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Sediment Remediation Impacts on Macroinvertebrate Community Structure: Assessing the Success of Urban Stream Restoration

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SEDIMENT REMEDIATION IMPACTS ON MACROINVERTEBRATE COMMUNITY STRUCTURE: ASSESSING THE SUCCESS OF URBAN STREAM RESTORATION

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By

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To

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“When one tugs at a single thing in nature, he finds it attached to the rest of the world.”

John Muir

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ABSTRACT

SEDIMENT REMEDIATION IMPACTS ON MACROINVERTEBRATE COMMUNITY STRUCTURE: ASSESSING THE SUCCESS OF URBAN STREAM RESTORATION

by Laurie Beth Nederveld

Land use practices altering the natural landscape have resulted in the widespread degradation of stream ecosystems and the need for urban stream restorations. While a number of studies have evaluated the success of these stream restoration efforts, few have assessed the recovery of macroinvertebrate communities following the remediation of contaminated sediments. The purpose of my study was to evaluate the impact of sediment remediation activities on macroinvertebate abundance, diversity, and richness to determine the success of stream restoration in Ruddiman Creek, a small stream in the Muskegon Lake watershed. During my investigation, macroinvertebrate samples were collected from all available habitat types at three study sites and three reference (control) sites using a Before-After Control-Impact (BACI) sampling design. Ryerson Creek, an urban system considered less disturbed with respect to heavy metal and organic contaminants, served as a reference stream within the Muskegon Lake watershed. Physical measurements, chemical analyses of water samples, and hydrologic measurements in Ruddiman and Ryerson Creeks were used to assess habitat and water quality changes as a result of remediation activities. This investigation concluded that although remediation activities resulted in a significant initial decline in macroinvertebrate abundance, diversity, and richness, the macroinvertebrate community recovered to pre-remediation conditions rapidly. After approximately one and a half years of recovery, stream quality of study sites had not approached reference conditions. The family-level biotic index (FBI), however, suggested marked improvement in stream quality, as indicated by a greater abundance of sensitive taxa (%) and a richer macroinvertebrate community. My findings suggest that chronically degraded water quality and hydrologic impairments continued to negatively influence the macroinvertebrate community and that additional restoration activities are needed to improve the ecological integrity of the Ruddiman Creek watershed.

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CHAPTER I

INTRODUCTION

Streams flowing over landscapes modified by anthropogenic activity are often subject to varying degrees of impairment. Industrial, urban, and agricultural practices directly and indirectly degrade aquatic ecosystems (Allan and Flecker 1993, Lenat and Crawford 1994, Johnson et al. 1997, Stepenuck et al. 2002, Davis et al. 2003, Roy et al. 2003, Cooper et al. 2009). As a result, urban streams are often characterized by unnatural flow regimes, excessive fine sediment and nutrient loads (e.g. phosphorus and nitrogen), and accumulations of heavy metals and organic contaminants (Lenat and Crawford 1994, Trimble 1997, Paul and Meyer 2001, Meyer et al. 2005, Tang et al. 2005, Walsh et al. 2005). Impacts to stream invertebrates can include growth reductions (Mattingly et al. 1981, Broekhizen et al. 2001), bioaccumulation of metals and organic contaminants (Meador et al. 1995, Cain and Luoma 1998, Beltman et al. 1999, Maret et al. 2003), diminished reproduction (Mulvey and Diamond 1991), increased drift (Poff and Ward 1991, Shaw and Richardson 2001), and altered competition behavior (Vuori 1994). Subsequently, macroinvertebrate communities exposed to degraded environmental conditions will exhibit decreases in abundance, diversity, and richness (McElravy et al. 1989, Scrimgeour and Winterbourn 1989, Stepenuck et al. 2002, Roy et al. 2003, Kaller and Hartman 2004). Because macroinvertebrate communities are sensitive to environmental degradation, macroinvertebrate assessments are effective indicators of general habitat and water quality (Jones and Clark 1987, Muotka et al. 2002, Schmidt et al. 2002, Fitzpatrick et al. 2004, Brown et al. 2005) and have been used extensively in this capacity (Carter and Resh 2001, Bonada et al. 2006, Olsen and Peck 2008). Macroinvertebrate assessments have often been preferred over physical and chemical analyses alone because they offer a more reliable indication of long-term, rather than just immediate, stream condition.

Long-term abundance, diversity, and richness of macroinvertebrate communities have been shown to increase following stream remediation when populations were previously stressed by heavy metal and organic contaminants (Hoiland et al. 1994, Nelson and Roline 1996, Adams et al. 2005). Sediment removal, however, can often result in an immediate degradation of macroinvertebrate communities (Bonvincini et al. 1985, Quigley and Hall 1999, and Gilkinson et al. 2005). Kelaher et al. (2003) demonstrated the potential for sediment removal activities to cause unpredicted habitat changes resulting in long-term alterations to macroinvertebrate communities. While a number of studies have documented impacts to macroinvertebrate community composition by anthropogenic contaminants, few studies have described macroinvertebrate recovery after the remediation of contaminated sediments.

My study stream, Ruddiman Creek, drains an urbanized watershed located in Muskegon County, Michigan, USA, and flows to a coastal drowned river mouth of Lake Michigan. Due to degraded stream conditions and public health concerns expressed by residents, the U.S. EPA Great Lakes National Program Office and the MDEQ conducted a 14.2 million dollar project to dredge and remove contaminated sediments in the Ruddiman Creek watershed. In addition to sediment remediation, limited hydrologic improvements were completed including the construction of a detention basin and restoration of braided stream patterns. Between September 2005 and April 2006, Ruddiman Pond and seven sections of the main branch of Ruddiman Creek were dredged removing 68,477 m^3 of contaminated sediments (Janesak 2006). The primary objective of this remediation project was to “reduce the relative risks to humans, wildlife, and aquatic life” (Hilgeman 2005).

The objective of my study was to evaluate the impact of sediment remediation on the biotic community of Ruddiman Creek, using macroinvertebrates as the primary indicator. We compared trends in Ruddiman Creek to Ryerson Creek, a reference (control) stream used to control for temporal variability within the region. Ryerson Creek was a system also impacted by urbanization but considered less disturbed with respect to heavy metal and organic contaminants. Remediation of Ruddiman Creek was intended to improve existing stream conditions through the dredging and removal of contaminated sediments. My investigation used macroinvertebrate collections, physical measurements, chemical analyses of water samples, and hydrologic measurements in Ruddiman and Ryerson Creeks to evaluate impacts of remediation activities on stream condition. I hypothesized that anthropogenic contaminants were the main factor contributing to the degraded macroinvertebrate community, and therefore, remediation of contaminated sediments in Ruddiman Creek would result in an increase in macroinvertebrate abundance, diversity, and richness compared to pre-remediation and reference conditions. Since only a limited degree of hydrologic restoration was performed, flashy stream condition had the potential to impact macroinvertebrate community after sediment remediation. My investigation builds on the findings of Cooper et al. (2009) for nearby Little Black Creek, which demonstrated measurably degraded macroinvertebrate communities, attributed to contaminated sediments, and the need for remediation efforts. This investigation provides important information concerning the success of urban stream restoration and will assist in determining whether further restoration strategies are necessary.

CHAPTER II

MATERIALS AND METHODS

STUDY AREA

Study sites were located in the Ruddiman Creek watershed (13.0 km^2), an urbanized area located primarily within the city of Muskegon (Fig. 1). A mix of residential (54%), commercial (20%), industrial (11%), and transportation (1%) development covered the landscape, once dominated by white pine – white oak forests (MSU RSGIS 1998). Natural features of the watershed included three stream reaches (north branch, 0.55 km; west branch, 2.14 km; and main branch, 3.09 km), a pond (0.04 km^2), and several forested, emergent, and shrub-scrub wetland areas (0.12 km^2). Roughly 44% of the main branch was enclosed in storm sewer before emerging from a 2.5-meter outfall. The main branch flowed into Ruddiman Pond before ultimately discharging into Muskegon Lake, a drowned river mouth lake. Three study sites were located on the main branch, upstream from Ruddiman Pond. Sites 2 and 3 underwent sediment remediation in December 2005 and February 2006, respectively, while Site 1 was influenced by upstream structures intended to moderate hydrologic extremes (Table 1).

Wastewater and stormwater discharges, improper hazardous waste disposal, and groundwater contamination contributed to the degradation and contamination of Ruddiman Creek (Rediske 2002). As a result of past land use practices, numerous pollutants were introduced into the stream, including benzo(a)pyrene, heavy metals (e.g. cadmium, chromium, and lead), and polychlorinated biphenyls (PCBs) (Snell

Environmental Group 2000, Earth Tech, Inc. 2002). Heavy metal and PCB concentrations exceeded the Michigan Department of Environmental Quality's (MDEQ) site-specific sediment quality criteria for human contact and aquatic life (Rediske 2004). In addition to sediment contamination, hydrologic instability and stormwater pollution had been identified as priority issues by local and state governments (Wuycheck 1989, Nederveld 2005). Due to the degraded warm water fishery and macroinvertebrate community, Ruddiman Creek was placed on the Michigan 303(d) List of Impaired Waters (Wuycheck and Creal 2002). Similar impairments to stream condition as a result of sediment contamination and degraded water quality had been observed in the nearby tributaries of the Mona Lake watershed (Cooper et al. 2009).

Ryerson Creek served as an urbanized reference stream during this investigation. Its watershed (21.0 km^2) was located partially within the city of Muskegon and included similar land use types as the Ruddiman Creek watershed, but had a greater undeveloped area (36% vs. 14% of the total watershed area) (MSU RSGIS 1998). Ryerson Creek's main branch was fed by two tributaries and flowed through a small pond feature before discharging into Muskegon Lake. Although Ryerson Creek also had been affected by nonpoint source pollution, it was considered less degraded than Ruddiman Creek in terms of heavy metal and organic contaminants. Three reference sites, with comparable characteristics to those on Ruddiman Creek (substrate, habitat, and location relative to stream mouth), were located on Ryerson Creek upstream of its confluence with Muskegon Lake.

MACROINVERTEBRATE SAMPLING

Macroinvertebrates (invertebrates>0.5 mm in length) were collected from Ruddiman and Ryerson Creeks following a sampling procedure originally applied in wetlands by Burton et al. (1999) and Uzarski et al. (2004). This methodology was modified for use in stream systems, similar to the methodology used by Cooper et al. (2009). Sample replicates were collected from available habitat types at each site. Habitats were of three potential types: 1) *Typha*, 2) overhanging riparian vegetation (e.g. *Phalaris arundinacea*, *Impatiens capensis*), and 3) floating/submergent vegetation (e.g. *Elodea canadensis, Nymphaea odorata,* and *Potamogeton foliosus*). The distribution of samples collected at each site was proportional to the amount of available habitat. At Sites 2 and 3 on each stream, three replicate samples were collected for each habitat type since habitats were approximately equally distributed. At Site 1 on each stream, six replicates were collected from riparian habitat (2/3 of available in-stream habitat) and 3 replicates were collected from floating/submergent habitat (1/3 of available in-stream habitat) (Table 2). Thus, a total of nine sample replicates were collected from each site on five sample dates: August 2005 (one month prior to dredging activities), May 2006 (one month after dredging was complete), August 2006, May 2007, and August 2007. No more than eight days elapsed between sample collections from Ruddiman Creek and Ryerson Creek during any collection period.

Field crews used D-frame dip nets containing a 0.5-mm mesh to collect macoinvertebrates. To ensure sampling of all localized habitats, sampling involved sweeps from the streambed through the entire water column, while keeping in contact with the vegetation. When present in the immediate vicinity of a sample replicate location, gravel and cobble over 5 cm in diameter were hand washed in dip nets to dislodge macroinvertebrates. Contents of dip nets were emptied into white pans and organisms were picked from each sample replicate for one-half-person-hour. After picking during this time period, organisms were tallied and picking continued to the next multiple of 50, unless the nominal maximum of 150 organisms had been reached (Burton et al. 1999, Uzarski et al. 2004). If counts were well below 50, 100, or 150 organisms after one-half-person-hour, then picking continued until the next multiple of 25. This ensured that enough organisms were picked to provide a representative sample, but meant that sampling effort tended to be greater at lower densities. Collected specimens, including semi-aquatic insects, were preserved in 70% ethanol and later sorted and identified to family. Exceptions included taxa more difficult to identify, Oligochaeta and Hydrachinida, which were identified to order. Taxonomic keys developed by Chu (1992), McCafferty (1998), Merritt and Cummins (1996), and Thorp and Covich (2001) were used for identification.

MACROINVERTEBRATE METRIC CALCULATIONS

Since macroinvertebrate community composition did not differ by habitat type in initial non-metric multidimensional scaling (NMDS) analyses ($p>0.05$), macroinvertebrate metrics were calculated using composite sample replicates. A composite sample replicate included one sample replicate from each of the three available habitat types, resulting in three composite sample replicates per site per sample date.

Since sample replicates could include 25 to 150 organisms, based on the sampling methodology, composite sample replicates could range from 75 to 450 total organisms.

Macroinvertebrate data were used to calculate relative macroinvertebrate abundance, sensitive taxa abundance (%), and taxon diversity and richness. Catch per unit of effort was used as a measure of relative macroinvertebrate abundance. Family-level tolerance scores (Hilsenhoff 1988, Bode 1988, Bode et al. 2002) were used to differentiate sensitive taxa (tolerance score 0 to 5) from tolerant taxa (tolerance score 6 to 10). A range of diversity statistics, Pielou's Evenness Index (J) (Pielou 1975), Simpson's Diversity Index (1/D) (Magurran 1988), and Shannon's Diversity Index (H') (Magurran 1988), were chosen to ensure the best estimate of macroinvertebrate diversity. Diversity metrics were chosen to include statistics biased toward richness and evenness (and dominance). Taxon richness was calculated to assist in interpreting diversity statistics and represented the total number of taxa per composite sample replicate.

A family-level biotic index (FBI), as described by Hilsenhoff (1988), was utilized to assess stream condition over time based on macroinvertebrate composition. The FBI is intended to be a rapid, field-based assessment, weighting the relative abundance of each family by its tolerance score to determine a total community score. Tolerance scores for families of stream arthropods in the western Great Lakes region were used (Hilsenhoff 1988), and supplemented with values for northeastern streams (Bode 1988, Bode et al. 2002). Both sets of family-level tolerance scores were on a ten-point scale. Tolerance scores corresponded to a family's sensitivity to poor habitat and water quality; lower scores indicated better stream condition. Family-level identifications have been used in previous investigations of stream condition (Linke et al. 1999, Mattsson and Cooper 2006). Although finer taxonomic resolution can be more useful, because tolerance can change within a genus (Resh and Unzicker 1975, Hilsenhoff 1987), coarser resolutions show similar patterns in response to habitat and water quality gradients (Somerfield and Clarke 1995, Vanderklift et al. 1996), especially in degraded environments (Olsgard et al. 1998).

Since Site 3 on Ruddiman Creek exhibited strong wetland characteristics prior to stream restoration (sites were heavily influenced by adjacent *Typha*-dominated marshes), an index of biotic integrity (IBI) (Uzarski et al. 2004) developed for Lake Michigan fringing coastal wetlands was applied at Site 3 on both streams to compare with FBI scores. Site 3 on Ryerson Creek was also influenced by adjacent *Typha*-dominated marshes throughout the investigation. The invertebrate-based Wetland IBI specifies metrics by vegetation zone. Since the *Typha* zone metrics from the original Wetland IBI (Burton et al. 1999) were eliminated in the revised Wetland IBI (Uzarski et al. 2004), the metrics specified for the inner *Scirpus* zone were used. This zone was characterized by dense *Scirpus*, limited *Pontedaria* and submergents, and a lack of wave action, and therefore, was the most similar to the vegetation observed at Site 3 on each stream. Family-level macroinvertebrate data was used to calculate all 13 specified metrics, including Odonata taxa richness, relative Gastropoda abundance (%), and Shannon's Diversity Index (H'), among others.

PHYSICAL AND CHEMICAL MEASUREMENTS

Physical habitat measurements included visual estimates of substrate composition, woody debris cover, and in-stream vegetation cover. Substrate assessments were based on the visible substrate layer, and included estimates of sand, fine and coarse particulate organic matter, and coarse fragments (>2 mm). Woody debris cover included dead woody material over 1.0 cm in diameter. Physical parameters were recorded at a representative 0.1-m^2 area within each macroinvertebrate sample replicate location on six dates: November 2005 (i.e. one and three months prior to dredging at Sites 3 and 2, respectively), May 2006, August 2006, November 2006, May 2007, and August 2007.

Chemical parameters were recorded at four locations on each stream during macroinvertebrate collection dates and three storm events: September 7, 2007 (10-month storm), June 5, 2008 (1-year storm), and September 4, 2008 (10-year storm). A Hydrolab DataSonde 4a (Hydrolab Corporation, Loveland, Colorado) was used to determine DO, DO saturation (%), oxidation-reduction potential (ORP), pH, specific conductance, temperature, and total dissolved solids (TDS). Water samples were collected in 1-liter acid-washed polyethylene bottles and analyzed for alkalinity, ammonium-N, chloride, nitrate-N, soluble reactive phosphorus (SRP), sulfate, and total phosphorus (TP). One duplicate water sample was collected from one randomly chosen location on each stream on each sample date. Repeated measurement errors were 3.0% for alkalinity, 7.6% for sulfate, 8.5% for nitrate-N, 9.2% for ammonium-N, 11.0% for SRP, 12.9% for TP, and 13.3% for chloride. Laboratory analytical procedures and quality assurance/control followed recommended procedures outlined in Standard Methods for the Examination of

Water and Wastewater (APHA 1998). Laboratory detection limits were 0.01 mg/L for nitrate-N, SRP, and TP, 0.02 mg/L for ammonium-N, and 1 mg/L for chloride and sulfate. Matrix spikes and matrix spike duplicates for all analytes were analyzed at a frequency of 10% with precision limits of ±15% relative standard deviation and accuracy control limits of 90-110% recovery.

HYDROLOGY

To compare variations in streamflow rates and volumes in response to storm events, measured hydrographs for Ruddiman and Ryerson Creeks were constructed using field data obtained from Site 3 on each stream. These sites were located near the mouth of each stream, had the potential to demonstrate the flashiness of the system, and were not influenced by Muskegon Lake levels. A Marsh-McBirney Flo-Mate Model 2000 Flow Meter and top-setting wading rod were used to measure stream velocities and water column depths on a uniform and stable reach at Site 3 on each stream. Measurements were taken during the five macroinvertebrate collection dates (base flow conditions), one date each in September 2007 and September 2008 (10-month storm and 10-year storm), and two dates in November 2008 (base flow conditions). Stream discharge was calculated using the midsection method (Hauer and Lamberti 2006).

In-situ Level TROLL 300 data-loggers were installed at Site 3 on each stream to record stage height continuously for two months during fall 2008. Field measurements of water column depth were used to calibrate stage height data from the water level recorder. Rainfall data for this period were obtained from the Muskegon County Airport weather station (43°10’12”N, 86°14’9”W), located approximately 7 km southeast of the sites on Ruddiman Creek. If stage height data indicated a storm event, but rainfall was not recorded at the Muskegon County Airport, rainfall data from the Muskegon Yacht Club’s weather station (43°13'7"N, 86°19'21"W) was used. A La Crosse Technology Weather Station (model 2317U) was maintained at this site, located on Lake Michigan’s shoreline, 4 km northwest of the sites on Ruddiman Creek.

Using stage height and discharge data collected in the field, a stage – discharge relationship was derived. This rating curved allowed for discharges to be predicted at stages other than those measured. Hydrographs for Site 3 on each stream were constructed for November 11 to 17, 2008, and September 2 to 7, 2008. During these time periods, minor storm events (≤ 1.1 cm within a 1 to 11-hour duration) and a 10-year storm event (8.9 cm within a 22-hour duration) were observed.

STATISTICAL ANALYSES

Substrate composition data were not statistically analyzed since these data were meant to be descriptive only. To compute inter-rater reliability estimates for the three field crews, intraclass correlation coefficients were calculated using relative macroinvertebrate abundance count data (SPSS version 14.0, Chicago, Illinois). This analysis was possible because field crews 1, 2, and 3 collected corresponding sample replicates 1, 2, and 3; for example, field crew 1 always collected sample replicate 1. Intraclass correlations ranged between 0.74 and 0.81, indicating that composite sample replicates were highly correlated and, therefore, field crews were consistent in their collection methods.

One-way repeated measures ANOVA was used to analyze water quality parameters (SPSS version 14.0, Chicago, Illinois); stream was treated as a fixed factor, sample date was treated as a repeated measure, and the four samples taken from different sites on each stream were treated as replicates. Two-way repeated measures ANOVA was used to analyze sensitive taxa abundance (%), taxon diversity and richness, FBI and Wetland IBI scores, and woody debris and vegetative cover (%). Composite sample replicates explained the variability within sites, the experimental factor. For these analyses, stream was treated as a fixed factor, sites were nested within the stream variable, and sample dates were treated as a repeated measure. Shapiro-Wilk tests were used to test for normality, $p>0.05$. When sphericity could not be assumed (Mauchly's test statistic was significant, $p<0.05$) the Greenhouse-Geisser corrected F-statistics were used. Means were compared using Bonferroni post-hoc tests (SPSS version 14.0, Chicago, Illinois). Differences were considered significant when $p<0.05$.

Since my sampling methodology followed a Before-After Control-Impact paired design (BACIP) (Stewart-Oaten et al. 1986, Smith 2002), I analyzed differences in FBI scores following the BACIP model to assess the impact of sediment remediation activities on stream quality. The BACI approach tests whether a potential change in the environment is due to a stressor rather than temporal or regional variability. The BACIP analysis of FBI scores was used in addition to two-way repeated measures ANOVA test on actual FBI scores to evaluate score differences between streams and sites over time.

One-way ANOVA was used to test whether FBI score differences between the control stream and the impact stream changed following remediation activities. Since sites explained the variability within streams, FBI scores for composite sample replicates were averaged by site. Sample date was treated as a fixed factor rather than a repeated measure due to the small sample size (n=3) after averaging by site. Levene's test was used to assess equality of variance. Shapiro-Wilk tests were used to test for normality, $p>0.05$. In addition to comparing stream differences by date, site differences were also compared using a separate analysis. One-way repeated measures ANOVA was used to evaluate differences in FBI scores among control sites and impact sites. Since composite sample replicates explained the variability within sites, FBI scores for composite sample replicates were analyzed. Sample date was treated as a repeated measure and site was treated as a fixed factor.

NMDS (Clarke 1993) was used to measure dissimilarity in macroinvertebrate composition among sites over time. To compare the immediate effects of remediation activities, August 2005 and May 2006 macroinvertebrate community compositions (based on individual taxa abundance (%)) were compared. Similarly, August 2005 and August 2007 communities were compared to assess overall changes during the project investigation. PC-ORD version 5.0 (McCune and Mefford 2006) was used to compile NMDS ordination plots. Analyses were completed with the Bray–Curtis distance measure, 500 maximum iterations, an instability criterion of 1E-8, six starting axes, 250 real runs, and 250 randomized runs using the Monte Carlo test. Dimensionality was selected based on the lowest final stress value among the best solutions for each dimension. Final stress values of selected dimensions were lower than that for 95% of the randomized runs.

Permutational multivariate ANOVA, or PERMANOVA, (Anderson 2001, McArdle and Anderson 2001) was used to determine if changes in macroinvertebrate composition, as indicated by NMDS ordinations, were significant, $p<0.05$ (Anderson 2001). For this analysis factors were crossed, stream and date were treated as fixed, and sampling sites were nested within streams. Probabilities were based on unrestricted permutation of raw data (4,999 permutations) and Bray-Curtis dissimilarities. Pairwise comparisons (999 permutations) were used to determine significant differences among means. Similarly, PERMANOVA also was used to analyze for differences in relative macroinvertebrate abundance using count data.

CHAPTER III

RESULTS

PHYSICAL AND CHEMICAL CHARACTERISTICS

Streambed sediments of sampled habitats in Ryerson Creek were primarily sand overlain with fine particulate organic matter and, to a lesser extent, coarse particulate organic matter (Fig. 2b). Coarse fragments were observed in minimal amounts, typically at Sites 2 and 3. Substrate composition was similar in Ruddiman Creek (Fig. 2a). Coarse fragments were found in the greatest amounts at Site 2, where cobble had been placed during sediment remediation. Substrate alterations resulted in greater habitat heterogeneity in Ruddiman Creek, but over time the deposition of fine sediment from upstream sources began to cover the cobble habitat (Nederveld, personal observation).

Woody debris cover between streams was comparable over the study period (Fig. 3, Table 3). Prior to remediation activities, in-stream vegetation cover at Ruddiman Creek sites was more extensive than at Ryerson Creek sites (Fig. 4, Table 3). As a result of remediation activities, *Typha*, floating/submergent, and riparian cover were significantly reduced at all Ruddiman Creek sites. By August 2006, riparian vegetation recovered at Ruddiman Creek sites and was comparable to Ryerson Creek sites, while recovery of T*ypha* and floating/submergent vegetation was not observed until the end of the investigation in August 2007.

During base flow conditions, inorganic contaminants (total dissolved solids and sulfates) were typically greater in Ruddiman Creek as compared to Ryerson Creek (Table

4, 5). Concentrations of these inorganic contaminants became reduced during storm flows, a trend less apparent in Ryerson Creek, which experienced smaller storm flow volumes relative to catchment size. In both study and reference systems, storm flow events resulted in elevated TP and SRP concentrations and undersaturation of DO; DO supersaturation was typically observed during base flow conditions.

HYDROLOGY

During the project investigation, base flow rates were typically greater in Ryerson Creek, the larger catchment. At comparable sites (Site 3), Ryerson Creek's mean base flow rate was 0.13 $\pm$ 0.03 $m^3 s^{-1}$ (mean $\pm$ SE), while Ruddiman Creek's mean was 0.06 $\pm$ 0.01 $m^3 s^{-1}$. According to the rating curves developed for each stream, predicted base flow rates for stage heights observed at Site 3 on each stream, ranged from 0.02 to 0.15 $m^3 s^{-1}$ for Ruddiman Creek and 0.05 to 0.35 $m^3 s^{-1}$ for Ryerson Creek.

Storm events observed during the study period resulted in a rapid response in Ruddiman Creek's measured hydrograph, while Ryerson Creek showed a more prolonged response for storms of similar intensity and duration (Fig. 5, 6). Ruddiman Creek's hydrograph also had a steeper recession limb indicating that it drained more rapidly between periods of rainfall. While minor storms typically produced comparable peak flow rates between streams (Fig. 5), larger storms produced substantially greater peak flow rates in Ruddiman Creek (Fig. 6). For the 10-year storm event occurring on September 4, 2008, Ruddiman Creek initially peaked at 0.89 $m^3 s^{-1}$, but Ryerson Creek's discharge only peaked at 0.46 $m^3 s^{-1}$. Peak flow rates in Ruddiman Creek reached their greatest rate 19 hours later at 1.3 $m^3 s^{-1}$, while Ryerson Creek never exceeded 0.54 $m^3 s^{-1}$.

MACROINVERTEBRATE COMMUNITY STRUCTURE

Macroinvertebrate taxa collected during the project investigation represented four phyla (Annelida, Anthropoda, Mollusca, and Platyhelminthes) and eight classes. Forty-six taxa were collected in Ruddiman Creek, while forty taxa were collected in Ryerson Creek. In each stream, Gammarids dominated sample collections and the three most abundant taxa on any given date accounted for at least 2/3 of the total collection (Fig. 7).

The community composition of Ruddiman Creek, dominated by Gammaridae and Chironomidae at pre-remediation, became dominated primarily by Chironomidae, Oligochaeta, and Gammaridae in May 2006 directly following remediation (Fig. 7a). By August 2007, the macroinvertebrate community composition resembled that of pre-remediation, however, Chironomidae and Physidae (tolerance score range 6-8) represented a smaller percentage of sample collections and Gammaridae, Haliplidae, and Planariidae (tolerance score range 4-6) represented a greater percentage. In comparison, sample collections from Ryerson Creek were dominated by Gammaridae and, to a far lesser extent, Asellidae (Fig. 7b). Gammaridae (tolerance score = 4) represented the vast majority of the sensitive taxa collected in Ryerson Creek. These organisms were observed to be typically larger and more robust in the reference stream in contrast to the study stream.

The NMDS ordination (a two-dimensional solution), comparing macroinvertebrate community structure of Ruddiman and Ryerson Creeks between August 2005 and May 2006, revealed distinctly different community compositions ($p<0.01$, PERMANOVA). The ordination plot (Fig. 8) demonstrated a shift in community composition at Ruddiman Creek sites after stream remediation, resulting in a community that was less similar to that of Ryerson Creek sites. Samples collected from Site 1 on Ruddiman Creek, which was not dredged, were different from the remaining collection in May 2006 following remediation. The NMDS ordination (a two-dimensional solution), comparing macroinvertebrate community structure between August 2005 and August 2007, indicated that Ruddiman Creek sites had become more similar to Ryerson Creek sites after one and a half years of recovery (Fig. 9), but that community compositions were still markedly different ($p<0.02$, PERMANOVA).

MACROINVERTEBRATE ABUNDANCE

Relative macroinvertebrate abundance (counts per sample) represented a catch per unit effort. According to the sampling methodology, composite sample replicates could range from 75 to 450 total organisms, but the actual range was 59 to 546 organisms. Relative macroinvertebrate abundance was comparable between streams prior to remediation activities (Fig. 10a, Table 6). In May 2006 after the remediation of Ruddiman Creek, however, abundance counts significantly declined at Site 3, the most heavily remediated site (Fig. 10b, Table 6). While macroinvertebrate abundance at Sites 1 and 2 were primarily unaffected in May 2006, the percentage of sensitive taxa (taxa with tolerance scores between 0 and 5) declined, especially at Site 1 (Fig. 11b, Table 6). A second significant decline in relative macroinvertebrate abundance was observed following a storm event in May 2007 when abundance counts at all three Ruddiman Creek sites, especially Sites 1 and 2, declined noticeably in comparison to the previous sample date (Fig. 10b, Table 6). At the conclusion of the project, after one and half years of recovery, relative macroinvertebrate abundance in Ruddiman Creek was comparable to levels at pre-remediation and those observed in Ryerson Creek (Fig. 10a, Table 6). The abundance of sensitive taxa (%), however, had markedly grown between August 2005 and August 2007 ($p<0.01$, Bonferroni), a trend not observed in the reference stream (Fig. 11a, Table 6).

MACROINVERTEBRATE DIVERSITY

Diversity of the macroinvertebrate community inhabiting Ruddiman Creek was markedly greater than that of Ryerson Creek throughout the investigation, except in May 2006 following stream remediation (e.g. Fig. 12a, Table 6). Greater diversity within Ruddiman Creek corresponded to greater overall taxon richness and evenness of the macroinvertebrate community. Taxon evenness within Ruddiman Creek was fairly constant over time (Fig. 13a, Table 6), and did not significantly differ between August 2005 and August 2007 ($p>0.05$, Bonferroni). Mean taxon richness, however, grew from 14.3 in August 2005 to 19.3 in August 2007 ($p=0.01$, Bonferroni) to include additional Hemiptera and Coleoptera taxa, while mean richness in Ryerson Creek ranged only from 11.1 to 12.6 ($p>0.05$, Bonferroni) during this period (Fig.14a, Table 6). Regarding

Ruddiman Creek study sites, Site 3 was significantly more diverse than Sites 1 and 2 initially but by August 2006 was comparable to Sites 1 and 2.

INDICES OF BIOTIC INTEGRITY

FBI scores indicated better stream condition in Ryerson Creek as compared to Ruddiman Creek throughout the two-year investigation (Fig. 15a, Table 6). The macroinvertebrate community of Ryerson Creek demonstrated “good” to “fair” stream quality, while macroinvertebrates inhabiting Ruddiman Creek signified “fair” to “fairly poor” stream quality.

In May 2006 following stream remediation activities, both FBI and Wetland IBI scores indicated a decline in habitat quality at Ruddiman Creek sites (Fig. 15b, Table 6 – Sites 1 and 2, Fig. 16, Table 6 – Site 3), but degradation was not significant (Fig. 17, Table 7). Recovery of Ruddiman Creek’s macroinvertebrate community in August 2006 following remediation activities was rapid and substantial (Fig. 17a, Table 7), especially at Site 1 (Fig. 17b, Table 7), which was not dredged. During post-remediation, FBI scores indicated that Ruddiman Creek sites experienced significant improvement in stream quality between August 2005 and August 2007 ($p<0.01$, Bonferroni). Based on FBI and Wetland IBI scores, Site 3 fell between “fair” (Fig 15b, Table 6) to “moderately impacted” (Fig. 16, Table 6) habitat quality at the end of the investigation in August 2007. Site 2 remained in fair condition, while Site 1 conditions had improved demonstrating the highest stream quality (Fig. 15b, Table 6). Reference and study sites farther upstream showed higher water quality than sites farther downstream.

According to FBI scores at pre-remediation, Site 3 on Ruddiman Creek was markedly degraded in stream quality ("fairly poor") as compared to Sites 1 and 2 at pre-remediation (Fig. 15b, Table 6), despite the fact that Site 3 demonstrated the greatest habitat heterogeneity in comparison. Wetland IBI scores, however, indicated that Site 3 on Ruddiman Creek was only "mildly impacted" suggesting Site 3, which exhibited strong wetland characteristics, was the least degraded study site initially (Fig. 16, Table 6). In May 2006 immediately following the dredging and removal of sediment the Wetland IBI, unlike the FBI, indicated a decline in habitat quality at Site 3 (Fig. 15b, 16, 17b; Table 6, 7). In August 2006, following stream remediation, both the FBI and Wetland IBI suggested habitat recovery, but FBI and Wetland IBI trends differed during post-remediation. The FBI indicated a slight decline in habitat quality after August 2006 but signified overall improvement by the end of the investigation (Fig. 15b, 17; Table 6, 7), while Wetland IBI trends indicated Site 3 was still in the process of recovery in August 2007 (Fig.16, Table 6).

Ruddiman Creek site trends in FBI scores followed site trends in Gammaridae populations, with few exceptions. That is, if Gammaridae percentages increased at a site between dates, FBI scores indicated an improvement in stream quality. Although Gammaridae percentages were significantly greater in the reference stream, Ryerson Creek values did not follow this trend. Rather, values corresponded to site trends in sensitive taxa abundance (%), with few exceptions (Fig. 11a, Table 6). This taxa group consisted primarily of Gammaridae, Baetidae, and Haliplidae.

CHAPTER IV

DISCUSSION

While the macroinvertebrate community inhabiting Ruddiman Creek was more diverse and rich (Fig.12a, 14a; Table 6) in comparison to the reference stream prior to stream remediation, it typified a more degraded aquatic system. The NMDS ordination (Fig. 8) revealed that macroinvertebrate compositions of the study and reference streams were distinct. While the reference system was dominated by taxa considered sensitive to water and habitat quality degradation (primarily Gammaridae), Ruddiman Creek was dominated by tolerant organisms (Fig. 11a, Table 6). Similarly, other studies have observed reduced abundances of sensitive taxa in response to activities related to urban development (Stepenuck et al. 2002, Davis et al. 2003, Roy et al. 2003) and sediment contamination (Cooper et al. 2006). The FBI revealed that macroinvertebrate community composition in Ruddiman Creek was indicative of degraded water quality conditions, while the community composition of the reference stream indicated higher stream quality (Fig. 15a, Table 6). Findings indicated that greater diversity of the macroinvertebrate community does not always imply better stream condition when the community is dominated by taxa tolerant to poor habitat and water quality. Since disturbance does not always result in a strong effect on species diversity (Mackey and Currie 2001), I suggest a note of caution when using diversity metrics as a sole indicator of stream condition.

Following the dredging and removal of sediment in Ruddiman Creek, subsequent changes to macroinvertebrate community structure (Fig. 7a, 8) indicated immediate degradation of stream quality (Fig. 15a, Table 6), but changes were not marked (Fig. 17a, Table 7). Since remediation activities were the most substantial at Site 3, greater reductions in relative macroinvertebrate abundance at this study site were expected (Fig. 10b, Table 6). All three study sites experienced an immediate reduction in the percentage of sensitive taxa (primarily Gammaridae) leaving sample collections at study sites dominated by smaller numbers of tolerant organisms (Fig. 11b, Table 6). Similarly, other studies (Bonvincini et al. 1985, Quigley and Hall 1999, and Gilkinson et al. 2005) have found significant and immediate changes to macroinvertebrate community structure as a result of dredging activities. Despite initial declines in macroinvertebrate abundance, recovery was rapid. Harvey (1986) also observed rapid recovery of invertebrate communities after stream substrate alterations by suction dredging. Rapid recovery in Ruddiman Creek was attributed to recolonization of dredge areas by macroinvertebrates from undisturbed areas. Bonvincini et al. (1985) and Gjerløv et al. (2003) also observed rapid recolonization of denuded substrate by benthic macroinvertebrates in response to disturbance.

Post-remediation changes at Site 1, which did not undergo dredging, were attributed to upstream remediation activities (Table 1). Construction activities likely resulted in the downstream propagation of fine sediment. Deposition of this fine sediment had the potential to smother the streambed accounting for reductions in aquatic vegetation (Edwards 1969, Brookes 1986) and changes in macroinvertebrate community structure (Wood and Armitage 1997, Shaw and Richardson 2001, Kaller and Hartman 2004, Rabeni et al. 2005). Results demonstrated that upstream remediation activities were able to significantly impact areas approximately 300 meters downstream. Subsequent improvements in stream condition at Site 1 (Fig. 15b, 17b; Table 6, 7), after one and a half years of recovery, are ascribed to the installation of upstream structures intended to reduce hydrologic extremes. The capability of detention basins to effectively regulate stormwater flows has been documented (Roesner et al. 1988). Appropriate detention basin sizing, however, has been shown to be critical for minimizing stormwater runoff impacts to receiving waters (Heitz et al. 2000). Although conditions at Site 1 improved after sediment remediation, and are attributed to hydrologic improvements, hydrologic instability continued to impact the system. Results suggested that the detention basin may be too small to address the systemic hydrologic fluctuations observed in Ruddiman Creek in response to major storm events.

Although Site 2 demonstrated slow but steady improvement during post-remediation between August 2006 and August 2007 (Fig. 15b, Table 6), its recovery appeared to stagnate relative to the other two study sites, possibly due its location downstream of the Glenside Boulevard stream crossing. Road/stream crossings and associated road networks are known to negatively impact aquatic ecosystem processes by influencing peak flow rates and sediment transport, as well as contributing heavy metals and organic contaminants (Jones et al. 2000, Trombulak et al. 2000). The two parallel culverts upstream of Site 2 produced a strong, channelized flow that scoured the streambed during major storm events and caused redeposition of sediments downstream (Nederveld, personal observation). The impact of the road/stream crossing upon the aquatic ecosystem may limit the potential for further stream improvement in stream condition at Site 2.

According to the FBI, Site 3 was the most degraded study site at pre-remediation (Fig. 15b, 17b; Table 6, 7), but the Wetland IBI indicated high habitat quality (Fig. 16, Table 6). Site 3 not only exhibited complex habitat heterogeneity at pre-remediation, but strong wetland character. Wetlands are characterized by anaerobic soils, and therefore, naturally low DO levels (White 1985, Mitsch 1989). Davis et al. (1999) found that natural stream stressors, such as low DO, could render a biotic index ineffective at distinguishing between reference and impact sites. The lowest DO % saturation values were typically observed at Site 3 on Ruddiman Creek (Table 4, 5). It seemed likely that the poor habitat quality conditions indicated at Site 3 by the FBI were due to the influence of adjacent wetlands and not greater stream degradation. High habitat quality conditions indicated by the Wetland IBI at Site 3, therefore, seemed more probable. Site 3's growing departure from reference conditions based on the FBI during post-remediation after August 2006 (Fig. 15b, 16, Table 6, 7), may have been the result of returning hydrophytic vegetation.

Overall improvement in habitat quality at Site 3, as indicated by the FBI (Fig. 15b, 17b; Table 6, 7), was presumably a result of the removal of contaminated sediments. Heavy metal concentrations have been shown to influence the structure of macroinvertebrate communities (Winner et al. 1980, Clements 1994, Clements et al. 2000, Maret et al. 2003, Pollard and Yuan 2006, Doi et al. 2007). After the reduction of heavy metals and organic contaminants, macroinvertebrate communities have demonstrated recovery (Hoiland et al. 1994, Nelson and Roline 1996, Adams et al. 2005). Post-dredge samples collected from the main branch of Ruddiman Creek in 2006 were reported as meeting MDEQ's site-specific sediment cleanup criteria for cadmium, chromium, lead, PCBs, and benzo(a)pyrene, with the exception of average PCB concentrations at two dredge areas upstream of study sites (Janesak 2006). Despite sediment contamination improvements at Site 3, unstable hydrology remained a concern based on hydrologic observations (Figs. 5, 6).

By the conclusion of the investigation in August 2007, the macroinvertebrate community inhabiting Ruddiman Creek was comparable to that of pre-remediation. In comparison to Ryerson Creek, Ruddiman Creek remained similar in abundance (Fig. 10a, Table 6), greater in diversity and richness (Fig. 12a, 14a; Table 6), and remained more degraded in stream condition (Fig. 15a, 17a; Table 6, 7). While post-remediation changes were not substantial, only one and a half years of recovery were assessed. In an extensive literature review, Niemi et al. (1990) found most freshwater systems exposed to major disturbances recovered within three years, but that habitat alterations resulted in more prolonged recovery times. While Ruddiman Creek recovered to pre-remediation conditions quickly, reference conditions were not approached during the investigation. Nelson and Roline (1996) found that following the reduction of in-stream metal contaminants aquatic invertebrate communities were comparable to reference sites within approximately two years. Chadwick et al. (1986), however, found limited recovery ten years after reduced in-stream metal concentrations due to remaining metal-contaminated sediments. Since this assessment followed only approximately one and a half years of recovery, a long-term bioassessment is necessary to determine ultimate changes to the aquatic ecosystem of Ruddiman Creek as the result of sediment remediation.

Although stream quality did not approach that of Ryerson Creek after one and a half years of recovery, changes indicating limited stream improvement did occur. Macroinvertebrate community structure in Ruddiman Creek became more similar to that of Ryerson Creek (Fig. 9). Although Gammaridae dominated the initial and final Ruddiman Creek collections, the remaining community changed to include greater representation of sensitive species, including Haliplidae and Planariidae, and less representation of tolerant taxa, Chironomidae and Physidae (Fig. 7a). The macroinvertebrate community became markedly more rich (Fig. 14a, Table 6) and more abundant in sensitive taxa (%) (Fig. 11a, Table 6), indicating a significant improvement in stream quality (Fig. 15a, Table 6).

Water quality results suggested that Ruddiman and Ryerson Creeks typified the generally degraded water quality conditions often associated with urban streams (Lenat and Crawford 1994, Trimble 1997, Paul and Meyer 2001, Meyer et al. 2005, Tang 2005, Walsh et al. 2005). Substantial variation in DO % saturation values and elevated nutrient concentrations during storm flows, as well as inorganic contamination of Ruddiman Creek, were indicative of chronic water quality impairments in study and reference streams. DO % undersaturation is characteristic of elevated biochemical oxygen demand, while supersaturation is symptomatic of high rates of photosynthesis driven by nutrient enrichment (Correll 1998). Nutrient enrichment of urban watersheds, as a result of nonpoint source pollution, is a common impairment to aquatic ecosystems (Carpenter et al. 1998). Although water quality parameters were able to provide a “snapshot” of stream conditions at a specific place and time, I found they were not a good indicator of overall changes in stream quality following remediation. The results of my study point to the necessity of bioassessments in augmenting one’s understanding of the aquatic environment and stream condition (Shapiro et al. 2008).

While water quality conditions during storm flows did not differ substantially between streams, hydrologic responses did, suggesting that episodic habitat disruption was the primary cause of differences noted in macroinvertebrate communities. Measured hydrographs for Ruddiman Creek’s main branch peaked early and had relatively high peak flow rates in comparison to Ryerson Creek (Fig. 5, 6). Peak flow rates and volumes were substantial during the 10-year storm event observed on September 4, 2008. At the time of the investigation, over 4/5 the Ruddiman Creek watershed was developed (MSU RSGIS 1998) and 2/3 was drained by storm sewer infrastructure connected to surface waters. In comparison, the Ryerson Creek watershed, although urbanized on its western portion, had a significantly greater percentage (36% vs. 14% of total watershed area) of forests and open lands (MSU RSGIS 1998), as well as very limited storm sewer inputs. Because urban development tends to alter the hydrologic regime (Richards 1990, Poff et al. 1997), often increasing the magnitude and frequency of high flows (Konrad and Booth 2005), I attribute the hydrologic conditions observed in Ruddiman Creek to the substantial urbanization of the watershed.

Similar to investigations by Pratt et al. (1980) and Gray (2004), I found hydrologic alterations had the potential not only to alter the hydrologic regime, but to significantly impact macroinvertebrate communities. During the May 2007 collection, a previous high flow event, indicated by freshly undercut banks, had noticeably scoured the streambed at Ruddiman Creek sites. Macroinvertebrate abundances were similarly reduced at all study sites, consistent with a systemic effect. Elevated flow rates can disrupt aquatic habitat (Scullion and Stinton 1983, Gurtz et al. 1988, Wood and Armitage 1997) and subsequently dislodge, damage, or kill aquatic invertebrates (Sagar 1983, Feminella and Resh 1990). Despite evidence of some hydrologic improvement at Site 1, as a result of the installation of upstream structures, hydrologic fluctuations appeared to have the potential to influence macroinvertebrate community structure at Ruddiman Creek sites through habitat modifications. Restoring the hydrologic regime to more natural conditions will likely be necessary to improve ecological integrity within Ruddiman Creek. Likewise, Cooper et al. (2009) reported that hydrologic improvements were needed in addition to sediment remediation to enhance aquatic invertebrate communities of a similarly impacted system also located in western Michigan.

Ruddiman Creek was remediated primarily to reduce elevated levels of heavy metal and organic contaminants. My investigation sought to assess the impacts on the macroinvertebrate community from the dredging and removal of contaminated sediments to assess changes in stream condition. When both physical and chemical disturbances are present in streams, however, physical factors have been shown to have a more dominant role in structuring the macroinvertebrate community (Peeters et al. 2001, Carew et al. 2007). While heavy metals impact macroinvertebrate communities (Clements 1994, Clements et al. 2000, Pollard and Yuan 2006, Doi 2007, Cooper et al. 2009), my data show that it is difficult to attribute specific chemical contaminant impacts to a location when physical disturbances, such as extreme hydrologic fluctuations, are also present.

CHAPTER V

CONCLUSIONS

My investigation evaluated the success of the Ruddiman Creek remediation project in terms of its impact on the macroinvertebrate community. An assessment of the biotic community would otherwise not be directly evaluated. I observed an initial reduction in macroinvertebrate abundance, diversity, and richness of Ruddiman Creek after sediment remediation due to direct removal of organisms and habitat destruction. Macroinvertebrate metrics were very sensitive to this effect, whereas physical and chemical water quality measurements did not detect these impacts. After approximately one and a half years of recovery, stream quality of Ruddiman Creek had not reached reference conditions. The FBI suggested significant improvement in stream quality did occur, however, as indicated by a greater abundance of sensitive taxa (%) and a richer macroinvertebrate community. Further improvements in stream condition appear to be limited by chronically degraded water quality and hydrologic instability. Results of my study indicate that additional restoration actions are needed to improve the ecological integrity of Ruddiman Creek, and similarly impacted urban stream systems. Future restoration strategies will need to consider and address the interrelated and complex factors associated with sediment contamination, degraded water quality, and altered hydrology to effectively achieve further ecological improvement within this urban system.

Table 1. Remediation practices implemented between September 2005 and April 2006 at study sites. Ruddiman Creek, Muskegon County, Michigan, USA.

Site	Dredge Depth	Dredge Volume	Replacement Sediment (depth)	Upstream Remediation Practices	Riparian Area Restoration Activities
1	0 m	None	None	Dredging, detention basin, channel armoring, riprap wing dams (2)	None
2	0.31 m	176 m^3	Sand (0.15 m), 7.62-cm cobble (0.15 m)	Dredging, braided stream pattern, riffle structure	Seeding
3	0.61 - 1.83 m	931 m^3	Sand (0.15 m)	Dredging, riffle structure	Bank slope grading, seeding

Table 2. Vegetation types present at study and reference sites. Ruddiman and Ryerson Creeks, Muskegon County, Michigan, USA.

Stream	Site 1	Site 2	Site 3
Ruddiman Creek	floating/submergent, riparian	floating/submergent, riparian, *Typha*	floating/submergent, riparian, *Typha*
Ryerson Creek	floating/submergent, riparian	floating/submergent, riparian, *Typha*	floating/submergent, riparian, *Typha*

Table 3. Significance probabilities from two-way repeated measures ANOVA for vegetative and woody debris cover data. Ruddiman and Ryerson Creeks, Muskegon County, Michigan, USA.

	Source of Variation				
Variable	Date[1]	Stream[1]	Site(Stream)[1]	Date x Stream[2]	Date x Site(Stream)[2]
Vegetative Cover (%)	**<0.01**[a]	0.86	**0.01**	**<0.01**[a]	**<0.01**[a]
Woody Debris Cover (%)	0.17[a]	0.17	0.07	0.76[a]	0.41[a]

Significant p values (<0.05) are given in bold face.

[a] The Greenhouse-Geisser correction was applied to the degrees of freedom to produce a valid F-ratio since data violated the sphericity assumption.

[1] p values for main effects.

[2] p values for interaction effects.

Table 4. Mean water quality parameter values. Ruddiman and Ryerson Creeks, Muskegon County, Michigan, USA.

Stream	Date	Discharge ($m^3 s^{-1}$) ‡	Alkalinity (mg/L)	Cl (mg/L)	DO (mg/L)	DO% Saturation	NH_3-N (mg/L)	$N0_3$-N (mg/L)	ORP (mV)	pH	$S0_4$ (mg/L)	Specific Conductance (µS/cm)	SRP (mg/L)	TDS (g/L)	Temperature (°C)	TP (mg/L)
Base flow conditions																
Ruddiman	8/12/05†	0.1	46.7	77.6	6.4	74.8	0.1	1.0	339.0	7.9	36.9	557.2	0.0	0.4	22.0	0.1
	5/22/06	0.1	39.6	152.9	10.6	109.2	0.1	1.6	355.5	8.0	58.6	990.6	0.0	0.6	15.1	0.0
	8/14/06	0.0	40.9	150.8	9.4	102.0	0.1	1.2	346.5	8.0	46.9	1010.5	0.0	0.6	20.2	0.0
	5/21/07	0.0	39.0	194.3	13.7	139.7	0.1	1.3	331.0	8.3	50.8	1048.3	0.0	0.7	15.2	0.0
	8/13/07	0.0	33.2	140.8	7.5	85.5	0.2	1.2	337.5	7.8	45.3	870.1	0.0	0.6	20.2	0.0
	Mean	**0.1**	**39.9**	**143.2**	**9.5**	**102.2**	**0.1**	**1.2**	**341.9**	**8.0**	**47.7**	**895.3**	**0.0**	**0.6**	**18.5**	**0.0**
	SE	**0.0**	**2.2**	**18.8**	**1.3**	**11.1**	**0.0**	**0.1**	**4.2**	**0.1**	**3.5**	**89.6**	**0.0**	**0.1**	**1.4**	**0.0**
Ryerson	8/19/05†	0.1	61.6	148.4	7.0	74.4	0.0	1.1	568.3	7.6	25.6	840.0	0.0	0.5	17.2	0.1
	5/19/06	0.3	26.0	88.7	10.6	102.3	0.1	1.0	360.8	8.0	24.5	587.6	0.0	0.4	12.6	0.0
	8/21/06	0.1	35.4	158.8	8.3	87.1	0.1	1.2	369.0	7.8	32.1	887.2	0.0	0.6	16.7	0.0
	5/22/07	0.1	35.1	162.3	10.0	104.1	0.1	1.0	345.0	8.1	23.0	826.6	0.0	0.5	16.3	0.0
	8/14/07	0.1	36.6	179.5	8.0	83.4	0.1	1.2	359.8	7.9	31.8	908.5	0.0	0.6	16.2	0.0
	Mean	**0.1**	**38.9**	**147.5**	**8.7**	**90.3**	**0.1**	**1.1**	**400.6**	**7.9**	**27.4**	**810.0**	**0.0**	**0.5**	**15.8**	**0.0**
	SE	**0.0**	**6.0**	**15.5**	**0.7**	**5.7**	**0.0**	**0.1**	**42.1**	**0.1**	**1.9**	**57.6**	**0.0**	**0.0**	**0.8**	**0.0**
Storm flow conditions																
Ruddiman	9/7/07	0.5	8.7	16.3	6.9	79.5	0.2	0.4	333.5	7.7	8.0	135.1	0.1	0.1	21.5	0.1
	6/5/08	NA	34.5	12.0	7.1	75.9	0.1	0.2	313.3	7.5	4.5	106.2	0.0	0.1	17.6	0.1
	9/4/08	1.3	42.0	18.3	8.8	90.1	0.2	0.4	315.8	7.9	6.5	100.3	0.1	0.1	16.3	0.2
	Mean	**0.9**	**28.4**	**15.5**	**7.6**	**81.8**	**0.2**	**0.3**	**320.9**	**7.7**	**6.3**	**113.8**	**0.0**	**0.1**	**18.5**	**0.1**
	SE	**0.4**	**10.1**	**1.8**	**0.6**	**4.3**	**0.0**	**0.1**	**6.4**	**0.1**	**1.0**	**10.7**	**0.0**	**0.0**	**1.6**	**0.1**
Ryerson	9/7/07	0.5	11.6	39.3	5.5	61.6	0.1	0.4	336.0	7.5	8.5	254.7	0.1	0.2	20.5	0.2
	6/5/08	NA	43.0	28.5	5.5	58.3	0.1	0.3	317.0	7.2	6.5	171.3	0.0	0.1	17.3	0.1
	9/4/08	0.5	47.3	35.8	7.4	74.1	0.1	0.3	314.8	7.9	8.0	232.8	0.1	0.2	15.5	0.1
	Mean	**0.5**	**34.0**	**34.5**	**6.1**	**64.7**	**0.1**	**0.3**	**322.6**	**7.5**	**7.7**	**219.6**	**0.1**	**0.1**	**17.8**	**0.1**
	SE	**0.0**	**11.2**	**3.2**	**0.6**	**4.8**	**0.0**	**0.0**	**6.7**	**0.2**	**0.6**	**25.0**	**0.0**	**0.0**	**1.4**	**0.0**

† Pre-remediation sample date.

‡ Discharges by date represent actual values and not means.

Table 5. Significance probabilities from one-way repeated measures ANOVA for water quality data measured during base and storm flow conditions between August 2005 and August 2007. Ruddiman and Ryerson Creeks, Muskegon County, Michigan, USA.

	Source of Variation					
	Base flow conditions			Storm flow conditions		
Variable	Date[1]	Stream[1]	Date x Stream[2]	Date[1]	Stream[1]	Date x Stream[2]
Alkalinity (mg/L)	**<0.01[a]**	0.64	**0.03[a]**	**<0.01**	**0.01[a]**	0.49
Cl (mg/L)	**<0.01[a]**	0.72	**<0.01[a]**	0.08	**<0.01**	0.58
DO (mg/L)	**<0.01**	0.17	**<0.01**	**<0.01**	**<0.01**	0.43
DO% Saturation	**<0.01**	0.12	**<0.01**	**<0.01**	**<0.01**	0.47
NH_3-N (mg/L)	0.22[a]	0.07	0.23[a]	**<0.01**	**<0.01**	**0.01**
$N0_3$-N (mg/L)	**0.02[a]**	0.18	**<0.01[a]**	**<0.01**	0.62	**0.04**
ORP (mV)	**<0.01**	**<0.01**	**<0.01**	**0.04**	0.83	0.95
pH	**<0.01[a]**	0.19	**0.01[a]**	**<0.01**	**<0.01**	**0.01**
SO_4 (mg/L)	**0.04[a]**	**<0.01**	**0.01[a]**	**0.03**	0.31	0.71
Specific Conductance (µS/cm)	**0.01[a]**	**<0.05**	**<0.01[a]**	**<0.01**	**<0.01**	**<0.01**
SRP (mg/L)	0.31[a]	**0.01**	0.54[a]	**<0.01**	0.42	**0.01**
TDS (g/L)	**0.01[a]**	**<0.05**	**<0.01[a]**	**<0.01**	**<0.01**	**<0.01**
Temperature (°C)	**<0.01**	**0.01**	**<0.01**	**<0.01**	**<0.01**	**<0.01**
TP (mg/L)	**0.02[a]**	0.09	0.28[a]	**<0.01**	0.50	**0.03**

Significant p values (<0.05) are given in bold face.

[a] The Greenhouse-Geisser correction was applied to the degrees of freedom to produce a valid F-ratio since data violated the sphericity assumption.

[1] p values for main effects.

[2] p values for interaction effects.

Table 6. Significance probabilities from ANOVA and PERMANOVA for macroinvertebrate community data. Ruddiman and Ryerson Creeks, Muskegon County, Michigan, USA.

	Source of Variation				
Metric	Date[1]	Stream[1]	Site(Stream)[1]	Date x Stream[2]	Date x Site(Stream)[2]
Family-level Biotic Index[a]	**<0.01**	**<0.01**	**0.04**	**<0.01**	**0.03**
Pielou's Evenness (J)[a]	**<0.01**	**<0.01**	**0.02**	**<0.01**	0.08
Relative Abundance (Count)[b]	**<0.01**	**<0.01**	**<0.01**	**<0.01**	**<0.01**
Sensitive Taxa (%)[a]	**<0.01**	**<0.01**	0.15	**<0.01**	**<0.01**
Shannon's Diveristy (H')[a]	0.41	**<0.01**	**0.03**	**<0.01**	0.15
Simpson's Diversity (1/D)[a]	0.23	**<0.01**	0.07	**<0.01**	**0.03**
Taxon Richness[a]	**<0.01**	**<0.01**	0.06	**<0.01**	**0.02**
Wetland Index of Biotic Integrity[c]	**<0.01**	0.47	-	**<0.05**	-

Significant p values (<0.05) are given in bold face.

[a] Composite sample replicates analyzed using two-way repeated measures ANOVA.

[b] Composite sample replicates analyzed using PERMANOVA.

[c] Composite sample replicates for Site 3 on both streams analyzed using two-way repeated measures ANOVA.

[1] p values for main effects, if applicable.

[2] p values for interaction effect, if applicable.

Table 7. Significance probabilities from Before-After Control-Impact paired model for family-level biotic index scores. Ruddiman and Ryerson Creeks, Muskegon County, Michigan, USA.

	Source of Variation		
Metric	Date[1]	Site[1]	Date x Site[2]
Family-level Biotic Index[a]	**0.02**	**-**	**-**
Family-level Biotic Index[b]	**<0.01**	0.97	**<0.01**

Significant p values (<0.05) are given in bold face.

[a] Mean composite sample replicates by site analyzed using one-way ANOVA.

[b] Composite sample replicates analyzed using one-way repeated measures ANOVA.

[1] p values for main effects, if applicable.
[2] p values for interaction effect, if applicable.

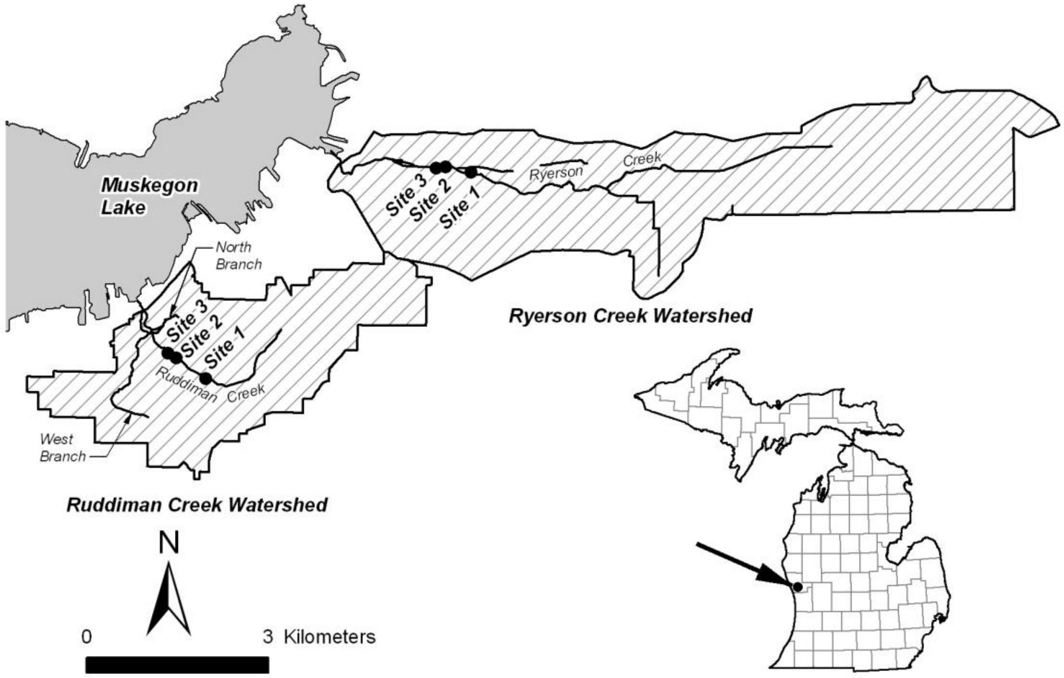


Fig. 1. Map of study and reference sites located within the Ruddiman Creek and Ryerson Creek watersheds, Muskegon County, Michigan, USA.

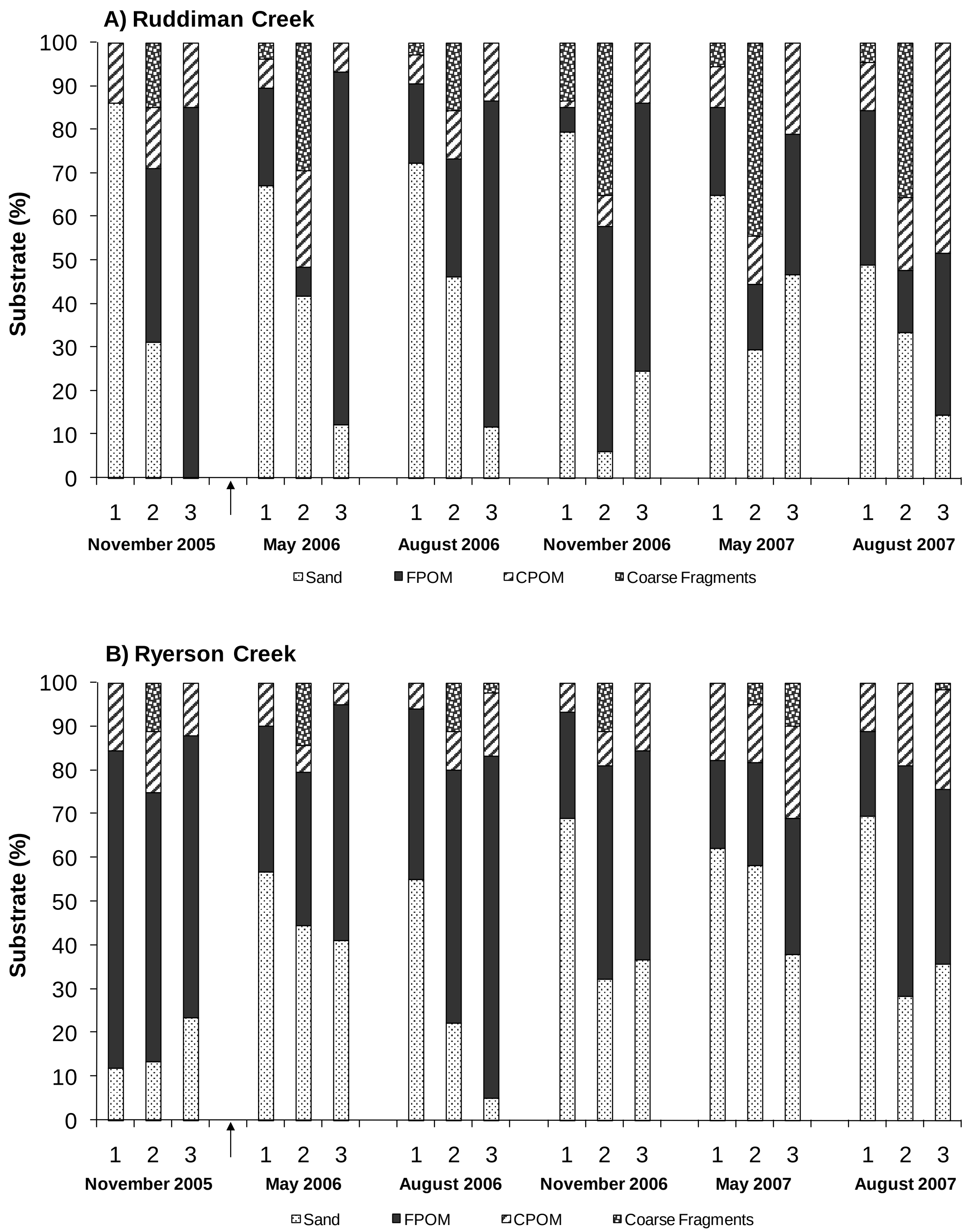


Fig. 2. Mean substrate composition at Ruddiman and Ryerson Creek sites, including sand, fine particulate organic matter (FPOM), coarse particulate organic matter (CPOM), and coarse fragments. Arrow (↑) indicates the division between pre and post-remediation.

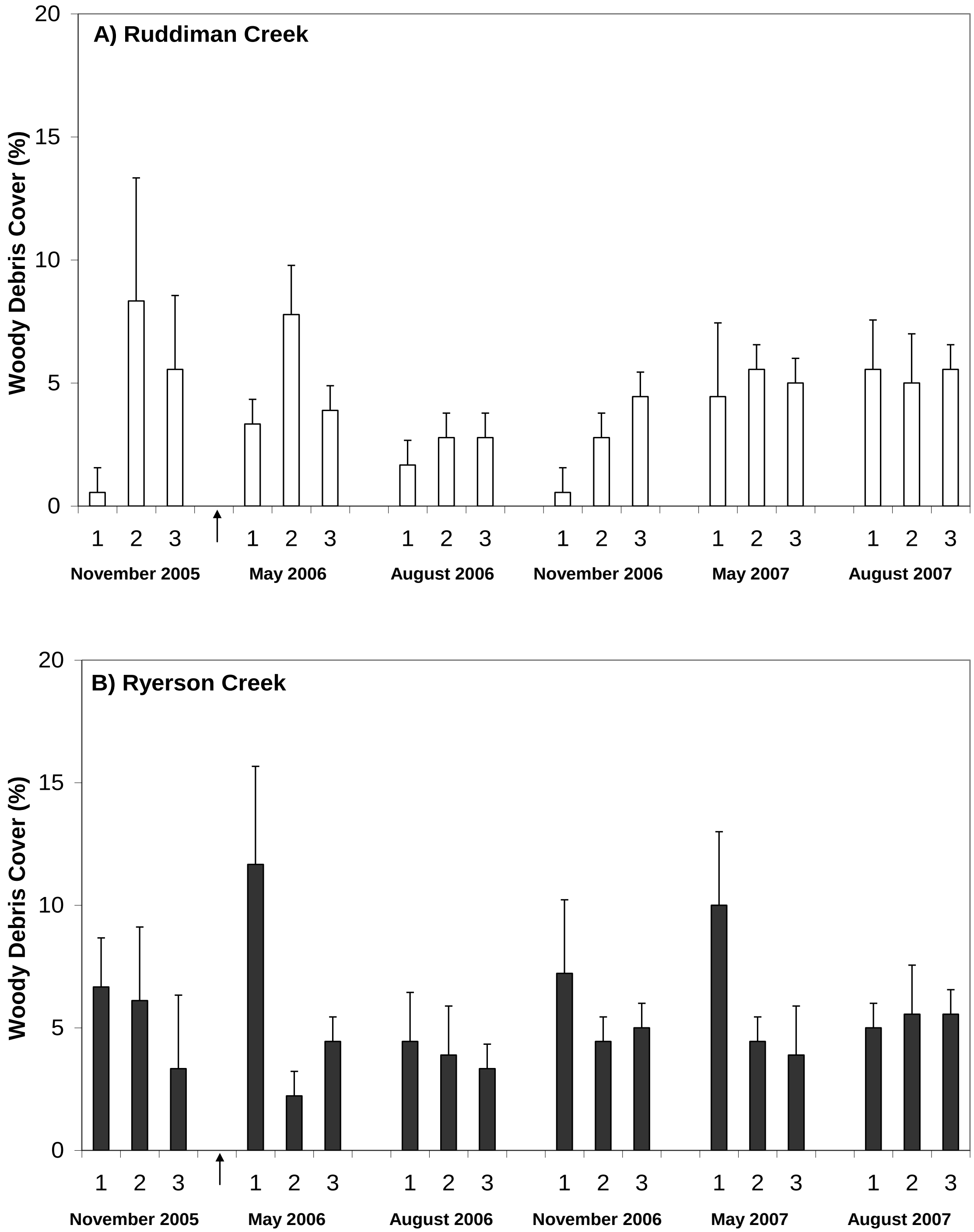


Fig. 3. Mean (±SE) woody debris cover at Ruddiman and Ryerson Creeks sites. No significant site x date interaction effect was revealed by two-way ANOVA. Arrow (↑) indicates the division between pre and post-remediation.

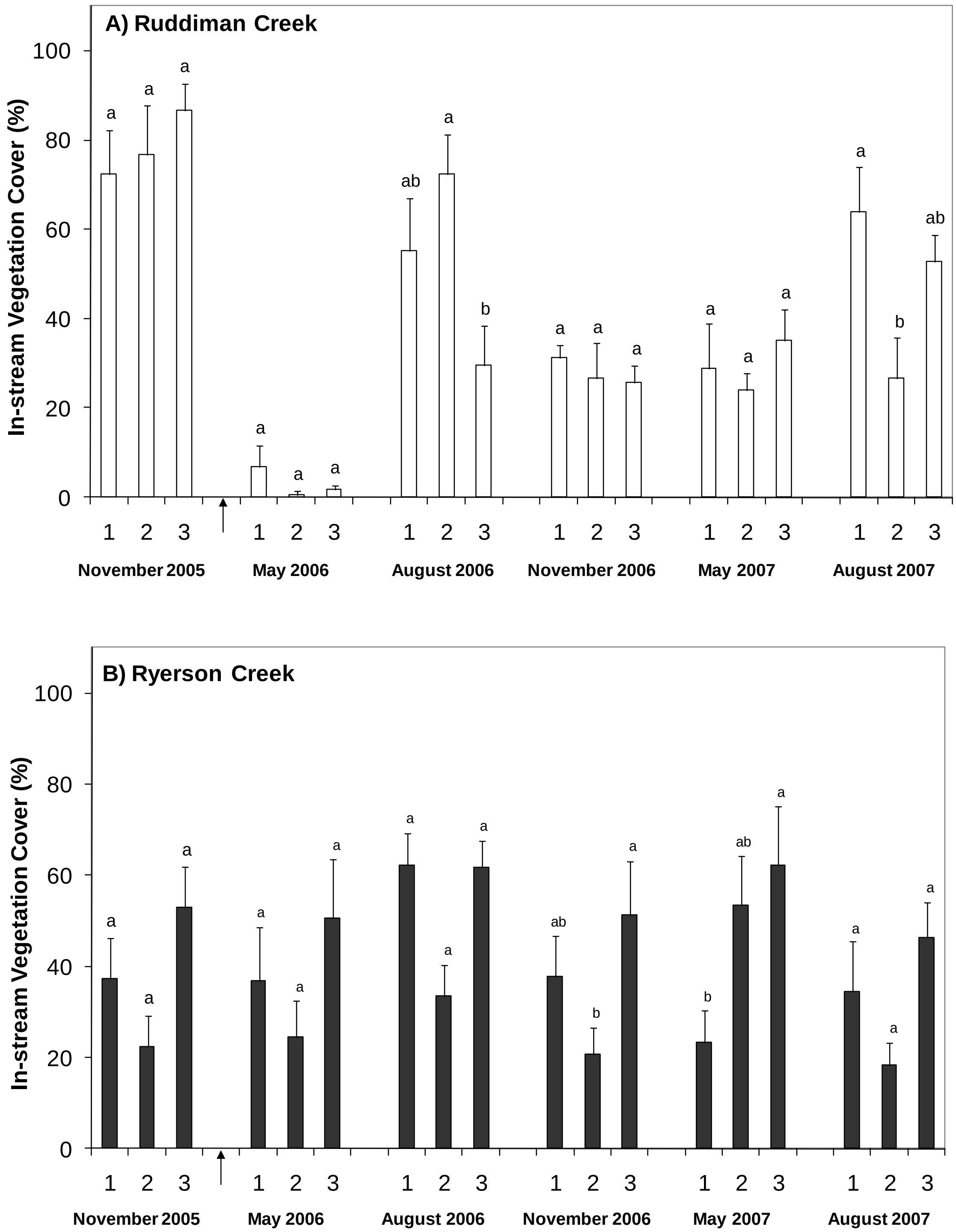
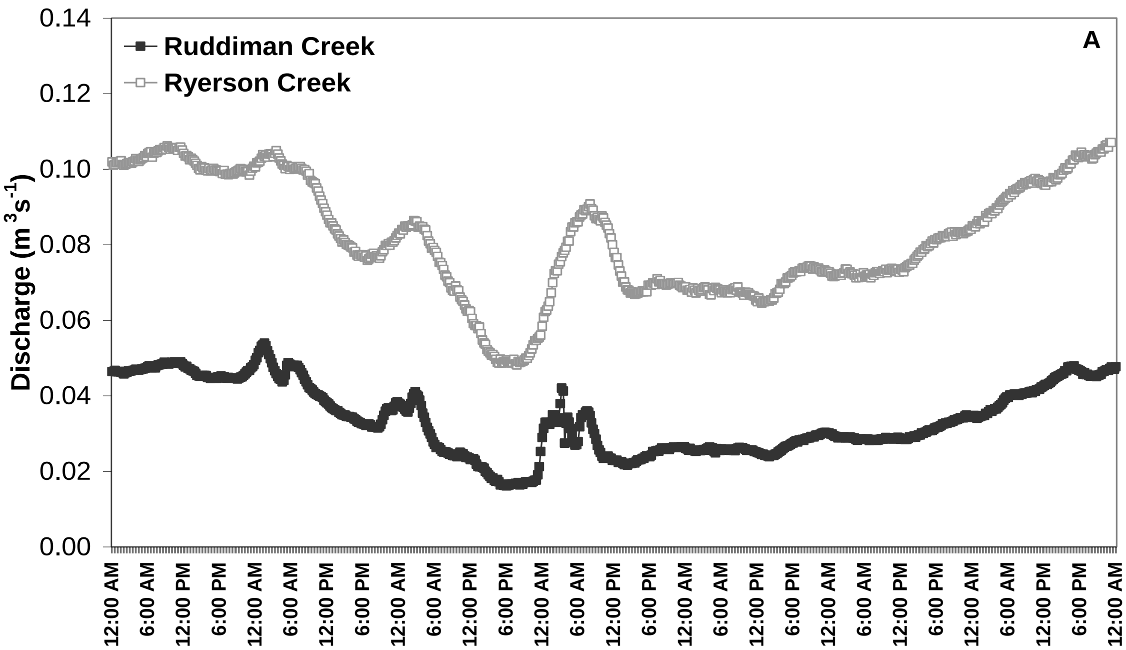


Fig. 4. Mean (±SE) in-stream vegetation cover at Ruddiman and Ryerson Creeks sites. Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.



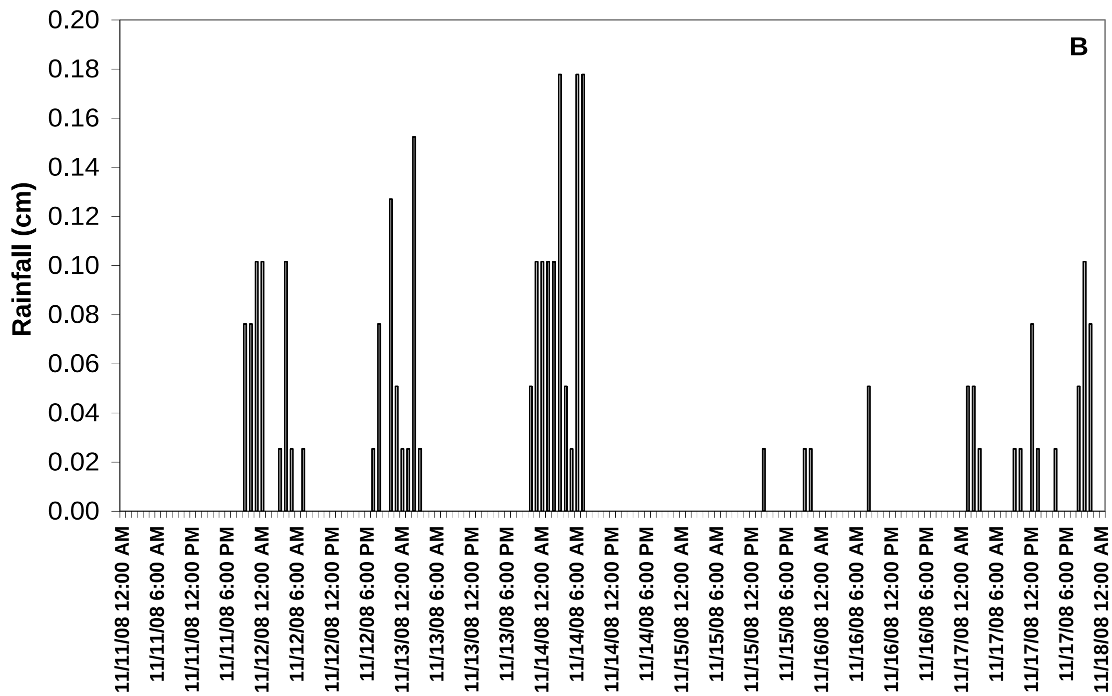


Fig. 5. Hydrographs (A) and rainfall data (B) for Site 3 on Ruddiman and Ryerson Creeks showing a response to minor storm events (≤ 1.1 cm within a 1 to 11-hour duration) between November 11 to 17, 2008.

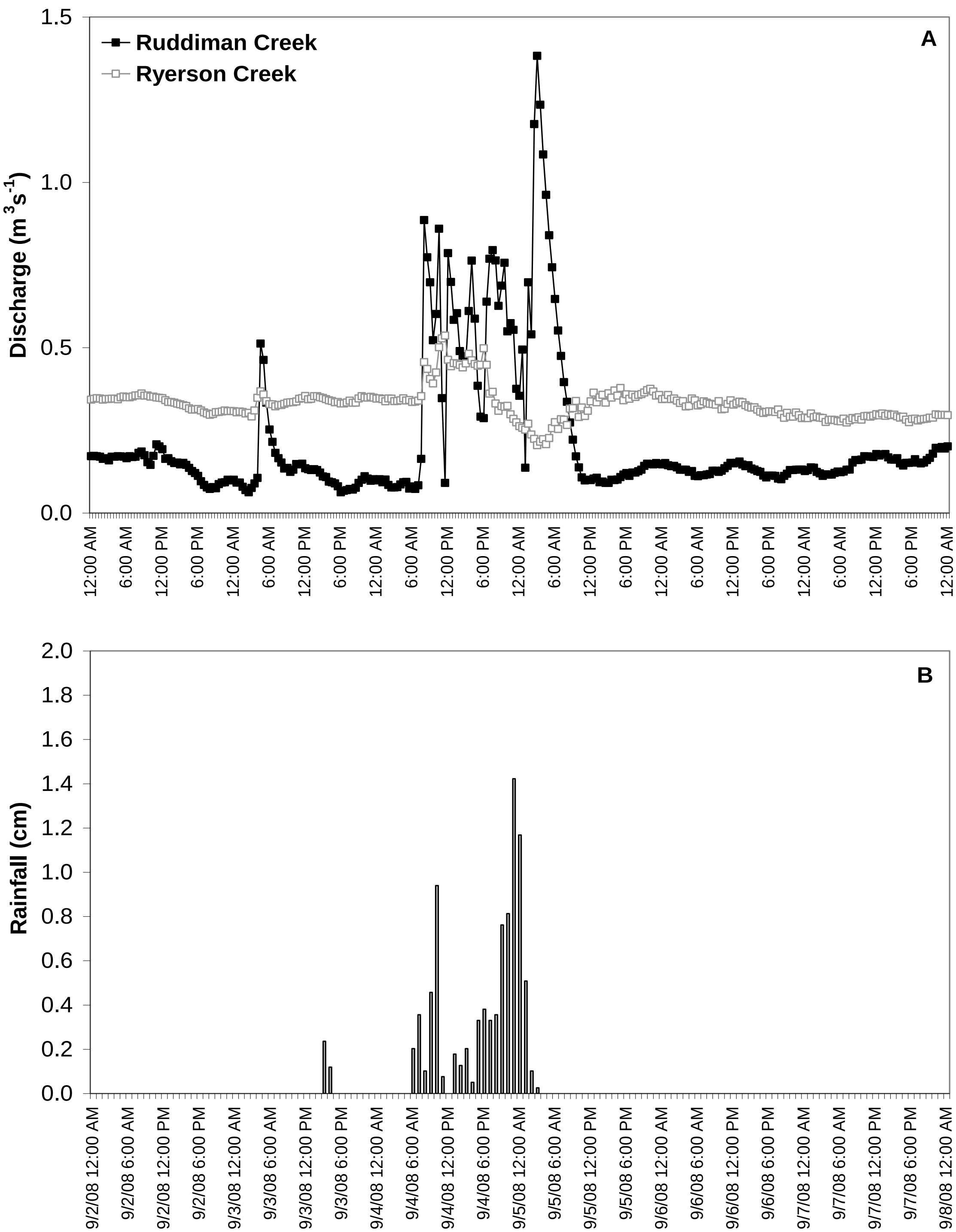


Fig. 6. Hydrographs (A) and rainfall data (B) for Site 3 on Ruddiman and Ryerson Creeks showing a response to a 10-year storm event (8.9 cm within a 22-hour duration) between September 2 to 7, 2008.

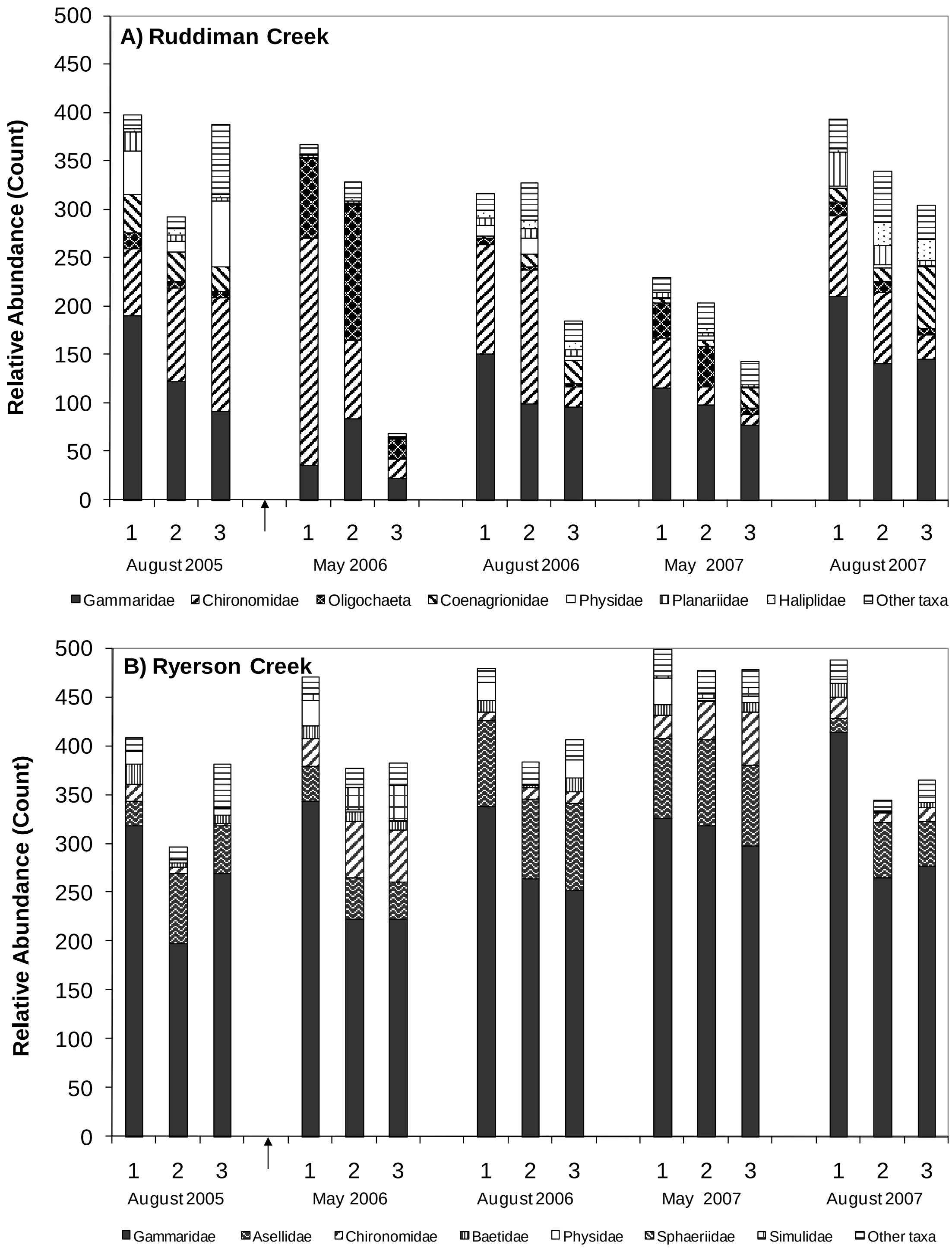


Fig. 7. Mean relative abundances of macroinvertebrate taxa collected from Ruddiman and Ryerson Creeks. Arrow (↑) indicates the division between pre and post-remediation.

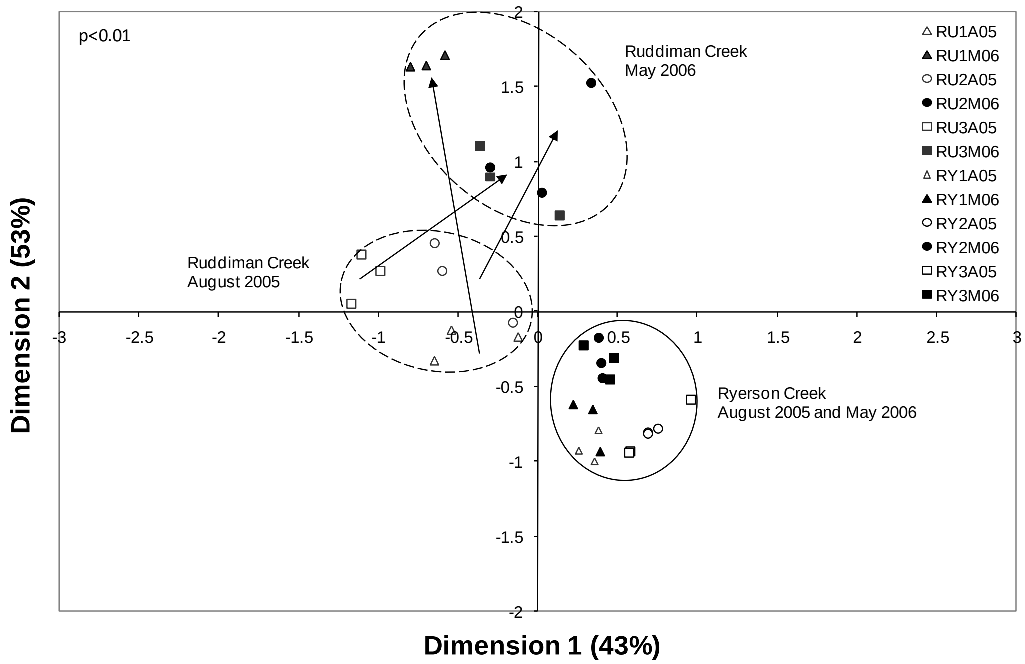


Fig. 8. Non-metric multidimensional scaling plot of macroinvertebrate community composition data collected from Ruddiman (RU) and Ryerson (RY) Creeks at study and reference sites (1-3) during August 2005 (A05) and May 2006 (M06).

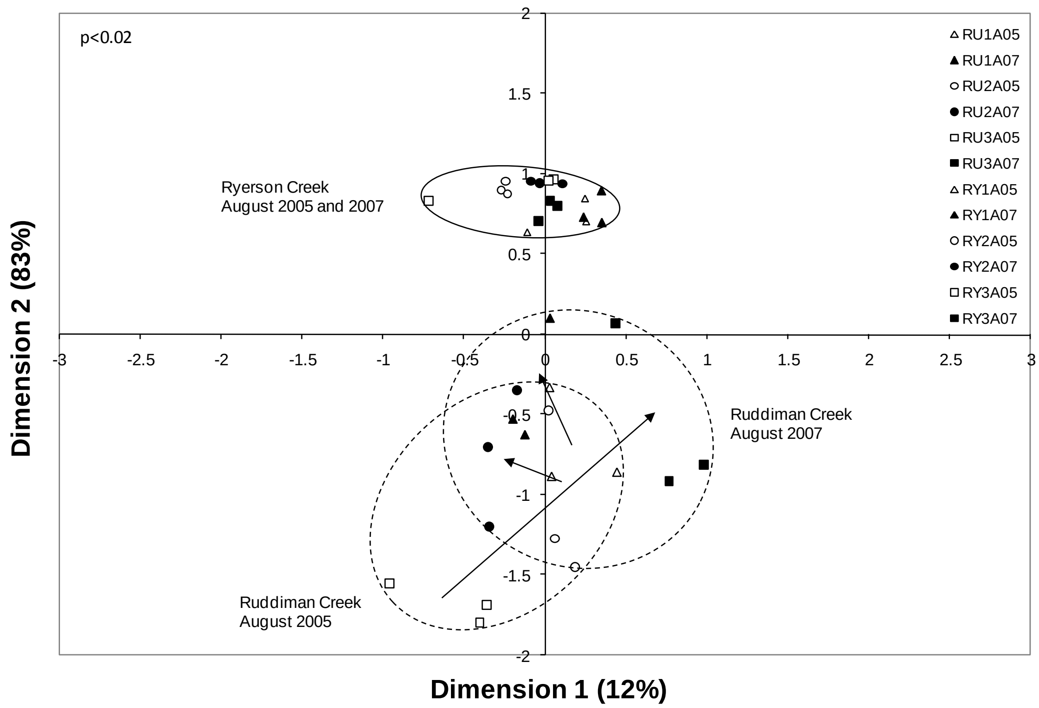


Fig. 9. Non-metric multidimensional scaling plot of macroinvertebrate community composition data collected from Ruddiman (RU) and Ryerson (RY) Creeks at study and reference sites (1-3) during August 2005 (A05) and August 2007 (A07).

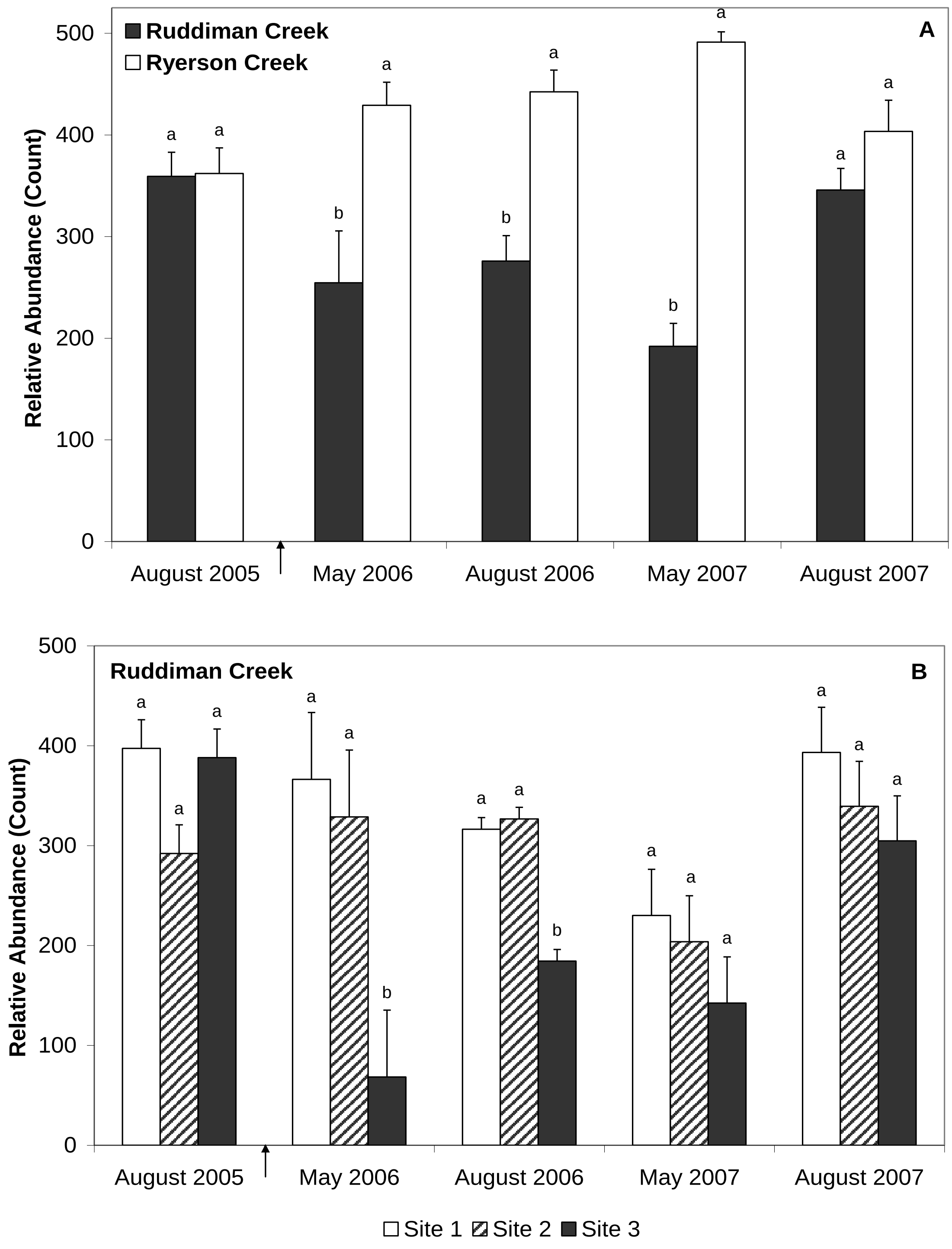
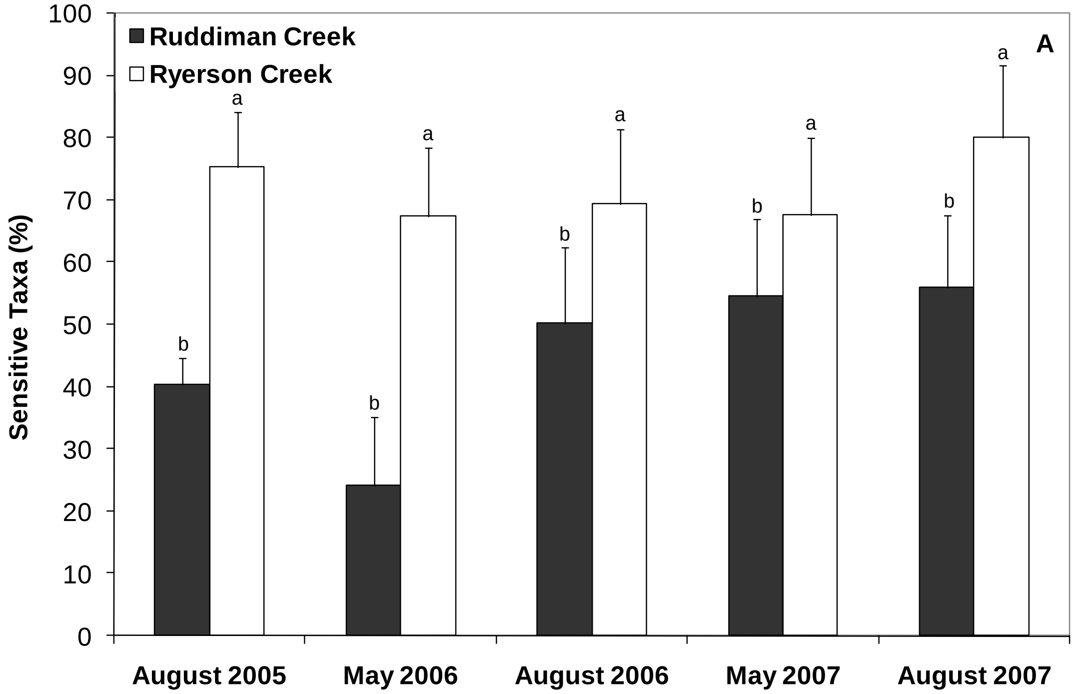


Fig. 10. Mean (±SE) relative abundance (mean counts per composite sample) of macroinvertebrates collected from Ruddiman and Ryerson Creeks (A) and Ruddiman Creek sites (B). Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.



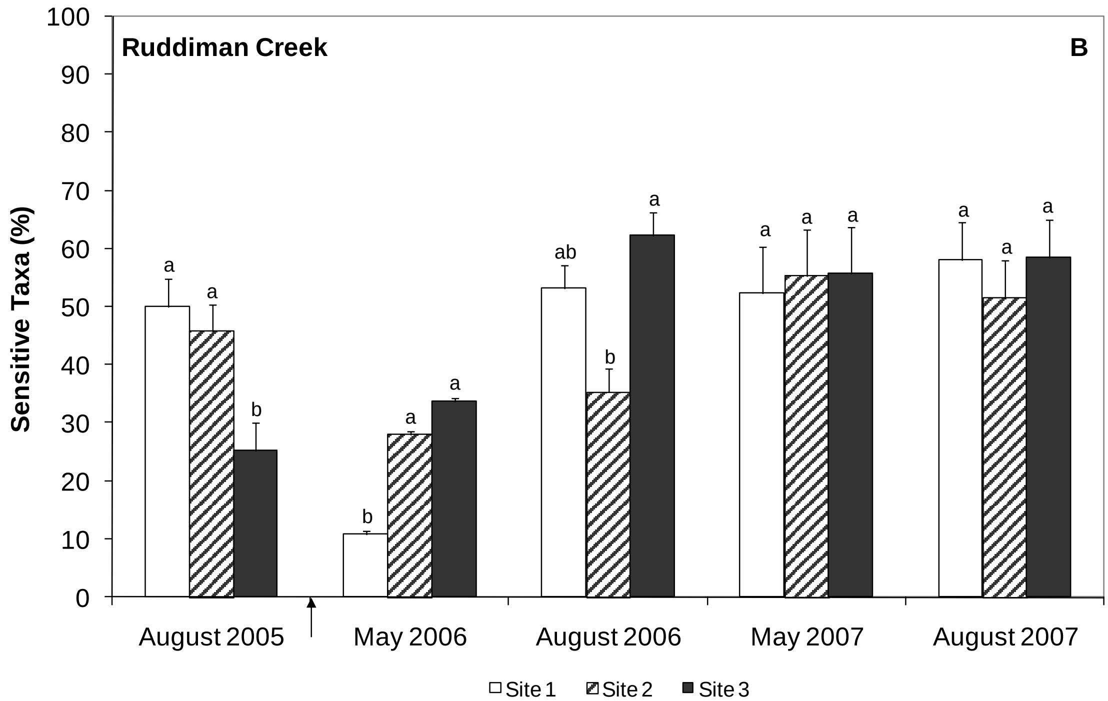


Fig. 11. Mean (±SE) composition percentage of sensitive taxa collected from Ruddiman and Ryerson Creeks (A) and Ruddiman Creek sites (B). Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

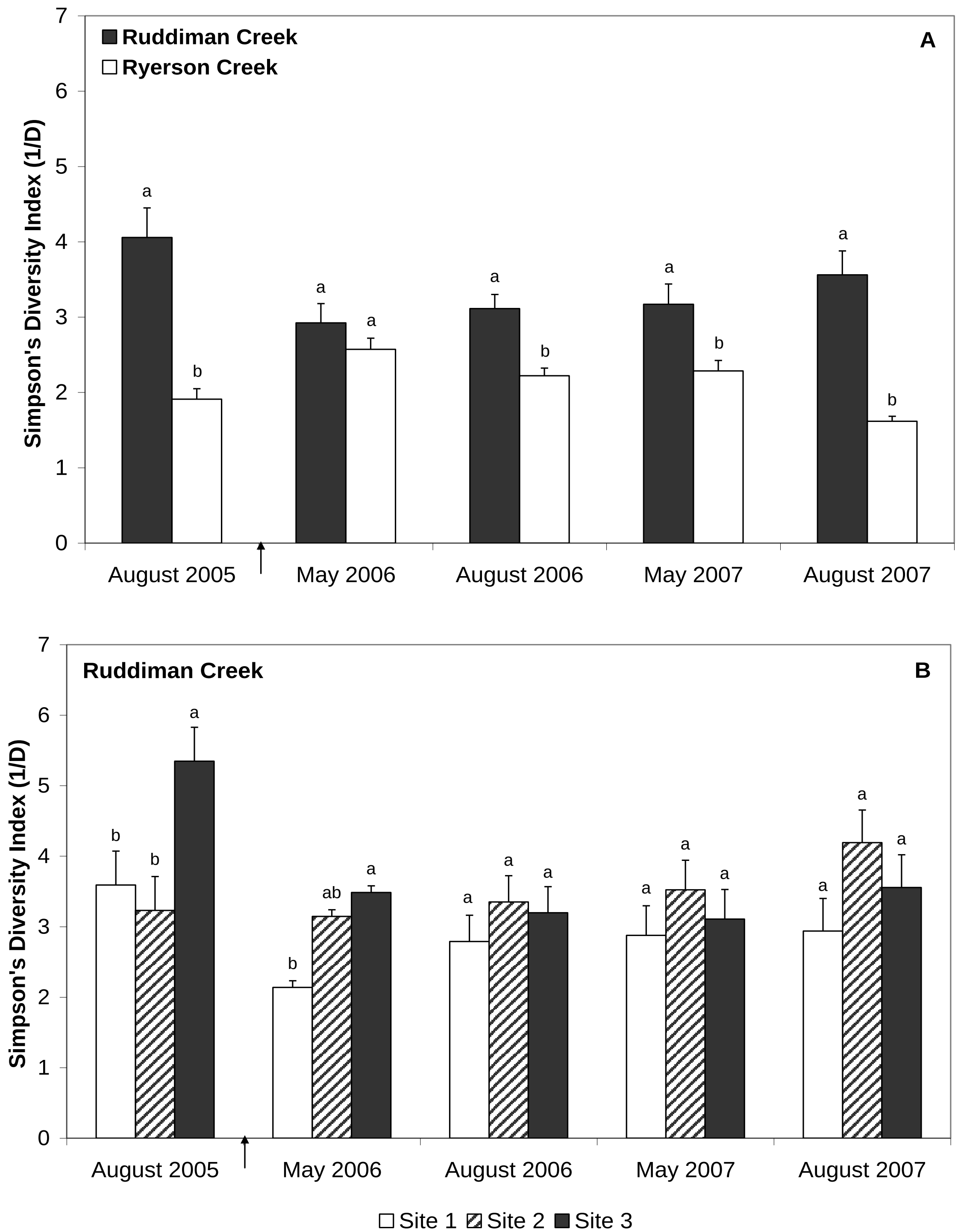


Fig. 12. Mean (±SE) Simpson's Diversity Index values for Ruddiman and Ryerson Creeks (A) and Ruddiman Creek sites (B). Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

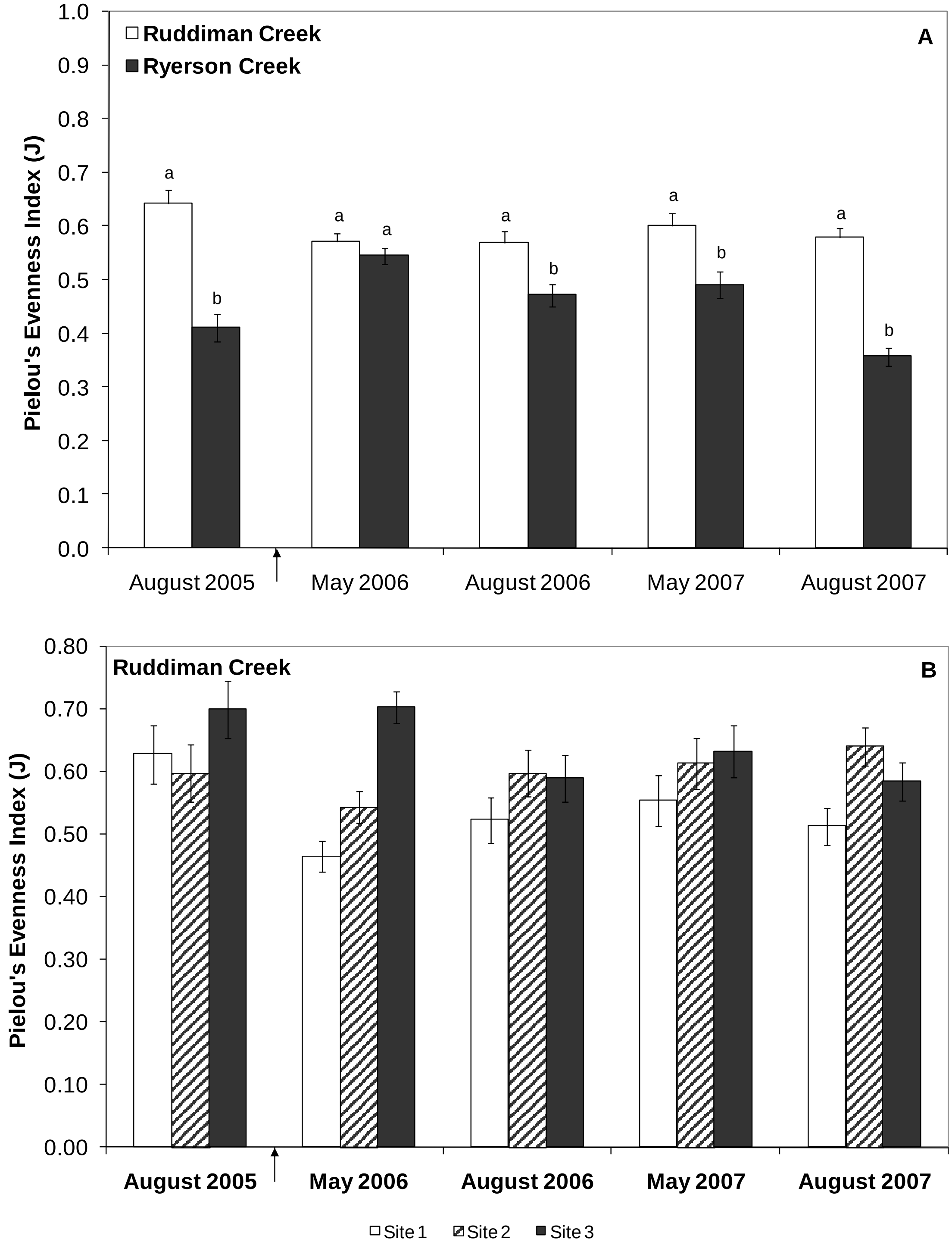


Fig. 13. Mean (±SE) Pielou's Evenness Index values for Ruddiman and Ryerson Creeks (A) and Ruddiman Creek sites (B). No significant site x date interaction effect was revealed by two-way ANOVA. Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

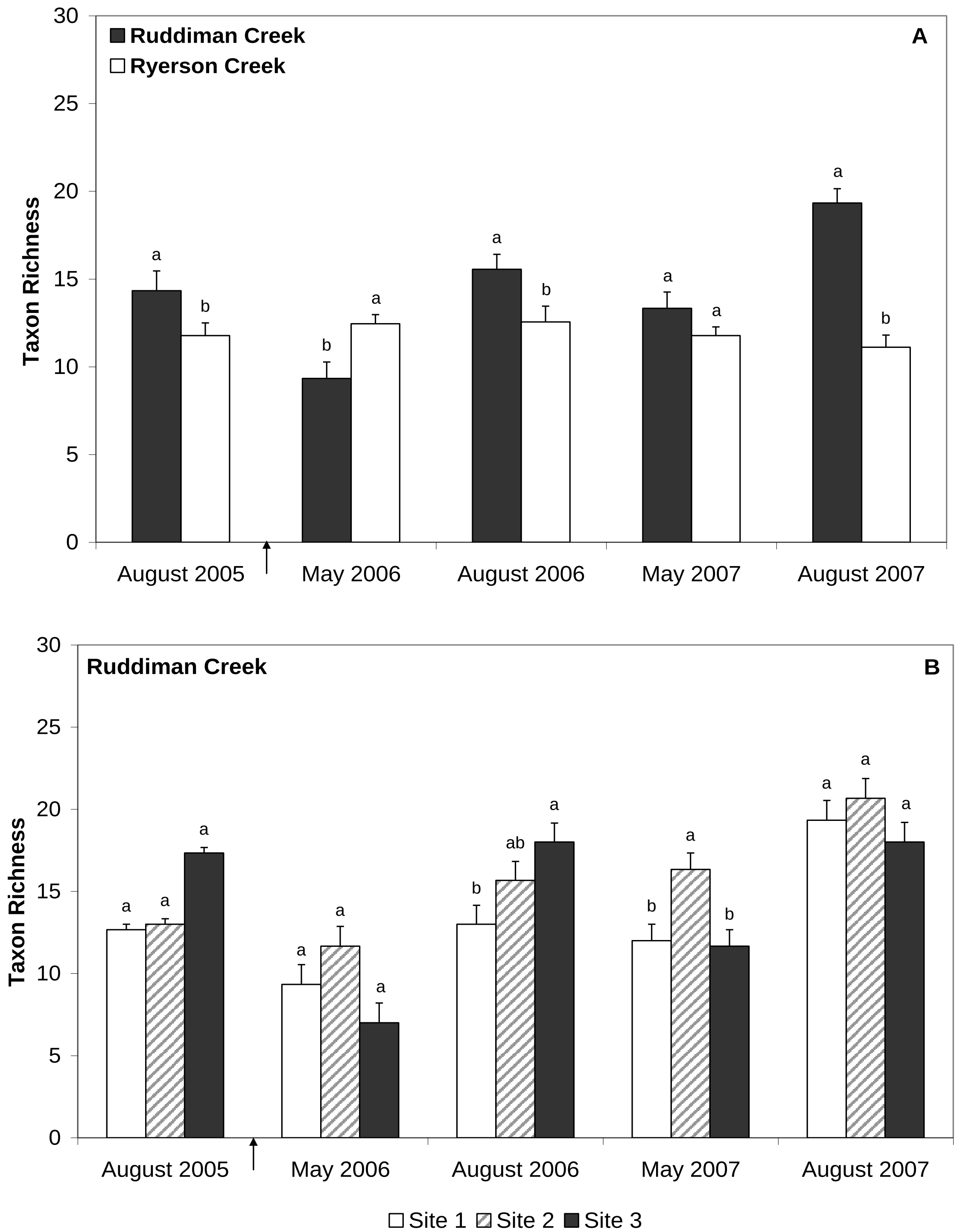


Fig. 14. Mean (±SE) taxon richness of macroinvertebrates collected from Ruddiman and Ryerson Creeks (A) and Ruddiman Creek sites (B). Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

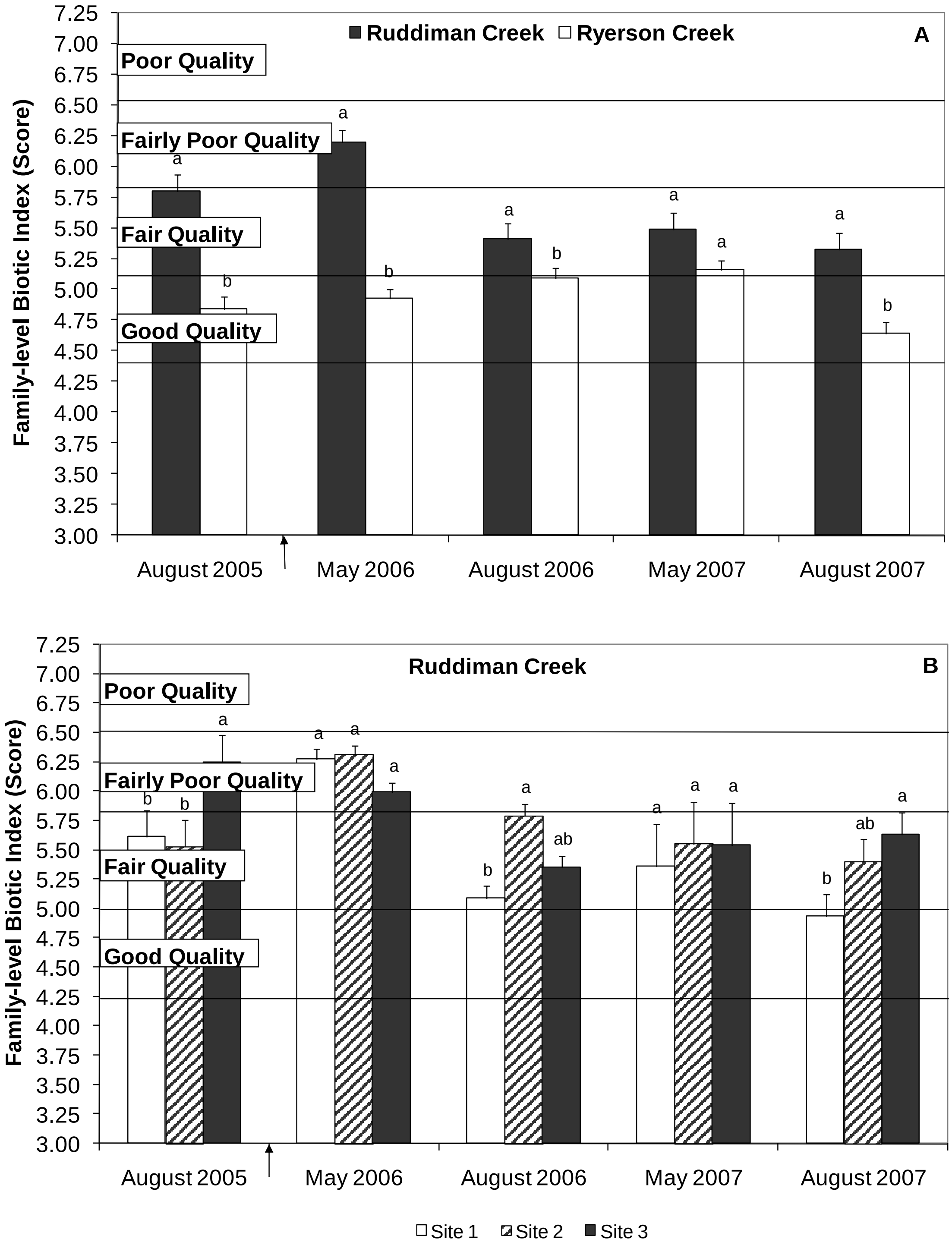


Fig. 15. Mean (±SE) family-level biotic index (Hilsenhoff 1988) scores of Ruddiman and Ryerson Creeks (A) and at Ruddiman Creek sites (B) indicating stream quality conditions. Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

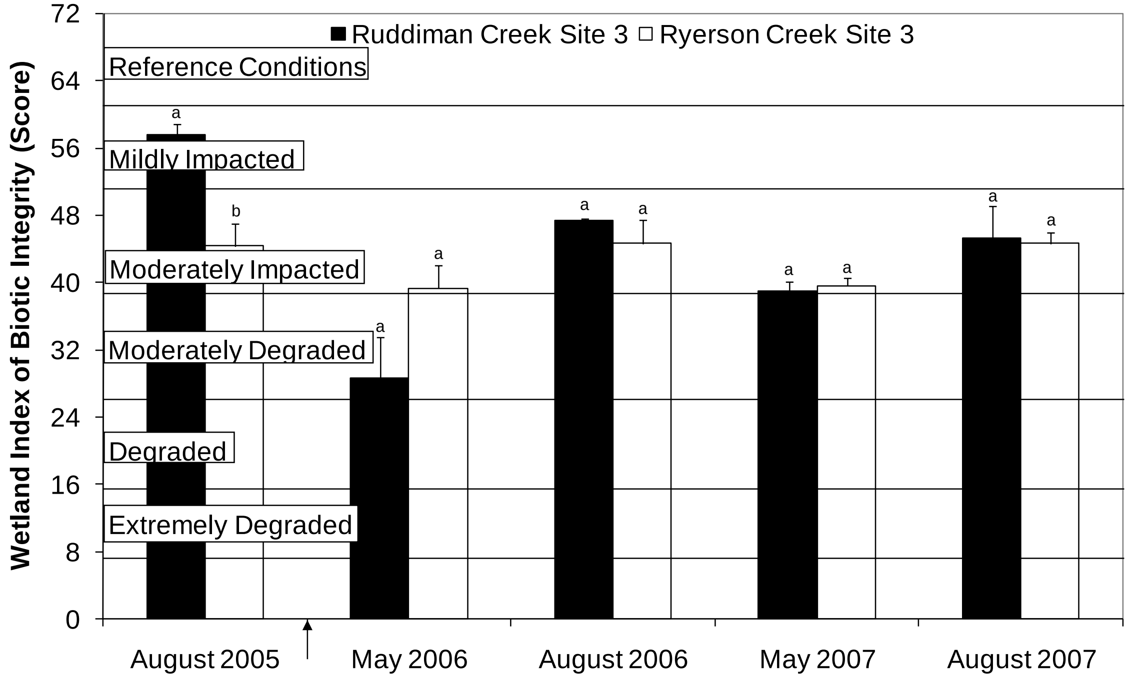


Fig. 16. Mean (±SE) Wetland Index of Biotic Integrity (Uzarski et al. 2004) scores determined for Site 3 on Ruddiman and Ryerson Creeks indicating wetland quality conditions. Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

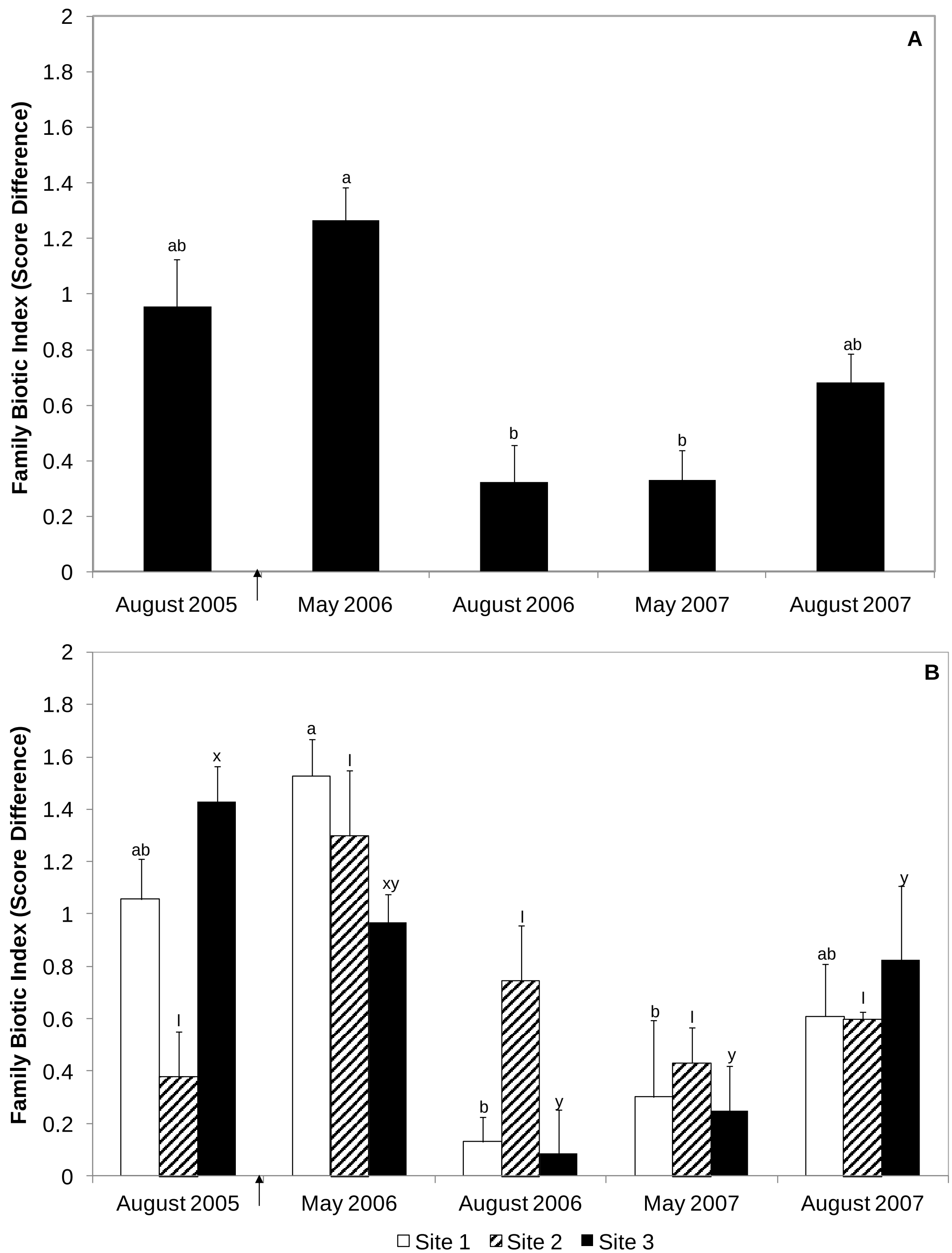


Fig. 17. Absolute mean (±SE) differences of family-level biotic index (Hilsenhoff 1988) scores between Ruddiman and Ryerson Creeks (A) and at Ruddiman and Ryerson Creek sites (B). Bars with different lettering within a sampling date differ significantly. Arrow (↑) indicates the division between pre and post-remediation.

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