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Stream Restoration in a Post-Agricultural System: Indirect Effects on Density and Secondary Production of Aquatic Insects

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LOYOLA UNIVERSITY CHICAGO

STREAM RESTORATION IN A POST-AGRICULTURAL SYSTEM:
INDIRECT EFFECTS ON DENSITY AND SECONDARY
PRODUCTION OF AQUATIC INSECTS

A THESIS SUBMITTED TO
THE FACULTY OF THE GRADUATE SCHOOL
IN CANDIDACY FOR THE DEGREE OF
MASTER OF SCIENCE

PROGRAM IN BIOLOGY

BY

LYLE S. DANDRIDGE JR

CHICAGO, ILLINOIS

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ABSTRACT

Restoration of stream habitats, with the goal of increasing biodiversity through increasing habitat heterogeneity, has been an ongoing trend in recent decades. Current investigations suggest most of those projects fail to significantly influence ecological structure and function when evaluated in light of their affect on species richness. In order to assess the “success” of restoration on a prairie stream in northern Illinois traditional metrics such as community composition and density were examined in addition to macroinvertebrate secondary production. Restoration of Nippersink Creek, McHenry County, Illinois was completed in 2000 and this study was conducted 8 years post-restoration. Benthic samples were collected from two replicate riffle sites in each reach type (restored and natural) during each season (Autumn, Winter, Spring, and Summer). Macroinvertebrate community composition was found to be similar in both reach types, with the exception of *Leucotrichia* (Trichoptera: Hydroptilidae), which occurred only in restored reaches. Insect abundance (mean (\pm SE)) was greater in restored reaches (R-1: 3197 (1169) individuals m^{-2} ; R-2: 3380 (1248) individuals m^{-2}) compared with natural sites (N-1: 793 (247) individuals m^{-2} ; N-2: 685 (320) individuals m^{-2}). Total annual secondary production of all species across restored sites was $56.5 \text{ g m}^{-2} \text{ yr}^{-1}$ and was significantly higher than in natural reaches (t test: $t(2) = 11.9$, $p < 0.05$) where annual production was $9.0 \text{ g m}^{-2} \text{ yr}^{-1}$. These results demonstrate that restoration of stream habitat heterogeneity had minimal effect on species richness, yet higher insect abundance and

annual secondary production in restored reaches relative to natural reaches may be attributable to restoration efforts. These data suggest secondary production estimates may be a valuable post-restoration assessment tool, as invertebrate density and turnover rate are important to bottom-up trophic cascades, as is invertebrate diversity.

CHAPTER ONE

INTRODUCTION

Ecosystem degradation and habitat loss via human interference have been of increasing concern in recent decades, as these problems coincide with logistic growth of the human population. Ecosystem degradation can be linked to population increase, and consequent land-cover, through a variety of mechanisms. These mechanisms range from habitat fragmentation to altering atmospheric composition enough to affect climate (Vitousek *et al.*, 1997; Pimm and Raven, 2000). Among these mechanisms, agricultural practices have some of the most profound effects. Global agricultural land use has transformed the terrestrial landscape across a range of locales, indicated by a 466% increase in cultivated land worldwide from 1700-1980 (Meyer and Turner, 1992). Agricultural practices at this scale have harsh implications for biodiversity, not only for plant species but also for soil invertebrates and microorganisms closely associated with crops (Swift and Ingram, 1996). Amid the apparent changes to the terrestrial environment brought on by agricultural practices, aquatic systems are also directly and indirectly impacted. For instance, runoff from croplands introduces fertilizers, herbicides and pesticides in quantities sufficient enough to alter natural lake and stream characteristics (Carpenter *et al.*, 1998; Graymore *et al.*, 2001; Weston *et al.*, 2004). Additionally, agricultural practices often require direct modifications to aquatic systems to increase yields. Stream channelization, which involves dredging a straight, deep

channel through what naturally occurs as a meandering stream section, is frequently used to facilitate the flow of irrigation/flood water away from crops. However, this type of modification plays a substantial role in ecosystem degradation.

Channelization of rivers and streams alters hydrology of lotic systems, and as intended, drastically reduces the water retention capacity of streams. This, however, has repercussions for stream nutrient retention (Stanley and Doyle, 2002; Bukaveckas, 2007), macroinvertebrate and fish species richness and abundance (Corbacho and Sanchez, 2001; Muotka *et al.*, 2002), and riparian vegetation, which affects bank stability (Hupp, 1992). Channelization presents a nationwide point of concern; however it disproportionately affects water quality at a smaller scale. Estimates suggest that about 55,000 km of U.S. streams were channelized during the 1930-1970s (Felleman, 1997), with over 80% of those modifications occurring in 15 states, including Illinois (Mattingly *et al.*, 1993). This has harsh implications for water quality and stream biota in Illinois.

Beginning in 1970, the Illinois Environmental Protection Agency (IEPA) was established and assumed responsibility for monitoring water quality in Illinois. Initial surveying of Illinois surface waters culminated in the IEPA 1972 Water Quality report, which stated that 11.3%, 54%, and 34.7% of surveyed streams were in poor, fair, and good condition, respectively (IEPA:<http://www.epa.state.il.us>). These designations are given based on the stream's capacity to meet all (good), some (fair), or none (poor) of its designated uses (e.g. supporting aquatic life, safe for primary contact, public/food processing water supply, etc). A more recent report suggests that, as of 2000, 0.8%, 39.9%, and 59.3% of surveyed streams were in poor, fair, and good condition,

respectively (IEPA:<http://www.epa.state.il.us>). These data indicate a clear trend in water quality improvement over the last three decades.

Trends in local water quality improvement generally stem from increased awareness and the subsequent reduction and prevention of human-induced environmental stressors. A synergistic effect on water quality improvement should be expected when awareness is used in conjunction with improvements to particularly degraded systems, in the form of restoration and maintenance projects. These projects have been occurring with increasing frequency in recent decades and have tended to focus on inland waterways because of the implications for commercial and game fish populations. Initially, the majority of these projects targeted single species, typically salmonids, and eventually broadened into an entire stream system approach (Palmer *et al.*, 2007). This system-wide methodology can involve modifications to the surrounding terrestrial environment (riparian zone) but typically focuses on in-stream modifications, often involving channel re-configuration (adding meanders) and coarse substrate inputs to increase habitat complexity (Palmer *et al.*, 2010).

Stream restoration has been implemented in many parts of the world, generally involving similar techniques with the goal of increasing habitat heterogeneity. Several studies have found a direct correlation between habitat heterogeneity and stream biodiversity (Allan, 1975; Williams, 1980; Muotka and Syrjanen, 2007). As most stream restoration projects aim to improve habitat for game fish, increasing biodiversity is expected to create effects that cascade up the trophic ladder to game fish taxa. Although these projects are quite effective at increasing habitat heterogeneity, it has been well documented that the effectiveness of most aquatic habitat restoration projects is not

evaluated with post-restoration measurements (Holmes, 1991; Kondolf and Micheli, 1995; Roni *et al.*, 2002, Hassett *et al.*, 2005). The lack of a systematic follow-up is due, in part, to a consensus that all restoration work is, in effect, beneficial to some degree, and also the difficulty associated with precisely evaluating lotic systems given their complex physical and chemical processes, coupled with extensive ecological interactions (Vannote *et al.*, 1980; Kondolf, 1995; Palmer *et al.*, 2005). In addition, optimum assessment of restoration success requires that project design includes a pre-project evaluation to establish “baseline” conditions and also the implementation of post-project evaluation. However, both of these types of evaluation typically are not incorporated into restoration projects (Kondolf, 1995). Without pre-project evaluation, investigators are restricted to using reference study sites to perform post-project assessments. Although this is common practice in many ecological studies, results may be inconclusive when this approach is used to evaluate stream restoration projects. Riverine systems are dynamic and may exhibit unique responses to disturbance and/or cyclical changes (Ward *et al.*, 2001; Palmer *et al.*, 2005), which can mask restoration effectiveness, and be unaccounted for when making comparisons with reference sites.

When post-project evaluation is performed, there has been a tendency to focus on the impact on game fish, especially salmonid taxa, with minimal data on macroinvertebrate community response. Gortz (1998) investigated how stream restoration (rocky substrate inputs and channel re-configuration) affected the macroinvertebrate community of a lake-outlet stream in Denmark. In comparing the restored and reference sites in this stream, Gortz (1998) found that restoration procedures resulted in no significant difference in species diversity; however there was an increase in

relative density of certain taxa. This study, however, only used quantitative data from two sampling dates, and variability in density estimates may have been attributable to temporal changes in aquatic macroinvertebrates populations rather than restoration effects. Muotka *et al.* (2002) compared macroinvertebrate community structure in restored, channelized, and unaffected (natural) stream sites in northern Finland. They concluded that community structure in channelized reaches was characteristically different than in restored and unaffected (natural) reaches, and that restored reaches required, on average, 8+ years to attain community structure similar to natural reaches. In all types of reaches studied, however, species richness was similar among reach types. Lepori *et al.* (2005) found similar results when they investigated fish and macroinvertebrate diversity between restored, channelized and reference reaches of the Ume River in Sweden and found diversity to be similar between treatments. All of these studies found that macroinvertebrate diversity did not increase as a result of restoration efforts, although this was an expected result of restoration in those stream systems.

In a recent and substantial meta-analysis, Palmer *et al.* (2010) found stream restoration that focused on increasing habitat heterogeneity resulted in minimal success in increasing species diversity. They found that out of 78 independent restoration projects monitored post-restoration, increased invertebrate species richness was demonstrated in only two. These findings have strong implications for the future direction of stream restoration because increased heterogeneity is one of the most common goals of U.S. and European restoration projects (Palmer *et al.*, 2010). Palmer *et al.* (2010) suggest that habitat heterogeneity may not be a major factor affecting species diversity, as it may be overshadowed by other contributing factors such as land use, watershed landscape

structure and food availability (Urban *et al.*, 2006; Kiffney and Roni, 2007). Despite the intended goal of increasing species diversity as a driving force behind many restoration projects, additional post-project evaluation parameters may serve to elucidate other indirect effects associated with increasing habitat heterogeneity.

Despite the recent focus on quantifying effects of increased habitat heterogeneity (via restoration) on invertebrate species richness in lotic systems, additional quantitative measures are lacking. Of those studies examining other quantitative measures, only overall densities are typically reported, with no estimates of secondary production. Invertebrate species richness is often seen as a standard by which restoration success can be determined for many restoration projects aiming to increase habitat complexity. As many of these projects aim at improving salmonid habitat (Roni *et al.*, 2002), increased invertebrate species richness is often viewed as a marker of improved habitat, by providing greater diversity of prey items for juvenile and adult fish. However, stream-dwelling salmonids are considered to be generalists (Hearn, 1987), thus invertebrate densities and biomass turnover rates might be more appropriate measures of improved habitat. Thus, secondary production estimates may serve as a more informative response variable to restoration events where restoration aims to improve fish habitat, as these estimates incorporate measures of both density and biomass turnover. Nippersink Creek, McHenry County, IL offers an opportunity to examine how benthic invertebrate secondary production responds to stream restoration in a post-agricultural stream system, where both restored reaches and natural (reference/least impacted) reaches are in close proximity.

This project was intended to assess whether restoration of Nippersink Creek, McHenry County, Illinois was reflected in invertebrate community structure and secondary production rates of aquatic insects. My specific goals were to measure if restored reaches differed from natural reaches in three respects: (i) invertebrate taxa and species richness, (ii) aquatic insect densities, and (iii) annual secondary production. In contrast to many previous post-restoration stream studies focusing on game fish populations (Jungwirth *et al.*, 1995; Zika and Peter, 2002; Pretty *et al.*, 2003), the focus of this study was on comparing and contrasting macroinvertebrate assemblages at replicate riffle sites of restored and natural reaches in a post-agricultural stream. Previous post-restoration studies have investigated macroinvertebrate community dynamics primarily as a function of species richness or invertebrate community structure (Gortz, 1998; Lepori *et al.*, 2005; Muotka and Syrjanen, 2007), while another study calculating secondary production (Entreken *et al.*, 2009) was not conducted in a prairie stream that underwent channel reconfiguration.

In the present study, I expected to find greater invertebrate taxa richness in restored riffles compared to natural areas. Both Andrade (2006) and Zack (2010) found the highest instance of invertebrate and fish taxonomic richness in restored areas of Nippersink Creek compared to natural reaches in previous studies. Similarly, aquatic insects were expected to be most abundant in restored reaches of Nippersink Creek, and consequently result in higher secondary production estimates for restored reaches relative to natural reach types. Previous post-restoration studies on Nippersink did not quantify insect density or secondary production estimates, however greater periphyton biomass and interstitial space was observed in restored riffle areas compared to natural areas

(Andrade, 2006; Zack, 2010). Both periphyton and interstitial space has been previously implicated in having a positive effect on density of insect taxa (Grubaugh *et al.*, 1997; Fairchild and Holomuzki, 2005).

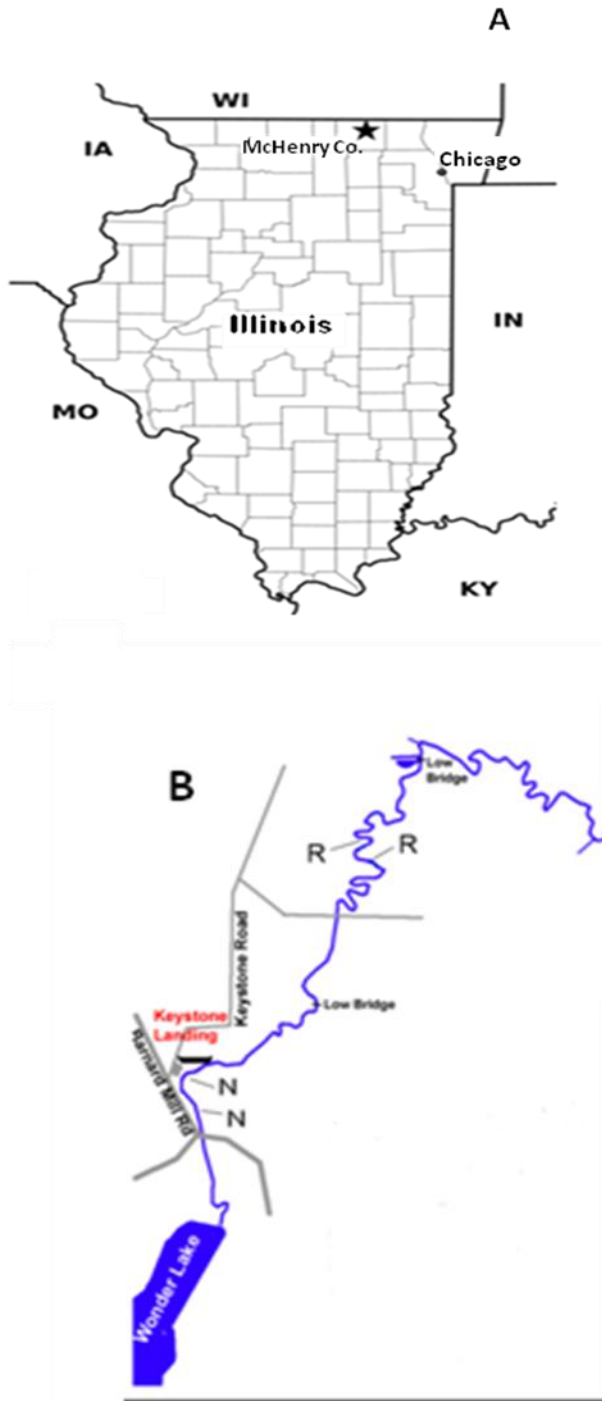
CHAPTER TWO

METHODS

Study Sites

This study was conducted in Nippersink Creek (42°22'30"N, 88°22'20"W), located in McHenry County, Illinois. The creek originates in Walworth County, Wisconsin and runs southeasterly into Wonder Lake, McHenry County, which was formed in 1929 upon construction of a dam. Nippersink Creek is the largest tributary of the Fox River, with a watershed of 355 km² in Illinois and 130 km² in Wisconsin. At the study area, located downstream of Wonder Lake, Nippersink Creek is approximately a fourth-order stream, although stream order is difficult to determine due to the influence of Wonder Lake (Figure 1). From Wonder Lake, Nippersink Creek continues approximately 37 km east towards Lake County, IL and the Chain O' Lakes area. Approximately 11 km of Nippersink Creek flows through Glacial Park, which is managed by the McHenry County Conservation District (MCCD). Within Glacial Park, formerly agricultural, downstream reaches have no canopy cover, with reed canary grass (*Phalaris arundinacea*) as the dominant riparian vegetation. The majority of upstream reaches are characterized by thick, deciduous canopy cover, consisting primarily of white oak (*Quercus alba*), eastern cottonwood (*Populus deltoides*), and crack willow (*Salix fragilis*). However, within upstream reaches of Nippersink Creek, riffle sites exist in areas with minimal or no canopy cover and these were the reaches selected for use in this

Figure 1. Study Site. A. Location of Glacial Park in McHenry County, Illinois. B. Part of the restored and natural (non-restored) sections of Nippersink Creek including the location of the four sampling sites. Abbreviations: R = restored riffle sites; and N = natural riffle sites.



areas with minimal or no canopy cover and these were the reaches selected for use in this study (hereafter referred to as upstream reaches). Water velocity and depth fluctuate considerably throughout the year in response to seasonal precipitation events and snowmelt. Stream discharge ranged from $0.6 \text{ m}^3 \text{ s}^{-1}$ (August 1988) to $57.8 \text{ m}^3 \text{ s}^{-1}$ (July 1993), with a 42-year average of $7 \text{ m}^3 \text{ s}^{-1}$ (USGS:<http://waterdata.usgs.gov/usa/nwis/uv?05548280>).

A portion of Nippersink Creek, within what is now Glacial Park, was channelized for agricultural purposes in the 1950's. In August 2000 a restoration project was completed that reconfigured the channelized portion of the stream to enhance stream habitat. This restoration project added an additional 1.6 km of meandering stream length and increased the proportion of larger and coarser substrates on the stream bed, in addition to implementing erosion control measures, and wetland and habitat restoration along Nippersink Creek.

Four riffle sites within Glacial Park were selected for this study. Two sites were in recently restored sections of Nippersink Creek (R-1 and R-2) and two were in natural (unchannelized) upstream sites (N-1 and N-2), which were never used for agricultural activities (Figure 1). The criteria for site selection were the presence of riffles, distance between sample sites, access to sites and similarity in canopy cover at specific riffle sites. Although upstream reaches of Nippersink Creek characteristically have thick canopy cover, natural riffle locations were selected in non-canopied areas. Although natural and restored riffle sites are spatially separated, this likely had little influence on the effectiveness of experimental treatments. Climate, photoperiod, and canopy cover are similar at the different riffle locations, and neither reach type is affected by tributary

input. However, both reach types are below Wonder Lake dam, which may have varying temporal and longitudinal effects on reach types. Restored (formerly channelized) riffle sections of Nippersink Creek do not allow for comparison with reference (natural) riffle sites where spatial segregation is avoidable, as no naturally occurring riffle sites exist within the restored portion of Nippersink Creek.

Field Sampling

Benthic macroinvertebrates were sampled every two weeks from October 2007 to October 2008. Macroinvertebrates were collected using a 0.09 m² Hess sampler, placed in labeled Whirlpak bags, and preserved in 70% ethanol for later sorting. Large rocks present in the sampling area were removed from the stream and macroinvertebrates were handpicked in the field. Three replicate samples were collected at each riffle site, on each sampling date, resulting in 12 samples per collection date. Sample replicates were taken from both midstream and stream margins on each collection date to ensure a comprehensive representation of invertebrate community diversity within the stream channel. Sample collections occurred at different areas within riffles on subsequent sampling dates to prevent population over-sampling at any location.

Laboratory Analyses

Initially, all 214 samples collected were to be sorted and analyzed, as previous studies in the area found low macroinvertebrate densities. However, preliminary sorting revealed relatively high macroinvertebrate densities. The majority of invertebrates to be sorted from samples consisted of very small organisms (≤ 3 mm). This resulted in an average sorting time of approximately two weeks for each non-split sample, depending on the amount of sediment present in a sample. To facilitate timely analyses, samples

were split and one sample per site for each of four dates (1 per season; 16 total) was selected for sorting and subsequent analysis: 16 November 2007 (Autumn), 28 January 2008 (Winter), 7 April 2008 (Spring) and 18 July 2008 (Summer). Seasonal sampling dates are treated as replicates of each reach type, which allows for statistical comparison between restored and natural reaches.

Benthic samples were split into 50% or 25% of the original sample (depending on the amount of material present in a sample) using a *Wildco* plankton splitter. Split samples were then sorted under a *Leica* stereomicroscope and 6.3-50X magnification. All insects used in secondary production analyses were identified, counted, and total body length measured to the nearest 1 mm. All other insects were counted and identified to either genus or species, whereas non-insect invertebrates and Chironomidae were only identified to class or order (non-insects), and family (Chironomidae). Insect and non-insect taxa in low abundance were excluded from production analysis; however these individuals were used to assess community composition and functional feeding group diversity between reach types. Chironomidae were omitted because the goal of this study was to assess non-chironomid secondary production, due, in part, to the time and difficulty associated with chironomid sorting and identification.

The five numerically dominant insect taxa, comprising 99.9% of all non-chironomid insects collected (“all insects” hereafter refers to non-chironomid insects), were chosen for secondary production analysis and identified to the lowest possible taxonomic level using Adler *et al.* (2004), Merritt *et al.* (2008) and Schuster and Etnier (1978) and assigned to functional feeding groups using Merritt *et al.* (2008). Taxa selected for secondary production analysis were *Simulium vittatum* (Diptera: Simuliidae)

and *Hydropsyche morosa*, *Hydropsyche betteni*, *Hydropsyche dicantha* and *Cheumatopsyche* spp. (Trichoptera: Hydropsychidae). Individuals in the genus *Cheumatopsyche* were not identified to species, as there is no reliable key for larval identification.

Individual wet and dry weights and ash-free dry mass of larvae were determined using preserved specimens of each numerically dominant taxon and each size class. Wet weight measurements were obtained after the organisms were air dried and weighed on a Sartorius RC 210 microbalance. The effect of ethanol preservation on larval biomass was assumed to be minimal because preservative effects on biomass loss occurs primarily in small (size-class) macroinvertebrates (Metzel, 2005), which contribute little to secondary production values (Benke, 1984). Specimens were then dried to a constant weight in a Fisher Scientific Isotemp Oven 200 series drying oven at 100°C for 24 hours, before being weighed again for dry weight mass. Specimens were then ashed in a Fisher Scientific Isotemp muffle furnace at 500°C for 2 hours, and placed in a dessicator until ash-mass measurements were taken. Mean individual biomass of each size class was determined by using the pooled biomass of multiple specimens and dividing by the number of specimens.

Regression Equations

Mean individual biomass for each size class was used to generate power regression equations relating total body length (TBL, mm) and biomass (AFDM, mg), using Microsoft Excel. All regressions used were significant at $p < 0.05$.

Secondary Production Estimates

Because cohorts were indistinguishable, secondary production rates for each taxon were calculated using the size-frequency method (Hynes and Coleman, 1968) with corrections by Hamilton (1969), Waters (1977) and Benke (1984). Negative values in the “times number of size classes” column were considered to be zeros, only for instances involving small size classes (simuliids: 0.1-3 mm; hydropsychids: 0.1-4 mm) where the previous size class had a positive value. These negative values most likely occur when individuals in smaller size classes are inadequately sampled (Benke and Wallace, 1980). Negative values for large size classes were incorporated into secondary production estimates. The basic size-frequency method assumes a developmental cycle of one year for aquatic life stages; therefore secondary production rates were corrected for the cohort production interval (CPI) for taxa exhibiting life histories different from one year. Multiplying the secondary production estimate by $12/\text{CPI}$, where CPI is the larval developmental time in months, yields a corrected estimate that is closer to the true annual production of a species.

CHAPTER THREE

RESULTS

Invertebrate Community Composition

Aquatic invertebrates collected in this study represented four phyla and ten classes. Among these invertebrates, insect taxa represented five orders and ten families (Table 1). Invertebrate species richness in restored sites was 23 and natural sites had a value of 22. Excluding one caddisfly taxon (Hydroptilidae: *Leucotrichia*), all other invertebrate taxa were represented in both reach types, and therefore reach types did not differ substantially with respect to species richness or community composition (Table 1). Pupal cases of *Leucotrichia* spp. were recovered only from boulder and large rock surfaces in restored sites, however no larvae were collected. Conversely, water striders (Hemiptera: Gerridae) were observed only in natural reaches, although they were not collected in benthic samples.

The relative composition of individual taxa, other than those used in secondary production analysis, could not be determined because Chironomidae and non-insects were not enumerated in this study. Chironomid densities, however, were high at sites from both reach types. Hydropsychid caddisflies and *Simulium vittatum* (Diptera: Simuliidae) comprised the largest populations of insect taxa in both restored and natural sites. Although non-insect taxa were not counted in this study, oligochaetes and bivalves were observed as the most prevalent non-insects collected in samples from both reach

Table 1. Benthic invertebrates collected in Nippersink Creek, McHenry County, IL with functional feeding group designations and observed presence by reach type (R= restored, N= natural).

TAXA	FUNCTIONAL FEEDING GROUP	REACH TYPE
Platyhelminthes		
Turbellaria	Predator	R , N
Mollusca		
Gastropoda	Scraper	R , N
Bivalvia	Filtering-collector	R , N
Annelida		
Oligochaeta	Gathering-collector	R , N
Hirudinoidea	Predator / Gathering-collector	R , N
Arthropoda		
Arachnida		
Hydracarina	Predator	R , N
Crustacea		
Branchiopoda		
Cladocera	Filtering-collector	R , N
Copepoda	Filtering-collector	R , N
Malacostraca		
Amphipoda	Shredder (detritivore)	R , N
Isopoda	Shredder (detritivore)	R , N
Insecta		
Ephemeroptera		
Baetidae		
<i>Baetis</i>	Gathering-collector / Scraper	R , N
Hemiptera		
Corixidae	Predator	R , N
Trichoptera		
Hydropsychidae		
<i>Cheumatopsyche</i>	Filtering-collector	R , N
<i>Hydropsyche morosa</i>	Filtering-collector	R , N
<i>H. dicantha</i>	Filtering-collector	R , N
<i>H. betteni</i>	Filtering-collector	R , N
Hydroptilidae		
<i>Leucotrichia</i>	Scraper / Gathering-collector	R
Coleoptera		
Dytiscidae	Predator	R , N
Elmidae	Gathering-collector / Scraper	R , N
Diptera		
Ceratopogonidae	Predator / Gathering-collector	R , N
Tipulidae		
<i>Dicranota</i>	Predator	R , N
Simuliidae		
<i>Simulium</i>		
<i>vitatum</i>	Filtering-collector	R , N
Chironomidae	Gathering-collector / Filtering-collector / Predator	R , N

were observed as the most prevalent non-insects collected in samples from both reach types.

Functional Feeding Groups (FFG)

The five numerically dominant insect taxa are primarily classified as filtering-collectors, and consequently this FFG comprised the majority of biomass in Nippersink Creek. The black fly *Simulium vittatum* (Diptera: Simuliidae) is classified as an obligate filtering-collector, whereas the four hydropsychid taxa are primarily filtering-collectors, although predation in the form of cannibalism has been documented within and on early instars of this family (Sherberger *et al.* 1977; Willis and Hendricks, 1992; Winterbourn and Harding, 1993). Other non-insect taxa abundant in samples, such as bivalves, copepods and oligochaetes are primarily filtering-collectors and gathering-collectors. Non-insect taxa in lower abundances, particularly amphipods, isopods and gastropods are mostly shredders and scrapers, and few predators were collected (Table 1).

Abundance and Life Histories

Densities of aquatic insect taxa were consistently higher at restored (R = restored) sites R-1 (mean (\pm SE): 3197 (1169) individuals m^{-2}) and R-2 (3380 (1248) individuals m^{-2}) compared with natural (N = natural) sites N-1 (793 (247) individuals m^{-2}) and N-2 (685 (320) individuals m^{-2}). Two hydropsychid taxa, *Cheumatopsyche* spp. and *Hydropsyche morosa*, and one simuliid species, *Simulium vittatum*, accounted for 92.2% of insects collected at all sites.

Hydropsyche morosa accounted for 33.4% of insects collected at all sites and was the most abundant species at sites R-1 (6384 (5335) individuals m^{-2}) and the non-restored, natural site N-1 (1339 (1,079) individuals m^{-2}) (Table 2). *Cheumatopsyche* spp.

Table 2. Mean secondary production parameters for all numerically dominant taxa at restored and natural reaches of Nippersink Creek, McHenry County, IL. Mean densities calculated from seasonal samples (n=4). Units for standing stock biomass (B) are mg ash-free dry mass and secondary production (P) estimates are mg ash-free dry mass yr⁻¹.

Taxon	Site	N (No. m ⁻²)(SD)	B (mg m ⁻²)	P (mg m ⁻² yr ⁻¹)	P/B
<i>Simulium vittatum</i>	R-1	3999 (613.7)	300	4306	14.4
	R-2	6992 (1132.4)	628	8669	13.8
	N-1	789 (163.9)	55	629	11.4
	N-2	988 (226.3)	44	599	13.5
<i>Cheumatopsyche</i> spp.	R-1	4595 (445.4)	2348	16470	7.0
	R-2	3198 (198.4)	1775	12957	7.3
	N-1	1333 (106.5)	615	4160	6.8
	N-2	1783 (300.6)	795	5853	7.4
<i>Hydropsyche morosa</i>	R-1	6384 (1090.2)	3902	30721	7.9
	R-2	5267 (727.9)	3147	25715	8.2
	N-1	1339 (233.8)	635	2293	3.6
	N-2	468 (71.3)	175	818	4.7
<i>H. dicantha</i>	R-1	702 (131.6)	755	5789	7.7
	R-2	1152 (177.1)	757	5883	7.8
	N-1	398 (72.3)	296	2394	8.1
	N-2	99 (23.4)	91	409	4.5
<i>H. betteni</i>	R-1	304 (56.4)	296	1801	6.1
	R-2	292 (91)	60	632	10.6
	N-1	105 (31.4)	83	565	6.8
	N-2	88 (22.9)	87	254	2.9

accounted for 27.1% of insects collected at all sites, and was the most numerically dominant taxon present at natural sites (N-1: (1333 (277) individuals m^{-2}), N-2: (1783 (1357) individuals m^{-2})), comprising 42.2% of insects collected from those reaches. The remaining hydroptychid taxa, *H. dicantha* and *H. betteni*, were present in lower densities at all sites, accounting for only 5.8% and 2.0% of insects collected, respectively. Both taxa were present in similar proportions in restored and natural sites.

All hydroptychid taxa were found to have a bivoltine life history. This determination was made based on the distribution of size classes and the presence of pupae and adults of the two most numerous hydroptychids, *Cheumatopsyche* spp. and *H. morosa* (Figure 2 and 3). Voltinism of *H. dicantha* and *H. betteni* were assumed to be the same as other hydroptychids because closely related taxa in stream systems displaying habitat and climate similarity exhibit comparable life history strategies (Benke *et al.*, 1984; Hauer and Benke, 1987). Hydroptychids were assumed to have a CPI of 6.5 months, by averaging the estimated life histories of the 9 month, over-wintering cohort and the shorter, 4 month cohort, during the rest of the year.

Simulium vittatum was the second most numerically dominant taxon among all sites, comprising 31.7% of all insects collected. Simuliids only accounted for 24.4% of insects collected at natural sites (N-1: (789 (\pm 393) individuals m^{-2}), N-2: (988 (\pm 458) individuals m^{-2})), however, *S. vittatum* was the numerically dominant taxon at site R-2 with a mean density of 6992 (2641) individuals m^{-2} .

Simulium vittatum was found to have either a trivoltine life history or two generations with cohort splitting between the spring/summer generation. Seasonal size-frequency distributions of total body lengths did not allow for distinction between these

Figure 2. Graph of total body length size-class distributions for *Cheumatopsyche* spp. in Nippersink Creek, IL, by sampling date. Autumn = 11/16/2007; Winter = 1/28/2008; Spring = 4/7/2008; Summer = 7/18/2008.

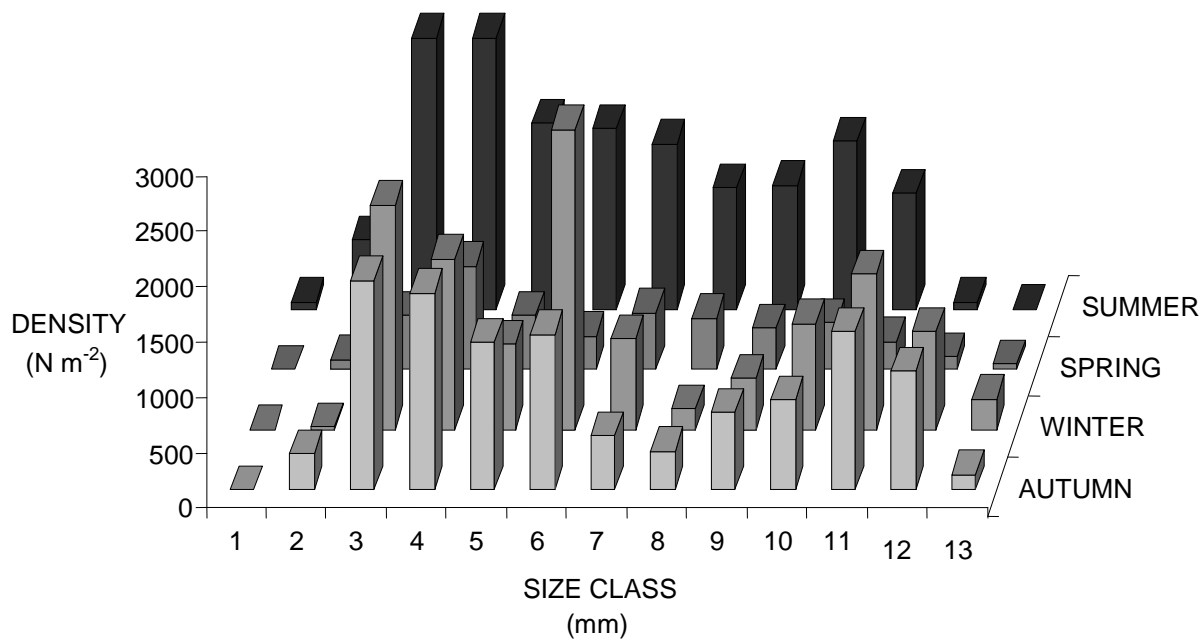
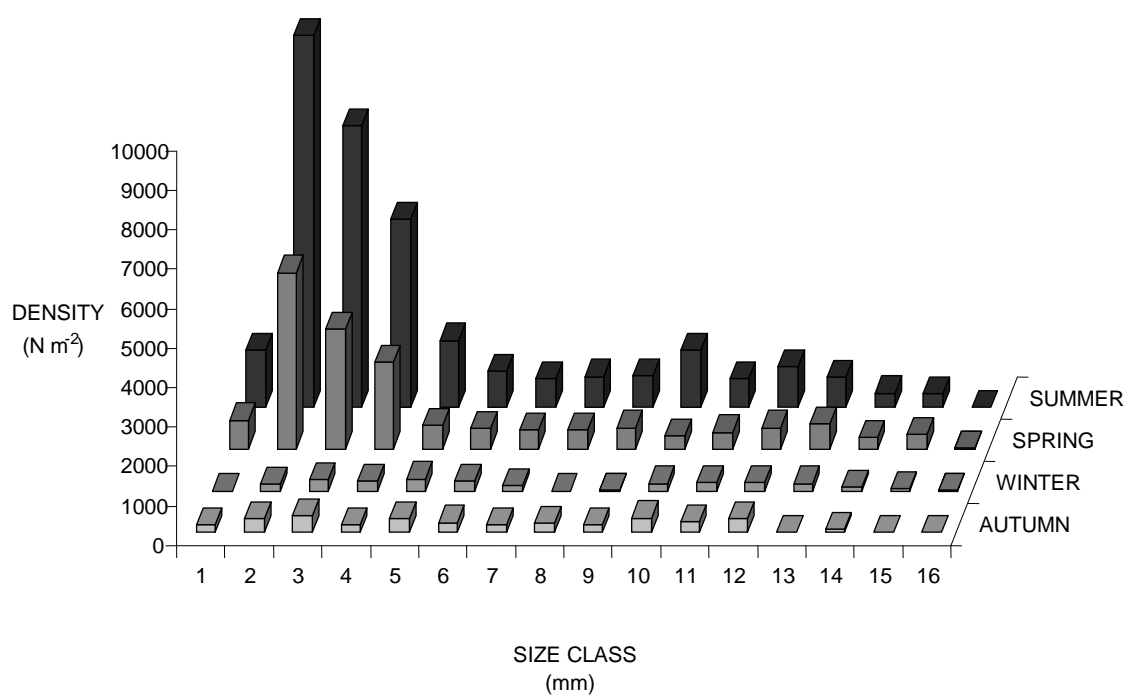


Figure 3. Graph of total body length size-class distributions for *Hydropsyche morosa* in Nippersink Creek, IL, by sampling date. Sampling dates are listed in Figure 2.



two life history scenarios, and was further complicated by the presence of smaller size classes during all seasons (Figure 4). The decision to use a trivoltine life history in the CPI correction for secondary production was made based on the presence of pupal stages in all seasons except winter. Size-frequency distributions, in conjunction with pupal presence suggests there were three generations, and a CPI of 5 months $((6\text{mo} + 5\text{mo} + 4\text{mo})/3)$ was used for this taxon.

Regression Equations

Power regression equations, relating total body length (TBL) and individual ash-free dry mass were generated and used to estimate the mass of individuals in each size class of *S. vittatum*, *H. morosa*, *Cheumatopsyche* spp., *H. dicantha*, and *H. betteni* (Table 3). Total body length and mass were strongly related for *S. vittatum*, *Hydropsyche morosa* and *Cheumatopsyche* spp. as evidenced by the power regressions. Power regression equations relating body length with biomass were also a strong fit for *H. dicantha*, whereas the equation for *H. betteni* had the lower predictive power, as this regression was generated without sufficient biomass data on smaller size classes and with fewer individuals (Table 3). Regression values are only valid within the range of data collected; however, the regression was still used in secondary production estimate calculations, as it was assumed that the relative contribution of smaller size classes to production estimates is minimal (Benke, 1984). Additionally, with the exclusion of site R-2, the majority of *H. betteni* collected were within the size range of individuals used to calculate the regression equation.

Secondary Production Estimates

The size-frequency method to estimate secondary production is exemplified for the

Figure 4. Graph of total body length size-class distributions for *Simulium vittatum* in Nippersink Creek, IL, by sampling date. Sampling dates are listed in Figure 2.

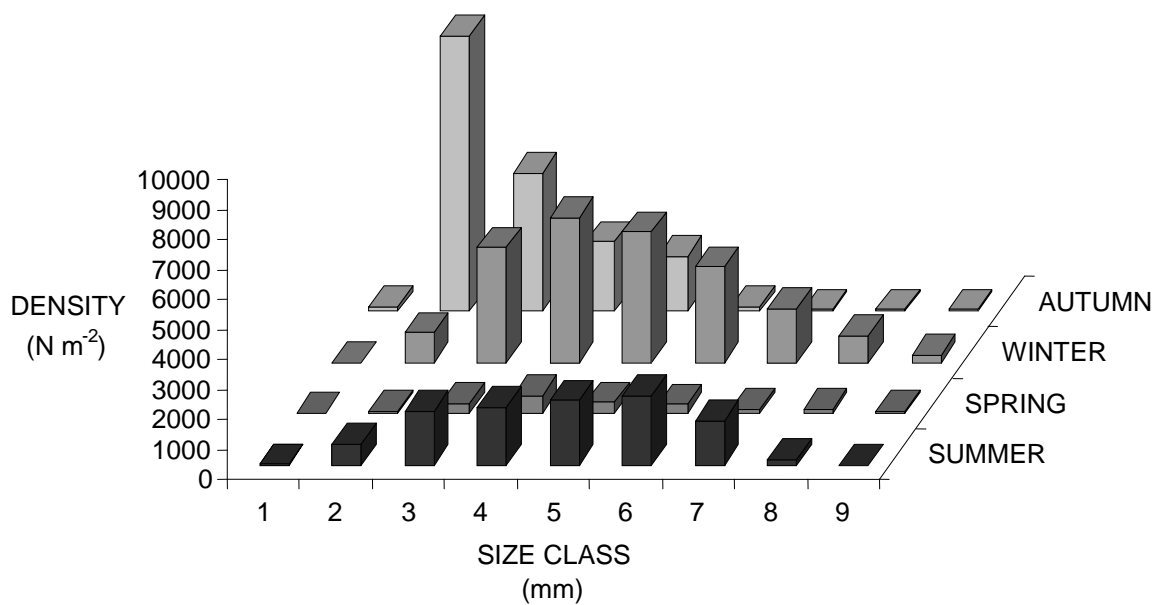


Table 3. Power regression equations relating total body length (TBL = x) and ash-free dry mass (AFDM = y) for all taxa used in secondary production analysis. n = number of individuals (n) used in calculation of each regression with the corresponding coefficient of determination (R^2).

TAXA	REGRESSION	n	R^2
<i>Cheumatopsyche</i> spp.	$y = 0.0011 x^{2.9188}$	313	0.9936
<i>Hydropsyche morosa</i>	$y = 0.0041 x^{2.5311}$	487	0.9845
<i>H. dicantha</i>	$y = 0.0018 x^{2.9438}$	126	0.9631
<i>H. betteni</i>	$y = 0.0029 x^{2.6057}$	20	0.8818
<i>Simulium vittatum</i>	$y = 0.0005 x^{3.4063}$	223	0.9604

hydrosychid species *Hydropsyche morosa* (Table 4). Annual production and mean standing stock biomass of each species were greater at restored sites than at natural sites for all taxa, except for *H. betteni* biomass (Table 2). Secondary production of *Hydropsyche morosa* was higher than all other taxa examined and was greatest at site R-1 ($30.7 \text{ g m}^{-2} \text{ yr}^{-1}$) (AFDM), which was similar to production at R-2 ($25.7 \text{ g m}^{-2} \text{ yr}^{-1}$) but over ten times greater than at N-1 ($2.3 \text{ g m}^{-2} \text{ yr}^{-1}$) and over 30 times greater than at site N-2 ($0.82 \text{ g m}^{-2} \text{ yr}^{-1}$). Annual P/B ratios for *H. morosa* were 7.9, 8.2, 3.6 and 4.7 at R-1, R-2, N-1, and N-2, respectively, with a mean P/B of 6.1. The observation of substantially higher P/B ratios in natural reaches than in restored reaches is unique to *H. morosa* in this system.

The pattern of higher secondary production estimates of *Simulium vittatum* in restored vs. natural reaches was similar to that observed for all hydrosychid taxa, except *H. betteni*, in that sites R-1 ($4.3 \text{ g m}^{-2} \text{ yr}^{-1}$) and R-2 ($8.7 \text{ g m}^{-2} \text{ yr}^{-1}$) had annual production values an order of magnitude greater than in N-1 ($0.63 \text{ g m}^{-2} \text{ yr}^{-1}$) and N-2 ($0.6 \text{ g m}^{-2} \text{ yr}^{-1}$). P/B values were similar across all sites, 14.4, 13.8, 11.4 and 13.5, at R-1, R-2, N-1 and N-2, respectively, with a mean P/B of 13.3.

Other numerically dominant hydrosychid species displayed a similar pattern of higher secondary production estimates at restored sites relative to natural sites.

Cheumatopsyche spp. had similar production values at site R-1 ($16.4 \text{ g m}^{-2} \text{ yr}^{-1}$) and R-2 ($13 \text{ g m}^{-2} \text{ yr}^{-1}$) but had 65% lower secondary production at sites N-2 ($4.2 \text{ g m}^{-2} \text{ yr}^{-1}$) and N-1 ($5.9 \text{ g m}^{-2} \text{ yr}^{-1}$). P/B ratios of *Cheumatopsyche* spp. were 7, 7.3, 6.8 and 7.4 at R-1, R-2, N-2, and N-1, respectively, with a mean P/B of 7.1.

Table 4. Example of secondary production rate calculation for Hydropsyche morose at site R-1 in Nippersink Creek, McHenry County, IL., showing cohort and annual production estimates. All measurements of biomass are ash-free dry mass.

Size-class	Density (No. m ⁻²)	Individual Mass (mg)	Biomass (mg m ⁻²)	No. Lost (No. m ⁻²)	Mass at loss (mg)	Biomass Lost (mg m ⁻²)	Times # size classes	
1≤2mm	175.38	0.01	2.01	-1391.33	0.03	-36.96	0.00	
2≤3mm	1566.70	0.04	65.31	222.14	0.07	15.48	232.22	
3≤4mm	1344.56	0.10	131.36	409.21	0.14	57.75	866.26	
4≤5mm	935.35	0.18	172.62	537.82	0.25	132.11	1981.58	
5≤6mm	397.52	0.31	121.92	187.07	0.39	72.47	1087.08	
6≤7mm	210.45	0.47	98.52	23.38	0.57	13.34	200.03	
7≤8mm	187.07	0.67	125.79	11.69	0.80	9.33	139.91	
8≤9mm	175.38	0.92	161.89	-23.38	1.07	-25.09	-376.41	
9≤10mm	198.76	1.22	243.13	-163.69	1.40	-229.08	-3436.24	
10≤11mm	362.45	1.58	571.16	140.30	1.78	249.72	3745.78	
11≤12mm	222.14	1.98	440.71	-58.46	2.22	-129.60	-1944.02	
12≤13mm	280.60	2.45	687.49	93.53	2.71	253.81	3807.08	
13≤14mm	187.07	2.98	556.90	81.84	3.27	267.79	4016.91	
14≤15mm	105.23	3.57	375.36	70.15	3.90	273.25	4098.71	
15≤16mm	35.08	4.22	148.13	35.08	4.22	148.13	2221.91	
Cohort Production = (uncorrected)							16640.80	mg m ⁻² yr ⁻¹
Annual Production = (Prod * 12/6.5)							30721.48	mg m ⁻² yr ⁻¹

Hydropsyche dicantha secondary production was higher in restored sites R-1 (5.8 g m⁻² yr⁻¹) and R-2 (5.9 g m⁻² yr⁻¹) than in natural sites N-1 (2.4 g m⁻² yr⁻¹) or N-2 (0.4 g m⁻² yr⁻¹). P/B ratios for *H. dicantha* were 7.7, 7.8, 8.1 and 4.5, for sites R-1, R-2, N-1 and N-2 respectively, with a mean P/B of 7. Conversely, secondary production of *Hydropsyche betteni* was greatest at R-1 (1.8 g m⁻² yr⁻¹) compared to lower production at other sites (R-2: 0.63 g m⁻² yr⁻¹, N-1: 0.57 g m⁻² yr⁻¹ and N-2: 0.25 g m⁻² yr⁻¹). *Hydropsyche betteni* P/B ratios were 6.1, 10.6, 6.8 and 2.9 for sites R-1, R-2, N-1 and N-2, respectively, with a mean P/B of 6.6.

A pattern showing higher secondary production in restored sites compared to natural sites, but with similar P/B ratios, was observed for *S. vittatum*, *Cheumatopsyche* spp. and, with the exclusion of site N-2, for *H. dicantha*. However, *Hydropsyche morosa* had annual P/B ratios in restored reaches approximately twice as high as annual P/B ratios in natural reaches. Distinct patterns in *Hydropsyche betteni* P/B ratios were not evident, as those values were highly variable among all sites.

Annual secondary production estimates for all taxa showed strong differences among restored and natural reaches, excluding *H. betteni* at site R-2 (Table 4). For example, mean annual production for all five species across restored sites was 56.5 g m⁻² yr⁻¹ and was significantly higher than in natural reaches (t test: t(2) = 11.9, p < 0.05) where annual production was 9.0 g m⁻² yr⁻¹. This six-fold difference in secondary production corresponded to higher mean densities for all taxa, and, with the exclusion of *H. betteni*, higher standing stock biomass, at restored sites compared to natural sites.

CHAPTER FOUR

DISCUSSION

Invertebrate Community Composition and Functional Feeding Groups

Stream systems in the Midwest region of the U.S. have been extensively affected by increased anthropogenic influence in the form of urbanization and agricultural practices. Agricultural land is often characterized by nutrient-rich runoff entering nearby stream systems. This phenomenon has been well-documented as having negative effects on water quality in receiving streams through altered nutrient dynamics (Omernik, 1976; Paerl, 1997; Smith *et al.*, 1999). In addition to nutrient-derived alterations to these natural systems, many stream channels undergo channel modifications to facilitate the drainage of crop irrigation water. This process often involves stream channelization, which occurs by the dredging of naturally occurring meandering portions of the stream into straight reaches. Alterations in stream community composition result from changes in land use, flow regime, channel form, water chemistry and riparian vegetation, and generally result in lower species richness (Richards *et al.*, 1993; Carter *et al.*, 1996).

Nippersink Creek was rated as a class “B”, a good water-quality stream, in 1993 according to the Index of Biotic Integrity (IBI) (McHenry County Defenders, 2002). Stream rating improved to “exceptional” in 1996 with an IBI score of 50 out of a possible 60. Since 1996, however, the rating has gradually declined to “good”, with a 12 point reduction in the IBI as of 2007 (Pescitelli and Rung, 2009). The period of this decline

overlaps with the major restoration project conducted on Nippersink Creek within Glacial Park, in August 2000. Although the overall IBI score of Nippersink Creek has declined since restoration, Andrade (2006) found artificial riffle areas had significantly greater invertebrate richness, abundance and biomass compared to runs that comprise the majority of Nippersink Creek. These artificial (restored) riffle areas contain more rock substrate and consequently greater habitat heterogeneity than runs and naturally-occurring riffle reaches of the stream. Past studies have found positive correlations between habitat heterogeneity and microhabitat diversity and refugia for benthic invertebrates, serving to positively influence invertebrate taxonomic richness and diversity in low-order temperate streams (Cummins *et al.*, 1966; Minshall, 1984; Brown, 2003). Results from the present study, however, indicate invertebrate taxonomic richness was similar in artificial (restored) riffle areas and natural (reference) areas. Similarly, other studies examining stream communities in post-restoration stream systems were unable to find significant differences in species diversity between restored and reference reaches (Gortz, 1998; Muotka *et al.*, 2002; Lepori *et al.*, 2005). In fact, Palmer *et al.* (2010) conducted a meta-analysis of restoration projects that had the implicit goal of increasing species diversity by increasing habitat heterogeneity. They found that only 2 of 78 projects were successful in accomplishing this goal.

Taxa richness was, however, lower in the present study compared to a previous study by Andrade (2006) on Nippersink Creek with similar levels of taxonomic resolution. Andrade (2006) observed a total of 43 taxa in riffle and run areas, with the greatest taxonomic richness occurring in riffle areas (35/43 taxa). Four of the six sites used in Andrade (2006) were the same or were in very close proximity to sites used in

this study, in which only 23 taxa were observed in riffle sites. The majority of taxa collected by Andrade (2006) that were not collected in this study were in the predator, gathering-collector and shredder functional feeding groups (FFG). Approximately half of the taxa collected by Andrade (2006) but absent from this study (Ephemeroptera: Isonychiidae, Tricorythidae, Caenidae, Potamanthidae; Odonata: Coenagrionidae; Plecoptera: Taeniopterygidae, Perlidae) are aquatic insects with life histories characterized by winged adult emergence during the spring/summer months. Mating pairs of Coenagrionidae (Odonata) were the only taxa from this list observed on the wing during the present study. However, the semi-monthly sampling regime used in this study may account for the absence of additional winged adults observed. Furthermore, many of the taxa observed by Andrade (2006) were found in low relative densities during the time of her study, which may have been difficult to collect given the seasonal sampling regime used. Andrade (2006) collected representatives from several mayfly families that sprawl or burrow in depositional substrates (Merritt *et al.*, 2008). Depositional substrate is closely associated with pool habitat, however, runs do provide some depositional areas, especially in comparison with riffles. As only riffle areas were surveyed during this study, there is greater likelihood that individuals preferring depositional areas would not be collected. Additionally, Andrade (2006) sampled via kick-net, which may have been a more suitable means of collecting any burrowing taxa present in riffle areas, relative to Hess sampling used in the present study. Thus, while the reduction in taxa observed in this study compared to that reported by Andrade (2006) may be due to the overall decline in species richness in Nippersink Creek, the influence of different sampling regimes should not be ruled out.

Taxa richness and functional feeding groups did not differ between restored and natural riffle sites, with the exception of one taxon. In addition, the filtering-collector FFG was most prevalent at both reach types, where sites for both reaches were downstream of Wonder Lake. This is consistent with other studies that have found the dominance of filtering-collectors below ponds or impoundments due to enriched (nutrient or zooplankton) seston (Chutter, 1963; Cushing, 1963; Spence and Hynes, 1971; MacFarlane and Waters, 1982; Parker and Voshell, 1983; Whiles and Dodds, 2001, Moore *et al.*, 2007). Filtering-collectors have feeding mechanisms adapted for exploiting food resources suspended in the water column.

The only taxon collected in restored reaches but absent from natural sites was *Leucotrichia* spp. (Trichoptera: Hydroptilidae). Although no *Leucotrichia* larvae were collected, pupal cases were abundant on large rocks and boulders in restored riffle areas. The presence of larger and coarser benthic substrates associated with these sites provide a better surface for pupal case attachment, especially for larvae in the tribe Leucotrichiini that affix cases to rocks in running water, unlike most other hydroptilids (Merritt *et al.*, 2008). Also, due to their sessile nature and feeding ecology, larval hydroptilids are known to be strongly associated with algal food resources in both lentic and lotic habitats (McAuliffe, 1984). Although periphyton biomass was not quantified in this study, qualitative assessment suggests greater algal biomass associated with coarser substrates present in restored sites. Furthermore, Andrade's (2006) previous study in Nippersink reported observing high levels of aquatic moss on substrates in restored riffle areas. Aquatic moss has been implicated as a substrate for epiphytic algae (Suren and Winterbourn, 1992), which was observed to be denser in restored reaches.

The relatively similar macroinvertebrate taxonomic composition in natural and restored reaches of Nippersink Creek suggests that restoration attempts resulted in a system where restored riffles displayed species richness similar to natural riffle areas, after a period of 8 years post-restoration. Edwards *et al.* (1984) investigated a post-restoration stream (channel reconfiguration/coarse substrate addition) in Columbus, Ohio and found that after monitoring at 4, 5 and 6 year intervals, macroinvertebrate communities were similar in natural (least impacted) and restored reaches. Friberg *et al.* (1998) evaluated a post-restoration stream (channel reconfiguration/coarse substrate addition) in Denmark, and found diversity indices fluctuated over time, yet overall invertebrate diversity was similar between restored and channelized reaches after a period of 6 years. Muotka *et al.* (2002) reported that it took 8+ years for restored streams (channel reconfiguration/coarse substrate addition) to reach the similar species richness as reference streams. These studies suggest that similar restoration efforts can have varying effects in different lotic systems, especially along a temporal gradient. In this study, 8 years post-restoration was sufficient for Nippersink Creek to develop similar species richness between reach types. However, as invertebrate communities may fluctuate with time (Friberg *et al.*, 1998), this study can only address reach type species richness during the study period.

Although species richness of Nippersink Creek has declined over the past 12 years, benthic insect community structure in riffles was similar in both reaches during this study. However, to explicitly determine the relative success or failure of restoration efforts on preserving or increasing taxonomic richness in Nippersink Creek, detailed pre-project information on species richness in the area designated for restoration is required.

The present study was restricted to comparisons between restored and reference (natural) reaches and must be interpreted in conjunction with the overall decline of taxonomic richness in the system. Both reach types are impacted by the Wonder Lake dam, and consequently natural reaches are not truly “natural”, as they are better described as “least-impacted”. Additional comparison between a channelized (unrestored) riffle in conjunction with restored and natural reach types would allow for better evaluation of how restoration affected species richness in this system. However, such an evaluation is not possible in Nippersink Creek because there are no true riffle reaches in channelized portions of the stream.

Abundance and Life Histories

Life histories of taxa used in the secondary production analyses in this study were difficult to discern given the limitations of using a small sample size. However, size-frequency distributions of insects used in secondary production analyses, in conjunction with observations of adult emergence and published studies on these taxa were used to conservatively estimate voltinism. Life histories were assumed to be similar between restored and natural reaches because both reach types were in close proximity and therefore experienced similar environmental conditions, including comparable levels of canopy cover and neither reach being influenced by tributary inputs. Yet, this does not account for potential influences of dam-release or groundwater inputs on water temperature. Site-specific water temperature measurements were not taken during this study; however Zack (2010) measured water temperature in Nippersink Creek at sites near both reach types (<0.8 km) during her study. In comparing the effect of differing habitat variables on fish communities between reach types, water temperature was not

found to be an important factor, indicating that temperature was not substantially different between reaches. Thus, bivoltinism in hydropsychid taxa can be justified at both reach types, as can multivoltinism (trivoltinism) for the simuliid taxon in Nippersink Creek.

Hydropsychid taxa in Nippersink Creek were estimated to have a bivoltine life history, due in part to pre-pupae/pupae recruitment and observed size frequency distributions (Figures 2 and 3). Pre-pupal larvae were defined as larvae possessing enlarged abdominal segments, to the extent that individual segments could no longer be easily distinguished, with head widths generally associated with the largest size classes. Pre-pupal and pupal larvae were only collected in Spring (7 April) and Summer (18 July), lending support to estimation of a bivoltine life history for these taxa. This observation is confounded by the abundance of smallest size class (1-4 mm) larvae at all sampling dates except Winter (28 January). Additionally, the majority of pre-pupal larvae collected in July were approximately 3-4 mm shorter than pre-pupae recovered in April (12-13 mm), and in some instances as much as 6 mm shorter. This pattern in size frequencies can best be explained by a bivoltine life history arising through cohort splitting. This scenario involves the presence of early summer hatchlings in which some individuals undergo a rapid life history during the warm season, and emerge and oviposit in late summer/early autumn. The other cohort undergoes a prolonged cycle involving overwintering as later instars that emerge early in the next spring. This scenario is supported by the observation of adult caddisflies in March (pers. obs.).

Cohort splitting has been documented for hydropsychid taxa in several studies (Benke and Wallace, 1980; Rutherford and Mackay, 1986; Winterbourn and Harding,

1993) and could have been more readily distinguished in Nippersink Creek if the sampling regime included more summer samples. A cohort splitting strategy would serve to maximize reproductive output during the warm season, and create the potential for trivoltinism for these taxa in Nippersink Creek. Trivoltinism, however, is fairly uncommon in hydropsychid taxa although it has been documented (Parker and Voshell, 1982; Alexander and Smock, 2005). Bivoltinism and partial bivoltinism also have been reported in a number of studies (Cudney and Wallace, 1980; Parker and Voshell, 1982; Sanchez and Hendricks, 1997). Despite those findings, studies of hydropsychid life histories generally report univoltinism, especially in small North American streams (Oswood, 1976; Mackay, 1986; Morin and Harper, 1986; Sallanave and Day, 1991) but also in lakes and rivers of subtropical locales (Winterbourn and Harding, 1993).

The limitations of assessing life history through analysis of samples by season are exacerbated by the possibility of asynchronous development in hydropsychid taxa. Further limitations arise from the use of total body length to assign organisms to size classes instead of using head-width measurements. Larval head-widths are a better indicator of instar and would further help to elucidate the presence of an asynchronously developing cohort by observation of individuals other than pre-pupae (Klingenberg and Zimmerman, 1992; Hutchinson, 1997; Oke and Oke, 2009). This presumption is more likely to lead to an underestimate of production in the stream rather than an overestimate. By sampling four times a year, the potential exists to have missed an entire cohort of rapidly developing individuals during the growing season. However, any error in magnitude of annual production estimates in this study will be consistent and will

therefore still allow for comparisons between reach-type production values because the same CPI was applied to hydropsychid taxa in both reach types.

The life history of *S. vittatum* was more difficult to determine than hydropsychid taxa in Nippersink Creek. Typically, life histories of dipterans are difficult to interpret because of rapid growth rates and asynchronous development, especially in warm water environments (Benke, 1984; Huryn and Wallace, 1986; Benke and Parsons, 1990). This is further complicated by low densities of the smallest size class of *S. vittatum* (0-1mm) on each sampling date (Figure 4), which was most likely attributable to inadequate benthic sampling and field processing.

The trivoltine life history estimated for *Simulium vittatum* in this system was based on size frequency distributions and presence of pupae (Figure 4). Pupae were recovered during every season except Winter (28 January) and small size-class (0-3 mm) individuals were abundant in all seasonal samples excluding Spring. Low recruitment of all size classes occurred in Spring (7 April), particularly in natural reaches. This apparently low recruitment may have resulted from the sampling date occurring between a period of recent emergence and egg hatching. This assumption is supported by a study in a lower Michigan stream where it was observed that *S. vittatum* began pupation in early March with emergence occurring in early April (Merritt *et al.*, 1978). During the present study, pupae were present in April, indicating that individuals were still emerging despite observations of high numbers of adults (pers. obs). Additionally, emergence data on 29 May 2008 indicate *S. vittatum* were emerging at this time. These data suggest the presence of a rapidly developing cohort between Spring sampling and the emergence event, as very few individuals of any size class were collected during Spring. These data

also illustrate the potential for a cohort that hatched in late Autumn/early Winter, and a mid-/late-Summer cohort as well.

The same limitations in deciphering life history patterns for hydropterygids (seasonal analysis, total body length size classes, etc.) apply to *S. vittatum*. An additional factor complicating the discernment of *S. vittatum* life history patterns results from black flies having much smaller body sizes than hydropterygids. This increases the likelihood of entire life cycles taking place within a very short time period, especially during the growing season. Rapid life cycles contribute to the potential for an underestimate of secondary production because rapidly developing larvae have greater potential for more rapid biomass turnover. Numerous studies, on a variety of taxa, have illustrated the negative correlation between body size and growth rate (Banse and Mosher, 1980; Gray, 1981; Jackson and Sweeney, 1995; Benke, 1998). In addition, *Simulium* taxa are widely known as a multivoltine clade, with a highly variable number of generations per year largely affected by temperature (Merritt *et al.*, 2008). Several species in this genus have been described as having CPI's of roughly 20 days (Benke *et al.*, 1984; Benke and Parsons, 1990), or developing to pupation in less than 30 days at water temperatures above 15°C (Reisen, 1975; Colbo and Thompson, 1978). Water temperature in Nippersink Creek remains well above 15°C from May-September (Vidales, 2001). Consequently, the conservative estimate of trivoltinism for *S. vittatum* may be an underestimate of the true voltinism for this taxon. However, Schwenneker (1985) conducted a study in a northern Indiana stream of similar order, and with similar climate to that observed in Nippersink Creek, and also reported a tri-modal pattern for *S. vittatum*. Thus, it is likely that *S. vittatum* voltinism is similar in the two streams.

Net-spinning caddisflies of the family Hydropsychidae, and black flies have been regarded as the most functionally important macroinvertebrates and most conspicuous groups of filter-feeders in streams and rivers (Benke and Wallace, 1980; Alexander and Smock, 2005). These filtering organisms are well-adapted at exploiting fine particulate organic matter (FPOM) suspended in the water column as a food resource, and consequently thrive in riffle areas below dams where seston is being exported, especially from a eutrophic lake (Macfarlane and Waters, 1982; Parker and Voshell, 1983; Mackay and Waters, 1986; Valett and Stanford, 1987; Alexander and Smock, 2005). Thus, it was not surprising that *S. vittatum* and hydropsychid taxa were found to be the most prominent groups of organisms present in riffles of restored and natural reaches of Nippersink Creek collected in the present study. Despite the abundance of hydropsychids and *S. vittatum* in both reach types, there were marked differences in overall abundance of these taxa among reaches. Densities of *Simulium vittatum* and *H. morosa* showed the greatest disparity between reach types, with both taxa being at least 4 times more abundant in restored than in natural reaches, whereas *Cheumatopsyche* spp. and *H. dicantha* were only about 2-3 times as abundant in restored reaches versus natural sites. Although *H. betteni* abundance was greater in restored reaches, this taxon was present in very low densities at both reach types.

Several factors may be responsible for differences in densities of filter-feeding taxa between reach types of lotic systems. These include the mesh size of the capturing net, water temperature, velocity, substrate preferences and biotic interactions (Alexander and Smock, 2005). Difference in water temperature was addressed in the previous section regarding community structure. Water velocity was not measured in this study,

but given the close proximity of reaches, lack of tributary inputs and site similarities, effects from this variable on a large scale can be considered negligible. However, filter-feeder distributions have been demonstrated as being highly influenced by microhabitat variables (small-scale velocities and hydraulic variables), of which their determination would exceed the scope of this project (Wetmore *et al.*, 1990; Rempel *et al.*, 2000; Brooks *et al.*, 2005). Biotic interactions were not investigated in this study, however all benthic taxa observed in restored sites were also present in natural sites, with the exception of *Leucotrichia* spp. Therefore, it is unlikely that the presence of *Leucotrichia* spp. would serve to increase the abundance of hydropsychid and simuliid taxa because *Leucotrichia* can monopolize substrate surfaces to the exclusion of other species (McAuliffe, 1984).

Cheumatopsyche spp. comprised the greatest proportion of filtering taxa at natural sites, which is contrary to the distribution expected if capture-net mesh size is the likely causative agent. Capture-net mesh size of hydropsychid species differ. In general *Hydropsyche* have a larger mesh size than *Cheumatopsyche*, due to larger body size (Parker and Voshell, 1983; Mackay, 1986). A major source of seston entering the stream system, near sampling sites, originates from the dam, in addition to the processing of allochthonous inputs from upstream riparian vegetation. These particles should decrease in size with increased distance from these sources, due to deposition. Thus, if this was a major factor influencing differences in density, *Cheumatopsyche* spp. would have comprised a greater proportion of filter-feeding taxa at downstream restored sites than *Hydropsyche*, because of greater efficiency in filtering smaller particles. This is the opposite of what occurred, as *Cheumatopsyche* spp. were the dominant taxon at natural

sites, which are further upstream, closer to the dam than restored sites. *Simulium vittatum* are smaller and have filtering mechanisms (labral fans) with finer retention capacity than the coarse nets used by hydropsychid taxa observed in Nippersink Creek, and therefore should be expected to filter even smaller particles than hydropsychids with greater efficiency (Parker and Voshell, 1983). Multiple studies support this notion by illustrating the ability of simuliid larvae to filter extremely fine particles (Wotton, 1976, 1977, 1978; Ciborowski *et al.*, 1997). Consequently, simuliids were greater in abundance at downstream restored sites and their distribution may have been a result of optimizing food-capture efficiency. However, if seston capturing efficiency was a contributing factor to the difference in densities between reach types, its contribution would be dwarfed by the influence of substrate heterogeneity.

Substrate preference is the most plausible explanation for the observed differences in insect densities between reach types. Hydropsychid taxa have been documented as using mats of vegetation to support their silken nets (Oswood, 1979; Parker and Voshell, 1983; Grubaugh *et al.*, 1997), and inhabiting the cracks and interstices between stones (McAuliffe, 1984; Fairchild and Holomuzki, 2005). Andrade (2006) observed greater periphyton biomass in restored reaches of Nippersink Creek. Qualitative assessment of periphyton during this study accounts for highest periphyton biomass associated with the large rocks and boulders added in restored reaches. Because simuliid taxa use their posterior hooks to anchor onto silken pads attached to the substrate (Merritt *et al.*, 2008), the coarse and rocky substrate associated with restored sites would provide more suitable attachment sites than that found in the largely silt and cobble dominated substrate of natural sites. In addition, larger rocks and boulders provide areas for black flies to find a

location with optimal hydrodynamics, which has been identified as a major component of simuliid distributions (Ross and Merritt, 1987; Zhang *et al.*, 1998). Thus, a difference in substrate heterogeneity is the most likely explanation for the dissimilarity in insect abundance among reach types.

Although increasing macroinvertebrate densities was not a specified goal of restoration in Nippersink Creek, insect densities in restored reaches were higher than those in natural reaches, presumably as a result of increased habitat heterogeneity. In the present study, *S. vittatum* was found at mean densities across seasons, of about 4000 individuals m⁻² and about 7000 individuals m⁻² at restored sites R-1 and R-2, respectively. Benke and Parsons (1990), reported mean densities of *Simulium* spp. in the Ogeechee River, Georgia of 2848 individuals m⁻² and 2954 individuals m⁻² in two years of their study. Benke *et al.* (1984) found densities of *Simulium* spp. in the lower Satilla River, Georgia of 7038 individuals m⁻². In the present study, mean densities of hydropsychid taxa were approximately 10,000 individuals m⁻² at restored sites. Benke *et al.* (1984) found total hydropsychid mean densities of 8287 individuals m⁻² in the lower Satilla River, Georgia. Similarly, Valett and Stanford (1987) also reported high densities of 6500 individuals m⁻² in the lake outlet of a coldwater stream in Glacier National Park, Montana.

Densities of hydropsychid taxa and *S. vittatum* were much lower in natural reaches than in restored reaches of Nippersink Creek. Mean densities of total hydropsychids were approximately 2800 individuals m⁻² and densities of *S. vittatum* were approximately 900 individuals m⁻² at natural sites. These density estimates are comparable to those reported for hydropsychid and *S. vittatum* taxa in temperate streams.

MacFarlane and Waters (1982) reported *Cheumatopsyche* sp. densities of 3628 individuals m^{-2} in Redwood River, Minnesota whereas Schwenneker (1985) observed *S. vittatum* densities of 74 individuals m^{-2} in Juday Creek, Indiana. Schwenneker (1985) did, however, report some difficulty in collecting smaller size classes of *S. vittatum*. Consequently, densities for those size classes were back-estimated to estimate secondary production. Thus, with more accurate sampling techniques, densities of *S. vittatum* reported by Schwenneker (1985) may be closer to density values estimated for *S. vittatum* in natural reaches of Nippersink Creek.

Post-restoration literature explicitly reporting densities for the numerically dominant taxa in the present study could not be found. Macroinvertebrate response to restoration events has only recently been of much concern, and tends to focus on indices of species richness or diversity. However, some post restoration studies have presented total macroinvertebrate densities for communities including other lotic taxa. Gortz (1998) observed mean densities of 10,120 individuals m^{-2} across three restored stream reaches, 4 years post-restoration (channel reconfiguration/gravel inputs). Entreken *et al.* (2009) reported mean densities of 3148 individuals m^{-2} across three Michigan streams monitored two years post restoration (woody debris inputs). Given that restoration projects have historically focused on fish habitat improvements, fish response to restoration is better documented in the literature. Roni and Quinn (2001) found significantly higher coho and steelhead salmon abundance in restored (large woody debris inputs) reaches, although dominant taxa varied with season. Muotka and Syrjanen (2007) found a two-fold increase in trout density in a Finland stream four years after restoration. Zack (2010) conducted a study on fish response to restoration in Nippersink

Creek, and also found highest fish abundance in a restored reach. Alternatively, Lepori *et al.* (2005) found mean densities of nine fish species to be similar when restored sites were compared to channelized reaches (corrected for channel area) of a stream in Sweden. These data show that restoration response may vary among taxa, often on a temporal scale, with responses generally influenced by the extent to which restoration affects an individual taxon's particular habitat. In this instance, aquatic insect densities appeared to be positively influenced by restoration techniques after a period of 8 years, despite restoration not being specifically geared towards that goal. Andrade (2006) did not specifically report densities, but also observed hydropsychid and simuliid taxa to contribute the most insect biomass across all seasons of her study, two years post-restoration. These combined studies provide some evidence that restoration has had a consistent impact on macroinvertebrate densities in restored reaches of Nippersink Creek.

Secondary Production Estimates

Previous studies have determined the adult life span of hydropsychid caddisflies to be relatively short and oviposition to take place within 2-3 days after emergence (Fremling, 1960; Jackson, 1988). Other studies found that several species of *Cheumatopsyche* had a minimum hatching period of 10 days (Sanchez and Hendricks, 1997), and that the pupation period for trichopterans was about 6 days (Parker and Voshell, 1983). This information on average non-productive days (days where an individual is not contributing to in-stream production), in conjunction with size class frequency distributions, resulted in an assumed CPI of 6.5 months for hydropsychids in this system. The CPI used in this study was similar to that reported by Sanchez and Hendricks (1997) for *Cheumatopsyche* spp. in low-order, Stroubles Creek, Virginia,

where those taxa were found to be partially bivoltine with a CPI of 225.3 days (~7.4 months). Additionally, Ross and Wallace (1983) found *Cheumatopsyche harwoodi* *enigma* to be partially bivoltine in the 1st-4th order Dryman Fork, North Carolina with a CPI of 234 days (~7.7 months).

Black flies have more variable life history strategies than hydropsychids, and thus it was more difficult to establish the minimum number of non-productive days. Previous studies have reported that black fly larvae can mature from hatchling to pupa in time spans ranging from one week to one year, depending on water temperature and species (Merritt *et al.*, 2008). However, Merritt *et al.* (1978) found *Simulium vittatum* to be multivoltine in a Midwest stream, with generations emerging in April, mid-June, late July and early September with some overlap in between. Because temperature is among the most important factors influencing larval development of simuliids (Merritt *et al.*, 2008), the observation of multivoltinism in *S. vittatum* in other Midwest streams suggests a similar life history is likely in Nippersink Creek. Using these studies, along with size-frequency data and pupae/adult observations, multivoltinism (trivoltinism) was assumed for *S. vittatum* in this system. A conservative CPI of 5 months was used for secondary production estimates. Compared to previous studies, the CPI assumed for *Simulium* in Nippersink Creek was much longer, however many of those studies occurred in subtropical climates, where these taxa display much faster growth rates (Benke *et al.*, 1984; Benke and Parsons, 1990). For instance, Hauer and Benke (1987) projected the longest CPI of *Simulium* spp. in the Ogeechee River, Georgia, would be 43 days (~1.4 months).

Annual secondary production estimates for hydropsychid taxa in both reach types, ranged from 0.25-30.7 g m⁻² yr⁻¹ (AFDM). Although the range of hydropsychid secondary production estimates in Nippersink Creek is very broad, it is consistent with estimates from other studies of hydropsychid taxa in low-order streams. Bowles and Allen (1991) also reported a broad range (0.01-68.5 g m⁻² yr⁻¹ (AFDM)) in secondary production estimates for curvivalpian, North American caddisflies. Mackay and Waters (1986) observed secondary production rates of 34.9 g m⁻² yr⁻¹ for multiple hydropsychid taxa below an impoundment in Minnesota. Secondary production estimates in restored reaches of Nippersink Creek were greater, in most cases by two-fold, than in natural reaches for all hydropsychid taxa (Table 4). The greatest difference in secondary production estimates between reach types was for *H. morosa*, which displayed a ten-fold higher production estimate in restored reaches than in natural reaches.

Annual secondary production of *S. vittatum* ranged from 0.6-8.7 g m⁻² yr⁻¹ (AFDM) across both reach types. Other than at one restored site, these values fall within the broad range of production estimates for black flies reported by Waters (1977) of 0.001-6 g m⁻² yr⁻¹ (dry weight). Additionally, Benke *et al.* (1984) found *Simulium* spp. production estimates on snag surfaces at two sites on the Satilla River of 43.9 g m⁻² yr⁻¹ and 12.9 g m⁻² yr⁻¹. The high production estimate of 43.9 g m⁻² yr⁻¹ reported by Benke *et al.* (1984) is much greater than *S. vittatum* secondary production during this study. However, the lower production estimate of 12.9 g m⁻² yr⁻¹ reported by Benke *et al.* (1984) is much closer to estimates for *S. vittatum* in restored reaches of Nippersink Creek during this study (R-1: 4.3 g m⁻² yr⁻¹; R-2: 8.7 g m⁻² yr⁻¹). Benke *et al.* (1984) only sampled snag surfaces, in an effort to establish the importance of snag habitats to macroinvertebrate

populations in stream systems. In the present study, samples were obtained from sediment and rock substrate only, in an effort to standardize sampling methods, as snag surfaces (riparian vegetation hanging into the water column) were present in restored, but not in natural reaches of Nippersink Creek. Therefore, production estimates in this study are likely underestimates, as snag surfaces are important substrates for filter-feeders (Benke *et al.*, 1984).

Estimates of macroinvertebrate secondary production, following stream restoration projects, are rather scarce in the literature. Consequently direct comparison of secondary production estimates from the present study with other post-restoration estimates was difficult. However, one study by Entreken *et al.* (2009) examined macroinvertebrate secondary production following addition of large woody debris. They estimated a $0.3 \text{ g m}^{-2} \text{ yr}^{-1}$ and $0.6 \text{ g m}^{-2} \text{ yr}^{-1}$ increase in secondary production estimates for filtering-collector taxa in 2 of 3 streams monitored 2 years post-restoration. A non-restoration study by Grubaugh *et al.* (1997) estimated secondary production across multiple FFG's along a 1-7th order river continuum. They observed that in cobble/gravel habitats with minimal plant biomass, total annual secondary production did not exceed $7 \text{ g m}^{-2} \text{ yr}^{-1}$, while cobble/gravel habitat with high plant biomass demonstrated estimates ranging from $15\text{-}364 \text{ g m}^{-2} \text{ yr}^{-1}$. These studies, across differing climates and stream orders support the view that greater habitat complexity, albeit natural or via restoration, can result in higher secondary production estimates for macroinvertebrates.

Because secondary production is the product of individual growth rate and standing stock biomass, many variables can influence production through their influence on these factors. These variables include temperature, food quality, habitat complexity,

photoperiod, and biological interactions (i.e., parasitism, competition, predation, etc.) (Fuller and Mackay, 1981; Krueger and Waters, 1983; Parker and Voshell, 1983; Benke 1984). Given the close proximity and similar riparian canopy conditions of sampling sites in both reach types, it is unlikely that differences in secondary production estimates between reach types were most heavily influenced by temperature or photoperiod. Biological interactions are unlikely to have contributed to differences in secondary production estimates between reach types because the same taxa are present in both reach types other than *Leucotrichia* spp in restored reaches. This taxon was discussed previously as more likely to lower abundance and secondary production estimates through competitive interactions with other taxa (McAuliffe, 1984).

Food quality is often implicated in distribution of filtering-collector taxa (Parker and Voshell, 1985; Mackay and Waters, 1986; Whiles and Dodds, 2001). Distributional patterns affect measures of biomass per unit area, which ultimately affects secondary production. In Nippersink Creek, hydropsychid taxa gut contents indicate that diets are comprised primarily of algae in Winter and Spring, and detritus in the Summer and Autumn (Vidales, 2001). For food quality to be a major factor in secondary production differences in Nippersink Creek, hydropsychid densities should display a positive relationship to proximity of an optimal food source. In this case, eutrophic Wonder Lake might be the most likely source of optimal food resources for hydropsychids in the portion of Nippersink Creek evaluated during this study. Seston released from eutrophic Wonder Lake is likely to have a high algal component, in addition to a high microbial load. This does not however rule out the possibility of additional sources of food arising from within Nippersink Creek, downstream of the dam. Algal particles may be more

available to filter-feeders nearer restored reaches, as dense canopy cover upstream of natural riffles likely prohibits growth of benthic algae. Benthic algae growing in natural riffle areas, which occur in open-canopy areas, may be available to filterers at natural sites if those algae become suspended in the water column. Conversely, algae originating in natural riffles will also be transported downstream, contributing to the pool of algal resources available to filter-feeders in restored riffles. Additionally, detritus may be composed of both feces of and/or remnants of upstream organisms and consequently may influence distributional patterns. If Wonder Lake is the optimal food source for taxa in this study, populations in natural reaches would have greater access to higher quality food than in restored reaches, as more algae and detritus should settle from the water column with increased distance from the dam, and no longer be available to filter feeders (Richardson and Mackay, 1991). For a similar reason, food quality also would not explain secondary production differences of simuliids in this system. Simuliid taxa have been documented in other systems as deriving at least 80% of their production from assimilated amorphous detritus (Wallace *et al.*, 1987), of which bacteria are a major component (Edwards, 1987). Water flowing from Wonder Lake should deliver high microbial content into Nippersink Creek, as the lake is eutrophic and by definition has elevated microbial activity, which should create a positive relationship between simuliid taxa density and proximity to the dam. As with hydropsychid taxa, such a relationship was not observed during this study. Simuliid taxa were more abundant in restored sites, which were further downstream from Wonder Lake than natural sites.

Because temperature, food quality, photoperiod, and biotic interactions are likely not primary causes of differences in secondary production between reach types, habitat

heterogeneity is the most likely mechanism explaining those differences. Increased habitat complexity in restored reaches is due to more interstitial space associated with coarse, rocky substrate, and greater algal biomass. Together these habitat attributes appear to have a positive impact on insect densities and consequently biomass of insects per unit area. This increased biomass was ultimately reflected in greater secondary production estimates. Grubaugh *et al.* (1997) found similar results when contrasting secondary production of similar macroinvertebrate communities in gravel habitats with and without high plant biomass. In that study, all FFG's, excluding scrapers, demonstrated a positive relationship between secondary production and plant standing-crop biomass. Natural riffle reaches used in this study were primarily composed of cobble/gravel habitat, whereas in restored reaches cobble/gravel habitat was augmented with additional rocky substrate and more plant biomass (Andrade, 2006). The augmented habitat of restored reaches resulted in more habitable area for macroinvertebrates which translated to higher insect biomass per unit area, and higher secondary production.

Cohort P/B ratios ranged from 1.6-6 for all taxa examined, with average P/B ratios of 4.6 and 3.5 for restored and natural reaches, respectively. The average P/B ratio for restored reaches is close to a ratio of 5 as reported by Waters (1979) for freshwater benthic insects. Multiple size classes on each date were absent in several samples from natural sites. This resulted in lower cohort P/B ratios for taxa in natural reaches, especially taxa that were found in low abundance during this study. Annual P/B ratios were higher, because all taxa considered for secondary production analysis were considered to be multivoltine.

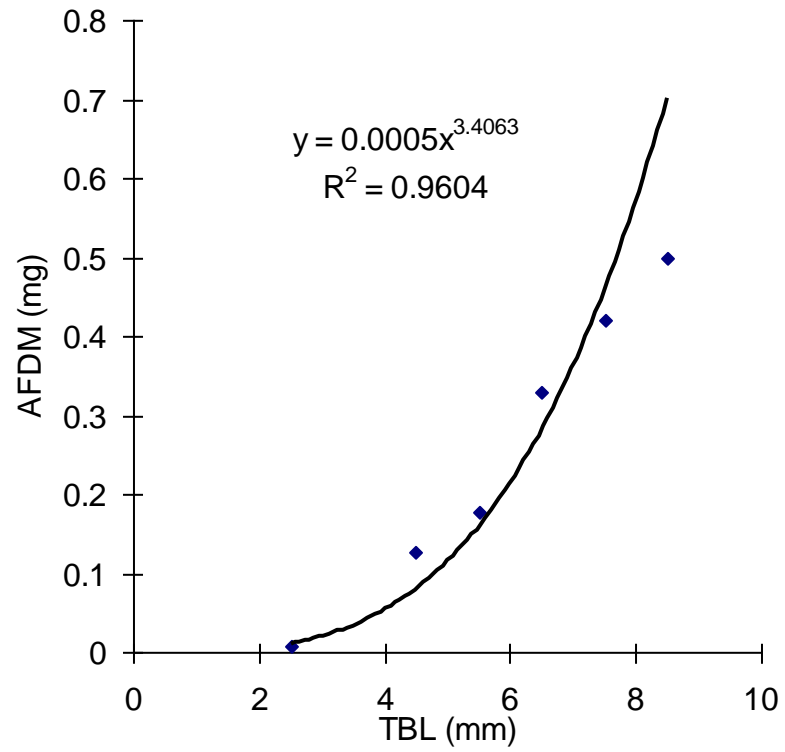
Simulium vittatum, *Cheumatopsyche* spp., and *H. dicantha* each had similar annual P/B ratios for both reach types. *Hydropsyche dicantha*, however, had similar annual P/B ratios except at site N-2. The annual P/B ratio for site N-2 was negatively skewed by very low densities of *H. dicantha* collected on all sampling dates. The similarity in P/B values between restored and natural reaches, for the aforementioned taxa, is notable given the substantial difference in their secondary production values among reach types. These data suggests that on a proportional basis, similar standing stock biomass resulted in similar secondary production values in both reach types. However, this was not the case for *H. morosa*, as restored reach P/B values were markedly higher than in natural reaches. This trend suggests that in restored reaches, individuals of lower biomass were contributing more to secondary production estimates whereas, in natural reaches, higher biomass individuals were contributing more to secondary production estimates. These data indicate that for *H. morosa*, individuals may have been accumulating biomass more slowly in restored reaches compared to natural reaches. This phenomenon may have been a result of intra-specific or inter-specific interactions associated with higher densities of individuals in restored sites. The most likely scenario resulting from intra-specific interactions, as *H. morosa* would have similar inter-specific interactions with taxa in restored reaches as in natural reaches, was due to similarity of taxa present in both reach types.

CHAPTER FIVE

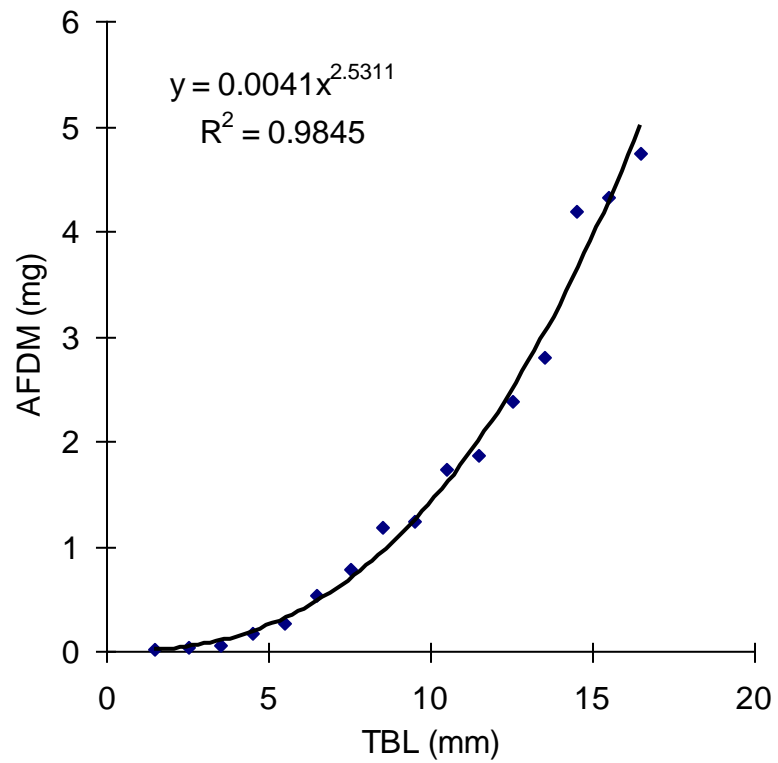
CONCLUSION

Stream restoration projects with the explicit goal of increasing species diversity by increasing habitat heterogeneity have been largely unsuccessful (Palmer *et al.*, 2010). The present study indicates that restoration efforts in Nippersink Creek were similarly unsuccessful in increasing species richness in restored reaches relative to natural reaches. Restored riffle areas did, however, have substantially higher macroinvertebrate densities and secondary production values compared to natural reaches. Although not the intended goal of the Nippersink Creek restoration project, restoration efforts resulted in riffle areas with macroinvertebrate densities and secondary production values analogous to values typically associated with highly productive systems. These conclusions are not necessarily applicable to other restored systems, as restoration techniques are varied and streams are dynamic systems. However, these data suggest that the traditional approach to evaluating stream restoration projects needs to be reconsidered, by incorporating metrics other than taxonomic diversity. For instance, abundance and secondary production of macroinvertebrates can be an appropriate measure of a restoration project's effect, especially concerning projects aimed at improving habitat for fish.

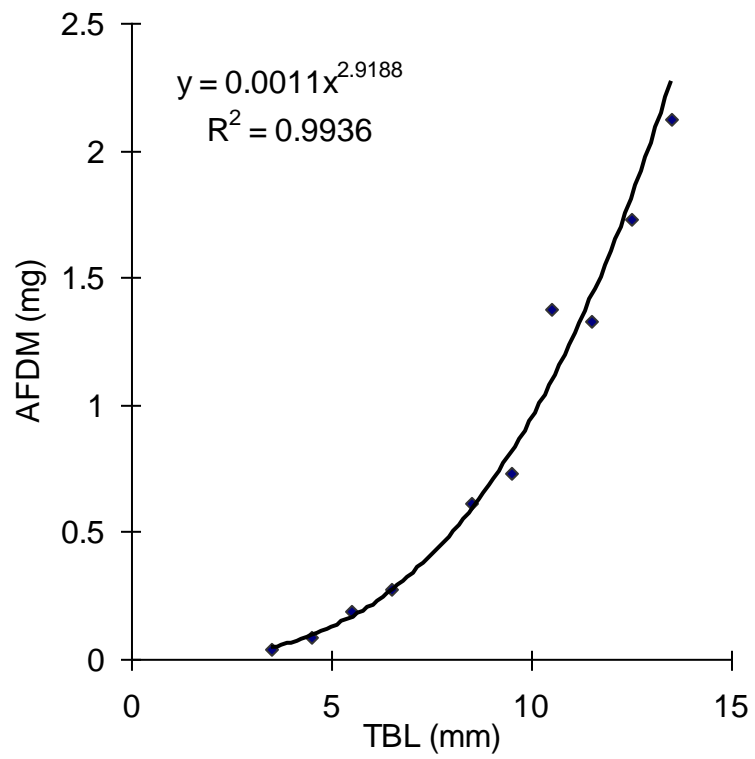
APPENDIX A:
LINEAR REGRESSION EQUATIONS



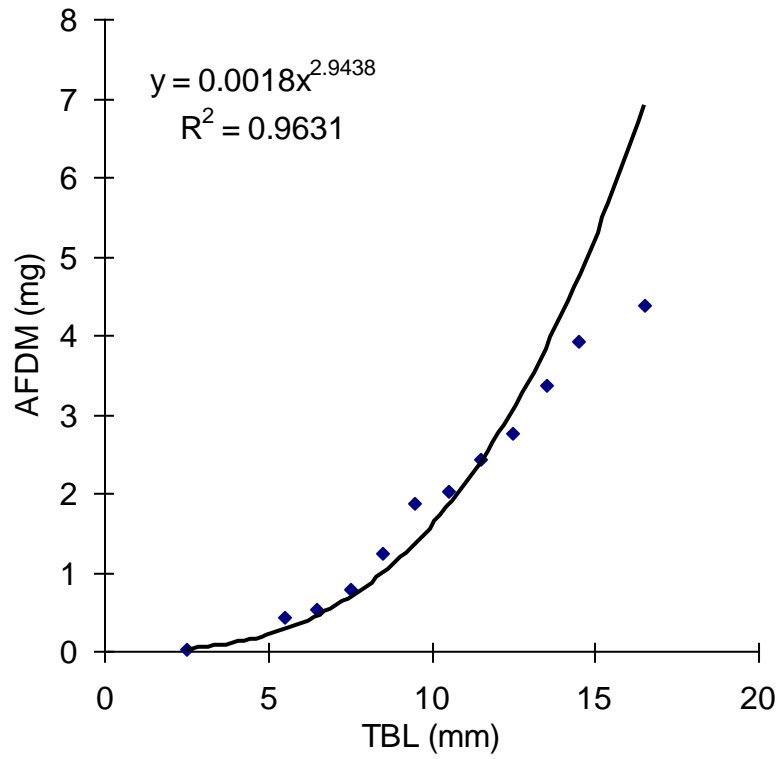
Length-mass regression for *Simulium vittatum*. TBL = total body length. AFDM = ash-free dry mass.



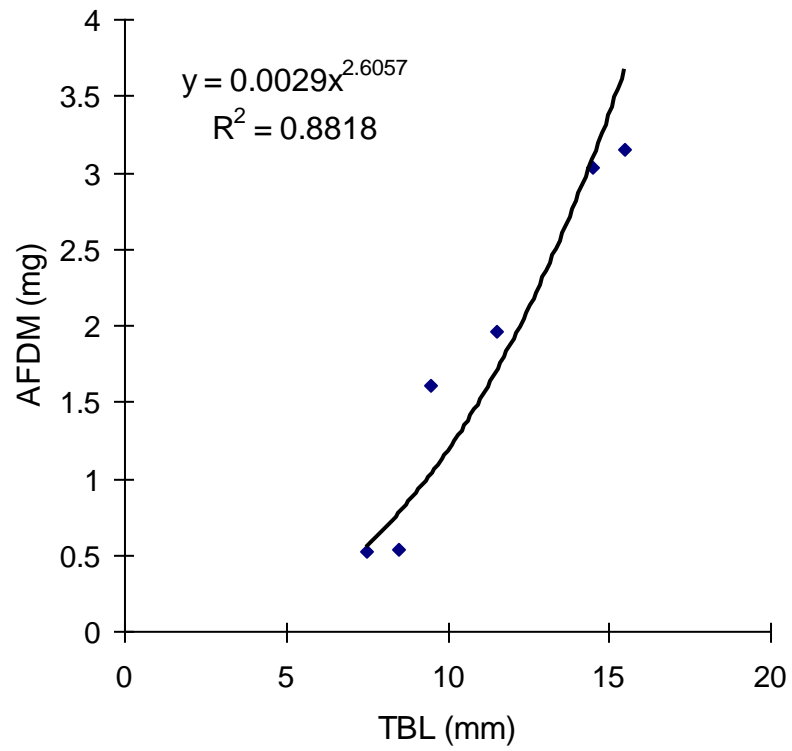
Length-mass regression for *Hydropsyche morosa*. TBL = total body length. AFDM = ash-free dry mass.



Length-mass regression for *Cheumatopsyche* spp. TBL = total body length. AFDM = ash-free dry mass.



Length-mass regression for *Hydropsyche dicantha*. TBL = total body length. AFDM = ash-free dry mass.



Length-mass regression for *Hydropsyche dicantha*. TBL = total body length. AFDM = ash-free dry mass.

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