored without addressing the underlying hydrological issues caused by the runoff (Walsh *et al.*, 2005).

A strength of nonpoint source pollution management in the Indian Mill Creek watershed is cooperation among jurisdictions and other stakeholders. The City of Walker recently developed a Stormwater Asset Management Plan for their stormwater systems (www.walker.city. Accessed March 23, 2018). This plan defines the goals of nonpoint source pollution management as meeting regulatory commitments, minimizing the risk of flooding and other hazards, removing combined sewers, planning for community development, and protecting the quality of receiving waters. They also maintain a GIS database of all pipes, manholes, catch basins, ditches, and outfalls in their system. The City of Grand Rapids has a Stormwater Master Plan with the purposes of flood mitigation, reducing pollution and sedimentation, protecting the environment, and improving the quality of receiving waters (www.grandrapidsmi.gov/Home. Accessed March 24, 2018). Kent County has a model stormwater ordinance, and other resources for stormwater management (www.accesskent.com/). Alpine Township has partnered with the Grand Valley Metropolitan Council, Kent Conservation District, and US. Department of Agriculture to implement a Regional Conservation Partnership Program in the Indian Mill Creek watershed, with the purpose of installing and maintaining best management practices for water resource protection in the watershed’s farmland (www.lgrow.org. Accessed March 24, 2018).

Representatives from these jurisdictions, along with residents and other stakeholders, regularly participate in meetings and activities of the Friends of Indian Mill Creek to address issues in the watershed (www.lgrow.org/indian-mill-creek. Accessed March 24, 2018).

Alternative Models

Various models can be used to estimate streambank erosion rates in a watershed. However, they often require extensive field data collection. Here we summarize four additional streambank erosion models that could have been used for this study and their limitations. We chose the GWLF-E model because it fits our purpose of assessing erosion risk in subbasins throughout the watershed, has attainable data, and doesn't require extensive field data collection.

The Bank Stability and Toe Erosion Model (BSTEM) from the National Sedimentation Laboratory will predict streambank erosion and loading rates based on hydrology and field measurements (Midgley *et al.*, 2012; Simon *et al.*, 2011). BSTEM is built in a spreadsheet and can evaluate bank stability over changing hydrological conditions (Simon *et al.*, 2011). The model uses field data about channel geometry and soil properties, including jet soil tests (Midgley *et al.*, 2012), which could make it more intensive to implement on a watershed scale.

The Bank Assessment of Nonpoint Source Consequences of Sediment (BANCS) model is widely used for stream restoration and estimating sediment yields (McMillan *et al.*, 2018; Sass and Keane, 2012). It uses the qualitative visual Bank Erosion Hazard Index (Rosgen, 2001) to estimate bank erosion rates. It however relies on visual estimates and an evaluation of the model deemed it uncorrelated with actual erosion rates (McMillan *et al.*, 2018).

The Soil and Water Assessment Tool (SWAT) uses the critical sheer stress equation to estimate the sediment loading of bank erosion in a watershed (Mittelstet *et al.*, 2017; Narasimhan *et al.*, 2017). Similar to the GWLF model, SWAT uses spatial data about land cover, soils, weather and slopes. It also incorporates data about channel morphology collected in the field (Mittelstet *et al.*, 2017). The SWAT model has been shown to reasonably account for complex streambank factors and estimate erosion rates that are similar to field measurements (Narasimhan *et al.*, 2017). However, it is physically-based and thus more difficult to use, requires extensive calibration with field data, and requires more complex datasets, than the GWLF model (Markel *et al.*, 2006; Shoemaker *et al.*, 2005).

The Dickinson-Scott model is a regression equation that is used to estimate streambank erosion rates (Dickinson and Scott, 1979). This model uses soil erodibility, an agricultural intensity index, and a hydraulic stability index to estimate lateral erosion rates (Dickinson *et al.*, 1989). The Dickinson-Scott model was developed to assess streambank erosion in agricultural catchments of southern Ontario and modified for lowland catchments in Germany (Dickinson and Scott, 1979; Kiesel *et al.*, 2009). Limitations are that it does not account for flow regime, bank slope, and bank vegetation (Kiesel *et al.*, 2009).

Study Limitations

The GWLF-E model predicted sediment loading from streambank erosion throughout the Indian Mill Creek watershed. These outputs are useful for managing sediment from streambank erosion. We found the MapShed model and GWLF-E model understandable and easy to use. One of the benefits of the GWLF-E model is this simplicity. However, this also introduces uncertainty into model outputs. The model could be overestimating stream discharge in our study stream because it was validated for Pennsylvania watersheds (Evans *et al.*, 2003) and not our study stream. The GWLF-E streambank erosion model assumes a uniform discharge, lateral erosion rate, bank height, and soil bulk density for the entire length of stream in a basin (Evans *et al.*, 2003). The bank height and soil bulk density are default values of 1.5 m and 1,500 kg/m^3. These can vary in nature, so some stream segments won't fit model predictions. We observed that incised segments of lower Indian Mill Creek can have taller banks of two or more meters, while small tributaries can have much shorter bank heights of less than a meter. Additionally, GWLF-E assumes that streambank erosion occurs at all discharges. Other studies suggest that streambank erosion occurs only after the force of discharge passes a certain threshold called the soil's critical sheer stress (Mittelstet *et al.*, 2017; Narasimhan *et al.*, 2017). We also introduced uncertainty through our treatment of soils in the urban land type and gaps in soil data availability in the methods. These uncertainties are important to weigh with the model's ease of use.

Conclusion

The GWLF-E model was used on 20 subbasins of Indian Mill Creek to predict runoff, field erosion, and streambank erosion using the MapShed plugin of MapWindow GIS with the GWLF-E model. The outputs can help managers identify critical areas for restoration and prioritize projects to reduce nonpoint source pollution. We recommended critical areas for management of runoff, field erosion, and streambank erosion based on model outputs. The ease of use of MapShed and the GWLF-E model could make them fitting for other Michigan watershed studies as long as model limitations are considered. Future research needs include investigations of critical catchments to further understand their contribution of water and sediment to Indian Mill Creek and use of the model to track implication of best management practices into the future.

Supporting information

Additional supporting information may be found electronically by contacting the corresponding author (myersda@mail.gvsu.edu). MapShed input data, GWLF-E input files, and GWLF-E output files are in Myers et al - Supplementary Data.zip. These files can be used with MapShed and GWLF-E to replicate the study.

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CHAPTER IV: MEASURING STREAMBANK EROSION: A COMPARISON OF EROSION PINS, TOTAL STATION, AND TERRESTRIAL LASER SCANNER

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4.1 ABSTRACT

Streambank erosion is important to watershed managers because it can be the dominant entry pathway of sediment to streams and also damages aquatic habitat for fish and invertebrates, riparian areas, and infrastructure. Bank erosion is difficult to measure and quantify and both models and field methods are needed to assess the significance of this important source. Our objectives were to 1) evaluate and compare three techniques for quantifying sediment pollution from streambanks: erosion pins, total station surveyor, and laser scanning, 2) spatially assess streambank erosion rates in the Indian Mill Creek watershed of Michigan, USA, and 3) relate streambank erosion results with a modeling study of nonpoint source pollution in the Indian Mill Creek watershed. We used an ANOVA with random blocks, Pearson Test for correlations, and percent difference metrics to compare techniques. We were unable to detect significant differences between measurement techniques (df=2/23, F=0.457, p=0.639). Total station and laser scanner data were correlated (R^2=0.79, p=0.003), but neither erosion pins and total station

(R^2=0.26, p=0.330) nor erosion pins and laser scanner (R^2=0.16, p=0.330) were correlated. Percent differences in bank erosion rates between techniques were large, with an average difference of 650% between erosion pins and total station, 596% between the laser scanner and erosion pins, and 1,275% between the laser scanner and total station. Banks with heavy vegetation had significantly lower average laser scan coverage of the bank after vegetation filtering (11.75%) than other banks (32.5%, p = 0.047). The terrestrial laser scanner collected high resolution data on barren streambanks with a clear line of sight, but the coarser erosion pin or total station techniques are preferable for vegetated banks because of better coverage. Differing results between techniques could be due to a combination of vegetation, undercut banks, and resolution. We also found that streambank erosion rates vary spatially in the watershed, with the lower reaches experiencing net deposition of sediment on the banks, and the middle watershed and agricultural headwaters experiencing net erosion that contributes to sediment loading in the creek. We estimate that streambank erosion contributes 2,020 Mg yr^{-1} of sediment to Indian Mill Creek, which is 28.5% of the total sediment load. This research is important for watershed managers addressing the sources of fish and macroinvertebrate community impairments in Indian Mill Creek and other watersheds that are degraded by excessive sediment.

Highlights

- Terrestrial laser scanner collects high resolution data on streambank erosion.
- Terrestrial laser scanner has limited data coverage for vegetated banks.
- Erosion pins and total station collect coarser data but work well with vegetation.
- Indian Mill Creek experiences net deposition in lower reach and erosion upstream.
- Streambank erosion adds 2,020 Mg yr^{-1} sediment to the creek, 28.5% of total load.

Keywords

Streambank, erosion, lidar, laser, sediment, watershed

4.2 INTRODUCTION

Background

Sediment pollution is a major concern for streams throughout the United States (Allan, 2004). It causes widespread degradation of aquatic habitat and reduces suitability for fish and macroinvertebrate communities (Allan, 2004; Paul and Meyer, 2001; Raleigh et al., 1984). Although sediment pollution can enter a stream through many pathways, the dominant pathway is often streambank erosion (Fox et al., 2016; Kiesel et al., 2009). Streambank erosion is natural in streams, but can be accelerated when there are disturbances caused by changing watershed land use (Allan, 2004; Paul and Meyer, 2001; Rosgen, 1994). Successful management of sediment pollution in a watershed requires an understanding of sources and entry pathways (Kronvang et al., 1997). Understanding the dynamic nature of streambanks is important to shoreline landowners threatened by retreating banks, engineers, water quality managers, and geomorphologists (Pyle et al., 1997). It also is important for projects involving stream restoration and Total Maximum Daily Load development (Resop and Hession, 2010). One difficulty with managing sediment pollution is that it is hard to quantify sediment loading from streambank erosion (Evans et al., 1993; Fox et al., 2016). Various techniques could be used for this purpose including erosion pins, total station surveying, and terrestrial laser scanning.

Streambank Erosion Measurement Techniques

Erosion pins are narrow pins that are installed horizontally in streambanks to measure the retreat of the bank over time (Kiesel et al., 2009). They are commonly used in streambank erosion studies (Kiesel et al., 2009; Lawler, 1993). An advantage of erosion pins is that they are suitable for a wide range of fluvial environments; they are also cheap and simple to maintain with no special equipment (Lawler, 1993). However, erosion pins can have difficulty accounting for spatial variability on a streambank (Lawler, 1993). They also can contribute to false positive erosion estimates because of bank destabilization during pin installation or turbulence caused by the pin (Lawler, 1993).

A total station is an electronic surveying instrument that combines horizontal angle, vertical angle, and distance measurement to map a structure or terrain (Keim et al., 1999; Resop and Hession, 2010). Total station surveys have been effectively used to see how the shape of a streambank changes over time from erosion or deposition (Keim et al., 1999; Resop and Hession, 2010). An advantage of the total station is that it can very accurately measure the location of a point on the streambank (Resop and Hession, 2010). A total station can also have disadvantages when used to survey streambank erosion. Total station data can be coarse and lack the point density needed to accurately model bank retreat and conditions (Plenner et al., 2016). Data collection with a total station can cause disturbance to the streambank (Resop and Hession, 2010). Overhanging banks can make total station surveys difficult. It can be nearly impossible to collect data beneath overhanging and undercut banks using a total station; there are no standard methods to account for the empty space below the overhang on topographic maps. Undercut banks have previously been ignored because of this difficulty, which causes error in the data (Keim et al., 1999). This is important because streams through urban areas experience a stage of channel widening from increased storm flows and water velocities (Paul and Meyer, 2001), which could increase the prevalence of undercut banks.

A laser scanner is a surveying instrument that uses lidar technology to create high resolution scans of a surface showing three dimensional topography (Resop and Hession, 2010; Wang et al., 2013). Lidar works by combining laser-based distance measurements with precise orientation to model a surface in three dimensions (Alho et al., 2009). It has many advantages compared with other streambank erosion measurement techniques. A main advantage of laser scanning is that it can detect small erosion rates along a streambank, bluff, or gully with as high as one millimeter resolution (Day, 2012; James et al., 2007; Lisenby et al., 2014). This gives managers more of an ability to control sedimentation at a watershed scale by measuring small erosion rates spread over an extensive stream system (Day, 2012). Though the technique provides superior measurement precision, optical issues with water reflection (Milan et al., 2007) and collecting data through vegetation and crenulated surfaces (Day, 2012) must be recognized. Terrestrial laser scanners have difficulties with measuring heavily vegetated streambanks (Heritage and Hetherington, 2007; Resop and Hession, 2010). Data collected with a terrestrial laser scanner can have missing data because of vegetation and other natural obstructions; this data could be interpolated to fill gaps, but the interpolation could cause errors so should only be used as a last resort (Brodu and Lague, 2012). Vegetation and other obstructions can be removed by special computer programs that classify the point cloud data from a terrestrial laser scanner into different classes. However, the complexity of natural surfaces and size of data files make vegetation classification difficult (Brodu and Lague, 2012). These large data files are difficult to process on desktop computer (Day, 2012). Heritage and Hetherington (2007) recommend a field protocol for using a terrestrial laser scanner to study fluvial morphology. This includes positioning the scanner to minimize the shadowing of obstructions like trees and vegetation, place targets for alignment with variation in all three dimensions, and repeating scans from the same positions.

Prior Comparison Studies

Previous comparisons between techniques to measure streambank erosion have provided valuable insights into difference and error. Resop and Hession (2010) compared a total station and terrestrial laser scanner for measuring streambank erosion along an 11 meter streambank of Stroubles Creek, Virginia, USA with six readings over two years. The bank was bare, with little vegetation. Estimates of bank retreat rate were 0.15 m yr^{-1} with the laser scanner and 0.18 m yr^{-1} with the total station, thus a relative error of 20%. They found that the laser scanner was quicker to use and did not disturb the streambank like the total station. However, processing the laser scanner data was difficult because of the size and complexity of data files. By comparing data points between the two methods, they found a mean bank retreat difference of 0.018 m, standard deviation of 0.020 m, and that 63% of total station points were within 0.02 m of the laser scanner data. Estimates of volumes of soil erosion from streambanks between the two techniques had an average difference of 109%, with a range from 7% to 373%. The cause of these differences was likely because of the different resolutions of the total station and laser scanner. Aside from some instances where an undercut bank clearly affected total station data, Resop and Hession did not find any systematic differences between the results of the total station and laser scanner on their bare bank.

Day et al. (2013) compared a terrestrial laser scanner with analyses of georeferenced aerial photography for measuring erosion of bluffs in the Le Sueur watershed of Iowa, USA.

Eroding banks were digitized from aerial photographs for 243 bluffs, while laser scans were taken of 15 bluffs, and results were extrapolated to 480 bluffs. These bluffs were large enough be identified using 3 m resolution elevation data and with a height up to 160 m. The study found an average erosion rate of 0.020 m yr^{-1} with the laser scanner and 0.14 m yr^{-1} from aerial photographs. It also found an average difference of 36% between sediment loading measurements from the two techniques. Eltner et al. (2013) compared a terrestrial laser scanner with photogrammetry on an unmanned aerial vehicle (UAV) for measuring bank erosion in two European catchments. They found that the point clouds of the laser scan and UAV photogrammetry differed by an average 3.1 to 18.0 mm, depending on the camera and software used for photogrammetry. Although we did not interpret aerial photography or include photogrammetry data in our analysis, the findings of Day and Eltner are relevant because they demonstrate the comparability of laser scanning with traditional techniques. Ours is the first study to compare erosion pins, a total station, and a terrestrial laser scanner on the same banks.

Objectives

Our objectives were to 1) evaluate and compare three techniques for quantifying sediment pollution from streambanks: erosion pins, total station surveyor, and laser scanning, 2) assess the spatial distribution of streambank erosion rates in the Indian Mill Creek watershed of Michigan, USA, and 3) estimate the annual rate of sediment loading in the watershed from streambank erosion and compare with modeled estimates. This research benefits watershed managers in addressing fish and macroinvertebrate community impairments in Indian Mill Creek and other watersheds that are degraded by excessive sediment. An ability to better quantify erosional bank loss is also important for owners of houses, farms, sewer lines, roads, and other infrastructure along streams who need to realize how much bank they're losing to protect themselves from damages due to eroding banks.

4.3 METHODS

Site Design

Indian Mill Creek in Kent County, Michigan, USA (HUC 040500060504) is a tributary to the Grand River and is 18.5 km long with a 44 km^2 watershed. The creek resides in the Southern Michigan Northern Indiana Till Plains ecoregion, characterized by irregular plains, cropland, pasture, and oak/hickory/beech/maple forests (Omernik, 1987). The watershed land cover is predominately urban (43%) and agricultural (39%), with commercial and residential development in the lower watershed, natural and urban lands in the middle watershed, and farmland and orchards in the upper watershed (LGROW, 2011). This land cover pattern affects the distribution of erosion risk in the watershed. The National Weather Service classifies the area as a humid continental climate with distinct summers and winters and fairly even distribution of precipitation throughout the year (www.weather.gov). A total of 28.5 km of streams were identified in the watershed using a Geographic Information System (GIS). Nine sites were chosen for this study (Figure 4.1). Four sites were in the lower urbanized parts of Indian Mill Creek, three sites were in the upper farmland, and two sites were on tributaries. Within each property, an 18 meter section of stream was chosen, based on a balance between an open channel for laser scanning and being representative of the reach, and then split into the left and right banks while looking in a downstream direction. Erosion pins were installed at all eighteen banks (minimum = 4 pins, average = 7.3 pins, maximum = 20 pins per bank), total station surveys were performed at sixteen, and laser scanning was performed at ten. The reason that laser scans were performed at fewer banks is that we were limited by time and financial resources to scan ten banks, while we had greater liberty with erosion pins and the total station coverage. The presence/absence of undercut banks and heavy vegetation at each bank also was noted.

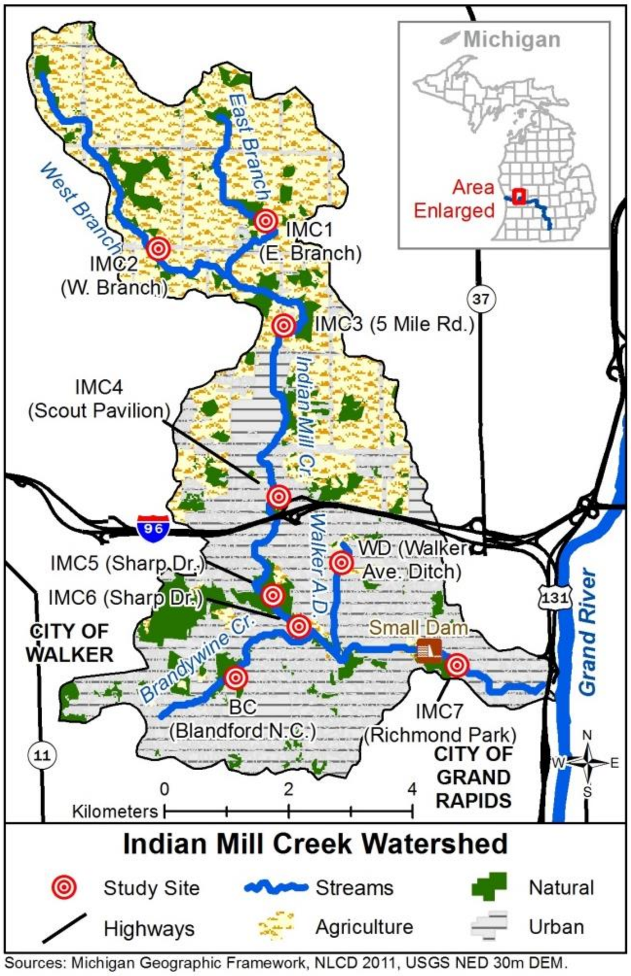


Figure 4.1. Study area map of the Indian Mill Creek watershed with features, land cover, and sites.

Erosion Pins

A total of 137 erosion pins were installed at the eighteen banks. Our design was based on those used in prior studies (Kiesel et al., 2009; Laubel et al., 1999; Lawler, 1993). Prior to installing erosion pins, the 18 meter stream section was divided into three six-meter subsections using a measuring tape. Erosion pins were carefully installed in the streambank in the middle of each subsection, on both banks. One to three pins were installed at each location evenly spaced up the bank, depending on the height of the bank. One erosion pin was placed for approximately every meter of bank height. Extra pins were installed if there were visible changes in erosion that were not otherwise captured by the design, such as the vertical transition between an undercut bank and vegetated slope.

Erosion pins were measured from the tip of the pin to the streambank using a measuring tape to the nearest 0.5 cm. The average of measurements from the top and bottom of the pin was used to account for bank slope. Where there was a horizontal angle to the bank, the left and right sides of the pin also would be measured and included in the average. Erosion pins were measured monthly from May to September 2017, with two additional measurements following rain storms, then April to May 2018. The spread of erosion pin data at each site was analyzed using R3.3.2 (R Core Team, 2016). Estimates of the volume of soil loss were calculated from the erosion pin data following methods of Palmer (2008) and Zaimes et al. (2005). Change in bank volume per meter of stream length was calculated for each bank at each site by multiplying the average erosion pin value by the bank height, taken from total station data. Overall change in volume of the study bank could be estimated by multiplying this rate by the 18 meter site length.

Total Station

The first step of the total station surveys was to set four control points at each site using a Trimble Geo7x Global Positioning System (GPS) with Zephyr external antenna. The purpose of these control points is to tie into the NAD 1983 UTM Zone 16N projected coordinate system and orient the total station. Control points were two foot rebar stakes driven into the ground and marked with orange tape or a cap. TerraSync 5.86 software was used to collect data. All GPS data were post-processed in Pathfinder Office using data from the Grand Rapids Continuously Operating Reference Station.

A Topcon GPT-3107W total station theodolite on tripod with SurveyPro software was used to survey streambank shape. The instrument would be set up on one of the control points and backsighted to the farthest point for the most accurate orientation. When the instrument needed to be moved, a temporary control point would be created by pushing a marker into the ground, and the previous point would be checkpointed to determine error during movement of the total station. To collect points, a reflector prism was used on top of a staff with bubble level. If there was an undercut bank that wasn't reachable, the horizontal distance between the prism staff and the back of the undercut was noted. However, data for undercut banks were not incorporated into erosion estimates because of the inability of our virtual model files to account for overhanging bank shape.

The site design for the total station surveys was based on methods of Keim et al. (1999) and Resop and Hession (2010). Seven transects were performed along each bank over the 18 meter site, at the 0, 3, 6, 9, 12, 15, and 18 meter marks. The 3, 9, and 15 meter marks coincided with erosion pin locations. In each transect, sideshots for the top of the bank and toe were collected. Then, two to three shots were taken evenly spaced along the bank, depending on its size and variability. These shots were taken at erosion pins during the 3, 9, and 15 meter transects, at the location where the pin met the streambank.

Total station data were exported as a CSV file to a computer using Windows Mobile Device Center 6.1 and imported into ArcMap software. Then, xy data was displayed and the data was exported as a shapefile. A separate file was created for each streambank using the Select tool of ArcToolbox. Then, a 3D TIN file was created using the Create TIN tool and Delaunay Triangulation. The TINs were cropped using the Delineate TIN Data Area Tool of 3D Analyst if superfluous data needed to be cleaned up to increase quality. The volume of soil gain or loss between the 2017 and 2018 TIN streambank models was then calculated using the Surface Difference Tool of 3D Analyst. This value was divided by the length of the study site to estimate change in volume per meter of stream per year.

Terrestrial Laser Scanner

One to two banks were surveyed at each site with a FARO Focus3D terrestrial laser scanner in 2017 and a Trimble TX8 scanner in 2018. These ten banks were chosen to try to incorporate representative conditions and have clear visibility for the scanner. Three survey markers were placed along the bank, as far apart as possible without sacrificing visibility. The lidar target spheres were placed on these markers. These markers act as control points, and were surveyed with the total station so the laser scan results can be projected in a Geographic Information System (GIS). To ensure that all three spheres were visible from the apex of the tripod, brush was pushed aside, cut with a knife or machete, held back, or sat on for the length of the survey.

Next, a preliminary low-quality scan was taken to adjust the horizontal and vertical scan limits to fit the desired area of the bank. Prior to the full scan, the resolution and quality were set to the desired levels. We used 1:1 resolution and 2x quality and color image for the FARO scans, and Level 3 quality for the Trimble scans. These levels were chosen because they were successfully used by the Annis Water Resources Institute previously (Kurt Thompson, personal communication) or recommended for the purposes of our study (Mark Tenhove, personal communication) as a balance between high quality data and manageable file size. Laser scans from both instruments were processed using CloudCompare software, although the Trimble scans first had to be exported to a compatible .LAZ format using Trimble RealWorks 10.4.3 software. The FLS plugin was used to import FARO files to CloudCompare. Excess data was cut out and scans were aligned by the target spheres. At the IMC7 and IMC1 sites, target markers disappeared over the year so the alignment incorporated sturdy points on wood or metal structures at the site, and manual alignment was needed for IMC7. The CANUPO plugin (Brodu and Lague, 2012) and veget_LongRange.prm filter (Lague et al., 2013) were then used to filter vegetation from the scans. This vegetation filter and resolution was chosen because it gave the most accurate classification of filters and resolutions we experimented with and was within the processing capabilities of our computer. Other filters we experimented with were otira_vegetsuper.prm and otira_vegetsemi.prm (Brodu and Lague, 2012), as well as vegetRangiCliff.prm and vegetTidal.prm (Lague et al., 2013) . Volume change of streambanks between 2017 and 2018 was calculated by bringing the scans back into Trimble RealWorks and using the Volume Calculation tool with horizontal difference and 10 cm resolution. The percent of laser scan coverage from these volume outputs was calculated by dividing the scan area occupied by bank in both 2017 and 2018, facing the bank directly and horizontally from the stream, by the total gridded area of the file. The difference in laser scan coverage between banks with and without heavy vegetation was analyzed using a Shapiro-Wilk test to confirm normal distribution ($p=0.110$ without vegetation, $p=0.547$ with vegetation), followed by a t-test in R 3.3.2.

Statistical Comparisons and Visualization

Statistical tests for differences and correlations were performed in R 3.3.2 using data from the ten banks that had laser scans. Prior to statistical analyses, the IMC6 right bank was removed because it was deemed an outlier for the laser scan tests, being 4.3 times higher than the second highest measurement, and affecting the normality of the data. Shapiro-Wilk Tests were used on the erosion pin, total station, and laser scanner volume change estimates to determine normality. Data from all three techniques were found to be normally distributed ($p = 0.977$, 0.964, and 0.746). Differences between techniques were tested using ANOVA with randomized complete block design, with estimates of erosion rate as values, techniques as groups, and sites as blocks. Plots of normal Q-Q and residuals vs. fitted values were interpreted and suggested that the ANOVA was appropriate to use over data transformations or nonparametric alternatives. A similar ANOVA test was used by Purvis and Fox (2016) to analyze the influence of riparian buffers and time period on erosion rates. Correlations between techniques were tested using Pearson Tests with Holm p-value adjustments for multiple comparisons. Percent differences between volume results of the laser scanner and total station techniques were calculated following the methods of Resop and Hession (2010), who took the difference between laser scan and total station results, then divided it by the laser scan result. We calculated the percent difference for laser scan and erosion pin results, and for erosion pin and total station results, in the same fashion. The IMC4 (L) bank was removed from the percent difference analysis because it was an outlier with high total station error and less than 1% laser scan coverage after vegetation filtering. A dot chart created in R 3.3.2 was used to visualize bank erosion results between sites and techniques.

Basinwide Estimates

Basinwide estimates of sediment loading from bank erosion in the watershed were calculated separately from erosion pin, total station, and laser scanner data. These were calculated by multiplying the bank erosion rate per meter of stream length ($m^3\ m^{-1}\ yr^{-1}$) by the entire length of streams in the watershed (28,500 m) by an average soil bulk density of eroding streambanks 1,500 kg $(m^3)^{-1}$ (Evans et al., 2003). We used erosion pin data to compare basinwide estimates with other studies because the erosion pins had more sites and were versatile with no limitations in coverage.

4.4 RESULTS

Site Conditions, Erosion, and Deposition

Our study documented streambank conditions, volumetric changes using three erosion measurement techniques, and coverage of the laser scan data (Table 4.1). Negative bank volume change represents net erosion over the study period, while positive change represents net deposition. NA's exist in total station and laser scanner data where a bank was not surveyed for logistical reasons. There was no discernable relationship between undercut banks and total station results biased toward deposition. This could be because the bias from undercut banks was relatively small compared to the spread of total station data.

Statistical Comparisons between Techniques

The ANOVA showed that there were no detectable differences between streambank erosion measurement techniques (df=2/23, F=0.457, p=0.639). Correlation tests found no significant correlations between erosion pin and total station data (R^2=0.26, p=0.330) or erosion pin and laser scanner data (R^2=0.16, p=0.330; Figure 4.2). However, there was a significant correlation between total station and laser scan data (R^2=0.79, p=0.003).

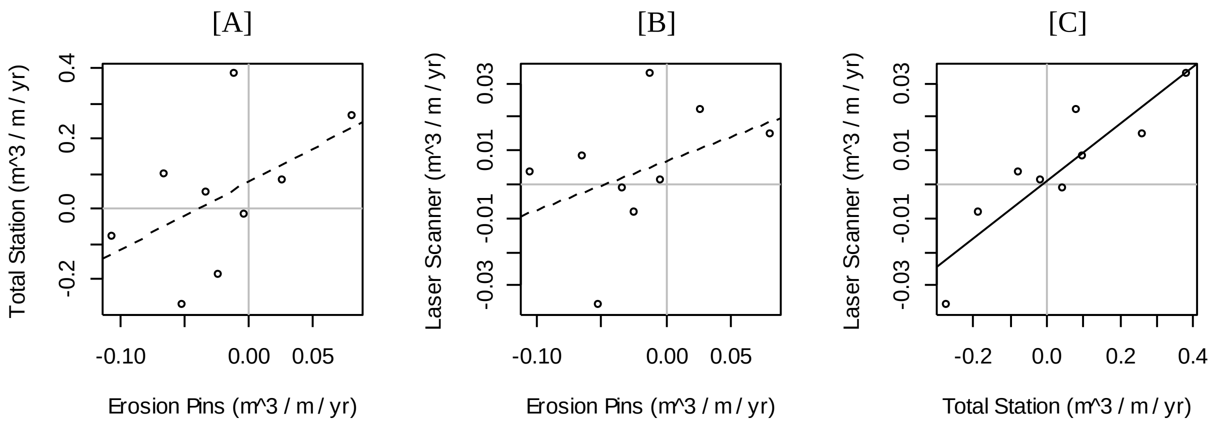


Figure 4.2. Correlations of bank volume change rate estimates between [A] erosion pins and total station (R^2=0.26, p=0.330), [B] erosion pins and laser scanner (R^2=0.16, p=0.330), and [C] total station and laser scanner (R^2=0.79, p=0.003) for nine sites in the Indian Mill Creek watershed. Solid line indicates significant correlation.

Vegetation Filtering

The terrestrial laser scanner performed well on barren streambanks with a clear line of sight. It was able to collect high resolution, high quality data for these banks with little hassle. However, for vegetated streambanks or where the streambank is otherwise obscured, the laser scanner had large data gaps. Banks with heavy vegetation had significantly lower average laser scan coverage after vegetation filtering (11.75%) than other banks (32.5%, p = 0.047). These banks were most common in the agricultural headwaters, which had been cleared of woody vegetation and thus had substantial growth of herbaceous plants, even in the spring when we surveyed. Laser scanner data could thus be underestimating change in bank volume because potential erosion of banks behind vegetation, roots, and other obstructions was not accounted for. This is especially true at the IMC4 (L) bank (Figure 4.3 G), where only 0.5% of the bank had coverage. This site was characterized by large masses of roots and overhanging vegetation that obscured the bank and were removed by the vegetation filter. The low rate of volume change for this bank could be an effect of the low coverage because the data gaps make it unclear from laser scanner data what change in bank shape is occurring under the vegetation. The ability for the laser scan and vegetation filter to produce high coverage along vegetated streambanks is a significant limitation of the technique. As far as we know, there is no standard for when coverage becomes too small to reliably use laser scan data. The site with the highest percent laser coverage, IMC6 (R), was a steep bank under forest canopy that was mostly clear of small vegetation growth and other obstructions. The IMC7 (R) bank was assigned a classification of no heavy vegetation because open banks were observed; however, patches of shrubbery and exposed roots could still be responsible for the low laser scan coverage. NA's exist in laser scan coverage data where a bank was not surveyed for logistical reasons.

Table 4.1. Site Conditions, volumetric results, and laser scan coverage for study streambank in the Indian Mill Creek watershed.

	Conditions		Change in bank volume ($m^3\ m^{-1}\ yr^{-1}$)			
Site (Bank)	**Undercut Banks**	**Heavy Vegetation**	**Erosion Pins**	**Total Station**	**Laser Scanner**	**Laser Coverage (%)**
IMC7 (L)	No	No	0.081	0.264	0.015	21.4%
IMC7 (R)	No	No	0.027	0.081	0.022	29.8%
IMC6 (L)	Yes	No	-0.004	-0.065	NA	NA
IMC6 (R)	Yes	No	-0.082	-0.111	0.155	60.1%
IMC5 (L)	Yes	No	-0.105	-0.078	0.004	24.4%
IMC5 (R)	Yes	No	-0.065	0.098	0.008	38.6%
IMC4 (L)	Yes	Yes	-0.034	0.047	-0.001	0.5%
IMC4 (R)	No	No	0.078	0.424	NA	NA
IMC3 (L)	No	No	-0.070	NA	NA	NA
IMC3 (R)	No	No	-0.048	NA	NA	NA
IMC2 (L)	No	Yes	-0.003	-0.018	0.001	5.6%
IMC2 (R)	Yes	No	-0.066	-0.111	NA	NA
IMC1 (L)	Yes	Yes	-0.034	-0.055	NA	NA
IMC1 (R)	Yes	Yes	-0.052	-0.273	-0.036	11.9%
WD (L)	No	Yes	0.003	0.046	NA	NA
WD (R)	Yes	Yes	-0.024	-0.186	-0.008	29.0%
BC (L)	No	No	-0.016	0.100	NA	NA
BC (R)	Yes	No	-0.011	0.383	0.033	20.5%

Figure 4.3. Photos of the 18 study streambanks in the Indian Mill Creek watershed, labeled by figure letter, site name and left (L) or right (R) bank. Photos [A] through [H] are in the lower watershed through urban and forested land cover, [I] through [N] are in the upper watershed through farmland, and [O] through [R] are along tributaries.

Comparative Analyses of Techniques and Sites

The dot chart shows that study streambanks experienced net deposition (positive volume change), net erosion (negative), little change in bank volume (points near zero), or a mixture depending on the technique (Figure 4.4). Sites are labeled with name and left (L) or right (R) bank and are ordered from lowest reach (IMC7) to headwaters (IMC1), followed by the two tributary sites. The presence of heavy vegetation (HV) or undercut banks (UB's) is noted under the site name to visualize the effects of these conditions on estimates of bank volume change. The following analysis of the chart is split into lower watershed, upper watershed, and tributary sites. Percent differences between techniques were substantial, with an average difference of 650% between erosion pins and total station data, 596% between the laser scanner and erosion pins, and 1,275% between the laser scanner and total station (Table 4.2). Bank photos are presented for reference in Figure 4.3.

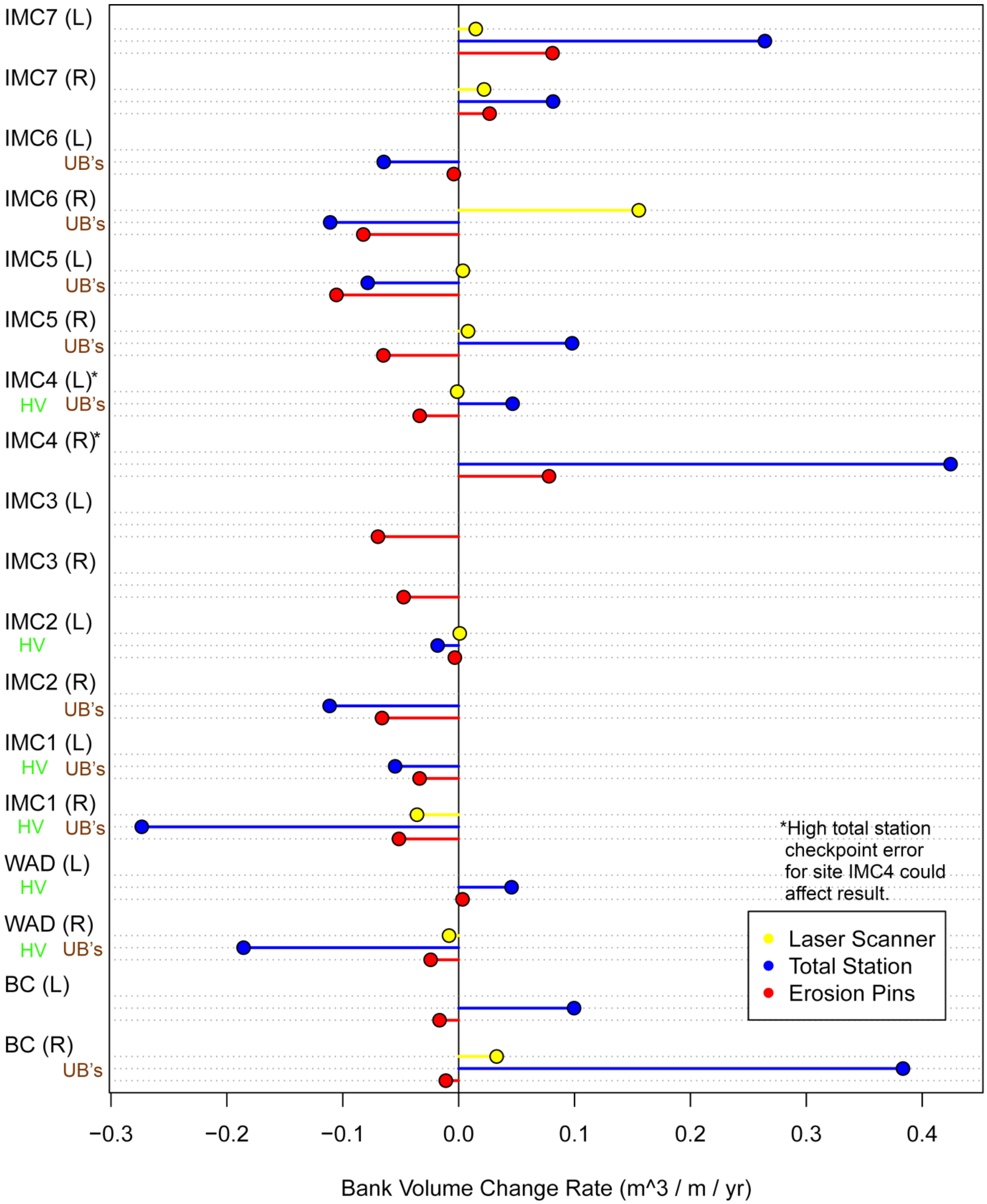


Figure 4.4. Comparison of results from techniques used to measure streambank erosion in the Indian Mill Creek watershed 2017-2018. Positive values indicate net deposition while negative values indicate net erosion being measured. Presence of heavy vegetation (HV) or undercut banks (UB's) is noted under site names.

Table 4.2. Percent difference in volume results for techniques to measure streambank change in the Indian Mill Creek watershed, calculated only for sites that had all three techniques used, following methods in Resop and Hession (2010). Reference Table 4.1 for absolute values.

Site (Bank)	Erosion Pins and Total Station	Laser Scanner and Erosion Pins	Laser Scanner and Total Station
IMC7 (L)	226%	449%	1,692%
IMC7 (R)	205%	22%	271%
IMC6 (R)	35%	153%	171%
IMC5 (L)	26%	3,003%	2,260%
IMC5 (R)	251%	904%	1,111%
IMC4 (L)	238%	2,511%	3,715%
IMC2 (L)	448%	466%	2,106%
IMC1 (R)	430%	43%	661%
WD (R)	668%	191%	2,136%
BC (R)	3,559%	134%	1,070%

Lower Watershed Sites (IMC7, IMC6, IMC5, and IMC4)

Sites in the lower watershed experienced either a positive volume change (deposition) or negative (erosion) depending on the bank and technique. Erosion pins, total station, and laser scanner all documented deposition of sediment at both IMC7 banks (Figure 4.3A and B), although there was considerable percent difference between rates. Erosion pin and total station results were similar for the IMC6 (L) site, showing only slight bank erosion. At the IMC6 (R) site, the laser scanner measured high deposition of sediment on the bank, while the total station and erosion pins both measured substantial erosion. The high value of the laser scanner measurement at this site caused the percent differences to still be under 200%, as it is in the denominator of the calculation. One explanation for the difference in measurements here could be that erosional areas were shadowed by leafy shrubs at the site, creating a gap in the laser scanner data.

At both IMC5 banks, the laser scanner documented very little change in bank volume, even though there was substantial undercutting and slumping along both banks (Figure 4.3 E and F), documented by erosion pins. The total station estimated a bank change rate of about -0.08 m^3 m^{-1} yr^{-1} for the IMC5 (L) bank, which is fairly consistent with erosion pin data (26% difference). However, for the IMC5 (R) bank, the total station estimated nearly 0.1 m^3 m^{-1} yr^{-1} of deposition on the bank, which was a 251% difference. A likely reason for the disparity is that the entire right bank is undercut and the lip of it has been pushed up in places; erosion pins were still able to collect data in the undercut, but the total station with TIN file format was only able to collect data on the top of the bank surface.

At the IMC4 (L) bank, erosion pins estimated slight erosion, while the total station estimated slight deposition. The difference between these estimates could once again be the undercuts that extend the entire length of the study bank (Figure 4.3 G). The erosion in these undercuts is measurable by the erosion pins, but the total station technique only collects data above the lip, missing the erosion underneath. The very low percent of laser scan coverage because of roots and vegetation (0.5%) likely explains the low estimate of bank change from the laser scanner. The IMC4 (R) bank had a disparity where the total station predicted very heavy deposition of sediment on the bank, but the erosion pins only measured slight deposition. An explanation for this difference is that erosion pins are limited in their ability to measure change when there is a pile of sediment dumped on the bank (Figure 4.3 H) that the total station can effectively map the surface of.

Upper Watershed Sites (IMC3, IMC2, and IMC1)

Agricultural sites in the upper watershed experienced primarily erosion along the banks. The IMC3 site had a decent amount of bank erosion measured by erosion pins along both banks (Figure 4.3 I and J). At this site, a constricting culvert under a driveway, large willow that has fallen across the creek, and runoff from agricultural areas upstream could explain the erosion that is occurring by altering the local and watershed's hydrology to scour the banks. At the IMC2 site, the left bank had pretty consistent measurements of bank change between all three techniques, showing slight erosion, although it is likely that the low estimates of bank erosion inflated the percent difference between techniques. This low erosion rate makes sense because the site is along a field but with a vegetated riparian buffer of approximately ten meters to protect the banks. The erosion pins and total station estimates for the IMC2 (R) bank both show erosion occurring. This bank was along a lawn with no riparian buffer and was visibly eroding (Figure 4.3 L).

The IMC1 site was also experiencing visible erosion that was documented by all three techniques at the right bank, and both techniques used at the left. The total station estimated a much higher erosion rate at the right bank than the laser scan and erosion pins (difference of 661% and 430%), which could be because of the resolution and coverage of the data. This bank was heavily vegetated and had low laser scan coverage. Differences could also be due to the shape of the bank, which was complex with many bends, slumps, and a couple large barren areas (Figure M and N). Differences could also be affected by a high checkpoint error documented from the total station, possibly due to unstable soil conditions for the tripod (see Estimates of Error section).

Tributary Sites (WD and BC)

Small tributaries in the watershed experienced a mixture of erosion and deposition. The WD site was along a meander, which explains why the left bank inside the bend had measured deposition, while the right bank on the outside of the bend had measured erosion (Figure 4.3 O and P). The total station could have estimated more erosion for the WD (R) bank than the pins and laser scans (differences of 668% and 2,136%) because the laser scans had data gaps, likely due to shrubs and herbaceous vegetation that got in the way, and because the erosion pins had lower resolution data that could have missed eroding areas. The BC site banks had erosion from the pin data, but substantial deposition estimated by the total station. The laser scan on the right bank showed minor deposition. We observed deposition of sediment on the bed of Brandywine Creek and the toe of the banks at the BC site during the study, as well as evidence of powerful flows during storms that pushed down grass in the floodplain (Figure 4.5). Undercut banks could also explain why the total station estimated more deposition and increased volume of the banks (Figure 4.3 Q).

Figure 4.5. Evidence of high flows in Brandywine Creek floodplain in the Indian Mill Creek watershed, with rain gauge in foreground.

Estimates of Error

Total station end checkpoint error data show that the measurements are likely to vary on the order of millimeters or a few centimeters (Table 4.3), with an absolute average of 5.5 cm (standard deviation 11.7 cm). The high 2018 checkpoint elevation error introduces uncertainty into the total station results for the IMC4 site. We presume that this error occurred because the tripod was set in soft muddy soil that was not the most stable, causing the instrument to tilt during the survey. It could also have been from a recording error because both the northing and easting error were small. Laser scanner alignment error had an average of 0.7 cm (standard deviation = 0.4 cm), suggesting that the data between years typically errs by less than a centimeter.

Table 4.3. End checkpoint error data from the total station surveys showing how much the instrument erred between the beginning and end of a streambank survey, along with alignment error from laser scanner targets in the Indian Mill Creek watershed 2017-2018. The IMC4 site was measured with only erosion pins so is not included.

	Checkpoint Error 2017 (m)			**Checkpoint Error 2018 (m)**			**Laser Target Alignment Error (m)**		
Site	*Northing*	*Easting*	*Elevation*	*Northing*	*Easting*	*Elevation*	*1*	*2*	*3*
IMC7	0.008	0.006	0.064	0.009	0.011	-0.006	0.008	0.010	0.002
IMC6	0.004	0.001	-0.004	0.392	-0.016	-0.005	0.006	0.004	0.009
IMC5	No data	No data	No data	-0.242	-0.356	-0.008	0.005	0.009	0.002
IMC4	-0.012	0.021	-0.001	0.006	0.038	-0.577	0.005	0.011	0.006
IMC2	-0.002	-0.012	-0.007	-0.023	-0.023	-0.021	0.002	0.002	0.002
IMC1	0.004	-0.001	0.004	0.020	-0.050	0.084	0.013	0.004	0.011
WD	0.022	0.022	-0.014	-0.034	-0.012	0.024	0.014	0.012	0.011
BC	No data	No data	No data	0.054	-0.018	-0.051	0.011	0.011	0.011

Basinwide Estimates

Overall, an average bank volume change rate of -0.024 m^3 m^{-1} yr^{-1} was estimated from erosion pin data (Table 4.1), with a standard deviation of 0.049. Both the total station and the laser scanner were more preferential toward deposition of sediment on streambanks, with an average bank volume change of 0.034 and 0.019 m^3 m^{-1} yr^{-1}, and standard deviation of 0.187 and 0.049. The high standard deviation and bank change rate from total station data is due in part to the right bank of the IMC4 site (Figure 4.3 H). This bank is the inside of a meander bend; we witnessed heavy deposition of sediment on the bank that makes the estimate not seem unreasonable. This deposition was also documented with erosion pin data, though not as heavily. The results of the laser scanner showing deposition of sediment on most banks could be due to data gaps from vegetation and other obstructions that shadowed eroding areas.

Assuming the average erosion rate of our eighteen study banks from erosion pin data (0.024 m^3 m^{-1} yr^{-1}) represents the average bank erosion rate for the 28.5 km of streams of the Indian Mill Creek watershed, we estimate from erosion pin data that bank erosion contributes 1,346.5 cubic meters of sediment per year to Indian Mill Creek. Multiplying by an average soil bulk density of eroded sediment of 1,500 kg/m^3 (Evans et al., 2003), we estimate that streambank erosion contributes an annual load of 2,020 Mg of sediment per year to Indian Mill Creek.

4.5 DISCUSSION

Comparison of Techniques

We evaluated and compared three techniques for measuring streambank erosion: erosion pins, total station, and terrestrial laser scanner. We were unable to detect significant differences between measurement techniques, and found a significant correlation only between total station and laser scanner data. Percent differences between techniques were large. Thus, when designing a streambank erosion study, results between different techniques of measuring bank erosion could have limited comparability, and thoughtful selection of a technique becomes very important depending on riparian conditions.

Our results show that selection of a streambank erosion measuring technique should be dependent on the resources available, desired resolution of data, and site conditions. Terrestrial laser scanning has high resolution and can detect small erosion rates with sub-centimeter error, especially on open streambanks with little vegetation. The scanner itself is easy to use, requiring little more than the press of a couple buttons to take a high resolution scan. However, the cost of the laser scanner would make it unusable for many watershed studies. Additionally, training with special point cloud processing software, and ideally Geographic Information Systems, is necessary to process the laser data. This software often requires computers that are more powerful than the typical home desktop. The terrestrial laser scanner performed well on barren streambanks with a clear line of site, such as the right bank of IMC6 that had the highest coverage of bank area. However, there were large data gaps and limited coverage when vegetation or other obstructions obscured the bank. This lack of coverage introduces uncertainty into the estimates of bank erosion because it is unclear how the bank is changing behind the vegetation. We recommend using the laser scanner only for bare banks with limited vegetation cover. If vegetated banks must be scanned, we recommend scanning them in early spring directly after snowmelt before vegetation has become established. We do not recommend removing vegetation from the banks because this could affect bank stability.

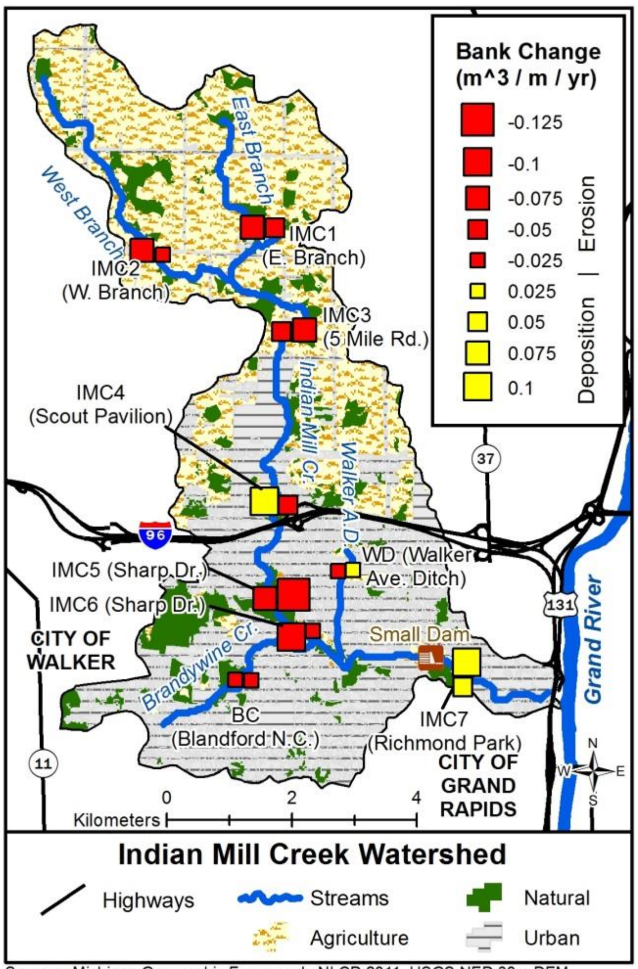
The total station or erosion pins are preferable techniques for vegetated banks. The pointed staff and reflector of the total station allowed us to collect data for points obstructed by vegetation. Similarly, erosion pins can be installed and measured on vegetated banks without loss of data. In general, erosion pins are the cheapest and easiest technique to measure streambank erosion. They can be installed and monitored for $1-2 per pin and do not require expensive equipment or familiarity of special software. However, they provide very low spatial resolution, as our transects were spaced three meters apart with approximately one pin per meter bank height. We also observed that there can be minor destabilization of the bank while installing and checking the pins. The total station works effectively for barren or vegetated streambanks. However, it requires skill with surveying, familiarity with the instrument and special software, and may not always be available to watershed groups. Additionally, minor bank destabilization can occur when using the staff and prism to collect data.

The total station does not work effectively for undercut banks using the methods we performed, ignoring the space under the overhang in its entirety. Undercut banks were documented at the IMC6, IMC5, IMC4, IMC2, IMC1, WD, and BC sites. Although it is unclear how much they affected erosion estimates as a whole, these undercuts shifted total station data at these sites toward deposition because the undercutting erosion was ignored in the TIN model. Total station results also had a larger spread of data than the other techniques. While results from the laser scanner and erosion pins tended to show change less than 0.1 m^3 m^{-1} yr^{-1}, the total station results were more variable, estimating changes in bank volume up to 0.2 to 0.4 m^3 m^{-1} yr^{-1} toward erosion or deposition (Table 4.1, Figure 4.3). The BC site right bank, IMC7 left bank, and IMC4 right bank all had high deposition documented with a total station that was not consistent with laser scanner and/or erosion pin results. The lack of erosion measurements from undercut banks could contribute to this deposition bias. On the other hand, the IMC1 right bank and WD right bank had relatively high erosion rates from total station data. An explanation for these rates could be from heavily eroding banks that were measured with the total station, but could have been between erosion pin transects or hidden from the laser scanner behind vegetation. Ultimately, the choice of technique for measuring streambank erosion should depend on the goals of the project and the resources available.

Resop and Hession (2010) noted that measurement of bank erosion can involve large errors and uncertainty. They did not find any systematic differences between results of total station surveys and laser scans, aside from some instances where the total station could not collect data beneath an undercut bank. Our study supports this, as the ANOVA was unable to detect significant differences between the laser scanner, total station, and erosion pins. Resop and Hession found that volumes of soil erosion from their study streambank estimated by the total station and laser scanner had an average difference of 109%, with a range from 7% to 373%. That is much smaller than what we experienced between the laser scanner and total station, which had an average difference of 1,275% with a range from 171% to 2,260%. Vegetation and other complexities along our banks are likely responsible for this greater range of differences; the bank that Resop and Hession studied was bare, with little vegetation.

Spatial Distribution of Bank Erosion

We assessed the spatial distribution of streambank erosion rates in the Indian Mill Creek watershed. The lower watershed experienced net deposition of sediment along the banks (Figure 4.6), as noted by researchers who observed heavy sand deposition on the IMC7 banks. The IMC4 site in the middle watershed experienced erosion on the left bank but deposition on the right. This is likely because the site was along a meander bend, with the outside on the left and inside on right. The WD site, also along a meander, experienced erosion on the right bank but deposition on the left. All other sites experienced net bank erosion and contributed to sediment loading in the Indian Mill Creek watershed. The highest rates of bank erosion from erosion pin data were at the IMC6 and IMC5 sites, which are along a high gradient reach of the creek as it descends the Grand River valley.



Sources: Michigan Geographic Framework, NLCD 2011, USGS NED 30m DEM.

Figure 4.6. Spatial distribution of erosion (red) and deposition (yellow) rates for study streambanks using erosion pin results.

Estimation of Sediment Loading

We estimated the total load of sediment entering Indian Mill Creek from streambank erosion and compared it to results from a concurrent study of field and streambank erosion rates (See Chapter III and Figure 3.8). This concurrent study used the Enhanced Generalized Watershed Loading Functions (GWLF-E) model (Evans et al., 2003) for the time period 1997-2010. The model predicted that average annual sediment loading from streambank erosion in the Indian Mill Creek watershed during that time period is 1,031.3 Mg/yr, while annual sediment loading from field erosion is 5,077.9 Mg/yr. Our estimate of the contribution of sediment loading to Indian Mill Creek from the erosion pin data was 2,020 Mg yr^{-1}. This is roughly double the streambank erosion predictions of the GWLF-E model. The difference between our estimate and modeled predictions could be because the GWLF-E model was validated by watersheds in Pennsylvania that could have different conditions than Indian Mill Creek. Stream discharge data collected with a flow meter suggest that GWLF-E, although not calibrated to Indian Mill Creek, follows the same pattern of increasing discharge toward the outlet of the creek, but may be overestimating discharge in subbasins by a factor of 2.8 to 11.0. The difference could also be that our eighteen study banks sample only a small proportion of the overall length of bank in Indian Mill Creek, which introduces uncertainty into the estimate. We decided not to use results of estimates of sediment loading in the watershed from the total station and laser scanner because they incorporated fewer sites than erosion pins and had more uncertainties due to undercut banks, issues of the tripod on squishy soil, and bank coverage. Both these techniques estimated an average bank volume change in the watershed that was positive, suggesting that more sediment is deposited on banks in the watershed than is removed by erosion, which seems unlikely and could be an effect of the uncertainties and limitations of the techniques. Our best estimate of sediment loading from bank erosion in relation to the GWLF-E field erosion estimate suggests that streambank erosion contributes 28.5% of the annual total sediment load to Indian Mill Creek. This is a substantial portion of the sediment load and is almost certainly affecting the quality of aquatic habitat, fish, and macroinvertebrate communities in the Indian Mill Creek watershed.

Previous studies have demonstrated that streambank erosion can be a large source of sediment loading in a watershed, though there can be a considerable degree of variability between watersheds (Sekely et al., 2002). Kiesel et al. (2009) estimated for a lowland catchment in Germany that 71% of the sediment load was from streambank erosion. The catchment was relatively flat but had a large amount of agriculture along the creek. Kiesel found this estimate to be plausible because it was similar to estimates for other European catchments. Evans et al. (2003) modeled the contribution of streambank erosion to 28 Pennsylvania watersheds using the GWLF-E model and estimated that eroding banks contribute between 4.8% and 78.6% of the total sediment loads to those watersheds, with an average of 17.9%. Fox et al. (2016) reviewed fourteen studies of streambank erosion and suspended sediment loading, and found that bank erosion contributions range from 7% to 92% of the suspended sediment load in the study watersheds. Beck et al. (2018) estimated that bank erosion contributes 4% to 44% of annual suspended sediment loading in an Iowa, USA watershed. Our estimate that 28.5% of the total sediment load in Indian Mill Creek comes from eroding banks seems reasonable compared with these studies.

Controlling Streambank Erosion

Sediment is a major cause of water quality impairment worldwide (Narasimhan et al., 2017). Throughout the United States, sediment pollution is the second highest cause of water quality impairment, impairing the quality and habitat of 225,000 km of streams (USEPA, 2016). In Michigan alone, sediment pollution has an enormous effect on aquatic life and impairs the quality and habitat of nearly 2,000 miles of streams (USEPA, 2016). Sediment is the greatest pollutant by volume to enter streams in both the United States and Michigan (Bernard et al., 1996; NOAA, 1978). Sediment is also notorious for carrying attached phosphorus pollution into surface waters (Miller et al., 2014). Our study and others show that streambank erosion can be a major contributor to sediment loading in a watershed. Streambank erosion occurs naturally in streams, but can be accelerated because of disturbances caused by changing watershed land use (Allan, 2004; Paul and Meyer, 2001; Rosgen, 1994). However, streambank erosion is often absent from management regulations such as total maximum daily loads (McMillan et al., 2018). Thus, it is important for watershed managers and regulators to understand and control streambank erosion in watersheds threatened by sediment loading.

Control of streambank erosion in Indian Mill Creek should focus on low impact development projects to reduce runoff from impervious surfaces. These projects can minimize the physical disturbances to a stream by erosive power storm flows, including bank erosion and incision (Walsh *et al.,* 2005). Sigdel (2017) found that an increase in stream discharge from impervious surfaces in lower Indian Mill Creek caused banks to erode and moved large amounts of bedload sediment. In urban areas, low impact development practices should be used to reduce the power of storm flows including a reduction in impervious surface area, bioretention basins, pervious pavement, stormwater detention basins, avoidance of floodplain development, wetland conservation, and vegetated swales (Southeast Michigan Council of Governments, 2008). In agricultural areas, Conservation Practice Standards should be implemented to control runoff and reduce nonpoint source pollution, including the Natural Resources Conservation Service guides of riparian cover (390, 391), filter strips (393), conservation cover (327), and residue and tillage management (329, 345). Once storm flows are controlled, a restoration plan for eroding banks and degraded riparian corridors should be developed and implemented. Benefits from riparian corridor restoration include improved bank stability and water quality, better habitat for fish and wildlife, and a greater aesthetic value (Anbumozhi et al., 2005). Guidance for this plan is available from the Natural Resources Conservation Service's Federal Stream Corridor Restoration Handbook (part 653 of the National Engineering Handbook). This handbook includes approaches for streambank stabilization and stream channel restoration, such as plantings and geotextile systems, based on site conditions of the stream corridor.

Conclusion

Sediment pollution is a major concern for streams throughout the United States (Allan, 2004). One difficulty in managing sediment pollution in streams is that it is hard to quantify sediment from streambank erosion. We evaluated the use of three techniques for measuring streambank erosion at nine sites in the Indian Mill Creek watershed: erosion pins, total station surveyor, and terrestrial laser scanner. We were unable to detect significant differences between measurement techniques, and found a significant correlation only between total station and laser scanner data. Percent differences between techniques were large. Each technique had advantages and disadvantages for measuring eroding streambanks, suggesting their application is highly dependent on watershed and site specific conditions. Erosion pins and total station surveying can be used in vegetated banks but have coarse resolution, while laser scanning has high resolution but cannot measure through vegetation. Ultimately, the choice of technique for measuring streambank erosion is very important and may depend on the goals of the project and the resources available. We also assessed how streambank erosion rates vary spatially throughout the watershed, with the most deposition occurring in the lower reach of Indian Mill Creek, and the most erosion in middle to upper reaches. Overall, we estimate that streambank erosion contributes 2,020 Mg of sediment each year to Indian Mill Creek, which is 28.5% of modeled sediment loads. This estimate is comparable with other studies, and shows that bank erosion is a substantial portion of the total sediment load and is almost certainly affecting the quality of aquatic habitat, fish, and macroinvertebrate communities in the Indian Mill Creek watershed.

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CHAPTER V: SYNTHESIS

Stream habitat, fish, and macroinvertebrate communities are impacted by environmental stressors in the Indian Mill Creek watershed of west Michigan, USA. In Chapter II, "Impacts of an Agricultural/Urban Land Cover Gradient in a Coldwater Stream," we learned that reduced habitat variability, riparian vegetation loss, increased sediment load, fine substrate from sedimentation, woody debris reduction, high water temperature, and episodic pollution events were significant variables that influenced the quality of biologic communities. Of the environmental stressors we studied, fine streambed substrate from sedimentation had the strongest relationship with degraded macroinvertebrate communities and was reducing the suitability of the watershed for trout habitat. We documented this pollution imbedding streambed substrate and reducing habitat variability throughout the watershed. Thus, sediment pollution is a major stressor of aquatic communities in the Indian Mill Creek watershed.

Two of the major sources of sediment pollution are an eroding landscape and eroding streambanks. Management of sediment pollution from these sources can be difficult because of the spatial and temporal variability in watersheds. Chapter III, "Watershed and Streambank Erosion Modeling in a Michigan, USA Stream Using the GWLF-E Model and MapShed GIS Plugin," used a watershed model to simulate the spatial distribution of these sediment sources in the watershed. We used the Enhanced Generalized Watershed Loading Functions (GWLF-E) model and MapShed plugin for MapWindow GIS to study these sediment sources in 20 subbasins of the Indian Mill Creek watershed from 1997-2015. We found that southwest subbasins had the highest rates of runoff because of impervious surfaces and urbanization. Field erosion was greatest in the lower watershed with steep slopes and erodible soils. The proportion of sediment load from streambanks and the lateral erosion rate increased in a downstream direction. Field erosion contributed a per-subbasin average of 0.5 to 2.5 Mg/ha/yr of sediment, while streambank erosion accounted for 0.2% to 50.1% of the subbasins' sediment yields. The GWLF-E model also predicted that streambank erosion increases from headwaters to mouth of Indian Mill Creek because the erosion rate is a function of stream discharge (Evans et al. 2003). This pattern is important because streambank erosion is a major source of sediment that degrades habitat and aquatic communities.

One difficulty with managing sediment pollution in a watershed is that it is hard to quantify sediment loading from streambank erosion (Evans et al. 2003; Fox et al. 2016). Chapter IV, "Measuring Streambank Erosion: A Comparison of Erosion Pins, Total Station, and Terrestrial Laser Scanner," evaluated and compared three techniques for this purpose. We were not able to detect significant differences between techniques (p=0.639), but found a significant correlation only between total station and laser scanner data (R^2=0.79, p=0.003). Percent differences between techniques were large, with an average difference of 650% between erosion pins and total station, 596% between the laser scanner and erosion pins, and 1,275% between the laser scanner and total station. Banks with heavy vegetation had significantly lower laser scan coverage after vegetation filtering (11.75%) than other banks (32.5%, p = 0.047). Differing results between techniques could be due to a combination of vegetation, undercut banks, and resolution. Thus, when designing a streambank erosion study, results between different techniques for measuring bank erosion could have limited comparability, and thoughtful selection of a technique becomes very important. We recommend using the laser scanner only for bare banks with limited vegetation cover, while the total station or erosion pins are preferable for banks with heavy vegetation or other obstructions. We do not recommend using a total station for banks with heavy undercutting because of data gaps in the undercuts. We also found that streambank erosion rates vary throughout the watershed. Streambank erosion rates didn't increase in a downstream direction as predicted by the GWLF-E model; instead, the lowest reach had net deposition of sediment on the banks coming from upstream sources. This sediment could have originated from eroding banks in the upper watershed's farmland or the middle watershed's steep gradient. By multiplying the average erosion rate from erosion pins by the 28.5 km of streams in the watershed, we estimate that 2,020 Mg of sediment enters Indian Mill Creek from streambank erosion annually. This is roughly double the GWLF-E model's estimate (1,031 Mg). When compared with the GWLF-E estimate of annual sediment from field erosion (5,078 Mg), we estimate that streambank erosion contributes 28.5% of the annual sediment load to Indian Mill Creek. Much of this sediment is ultimately transported to the Lower Grand River.

Sedimentation is a natural process that has been occurring in west Michigan over long geologic timescales. Beneath the landscape are thousands of feet of sedimentary rock layers that were deposited by ancestral lakes, rivers, and seas on top of a deep continental crust that is 1.7 billion years old (Whitmeyer and Karlstrom 2007; Gillespie et al. 2008). Beginning 65,000 years ago, series of glaciers advanced and retreated over the area that is now Indian Mill Creek (Churches and Wampler 2013). The retreat of these glaciers deposited coarse till over the landscape that today forms the hills of the upper watershed. Around 15,000 years ago, the last glacier retreated over the region. The Grand River Valley was carved out as it drained meltwater from large glacial lakes (Larson and Schaetzl 2001). Thus, streambank erosion has been occurring naturally in the region since this last glacial retreat, and is nothing new in the Indian Mill Creek watershed. Eroding banks occur naturally in streams and are affected by climate, geology, and topography (Rosgen 1994; Montgomery 1999).

Agricultural and urban land use in a watershed can reduce infiltration of precipitation and increase runoff, causing powerful flows that increase bank erosion (Paul and Meyer 2001; Allan 2004). These land uses are prevalent in the Indian Mill Creek watershed, which is 43% urban and 39% agricultural by land area (LGROW 2011). However, development of agricultural and urban land use here is recent relative to geologic time. Although the region has been settled by Native Americans for 2,000 years, pioneers only began settling Indian Mill Creek in the 1830's, around the time Michigan was first becoming a state (Grand Rapids Historical Society, History of Grand Rapids: Accessed May 14, 2017, http://www.grhistory.org/history_of_grand_rapids; Tuttle 1874). These settlers farmed the watershed and harvested timber. Much of the urbanization in the Indian Mill Creek watershed has only occurred within the last few decades.

It is important to consider time scales when studying the relationship between a stream and its watershed, as well as the response a streams to disturbance (Minshall 1988; Ward 1989). These changes can cause the stream to adjust its morphology as it progresses toward a new stable form (Rosgen 1994). The extent of bank erosion we documented in Indian Mill Creek is likely an effect of the creek's response to disturbance in the watershed from these land use changes. This could help explain why the GWLF-E model predicted a different pattern of bank erosion in the watershed than we documented with our measurement techniques, as the model does not incorporate the state of the stream nor temporal readjustment of morphology after disturbance into its calculations.

The data from the three chapters support the conclusion that altered hydrology is degrading the health of aquatic habitat, fish, and macroinvertebrates in Indian Mill Creek. Hydrologic alteration is the result of increased runoff to the creek from agricultural fields and urban stormwater, creating powerful flows that scrape away at streambanks and degrade habitat for aquatic communities. Agriculture and urban stormwater are listed as two out of the three top causes of stream impairments in the United States by the U.S. Environmental Protection Agency (USEPA 2000). Research shows that restoration of stream habitat and riparian conditions in urban streams, which also is needed for Indian Mill Creek, can be ineffective for recovery of aquatic life if the destructive impacts of these intense stormflows are not addressed (Walsh et al. 2005). Thus, restoration should focus on reducing the amount of runoff volume, rate, and sediment content from urban stormwater and agricultural runoff. In urban areas, this should take the form of low impact development practices as described by the Low Impact Development Manual for Michigan (Southeast Michigan Council of Governments 2008). These practices include a reduction in impervious surface area, bioretention basins, pervious pavement, stormwater detention basins, avoidance of floodplain development, wetland conservation, and vegetated swales. Based on the spatial results of the GWLF-E modeling, priority areas for these practices should be those that have the highest potential for runoff, such as the subbasins of Brandywine Creek. In agricultural areas, Conservation Practice Standards should be implemented to control runoff and reduce nonpoint source pollution. These standards, with Natural Resources Conservation Service guides in parenthesis, include riparian cover (390, 391), filter strips (393), conservation cover (327), and residue and tillage management (329, 345).

Once storm flows are reduced through these low impact development and best management practices, additional restoration should occur to improve aquatic habitat for macroinvertebrates and fish. Recommendations for restoration of the Indian Mill Creek watershed based on our research are outlined here:

1. <u>Control Stormwater by practices that encourage infiltration and storage</u>. The low impact development and best management practices described above should be continually used to

control stormwater flow into the creek. This control of runoff to maintain a more natural flow volume and timing is imperative to restoration of the watershed. Fast delivery of runoff to a stream constrains the integrity of biological communities and water quality; thus, restoring stream habitat and riparian conditions can be ineffective for recovery of aquatic life if the impacts of intense stormflows are not addressed (Walsh et al. 2005).

2. Develop a Watershed Plan. A stand-alone watershed plan should be developed for Indian Mill Creek that summarizes watershed conditions, identifies priority pollutants and critical areas, cooperatively develops goals and objectives, and outlines an action plan with realistic projects to control nonpoint source pollution (Brown et al. 2000).

3. Work Cooperatively. Continue working cooperatively to coordinate projects and restore the creek. Successful restoration of Indian Mill Creek is dependent on continued involvement of watershed organizations, local governments, researchers, and other stakeholders. The sustainability of nonpoint source pollution management in the Indian Mill Creek watershed depends on cooperation among jurisdictions and other stakeholders.

4. Restore Riparian Corridors. A restoration plan for the agricultural riparian corridors should be designed and implemented as per Natural Resources Conservation Service (2001) guidance. Poor riparian conditions contribute to streambank erosion, high water temperatures, increased pollutant loading, and decreased inputs of leaf litter and terrestrial invertebrates that provide energy for aquatic organisms (Delong and Brusven 1991; Magana 2001; Nakano and Murakami 2001; Allan 2004). Improvement of these conditions is essential for recovery of biological communities and can be done through conservation easements, vegetation buffers, and bank restoration (Natural Resources Conservation Service, 2001; Walsh et al., 2005b).

5. Improve Woody Debris Habitat. Woody debris habitat should be increased in both urban and agricultural areas; this restoration of woody habitats has been shown to increase the richness of macroinvertebrate taxa and functional groups (Lester et al. 2007).

6. Improve Habitat Variability. Riffle and pool habitat variability should be restored through dechannelization of the creek in agricultural headwaters and the lowest kilometer that has been artificially straightened by urbanization. A channelized Indiana stream that was experimentally restored by constructing riffle and pool habitats, adding woody debris, and reducing sedimentation saw a recovery of macroinvertebrates and fish within one year and remained high after five years of monitoring (Moerke et al. 2004).

7. Reduce Field Erosion. Conservation Practice Standards should be implemented in agricultural lands to reduce field erosion. These practices, with Natural Resources Conservation Service guide in parentheses, include conservation cover (327), filter strips (393), residue and tillage management (329, 345), and riparian cover (390, 391).

8. Control Streambank Erosion. Low impact development projects to reduce runoff should be the primary activities to reduce streambank erosion. These projects can minimize the physical disturbances to a stream by the erosive power storm flows, including bank erosion and incision (Walsh *et al.*, 2005). Additionally, a restoration plan for degraded riparian corridors in critical subbasins for streambank erosion control should be developed and implemented. Guidance is available from the Natural Resources Conservation Service's Federal Stream Corridor Restoration Handbook (part 653 of the National Engineering Handbook). This handbook includes approaches for streambank stabilization and stream channel restoration, such as plantings and geotextile systems, based on conditions of the stream corridor.

9. Manage Episodic Pollution Events. Provide additional monitoring and control of episodic pollution events in the watershed, including further toxicity research. These episodic pollution events in the lower, urbanized reach of Indian Mill Creek are one possible explanation for the low numbers of fish, particularly small-bodied, sedentary species. This highlights the need for further toxicity studies in the watershed.
10. Assess Road Stream Crossings. Perform a road-stream crossings inventory following Great Lakes Road Stream Crossing Inventory Protocol (US Forest Service et al. 2011) to assess the impacts of crossings on hydrology, sediment transport, and fish passage in the watershed. Poorly-designed crossings have been an impediment to fish passage in other Michigan streams (Briggs and Galarowicz 2013; Evans et al. 2015).
11. Monitor Watershed Health. A long-term monitoring program should be developed for stream habitat, water quality, and biological communities; one option is participation in the Michigan Clean Water Corps Volunteer Stream Monitoring Program (MiCorps 2006).

CHAPTER VI: EXTENDED REVIEW OF LITERATURE AND EXTENDED METHODOLOGY

6.1 EXTENDED REVIEW OF LITERATURE

Indian Mill Creek Biological Surveys: 1970's to Present

Indian Mill Creek is a designated coldwater trout stream (MDNR 1997). Biological surveys by the Michigan Department of Natural Resources (MDNR) and Michigan Department of Environmental Quality (MDEQ) have identified sediment deposition and streambank erosion as the primary cause of degraded biological communities (MDNR 1993). When a stream becomes too polluted and is no longer capable of supporting healthy aquatic life or a human use, it is considered impaired (92nd United States Congress 1972). A history of Indian Mill Creek surveys is outlined below.

Prior to the late 1990's, the fish of the creek were surveyed by the MDNR Fisheries Division. MDNR survey reports in the 1970's show that Indian Mill Creek had the potential to support brown trout, at least in the upper reaches (Division 1971). Later reports showed that the creek could also support natural reproduction of anadromous (swimming upriver to spawn) fish like salmon and steelhead (MDNR 1990). However, as of a 1990 report, very few brown trout were supported.

The MDNR and MDEQ have surveyed the fish, macroinvertebrates, and habitat of Indian Mill Creek since the 1990's (MDNR 1993). These surveys follow the GLEAS Procedure #51 method , which incorporates fish, macroinvertebrates, and habitat (MDEQ 1997). In 1991, a survey found steelhead in the creek, but no self-sustaining populations of other trout (MDNR 1993). Low numbers of macroinvertebrates were found. The creek was determined to be slightly to moderately impaired for fish, and moderately impaired for macroinvertebrates. Habitat was rated as moderately different from reference conditions, with elevated siltation and deposition, unstable stream flow, and excessive streambank erosion. Deposits of silt and clay greater than one foot deep were found in the upper watershed and linked to agricultural erosion. Sites throughout the watershed demonstrated the continued influence of this agricultural erosion at downstream sites. It was also noted that an increase in parking lots, highways, rooftops, and other impervious surfaces from urban development caused large increases in runoff to the creek, as well as channel erosion. The 1993 report stated that stream flow stabilization, habitat restoration, and erosion control are needed to protect the creek's coldwater stream designation.

In 1998, the MDEQ surveyed Indian Mill Creek following a fish kill. The Thornapple Valley Meat Company had lost ammonia refrigerant through a storm drain to the creek, killing all fish for two miles in the lower section of the creek (Hanshue 1998). Sites upstream of the kill were found to have acceptable macroinvertebrate communities, while the community downstream of the kill at Richmond Park was severely impaired. Habitat was evaluated as slightly impaired at all sites. Embedded substrate and high bedload was documented and linked to unstable streambanks during high flows.

An MDEQ report from 2005 documented that the headwaters of Indian Mill Creek had been greatly modified to drain the surrounding farmland (MDEQ Water Bureau 2005). The report also documented lots of impervious surfaces in the urbanized lower sections of the creek in Walker and Grand Rapids. This survey found many fish including rainbow trout and a salmon at Richmond Park, which showed the creek was acceptably meeting its coldwater designation. However, the macroinvertebrate community was rated as poor, and future evaluations were recommended to better study the impaired macroinvertebrate community.

In the mid-2000's, the West Michigan Environmental Action Council recommended the MDEQ survey Indian Mill Creek again to evaluate how the creek was supporting its coldwater fish and other aquatic life. The creek was then surveyed in 2009 at Richmond Park and Three Mile Road (MDEQ 2011). Very few fish were found at Richmond Park, dominated by suckers, indicating the fish coldwater fishery of the lower reaches is impaired. Sufficient fish were found at Three Mile Rd. to meet the designated coldwater fishery status, although only 4% were trout or salmon. The macroinvertebrate communities at each site were considered acceptable, and the habitat surveys found adequate aquatic habitat at each site. Water and sediment samples were tested at each site for metals and other pollutants. Richmond Park had high levels of some metals, but the levels were below United States Environmental Protection Agency criteria.

Climate Change and Streams

Extreme weather has become more common in the Great Lakes region over the last 50 years because of climate change (Bartolai et al. 2015). Trends of more frequent heavy storms and increasing air temperatures are expected to continue. These are the two drivers of climate change that affect streams (Robertson et al. 2016). Average air temperature increased by 0.7 degrees Celsius in the region from 1895 to 1999, while average annual precipitation increased by 10.7 centimeters (Bartolai et al. 2015). Average annual precipitation is projected to increase up to 20% by the end of the century. Climate change is expected to affect the transport of pollutants and water quality, through increased frequency of both floods and droughts. Total runoff is projected to increase 7-9%, with increases in winter and spring melt runoff, which will increase erosion. However, it is unclear whether overall stream flow will increase or decrease due to climate change because of its complex relationship with air temperature (evaporation) and precipitation (Robertson et al. 2016).

In October 2016, the United States federal government released a report about how to make America more resilient to climate change (White House 2016). Recommendations of the report include advancing science-based technology and tools to address climate change, integrating climate resilience into federal programs, and supporting climate resilience efforts in communities.

Ecological Facets of Streams

Streams can be conceptualized using four dimensions (Ward 1989). The first dimension is longitudinal and describes interactions between upstream and downstream. The second dimension is lateral and describes interactions between the stream channel and riparian and floodplain zones. The third dimension is vertical and describes interactions between the stream and groundwater. The fourth dimension is temporal, and describes interactions in the stream over time, such as the time it takes for the Indian Mill Creek fish community to recover after a large episodic pollution event.

The River Continuum Concept describes changes in a stream along the longitudinal dimension from headwaters to the mouth (Vannote et al. 1980). These include changes in energy sources and functional feeding groups of organisms. Along the continuum, stream headwaters have heterotrophic (consumption-driven) energy sources with the shredder and collector feeding groups, powered largely by leaves and other debris that fall in the stream. However, this isn't always the case, because headwaters can have primarily autotrophic (sun-driven) energy sources (Minshall 1978). Mid-reaches have autotrophic energy sources with collectors and scrapers, powered by the photosynthesis of algae and plants. Lower reaches of large streams then revert back to heterotrophy because the water is too deep and turbid to be powered primarily by an autotrophic energy source. The energy sources of an ecosystem are important because they fuel biological and chemical cycles, such as the carbon cycle (Dila and Biddanda 2015).

Also acting along the longitudinal dimension is the concept of nutrient spiraling (Newbold 1981). This concept describes how nutrients cycle between biotic and abiotic forms as they move downstream. The three compartments that the nutrients spiral through are water, particulates, and biota. Disturbance to a stream can alter the nutrient spiraling and change the efficiency and health of the stream. Organic matter retention is a process that affects nutrient cycling and the biological communities in streams (Brookshire and Dwire 2003). This retention is affected by discharge, channel morphology, stream woody debris, and riparian vegetation.

The concept of invertebrate drift also acts along the longitudinal dimension of streams. Stream macroinvertebrates will move either upstream or downstream (Allan 1995). This can happen from accidental dislodgement, or intentional drift to exploit food resources or colonize new habitat.

The Flood Pulse Concept describes changes in a stream along the lateral dimension between the river channel, riparian zone, and floodplain (Junk et al. 1989). This dimension is important for the cycling of nutrients in the stream ecosystem. It is also an important connection for subsidizing energy sources, because the stream channel and riparian zone have highs and lows of energy production throughout the seasons (Nakano and Murakami 2001). This dimension is also important for migration and spawning of fish (Junk et al. 1989). Hydrologic disturbances like dams and impoundments can upset lateral connectivity in a stream (Ward and Stanford 1995). These changes include destabilized streambanks and a loss of connectivity between a stream and its floodplain. This can cause further problems like a changing thermal regime, loss of biodiversity, desiccation of water bodies in the floodplain, and withering of alluvial vegetation. Further, the lateral effects of hydrological alterations vary depending on the type and location of the stream reach.

The Hyporheic Corridor Concept describes the vertical dimension of streams and the connection between the stream channel and groundwater that is under and around it (Stanford and Ward 1993). The hyporheic zone is an area under the streambed or in adjacent alluvial areas where there is an underground hydrological connection. This dimension is important for nutrient cycling, macroinvertebrate habitat, and links in a stream's food web.

The temporal dimension of streams is important to understand the response of streams to disturbances (Ward 1989). For example, it describes how the macroinvertebrate community of a stream responds after a disturbance like a dam. Hydrological alterations like dams and impoundments affect a stream ecosystem both upstream and downstream of the alteration (Nilsson et al. 2005). After an alteration, a stream could experience inundation, changing flow patterns, fragmented habitat, elimination of turbulence and riffle habitat, sedimentation, upset nutrient cycling, changing aquatic communities, and loss of species. Temporal responses in streams can vary greatly in scale depending on the process (Minshall 1988). For example, the movement of an aquatic insect can occur in seconds, while the movement of tectonic plates under the stream occurs over many millions of years.

Stream Morphology

The morphology (form and structure) of streams can differ by valley entrenchment, longitudinal gradient, width to depth ratio, velocity, flow, channel roughness, sediment characteristics, and sinuosity (Rosgen 1994). When one these factors changes, a series of channel adjustments can occur that results in a new form of the stream channel. The morphology of a stream is important to consider when planning engineering projects, improving fish habitat, and restoring streambanks.

One model for conceptualizing the dynamic nature of stream morphology is Lane's balance (Pollock et al. 2014). Lane's balance states that water discharge (Qw) and channel slope (S) are related to sediment load (Qs) and representative particle size (Ds) (Dust and Wohl 2012). The balance can be visually depicted as an actual balance, that shows whether aggradation (deposition in channel) or degradation (erosion) will occur under changing scenarios (Figure 6.1). This balance has been used by engineers, geomorphologists, and educators to understand and predict changes in stream morphology after disturbances. Specifically, the balance can be used to predict whether there will be erosion or deposition in a stream channel after an alteration as the stream searches for a new equilibrium (Pollock et al. 2014).

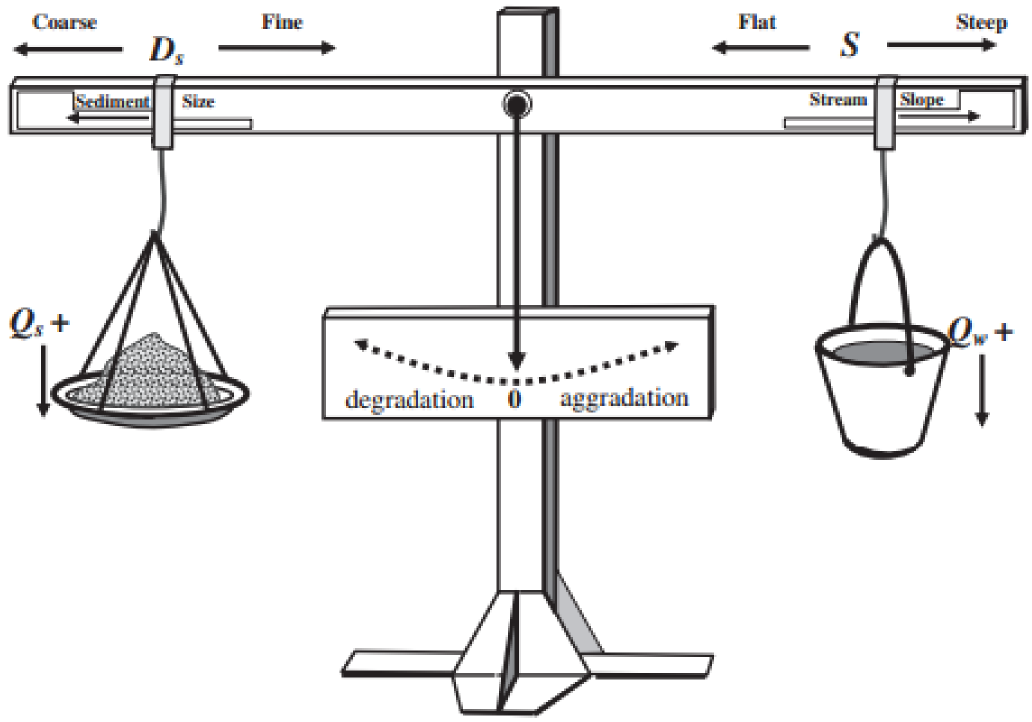

Figure 6.1. Lane's Balance (Dust and Wohl 2012, after Dr. Whitney Borland CSU).

The original Lane's Balance can be improved by modifications that relate changes in cross section shape, sinuosity, and bedform units (Dust and Wohl 2012). These modifications are the width /depth ratio of the channel cross section, elevation change, and sinuosity. This allows the model to conceptualize more complex channel adjustments than the original Lane's Balance.

Stream morphology can be very complex, making simplification difficult (Rosgen 1994). However, understanding morphology through a classification scheme is important for predicting river behavior, understanding flow and sediment relations, extrapolating site-specific data to unsampled stream reaches, and providing a standard way for professionals working with rivers to communicate.

Rosgen 1994 developed a classification of stream types based on stream morphology and processes. This fits into the temporal dimension of streams described by Ward (1989). This classification has three levels depending on the desired level of detail. The first level places a stream in one of seven major types based on geomorphic characteristics. These characteristics are stream gradient, cross section shape, floodplain shape, and sinuosity. For example, Type A streams have a steep slope, incised cross-section, narrow floodplain, and low sinuosity. These characteristics are important because they can drive other stream characteristics like bed features and abilities to transport debris. The second level further classifies streams by particle sizes in the channel, such as cobble, gravel, and sand. For example, Type A3 streams have Type A morphology with cobble substrate, while Type A4 streams have Type A morphology with gravel substrate. The third levels uses very fine ranges of morphological characteristics to further classify streams.

The Rosgen 1994 classification describes a natural dynamic continuum between stream types, especially in response to changes in morphology. Streams in this continuum experience adjustments in morphology over time, which can either be rapid or very slow. These changes can be caused by factors like flow, sedimentation, and bank stability. For example, if streambanks become destabilized, a stream will often follow a pattern of increasing channel width to depth, increasing slope of water surface, and decreasing sinuosity, with a resultant change in geomorphic classification. These dynamics are very important to consider for stream restoration. Restoration projects should not be designed to change a stream back to a form that no longer fits geomorphically. For example, placement of fish habitat structures could alter the morphology of a stream and degrade fish habitat.

Increased peak flows in streams from landscape development and runoff can change geomorphology (Paul and Meyer 2001). During development, erosion from the landscape increases the sediment load to a stream, causing the stream to aggrade (fill with sediment). Then, the amount of runoff increases following reduced infiltration and more impervious surfaces. This causes the streambanks to erode and the channel to widen and deepen. It also causes greater flooding. Soil loss form streambank erosion is the primary source of sediment to urban streams at this erosional stage (Paul and Meyer 2001).

Stream Habitat

Stream habitat surveys can be the building blocks for management planning and environmental monitoring (Dolloff et al. 1997). Habitat data is essential for evaluating the success of stream restoration and detecting change. Ecologists have recognized that the distribution and abundance of stream biota is in response to not one, but many interacting habitat variables (Raleigh et al. 1984; Poff 1997). Therefore, habitat surveys should incorporate many important abiotic and biotic aspects of stream ecosystems.

A problem often encountered with widely-used habitat surveys, such as the Representative Reach Extrapolation Technique, is that only short reaches of habitat are surveyed and extrapolations between them are likely to be inaccurate due to natural variation. The Basinwide Visual Estimation Technique (BVET) addresses this problem by sampling every stream segment in a study area, eliminating the need for extrapolation. It also allows for detailed maps of stream habitats to be produced (Hankin and Reeves 1988). One aspect of the Basinwide Visual Estimation Technique is a visual fish survey by divers that complements the habitat observations.

An important component of the Basinwide Visual Estimation Technique is an in-stream wood survey. Instream wood is an important habitat component that can affect stream hydraulics, sediment dynamics, and channel morphology (Lisenby et al. 2014). It does this by altering water velocity, changing erosion patterns, dissipating energy during powerful flows, and creating patchy hydrology like eddies and backwater areas (Ehrman and Lamberti 1992). Instream wood was studied in a Costa Rican watershed and was related to stream morphology and hydrology. It can also provide structure, retain organic matter, and form bedform units in streams (Brookshire and Dwire 2003; Cordova et al. 2007).

Large wood in streams also is important for habitat diversity (Nakamura et al. 2017) and can facilitate both sediment deposition and erosion while increasing the patchiness of stream habitat. A study in Japan found that dams, precipitation patterns, flow, watershed size, and latitude all affect the distribution of large wood in streams (Nakamura et al. 2017). The study concluded that monitoring large wood is very important for managing rivers and floodplains. Additionally, wood pieces that are shorter than the width of a stream channel are often controlled by the stream, but wood pieces larger than the stream channel can have a large influence on stream processes (Cordova et al. 2007). Large wood pieces also provide habitat for invertebrates (Ehrman and Lamberti 1992).

Large woody debris in streams can be surveyed by visually observing wood pieces, recording their size, and comparing to reference conditions. Cordova et al. 2007 state that the average abundance of large wood in Midwestern streams is 32.6 pieces per 100 meters, with large wood defined as pieces greater than ten centimeters in diameter and one meter in length.

Habitats and the organisms that live in them have many variables that can be assessed to determine how suitable the habitat is for the organism (Southwood 1977). The results of stream habitat surveys can be compared with habitat suitability indices and representative to assess the quality of stream habitat and guide restoration (Tip of the Mitt Watershed Council 2015). These indices provide objective, quantifiable methods to analyze how habitat conditions meet the life history and habitat requirements of the target organisms (Raleigh et al. 1984). They can be used to guide management decisions by planning habitat improvement projects and evaluating a project's impact. The habitat suitability index values range from 0.0 (unsuitable habitat) to 1.0 (optimal habitat) for the habitat variables. The indices have been developed and reviewed by professional biologists familiar with the target organisms.

Habitat suitability indices include many variables (Raleigh et al. 1984). Water temperature regime, substrate composition (silt, sand, gravel, etc.), stream bedform ratios (i.e. riffle to pool ratio), riparian vegetation, canopy cover, streambank stability, instream cover, and flow regime are some variables that are included in the indices. Water temperature is very important because target organisms often have narrow optimal temperature ranges, and very high temperatures can be lethal. For example, optimal water temperature for brown trout is between 12 and 19 degrees Celsius, and becomes lethal at 27 degrees Celsius. Stream substrate composition is important because it affects the aquatic invertebrate community that can inhabit the stream and the ability of fish to spawn. For example, optimal brown trout habitat has less than around 10% fine substrate in riffles and less than about 5% of fine substrate in spawning areas. Stream bedform units are important for the habitat requirements of different organisms. For example, aquatic invertebrates are more abundant and diverse in riffles than pools, and optimal habitat for brook trout has a ratio of riffle area to pool area of around 1:1 (Raleigh 1982). Riparian vegetation is important for controlling erosion, while canopy is important for keeping water temperatures within desirable ranges in small streams. Flow regime is important for the life histories of target organism and high peak flows can be destructive. For example, a base flow greater or equal to 5% of average daily flow is excellent for trout and salmon reproduction, while a base flow less than 25% of average daily flow is poor habitat for trout and salmon reproduction.

Stream temperature is an important component of habitat that affects biological communities. In Lower Michigan, stream temperature can be classified into three thermal regimes: coldwater, coolwater, and warmwater (Wehrly et al. 1999). A coldwater stream is defined as having July mean temperatures less than 19 degrees Celsius. Similarly, coolwater streams are 19 to 21 degrees Celsius, and warmwater streams are greater than 21 degrees Celsius. Stream temperature can also be classified into temperature fluctuation regimes: stable, moderate, and extreme. Stable streams have July temperatures that fluctuate less than 5 degrees Celsius, moderate streams fluctuate 5 to 9 degrees Celsius, and extreme streams fluctuate more than 9 degrees Celsius. Temperature is important because different fish have different optimum thermal regimes and temperature fluctuation regimes. For example, brook trout prefer coldwater streams with stable temperatures, while rock bass are in the category of warmwater with extreme fluctuation.

Streambank Erosion Measurement Techniques

Erosion pins are installed by being pushed horizontally into the streambank (Kiesel et al. 2009). Erosion of a streambank can be assessed over time by repeated measures of the distance between the bank and pin's tip. Increasing exposure length on the pin indicates erosion, while decreasing exposure length indicates deposition (Palmer 2008). The data can be used to measure lateral bank retreat or to calculate the volume or mass of soil loss over time (Palmer 2008). Bank retreat data from erosion pins can be used in the equation

Sediment Loss (kg/yr) = Bank Height (m) * Bank Length (m) * Recession rate (m/yr) * Soil Bulk Density (kg/m^3)

to estimate the total mass of soil lost due to streambank erosion (Palmer 2008). Erosion pin results are useful in watershed studies because they can be compared between categories of land use to analyze differences in streambank recession (Palmer 2008).

Total station data of streambanks is taken by setting the instrument over a control point and taking side shots of the bank (Keim et al. 1999). Different patterns of shots have been explored, where more shots allows for higher resolution but also takes more time. Keim et al. 1999 took shots at the top of the streambank, toe of the bank, and the lowest point of the stream bottom. The researchers also took shots throughout the slope of the bank wherever there was a change in topography greater than 15cm. The surveys were repeated once per year to analyze change in morphology. Another study in Virginia collected streambank data in cross sections, with on average five data points per cross section (Resop and Hession 2010). Points were collected at the top of the bank, water level, and erosion pin locations. The point data collected by a total station can be converted to a Digital Terrain Model, which can be analyzed in a Geographic Information System (Keim et al. 1999). A time series of these models can be used to examine sediment transport, erosion, and deposition. Total station data can be used to examine streambank stability, pool volume, and sinuosity (Keim et al. 1999).

Laser scanners can be mounted on all other sorts of frames and crafts, resulting in many different techniques to perform the scans. One of these techniques is aerial laser scanning, where a laser scanner is mounted on an airplane and flown over a site. This can provide accurate maps and detect small changes in topography (James et al. 2007). However, aerial laser scanning is less successful than other techniques if the intended target is under the forest canopy, which can cause inaccurate elevation inferences, and has trouble collecting data on vertical surfaces like eroding streambanks (James et al. 2007; Day 2012). A second technique for laser scanning is to use a backpack mobile laser scanner (Wang et al. 2013). This device has both a laser scanner and GPS attached to a backpack frame, to collect data while walking or wading. A third technique for using a laser scanner is to mount it on a boat. This can be effective for surveying large rivers. One European study mounted a laser scanner on a dinghy, and was able to survey six kilometers of riverbanks in just over an hour, while maintaining an accuracy of two centimeters (Alho et al. 2009). A fourth technique, terrestrial laser scanning, involves securing the laser scanner to a stationary tripod or frame (Lisenby et al. 2014). An advantage of mounting a terrestrial laser scanner to a frame is that the suspension allows the scanner to collect data on the stream channel at a more direct angle, reducing error (Lisenby et al. 2014). Another advantage is that, with this high resolution, laser scanning can model the sediment composition of a river (Wang et al. 2013). Sediment particles greater than 63 millimeters in size can be detected and classified in the laser point cloud. A third advantage is that laser scanning data can be used to estimate channel slope, width, discharge, and stream power (Biron et al. 2013). Cross sections of models can be used to analyze flow levels throughout a stream as well (Lisenby et al. 2014).

Prior to using a terrestrial laser scanner, it is important to set stationary targets so sequential scans of the same coverage can be linked together through alignment (Lisenby et al. 2014). A Virginia study used target alignment to align multiple scans from different locations at a site for better coverage (Resop and Hession 2010). Tying these targets into a projected coordinate system can allow the precise placement of data in a Geographic Information System (Milan et al. 2007).

Laser scanner data is collected as a point cloud and can be processed using powerful software, including Leica Geosystems Cyclone, InnovMetric PolyWorks, ArcGIS, and LAStools (James et al. 2007; Resop and Hession 2010; Day 2012; Lisenby et al. 2014). These software can filter out unwanted points so that the cloud represents the ground or bank surface and not vegetation or other obstructions (Day 2012; Lisenby et al. 2014). The software can also fill gaps in the data by placing a flat plane over the surrounding points (Day 2012). From the point cloud data, software can be used to create virtual models of a streambank, such as a Digital Elevation Model or Digital Terrain Model (James et al. 2007; Flener et al. 2013). Both types of model have similar functions. The data could also be converted to a Triangulated Irregular Network (TIN) model (Day 2012). The TIN model is beneficial because it has a three dimensional structure and models rugged crenulated surfaces well (Day 2012).

Models of repeat scans can be overlaid to estimate geomorphic change, such as erosion rate and volume of soil lost (James et al. 2007; Resop and Hession 2010). Volume of soil lost can then be multiplied by soil bulk density to get a mass of soil lost from streambank erosion (Thoma et al. 2005). A space time cube can be created to visually examine changing bank conditions over many survey events (Starek et al. 2013). Replicate same-day scans can be taken to estimate error of the laser scanning models (Day 2012).

Photogrammetry can be performed by aligning images from cameras mounted on tripods (Pyle et al. 1997). However, it can also be performed aerially. A camera mounted on an Unmanned Aerial Vehicle has been able to model the topography of a European floodplain (Flener et al. 2013). Pix4UAV Desktop and Agisoft Photoscan are two programs that can process photogrammetry data from photos to elevation models of eroding streambanks (Eltner et al. 2013). One type of model is a Digital Elevation Model (Pyle et al. 1997). Repeat photogrammetric models of streambanks can then be overlaid to estimate soil loss using bank retreat and volume analysis (Lawler 1993). The quality of a photogrammetric model can be determined quantitatively by analyzing the precision of the image matching at aligning points, comparing data with a ground survey, or by collecting replicate data of the same streambank. A photogrammetric quality study of Swiss streambanks found an average discrepancy of eight millimeters in the digital elevation models (Pyle et al. 1997). A study in Spain found that models processed with Pix4UAV software have sub-centimeter accuracy, while models processed with Agisoft Photoscan had accuracies of 0.6 to 4.1 centimeters (Eltner et al. 2013). However, the study found that AgiSoft Photoscan can produce high point densities of up to six points per square centimeter, while Pix4UAV has densities less than one point per square centimeter.

Photogrammetry is promising because it is easy to do and requires only a common digital camera. However, the expense of the software could be a deterrent. Additionally, it can be difficult to get clear enough data on streambanks because of low light or vegetation coverage. We suggest that researchers measuring streambank erosion with photogrammetry do so in the early spring when ground and canopy vegetation is sparse, on bright sunny days, and with the intent of photographing clear features like rocks and tree trunks to aid in data alignment.

Each technique has advantages and disadvantages for measuring eroding streambanks. Erosion pins are cheap and easy to maintain, but can have difficulty accounting for spatial variation in a streambank (Lawler 1993). A total station can accurately measure the location of a point on a streambank but has a low point density, can disturb banks, and has difficulty with overhanging banks (Keim et al. 1999; Resop and Hession 2010; Plenner et al. 2016). Laser scanning has high resolution and can detect small erosion rates, but has trouble with vegetated banks and requires special software that doesn’t work well on the typical desktop computer

(Resop and Hession 2010; Day 2012; Lisenby et al. 2014). Photogrammetry is non-intrusive and quick, but doesn't do well in poor light conditions (Lawler 1993; Pyle et al. 1997). Watershed modeling of streambank erosion can model sediment entering a river from different pathways using readily-available data, but has difficulty modeling complex factors affecting streambank erosion and can have a considerable degree of uncertainty (Evans et al. 2003; Kiesel et al. 2009). Ultimately, the choice of technique for measuring streambank erosion may depend on the goals of the project and the resources available.

6.2 EXTENDED METHODOLOGY

Streambank Erosion Measurement Techniques

Site Design

Nine sites were chosen for the study. Four sites were in the lower urbanized parts of Indian Mill Creek, two sites were in the upper farmland, and two sites were on tributaries. Properties for sites were chosen based on where we could get permission for access. Three of these sites were on public land, while five were private. Permission from all landowners was attained. Within each property, an 18 meter section of stream was chosen, based on a balance between an open channel for laser scanning and representative streambank conditions.

Control Points

Four control points were surveyed at each site at high, open locations where Global Positioning System (GPS) accuracy is best. These points were two foot rebar driven into the ground with orange tape or a cap. These points were used to tie into the projection and orient the total station.

Sites were surveyed into the projected coordinate system using a Global Positioning System. A Trimble Geo7x GPS with Zephyr external antenna was used, placed on a two meter bipod. The GPS sensors were calibrated each day prior to the first collection. The GPS was set to collect data for 15 to 30 minutes at each point. It collected one reading per second with a minimum accuracy set to 5cm. TerraSync 5.86 software was used to collect data. All GPS data were post-processed in Pathfinder Office using data from the Grand Rapids Continuously Operating Reference Station.

Control points were surveyed with a Topcon GPT-3000 total station. The total station is leveled and set on one control point. Then, it was back sighted to another control point for orientation to the NAD83 UTM Zone 16N projection. Elevations were measured above Mean Sea Level. After the total station was backsighted, other control points were check pointed to determine measurement error. Both reflector and reflectorless surveying are used, depending on the situation.

Erosion Pins

Prior to installing erosion pins, the stream section 18 meters was divided into three six-meter subsections using a measuring tape. Erosion pins were carefully installed in the streambank in the middle of each subsection, on both banks. One to three pins are installed at each location evenly spaced up the bank, depending on the height of the bank. One erosion pin was placed for every meter of bank height. Extra pins were installed if there were visible changes in erosion that were not otherwise captured by the pins, such as the vertical transition between an undercut bank and vegetated slope. Pins were pushed or hammered into the streambank until there were only a few centimeters sticking out.

Erosion pins were measured from the tip of the pin to the streambank using a measuring tape. The average of measurements from the top and bottom of the pin was used to account for bank slope. Where there is a horizontal angle to the bank, the left and right sides of the pin would also be measured and included in the average. This average gives a rough estimate of where the bank would be in the center of the pin. Erosion pins were measured monthly from May to September 2017, with two additional measurements following rain storms, then April to May 2018.

Total Station

A Topcon GPT-3107W total station theodolite on Wild GST05 tripod with SurveyPro software was used to survey streambank shape. The total station was tied into the NAD83 UTM Zone 16N projection by setting on control points, and backsighting to other control points, that were marked with the GPS. At each site, the GPS points were re-surveyed with the total station.

To set up the total station, the tripod would be placed over the survey marker. The legs would be pushed down and adjusted. Then, the theodolite would be firmly secured to the top of the tripod, and a plummet laser used to know it is centered over the marker. Coarse adjustments would be performed with the coarse bubble level and tripod legs. Then fine adjustments would be performed with the single-plane level and fine knobs. Once level, the plummet laser would be re-checked. The instrument would regularly be checked for levelness during use. Then the height of the instrument is measured with a tape measure to use for shooting points.

When the instrument needed to be moved, a temporary point would be created by pushing a marker into the ground. This point would be sideshot. Then, the instrument would be placed and aligned on top of the temporary point and re-oriented to a backsight point. The previous point would be check pointed to determine error during movement of the total station.

To collect points, a reflector prism was used on top of a staff with bubble level. The height of the instrument and height of the prism are measured multiple times over the day. The total station is sighted into the prism and a sideshot taken. Different descriptions are used including “bank” and “erosion pin”, depending on what data is being collected. If there is an undercut bank that isn’t reachable, the horizontal distance between the prism staff and the back of the undercut is noted. The streambank shape is then adjusted to reflect the undercut bank.

The site design for the total station surveys builds off the methods of Keim, Skaugset, & Bateman, 1999 and Resop and Hession 2010. Six transects were performed along each side of the bank over the 18 meter site, at the 0, 3, 6, 9, 12, 15, and 18 meter marks. The 3, 9, and 15 meter marks coincided with the erosion pins. In each transect, the top of the bank and toe are collected. Then, two to three shots are taken evenly spaced along the bank, depending on its size and variability. These shots are taken over erosion pins during the 3, 9, and 15 meter transects, at the location where the pin meets the streambank.

Markers for the lidar scanner and target spheres also were surveyed. At the end of the shot-taking, the previous marker was used as a check point to determine if there was any error from drift over the survey.

Total station data was exported as a CSV file to a computer using a USB cable and Windows Mobile Device Center 6.1. The CSV file was imported into ArcMap. Then, xy data were displayed and exported as a shapefile. A separate file was created for each streambank using the Select tool of ArcToolbox. A 3D TIN file was created using the Create TIN tool and Delaunay Triangulation. This TIN is the model of the streambank and was overlaid with repeat models to estimate soil loss from streambank erosion.

Terrestrial Laser Scanner

One bank was surveyed at each site with a FARO Focus3D terrestrial laser scanner in 2017, and a Trimble TX8 in 2018. The scanners were rented from Michigan Surveyors Supply Co. in Lansing, Michigan. This bank was chosen subjectively to try to incorporate representative conditions and have clear visibility for the scanner. Three survey markers were placed along this bank, as far apart as possible without sacrificing visibility. They were placed usually 2-5 feet upland from the bank, where they were visible from the other side but unlikely to erode away. These markers were a 2 foot rebar stake with a 2.5 inch flat aluminum cap on top. The lidar target spheres were placed on these plates. One additional marker was selected on the opposite bank at a location deemed the best visibility for the laser scanner, and marked with a plastic yellow survey cap. These markers act as control points, and were surveyed with the total station so the laser scan results can be projected in a geographic information system.

At each site, the preparations were set up prior to using the laser scanner. A previous site sketch, photos, and a metal detector were used to locate the four survey markers for the laser scanning. A target sphere was placed on the aluminum plate of each of the three target bank markers, directly in the center. A sturdy CRAIN tripod was placed over the fourth marker, leveled with a bubble level, and centered using a string-and-weight plummet. The apex of the tripod was placed high enough to see the opposite streambank over grass and brush. The Height of Instrument was measured from the top of the survey marker to the apex of the tripod and noted for later.

To ensure that all three spheres were visible from the apex of the tripod, brush was pushed aside. If brush cannot be pushed aside, it was cut with a knife or machete, held back, or sat on for the length of the survey. The minimum amount of brush was cleared to see targets, to minimize the effect on the study streambank.

The laser scanner was then carefully placed on the tripod and secured. The front of the scanner was pointed toward the bank, and the protective optics cover was removed. A 64 gb SD card was used to store data. The scanner was powered on. Then the sensors were set. The inclinometer was adjusted so the scanner was level. The compass was updated before the first scan of the day. Temperature was checked to be sure it was safe. Approximate latitude was entered. Elevation above mean sea level was entered to adjust the altimeter.

Next, a preliminary scan was taken. The resolution and quality were set to low settings, typically 1/8th resolution and 2x quality or less. Horizontal and vertical scanned area were set to near full. The preliminary scan then took about four minutes. The image was used to adjust the horizontal and vertical limits to fit the desired area of the bank, usually with some extra data on each side.

Prior to the full scan, the resolution and quality were set to the desired levels. We used 1:1 resolution and 2x quality with the FARO in 2017, which usually took a half hour scan. These levels were chosen because they were successfully used by the Annis Water Resources Institute previously (Kurt Thompson, personal communication). We then double checked that color image was turned on. Once ready, we hit the Start Scan button and stayed out of its way for 30 minutes. For 2018 with the Trimble unit, we used a Level 3 scan in a similar fashion.

At the end of the scan, the image on the scanner was checked to be sure the desired area was scanned and there weren't any obvious errors. Then the scanner was carefully disassembled and packed up. The target spheres and tripod were removed as well. The scanner and spheres were kept safe in sturdy travel cases when not in use.

At the end of the day, batteries were recharged and data was backed up using a Dell Inspiron laptop.

Photogrammetry

Photographs of eroding streambanks should be taken with good lighting and with enough overlap to minimize blind zones and construct geometry. A meter stick or distance between two known points should be included in the scan for later spatial reference. Approximately 60% of overlap is suggested for proper photo alignment (Agisoft 2017). However, more photos can allow for more accurate models. The direction of each photograph should be straight at the face of the bank, not rotated. The bank should take up maximum area in the photograph to ease processing.

A Canon Power Shot ELPH130 IS with infrared camera was used for the photographs. Photos were taken approximately once every meter along the streambank at a distance sufficient to achieve 60% overlap. Photos were taken with the top of the bank near the top of the picture frame, and the stream near the bottom of the picture frame. Photos were taken to be crisp and sharp. If needed, a tripod was used to stabilize the camera and avoid blurry or unfocused photos.

Photos were processed using Agisoft Photoscan and the workflow to align photos and make a dense point cloud, mesh, and texture. Each textured mesh was then exported as an .OBJ file and opened in CloudCompare. Streambank models were cropped and aligned in CloudCompare using precisely matching points between multiple models, such as root knobs. The meshes were then scaled using an object of a known distance, such as a log that had length measured with a tape in the field. The Volume Calculation tool of Trimble RealWorks was then used to calculate the change in volume between meshes.

Unfortunately, we were only able to calculate bank erosion using photogrammetry at one streambank, the right bank of the Upper Sharp Drive site. For all other sites, we were unable to generate a sufficient alignment of photos using the Align Workflow of Agisoft Photoscan. Likely reasons for this are low-light conditions and confusing bank images with lots of similar-looking leaves and soil.

Stream Survey Methods

Scour Chains

Scour chains were installed at the 9 meter mark of the 18 meter section. Three chains were installed laterally across the creekbed. One was installed in the thalweg, while the other two were halfway between the thalweg and bank. The scour chains were 24 inch chain links attached to a duckbill anchor by a clip. They were driven vertically into the streambed with a duckbill driver until just a few links were exposed. Where the creekbed was very narrow and less than 1.5 meters wide, scour chains would instead be installed in the middle of the channel at the 3, 9, and 15 meter marks. Scour chains were measured as how many links were exposed. Alternately if there was deposition, they will be measured as the vertical depth to the right-angle of the chain from the surface of the creekbed.

Pebble Counts

Wolman pebble counts with the zigzag method were performed by zigzagging from bankful to bankful in an upstream direction, randomly sampling a pebble at approximate seven foot intervals (Bevenger and King 1995, Figure 6.2). A total of one hundred pebbles were sampled. The pebbles were randomly sampled by reaching down to the toe of the wader boot with a forefinger and picking up the first pebble touched. A gravelometer was used to classify the sediment at each count into distinct grain sizes along the intermediate axis.

The size class of each pebble was noted in a field datasheet. Data were plotted as a cumulative frequency distribution of the size classes (Bevenger and King 1995). Grain size distribution was then analyzed for each site. Median bed particle size and small particle distributions are both useful for comparisons. Impacted reference and unimpacted stream reaches should both be surveyed to analyze impacts of land management.

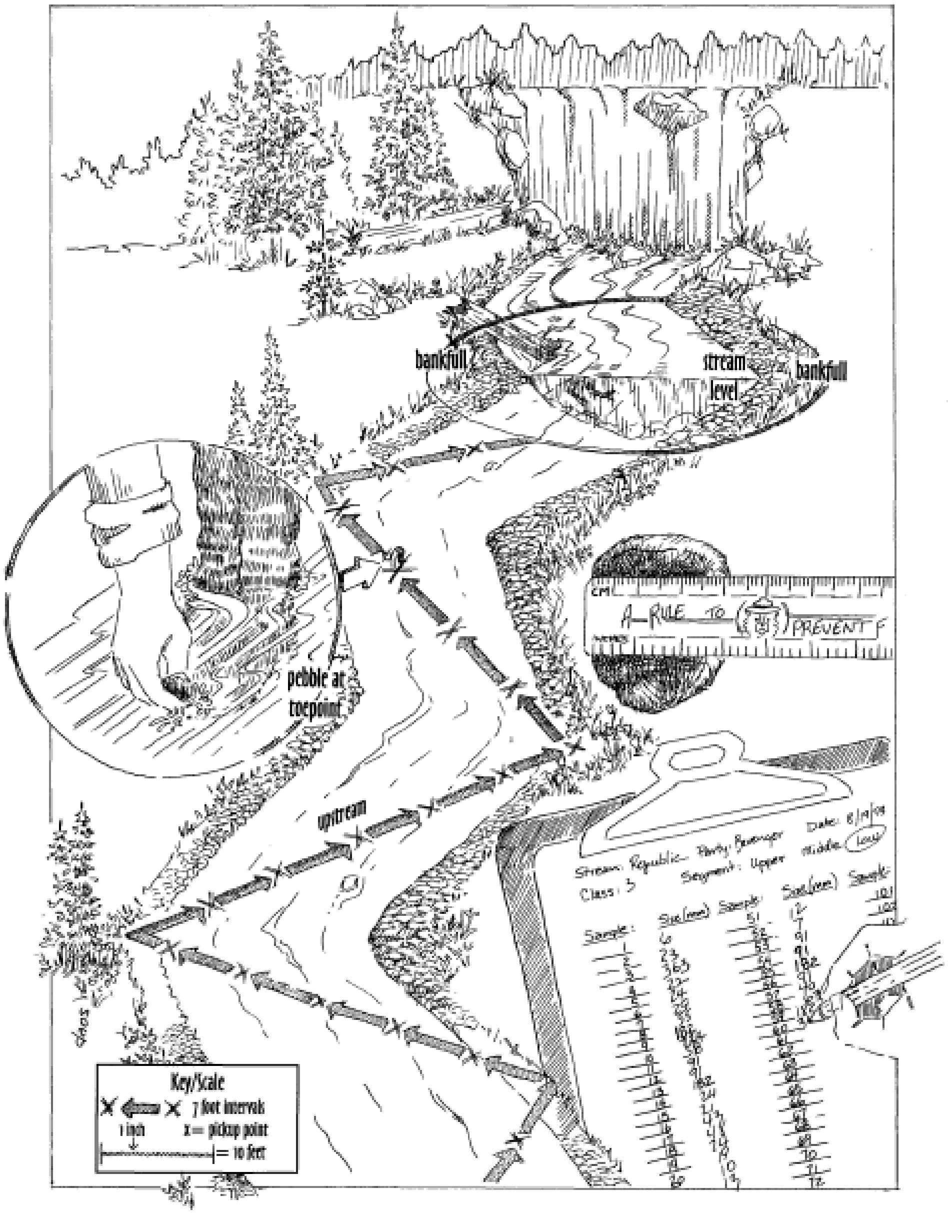

Figure 6.2. Zig-zag method for pebble counts (Bevenger and King 1995).

Stream Habitat Survey

A stream habitat survey was performed using the methodology of the United States Forest Service's Basinwide Visual Estimation Technique, modified by Tip of the Mitt Watershed

Council to fit in a Geographic Information System (Dolloff et al. 1993; Tip of the Mitt Watershed Council 2015). This survey was chosen because of its effectiveness at spatially analyzing stream habitat on a watershed scale. The creek was surveyed in an upstream direction to avoid clouding the water being observed.

The partitioning element used in the habitat survey was the stream bedform unit (Figure 6.3). These elements were spatially organized using a Global Positioning System. These units include runs, riffles, glides, pools, and cascades. Riffles were the units with the steepest slopes and shallowest depths, with a thalweg (deepest part of stream channel) that was not well defined (Vermont Agency of Natural Resources 2003). Runs were deeper than riffles but have a flatter slope and often well-defined thalweg. Pools were the deepest bedform units in a stream, with almost no surface-of-the-water slope, and are often located on the outside of meanders. Glides were found downstream of pools, with a negative bed slope but positive water surface slope, and an increase of velocity from the pool. A unique identifier was given to each bedform unit on the data sheet and in the Global Positioning System. Data of stream wetted width, bankfull width, average depth, and maximum depth were collected for each bedform unit. Additionally, data were collected about the following stream habitat components: substrate composition, riparian vegetation, and large woody debris. A photograph of the bedform unit was also taken and the photo number was noted on the datasheet. A list of equipment needed is in Figure 6.4.

The percentage of substrate composition was estimated for the substrate types of clay, silt, sand, gravel, cobble, boulder, and bedrock in the streambed. The sum each substrate type's percentage should equal 100%.

The size class and type of dominant riparian vegetation in each bedform unit was noted for the inner zone, outer zone, and canopy. The inner zone was the vegetation within a few feet of the stream channel, and the outer zone was surrounding vegetation. Dominate size class choices were grasses/sedges, shrubs, saplings, or trees. Dominate riparian types were grasses/sedges, alders, conifers, aspen, or hardwoods. Canopy was the cover over the stream, and had the options of partial shrub, partial tree, closed shrub, closed tree, and open canopy. When more than one size class was dominant at a unit, the largest class was chosen to represent the unit.

A count of large woody debris was performed in each bedform unit for three size classes using the Tip of the Mitt Watershed Council (TOMWC) method, and one class using the Cordova et al. 2007 method. The TOMWC method classes are 1) longer than ½ stream bankfull width and greater than six inches in diameter, 2) longer than ½ stream bankfull width and four to six inches in diameter, and 3) shorter than ½ stream bankfull width and greater than six inches diameter. Pieces aren't counted if they're longer than ½ bankfull width but narrower than four inches diameter, or shorter than ½ bankfull width and narrower than six inches diameter. The Cordova method counts all wood pieces greater than one meter in length and 10 centimeters in diameter.

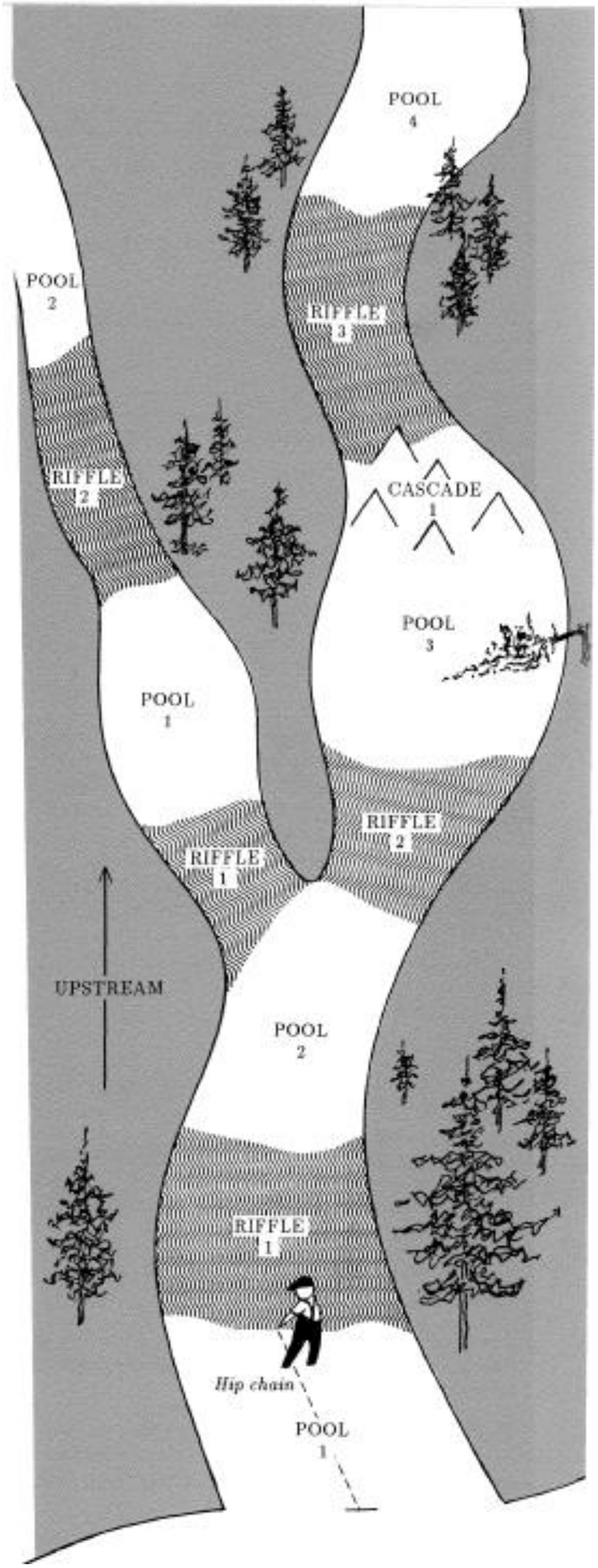

Figure 6.3. Stream Habitat Survey Partitioning (Dolloff et al. 1983).

Habitat Survey Equipment Checklist:

First Aid
Waders
Bug spray
Bug net
Long sleeve shirt
Sunscreen
Food
Water

GPS (checked out)
GPS batteries (charged)
Tape measure
Meter stick
Orange vest
Clipboard
Pencils
Camera
Conductance meter (optional)
Thermometer (optional)
Pocket knife

<u>Materials folder:</u>
Datasheets
Example datasheet
Dolloff et al. 1993
Habitat Survey Chapter of Stover Creek Plan
Bedform Units sheet (Vermont – App. M)
Size classes sheet (Pebble Count Methods)

Figure 6.4. Stream Habitat Survey Equipment Checklist.

Benthic Macroinvertebrate and Fish Surveys

Stream macroinvertebrate communities were surveyed following the Procedure 51 method used by the Michigan Department of Environmental Quality (MDEQ 1997). For consistency, Procedure 51 sampling should occur between June 1 and September 30 during stable, low or moderate flows. Sites should be selected to be representative of the stream and not be locally modified. Similarly, sampling should occur upstream of road crossing influences and avoid river mouths. If fish, macroinvertebrates, and habitat are all sampled in the same visit, they should occur in the order of fish first, then macroinvertebrates, and habitat last. Data was recorded on a datasheet.

Macroinvertebrate samples were taken from all habitats at a site using a dip net or hand picking. Sampling effort should be sufficient to be sure that macroinvertebrates found are representative of the stream reach. This should take around 20 minutes. Habitats to survey include silt, sand, gravel, cobble, boulder, vegetation, wood, and other structures. Both high and low velocity areas should be sampled. Collected macroinvertebrate samples should first all be placed in the same bucket with water. Prior to subsampling, the sample is visually scanned for giant water bugs, whose venomous bite could cause excruciating pain and paralysis with no known treatments (Haddad et al. 2010). The bucket was then subsampled with a small minnow net, which was used to select both small and large macroinvertebrates to limit bias. A total of 300 +- 60 macroinvertebrates were selected in the subsample (MDEQ 2008). When a subsample is collected with the minnow net, all macroinvertebrates in the subsample must be counted.

Then, forceps or a pipette were used to hand-select taxa from the macroinvertebrate sample that were not collected in the subsample and add them to the subsample. This post-subsample search should take 3-5 minutes. Special care was taken for snails that are difficult to dislodge. This ensures that the subsample represents all taxa. The numbers of each taxa identified from the subsample were recorded on a datasheet. Taxa that were found by searching post-subsample should be recorded by marking 1 on the datasheet and circling it. Nine metric scores were calculated based on the data, and these were summed to create a final score.

Macroinvertebrates that cannot be identified in the field were preserved in a 70% ethanol solution and identified in a lab using identification guides (Bouchard et al. 2004; Merritt et al. 2008). Macroinvertebrate indices were analyzed using the methodology outlined in Procedure 51.

Fish were sampled using a backpack electroshocker following safety procedures in an upstream direction. Per MDEQ (2008) protocol, sites were sampled with a single pass in a section of stream that was ten times the width of the stream. Fish were identified to species, enumerated, and released back into the stream. Sampling effort should be sufficient to sample all fish species at a site, with a goal of at least 100 individual fish per site. This will take 30 minutes and cover 100-300 feet of stream. Sampling is terminated if less than 100 fish are found after 45 minutes, and analyses are done with the smaller sample size. Fish were placed in replenished tubs of water to keep them alive, and the methods allow optional battery aerators, though we did not use them. Species, total length, and any anomalies were noted for each individual fish caught. Ten metrics were calculated based on the data, and a quantitative Fish Score was calculated as the sum of the metric scores. Sites were automatically considered poor if less than 50 fish were caught or anomalies were found on greater than two percent of fish. Additionally, the percentage of salmonids relative to total fish collected needs to exceed 1% for coldwater streams like Indian Mill Creek to meet their designation.

Stream habitat was also evaluated using the Procedure 51 method. Substrate, instream cover, channel morphology, and riparian and bank structure were all evaluated. Nine metrics were calculated off the habitat data, and those were summed to create a station habitat score. The quality of habitat was then rated as excellent, good, fair, or poor based off the habitat score.

Sediment Loading Study

Sampling locations for the sediment loading study were chosen to reflect the spatial variation in the Indian Mill Creek watershed: some sites were in the upper watershed's farmland, some sites were in the lower urban areas, and two sites were on tributaries. The study occurred monthly from May through September 2017, with two additional sampling events immediately after storms. Sampling involved three components: stream discharge, suspended sediment, and bedload. One replicate sample and one field blank were collected during six sampling events, for a total of six replicates and six blanks.

Discharge was estimated following methods in the Annis Water Resources Institute's Ruddiman Creek Quality Assurance Project Plan (Muskegon, MI, USA). Transects were established perpendicular to stream flow at each location to measure discharge. Water depth and velocity were measured at seven to ten equally-spaced points along the transects to form compartments. Water velocity was measured according to USGS protocols (Rantz 1982) using a Marsh-McBirney Flow Mate 2000 flow meter attached to a top-setting wading rod during each field visit. When water depth was less than 2.5 ft, velocity was measured at 0.6 x depth. When depth exceeded 2.5 ft, velocity was measured at 0.2 x and 0.8 x depth.

An average depth, velocity, and width were calculated for each compartment. These were multiplied together to get an individual discharge estimate for each compartment. The compartments were then summed to provide an estimate of total stream discharge.

Bedload subsamples (2-min duration) were collected using a 3”x3” Helley-Smith sampler at 3-7 equally-spaced points across the stream at each site (6-14 min total sampling time) following methods in the Annis Water Resources Institute’s Ruddiman Creek Quality Assurance Project Plan (Muskegon, MI, USA). The instantaneous bedload transport rate (Qb) in kg s^{-1} was calculated using the equation Qb=(Mb/T)*(1/N)*(W/0.076m). Mb is the total mass of bedload sediment in kg; T is the subsample duration in seconds; N is the number of subsamples; W is the wetted width in meters, and 0.076m is the width of the 3x3” Helley-Smith sampler opening.

Bedload samples were carefully processed at the Annis Water Resources Institute, keeping track of sample ID at all times. First, the initial weight of each filter paper was measured on a scale. A unique identifier was written on the filter in pencil that corresponds to each site. The bedload sample of each site was processed individually. The sample was rinsed from the net and bag into a 9”x13” glass tray. Then, the tray was poured over the filter using a vacuum flask and funnel to drain water. The tray and funnel were rinsed into the filter. Then, samples were dried in a 105F degree oven for one hour or longer, depending how long it took a sample to dry. Samples were cooled for one hour or longer. Then, the weight of a sample was measured on a scale and the initial weight of the filter was subtracted.

Water samples for suspended sediment concentration were collected in duplicate 500-ml polyethylene bottles in the center of the stream at mid depth and stored at 4°C. Sample holding times were consistent with Environmental Protection Agency recommendations (USEPA 1983). Suspended sediment concentration was analyzed by method 2540 D (Greensberg et al. 1992).

Filter papers were labeled. Then, the initial mass of the filter was taken using a tared scale. Next, the filter papers were placed on a vacuum filtration device. A known volume of water sample, usually around 500 mL, was run through the filter. The filtration device was rinsed with deinonized water between samples. The filter was then protected in a tinfoil purse and placed in a 105 degree Fahrenheit oven for one hour. It was then moved to a desiccator box to cool for at least one hour. Then, the final mass of the sample was recorded. Suspended sediment concentration was calculated by subtracting the initial mass by final mass and dividing by volume filtered.

Suspended sediment loading was estimated by multiplying the sediment concentration of the water sample by the stream discharge at each site. A conversion factor was applied so the function becomes milligrams per day.

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