Use of benthic macroinvertebrates to assess impacts of agricultural land use in nontidal coastal plain streams

Diane Katharine Barnes
Charlottesville, Virginia

B.S., James Madison University, 1998

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ABSTRACT

During recent history, anthropogenic activities in coastal watersheds have played a major role in increasing nutrient transport to rivers and offshore waters, often with detrimental consequences. Research on nutrient enrichment has focused primarily on lakes, rivers, and estuaries, while enrichment in coastal plain streams has been greatly understudied. Benthic macroinvertebrates are often extremely sensitive to enrichment, and as a result, are commonly used in biomonitoring of nutrient pollution. This study examined the impacts of agricultural land use in coastal streams on Virginia's Eastern Shore, through assessment of macroinvertebrate community structure and monitoring of a variety of chemical and physical parameters.

Nine catchments ranging in agricultural land use from 28 to 91 % of watershed area were sampled seasonally from June 2003 to March 2004. Water and sediments were analyzed for nutrients, chlorophyll, organic matter, and dissolved oxygen. To assess macroinvertebrate community structure, sediment cores were collected, sieved through 0.5 mm mesh, and organisms retained on the sieves were identified to genus where possible. Twenty-one metrics of community structure were calculated, including measures of taxonomic richness, taxonomic composition, tolerance, feeding roles, and $\delta^{15}N$ signatures.

Though nitrate fluxes were higher in watersheds with higher % agriculture, elevated loadings were not related to changes in autotrophic biomass, decomposition, or macroinvertebrate communities that were predicted to occur with increasing nutrient enrichment. Therefore, the sampling reaches are likely on the low end of the nutrient enrichment spectrum, perhaps because of depleted nutrient concentrations potentially resulting from low surface runoff and/or retention by riparian vegetation. However, the streams are probably towards the high end of the scale with respect to allochthonous organic enrichment. Increased inputs of riparian detritus were linked to elevated decomposition, increased abundance of macroinvertebrates, and a shift in community structure toward dominance by tolerant taxa, all patterns which were hypothesized to occur in response to nutrient enrichment. Overall, nutrient pollution from cropland agriculture in coastal watersheds with flat topography and forested riparian zones was insufficient to produce undesirable changes in nontidal streams. In these systems, local factors were more important than watershed land use in dictating macroinvertebrate community structure.

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INTRODUCTION

During recent history, anthropogenic activities in coastal watersheds, such as fertilizer application, commercial animal production, and discharge of wastewater, have played a major role in increasing nutrient transport to rivers and offshore waters (Vitousek et al. 1997). With continued population growth in coastal regions, human-induced nutrient transport is likely to accelerate in the coming years (Boynton et al. 1995). While considerable effort has been made to regulate and reduce point source nutrient pollution, such as discharges of municipal and industrial effluent, non-point source nutrient pollution has received much less attention (Howarth et al. 2000). Consequently, diffuse inputs of nitrogen (N) and phosphorus (P), particularly those resulting from agricultural land use, now constitute the major source of pollution in both freshwater and coastal waters around the world (Downes et al. 2002, Correll et al. 1992). Application of manures and synthetic fertilizers, deforestation, and intensive animal production are the primary agricultural activities leading to non-point source pollution of N and P (Smith et al. 1999).

Nutrient enrichment of streams, rivers, lakes and estuaries can cause reduction of plant and animal diversity, development of toxic algal blooms, lowered oxygen concentrations, fish kills, and loss of aquatic plant communities and coral reefs (Carpenter et al. 1998). Aquatic fauna are impacted by nutrient enrichment primarily as a result of changes in primary production and in the chemistry of water column and sediment. These changes potentially lead to reduced diversity and abundance, shifts in community composition, physiological changes, and mass mortality. The sensitivity of aquatic fauna to conditions of nutrient enrichment varies with individual organisms, due to differences in feeding habits, mobility, and life cycle (Mucha and Costa 1999, Harper 1992). As a result, measures of the structure of faunal communities can be used to assess the impacts of nutrient enrichment on aquatic ecosystems. Benthic macroinvertebrates, in particular, have been demonstrated to be useful biomonitoring tools for assessing impacts of enrichment in both freshwater and marine ecosystems.

Studies of nutrient enrichment have focused primarily on freshwater lakes and coastal estuaries, while enrichment in flowing waters has received much less attention, especially in small headwater streams (Young et al. 1999, Smith et al. 1999, Miltner and Rankin 1998, Peterson et al. 1985). Enrichment of nontidal coastal plain streams, in particular, is greatly understudied (US EPA 1997, Wright and Smock 2001, Kedzierski and Smock 2001). In addition, the majority of research on the structure and function of aquatic macroinvertebrate communities has focused on higher-gradient streams with rockier substrates (Wright and Smock 2001). Coastal plain streams are likely to respond differently to nutrient enrichment than their higher-gradient counterparts, as a result of lower velocities, less stables substrates, and fewer riffle habitats (US EPA 1997).

This study took place in the Virginia Coast Reserve (VCR), located on the southern portion of the Delmarva Peninsula on Virginia’s Eastern Shore. The VCR mainland consists of a number of adjacent, small catchments that are similar in size, shape, climate, and topography, but vary in percentage of land in agriculture. Each watershed contains one to several tidal creeks that serve as the primary linkage and route for transport of materials from terrestrial uplands to offshore lagoons (Lerberg et al. 2000). The unique structure of the VCR landscape enables analysis of the effects of agriculture on coastal plain streams using a multiple watershed approach within a relatively small regional area (Hayden et al. 2000).

The overall objective of this work was to examine the impacts of agricultural land use on headwater tidal creeks by using benthic macroinvertebrates as bioindicators of nutrient pollution. Macroinvertebrate community structure was assessed and related to a suite of chemical and physical variables. The specific hypothesis being tested was that the structure of benthic macroinvertebrate communities varies with percentage of land in agricultural use and associated nutrient enrichment.

BACKGROUND

Nutrient Transport and Watershed Land Use

Nutrients are transported from agricultural fields to surface water bodies directly through surface runoff, and indirectly by leaching through soils and transport by groundwater (Giupponi and Rosato 1999, Biggar and Corey 1970). Several factors might increase rates of nutrient flux from agricultural land, including increased contamination of runoff from fields and pastures, accumulation of nutrients in soils, changes in fertilizer and manure applications, and increases in erosion and runoff due to land alterations and vegetation removal (Withers and Lord 2002). Because nitrate is very soluble, the primary route of transport from agricultural fields is infiltration through the soil profile to the water table and delivery to streams in groundwater (Biggar and Corey 1970, Mainstone and Parr 2002, Summers et al. 1999). In contrast, inorganic P is extremely insoluble and tends to adsorb to particulates, so is primarily transported attached to soil particles in surface runoff (Mainstone and Parr 2002, Summers et al. 1999, Correll 1998). Though phosphorus leaching is generally less significant than runoff, it may be an important pathway for transport if soils are overloaded with the nutrient, and/or in sandy soils that have a low capacity to retain P (Mainstone and Parr 2002, Summers et al. 1999).

The groundwater component of stream flow is termed baseflow, and is essentially the background discharge between precipitation events (Hornberger et al. 1998). On Virginia’s Eastern Shore, low topographic relief and an extremely permeable aquifer reduce runoff and make baseflow the major component of stream discharge (Reay et al.

1992, Lowrance et al. 1997, Robinson and Reay 2002). As a result, groundwater transport is the most important route of nutrient delivery to streams in this area. The region is underlain by the Columbia aquifer, an unconfined aquifer with a shallow water table and highly permeable sediments that make it particularly susceptible to groundwater contamination and transport of pollutants to surface water bodies (Reay et al. 1992). Depth to the water table ranges from 0 to 3 m, and aquifer sediments are predominantly sand and gravel up to 30 m thick (Lowrance et al. 1997, Robinson and Reay 2002). The combination of intensive agricultural activities and an aquifer highly susceptible to pollution make nutrient influx to groundwater and streams a major concern in this area (Reay et al. 1992). In addition, groundwater has recently been recognized as an important source of nutrient delivery to coastal waters (Lapointe and Clark 1992, Robinson and Reay 2002), and enriched groundwater has been implicated as the main source of nutrients to VCR coastal lagoons (McGlathery et al. 2001).

Many studies have demonstrated the link between land use in watersheds and the concentrations of nitrogen and phosphorus in streams, rivers, and coastal waters. Reay et al. (1992), in research on the Eastern Shore, found that stream headwaters and shallow groundwater near agricultural areas had inorganic N and P levels up to two times higher than waters underlying forested lands. In addition, Stanhope (2003) established that nitrate concentrations and flux rates in coastal plain streams on the southern Delmarva Peninsula were inversely related to percentage of watershed area in forest. Similar results were found elsewhere in the Atlantic coastal plain, where fluxes of N and P increased significantly with increased percent cropland in subwatersheds of the Rhode River estuary (Correll et al. 1992). Positive relationships between agricultural land use and streamwater nutrient concentrations have also been demonstrated in the Atlantic-Gulf coastal plain (Lowrance et al. 1985), in watersheds of Fish River, Alabama (Basnyat et al. 1999), in six catchments in Scotland (Hooda et al. 1997), and in many other regions. Due to groundwater transport times and flow paths, historical land use is often more important in predicting water quality than is current land cover (Harding et al. 1998). Robinson and Reay (2002) measured average residence times of groundwater on the southern Delmarva Peninsula to range between 16 and 21 years. As a result, land cover data from 1988-1989 were used to assess impacts of agricultural land use on streams in the present study.

In addition to watershed land use, land cover in the riparian zone may play a role in nutrient transport to surface waters. Denitrification and uptake by vegetation in the riparian zone can promote loss of nitrate transported in both groundwater and surface runoff (Peterjohn and Correll 1984, Lowrance et al. 1997). In coastal plain watersheds, where groundwater moves in shallow flow paths, up to 90% of nitrate transport to streams may be removed by riparian forests (Lowrance et al. 2000). Research has also indicated that buffers of riparian vegetation may reduce flux of sediment and phosphorus to adjacent waters (Reed and Carpenter 2002, Peterjohn and Correll 1984, Lowrance et al. 1997.).

Conceptual Model of Nutrient Enrichment in Aquatic Ecosystems

Nutrient enrichment is defined as the process by which an increase in nutrient inputs causes excessive production of organic matter in aquatic systems, often with detrimental consequences (Howarth et al. 2000). The conceptual model of nutrient enrichment was developed primarily through studies of lake ecosystems, and has since been applied extensively to coastal water bodies such as estuaries and bays (Smith et al. 1999, Cloern 2001). Though the majority of research on freshwater enrichment has focused on lakes, nutrient pollution of streams and rivers is also of great importance, as the majority of flowing waters in the United States exhibit poor water quality from the standpoint of nutrient enrichment (Young et al. 1999, Smith et al. 1999, Miltner and Rankin 1998, Peterson et al. 1985). However, the applicability of the conceptual model to small flowing freshwater systems has not been extensively tested to date.

The foundation for development of enrichment is nutrient limitation of plant production (Figure 1). The initial response to increased nutrient loading is enhanced production of autotrophs that have rapid nutrient uptake and growth, often at the expense of those less able to take advantage of elevated nutrients (Harper 1992, Grall and Chauvaud 2002, Cloern 2001, Hessen et al. 1997). In particular, certain species of epiphytes, periphyton, filamentous macroalgae, and phytoplankton have been demonstrated to respond opportunistically to nutrient pollution (Mainstone and Parr 2002, Grall and Chauvaud 2002, Cloern 2001, Hessen et al. 1997). This elevated production may decrease available light for often light-limited, slower-growing macrophytes, and interfere with seed germination and seedling development (Mainstone and Parr 2002, Harper 1992, Hauxwell et al. 2001). Such changes in primary production ultimately lead to alterations in the composition of autotrophic communities, usually causing a decline in diversity, as species intolerant of low light, higher nutrients, or competition with opportunistic taxa disappear (Harper 1992).

Increased primary production results in elevated sedimentation of organic matter, which causes enhanced decomposition in the benthos. Intensified aerobic microbial

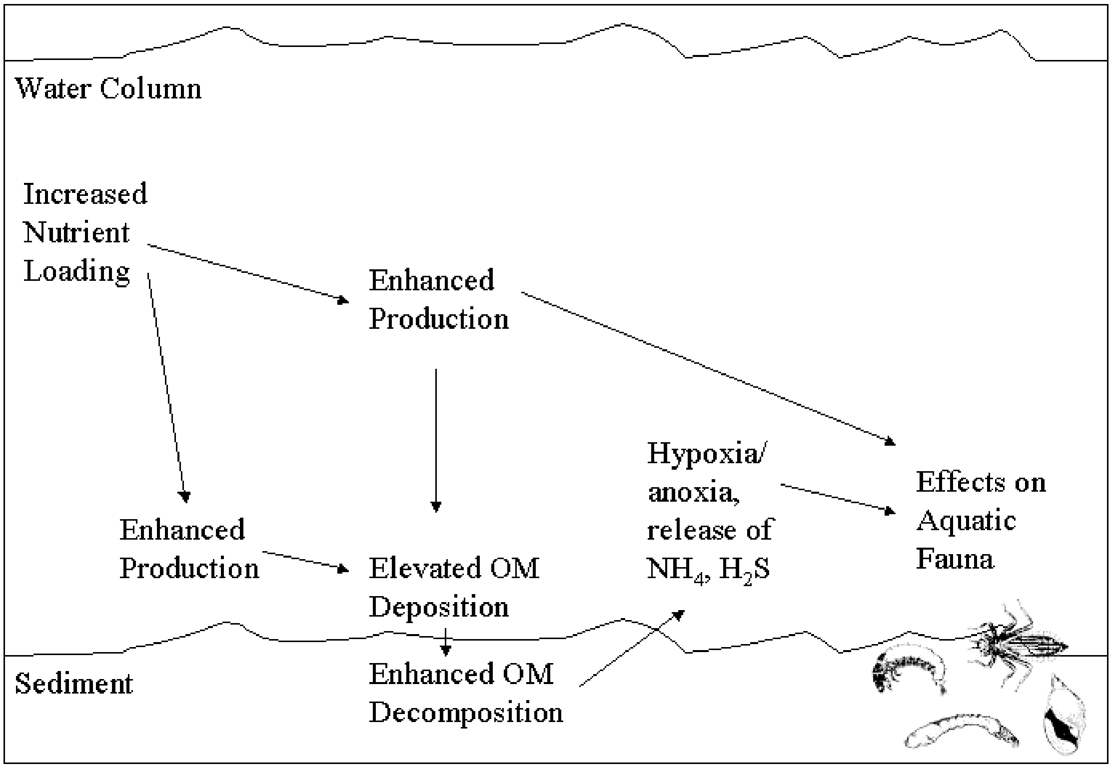

Figure 1. Conceptual model of nutrient enrichment (from Gray et al. 2002).

metabolism leads to decreased oxygen concentrations in the water column and sediments, as the biological oxygen consumption exceeds the oxygen supply (Cloern 2001, Meyer-Reil and Koster 2000, Harper 1992.). In particular, high sedimentation of materials with low C:N ratios (i.e. phytoplankton and fast-growing macroalgae) results in rapid decomposition of the organic material (Gray et al. 2002). Increased plant respiration associated with larger plant populations may also lower oxygen concentrations (Mainstone and Parr 2002). With continued nutrient enrichment, oxygen may be reduced to the point at which metabolism shifts from aerobic to anaerobic (Cloern 2001, Harper 1992). In the absence of oxygen, microbes use alternative electron acceptors (nitrate, manganese, iron, sulfate, and carbon dioxide) to oxidize organic material, resulting in release of hydrogen sulfide and/or ammonium, both of which are toxic to many aquatic organisms (Meyer-Reil and Koster 2000, Eyre and Ferguson 2002, Harper 1992). Anaerobic decomposition also increases release of phosphate from the sediments, potentially exacerbating conditions of nutrient enrichment (Harper 1992).

Nutrient Enrichment in Stream Ecosystems

The majority of studies of freshwater nutrient enrichment have focused on lakes and reservoirs. Because streams and rivers are flowing, and are often narrow and shaded watercourses, their response to nutrient pollution is typically different than that of lentic systems. While nutrient enrichment in lakes usually causes enhancement of phytoplankton production, most stream waters, especially headwater reaches, move too rapidly to produce significant phytoplankton growth (Harper 1992, Cushing and Allan 2001). As a result, nutrient enrichment in flowing waters is likely to lead to increased growth of tolerant periphyton or macrophytes rather than phytoplankton (Harper 1992, Mainstone and Parr 2002, Cushing and Allan 2001). In addition, flowing water transports nutrients downstream, so streams and rivers must receive continuous loadings to develop and sustain enrichment, while lakes may become eutrophic and remain in that state (Hynes 1970, Harper 1992). Furthermore, the relationship between nutrients and primary production in streams is confounded by a number of other factors which may affect production, including light, temperature, flow rate, substrate, flood frequency, and grazing (Young et al. 1999, Smith et al. 1999, Miltner and Rankin 1998). As a result, it was assumed for many years that other factors that can limit plant growth in flowing waters restrict the effects of nutrient pollution (Smith et al. 1999, Hynes 1970). However, evidence from a variety of sources indicates that streams and rivers are indeed susceptible to enrichment (Smith et al. 1999).

Several studies have demonstrated increased plant production associated with elevated inputs of phosphorus and/or nitrogen in flowing water systems. Though both nutrients can contribute to enrichment, phosphorus is generally in shortest supply, so is considered to be the nutrient with the most potential to limit plant growth in freshwaters (Carpenter et al. 1998, Mainstone and Parr 2002, Young et al. 1999). Experimental phosphate enrichment of an oligotrophic woodland stream in Tennessee resulted in dense growth of filamentous algae (Elwood et al. 1981, Rosemond et al. 1993). In addition, Peterson et al. (1983, 1985) observed stimulated periphyton production following P enrichment in an arctic river. Though fewer works have demonstrated N limitation, some enrichment experiments have shown similar results, suggesting that nitrogen may be the primary limiting nutrient in some systems at certain times of year (Smith et al. 1999, Rosemond et al. 1993). Many enrichment studies have also indicated that increases of both nitrogen and phosphorus often elevate production more than enrichment with either nutrient alone (Rosemond et al. 1993, Peterson et al. 1983, Smith et al. 1999, Hershey et al. 1988). A number of researchers, however, have failed to show relationships between nutrient addition and stimulated plant growth in flowing waters (Bernhardt and Likens 2004, Young et al. 1999, Madsen and Cedergreen 2002, Hessen et al. 1997). They have usually attributed these results to the initial nutrient status of the water, and to the plethora of confounding factors that can impact primary production in these systems.

Headwater streams have received less attention in nutrient enrichment studies than have larger, more autotrophic stream segments (Bernhardt and Likens 2004). In their pristine state, headwater reaches are typically forested, narrow, and well-shaded by riparian vegetation along the banks (Cushing and Allan 2001). In this environment, algal production is usually limited by the availability of light, and the growth of macrophytes is often limited by high water velocities, unsuitable substrate, and low nutrients (Cushing and Allan 2001, Harper 1992). As a result, allochthonous production of organic matter, primarily in the form of leaves, twigs, and branches from the surrounding canopy, is greater than autochthonous production, and the ecosystem is net heterotrophic (Cushing and Allan 2001, Hynes 1970). It should be noted that although periphyton production is usually small in these stream reaches, it may be an extremely important food source for secondary consumers (Bernhardt and Likens 2004).

Agricultural activities, however, can change the limiting forces on plant growth so that the response of headwater streams to accelerated nutrient inputs in agricultural watersheds is different than that of pristine systems. Though elevated nutrient inputs alone have been shown to cause enrichment in streams, clearing of vegetation and associated erosion often accompanying agricultural development may increase the susceptibility of the waters to enrichment. These practices may decrease shading, increase temperatures and turbidity, and reduce allochthonous inputs to streams, potentially enhancing algal production (Hynes 1970, Cushing and Allan 2001, Delong and Brusven 1992, Miltner and Rankin 1998). Combined with increased nutrient loading, these changes may cause a headwater reach to shift from being heterotrophic to autotrophic, with concomitant effects on the entire stream ecosystem (Peterson et al. 1985, Hynes 1970, Harper 1992, Delong and Brusven 1998).

Effects of Enrichment on Aquatic Fauna

A variety of physical, chemical, and biological factors determine the structure of macroinvertebrate communities in streams. Physical factors include current velocity, which directly affects substrate particle size, and relief, lithology, and presence of woody debris, all of which determine the diversity of microhabitats available for invertebrate colonization (Harper 1992, Malmqvist 2002, Wallace and Webster 1996, Hawkins et al. 1982, Cushing and Allan 2001). Water temperature and light are additional physical factors that may affect macroinvertebrate communities (Voshell 2002, Harper 1992, Cushing and Allan 2001). Chemical factors include dissolved oxygen, pH, hardness, and nutrients (Voshell 2002, Harper 1992, Cushing and Allan 2001). Biological factors are mainly those which dictate the food supply, such as composition of autotrophic communities and extent of riparian vegetation, in addition to factors such as competition and predation (Harper 1992, Wallace and Webster 1996, Hawkins et al. 1982). Of these physico-chemical and biotic variables, food availability and habitat structure are likely to be the most important factors influencing stream communities (Hawkins et al. 1982, Harper 1992, Hershey et al. 1988, Lammert and Allan 1999).

Macroinvertebrates inhabiting stream environments have evolved a varied array of mechanisms for obtaining food (Wallace and Webster 1996). To simplify the diversity of feeding methods in aquatic fauna, stream ecologists often employ Cummins' (1978) classification of aquatic invertebrates into functional feeding groups (FFG) according to mode of feeding. The major groups include shredders, organisms that feed on coarse particulate detritus (CPOM, >1mm in diameter), consume living vascular plant tissue, or bore into decomposing wood; gatherers, those that gather fine particulate organic material (FPOM, <1mm in diameter) on surfaces of sediment and surface films; filterers, organisms that feed by filtering FPOM in suspension; scrapers, those that graze attached periphyton from mineral and organic substrates; and predators, invertebrates that feed primarily on animal tissue (Cummins 1978, Voshell 2002, Wallace and Webster 1996).

Changes in primary production associated with nutrient enrichment impact sources, quantity, and quality of food, and these in turn directly impact consumers (Rosemond et al. 1993, Miltner and Rankin 1998). Shifts in the composition of autotrophic communities alter relative abundance and quality of various food supplies, impacting organisms adapted to subsist on certain sources. In general, increased benthic production generates increased food for scrapers, while elevated sedimentation of organic matter from enhanced pelagic production aids the gathering and filtering communities (Grall and Chauvaud 2002). Deposition of high quality organic matter, such as phytoplankton, supports collectors that occupy the shallow sediment depths, while sedimentation of poor quality organic matter benefits deep burrowing deposit-feeding fauna (Grall and Chauvaud 2002). In addition, enrichment impacts fauna directly by causing a decline in the diversity of autotrophic communities and reduction in macrophytic biomass, thereby reducing heterogeneity and quantity of habitat available for shelter and breeding (Mainstone and Parr 2002, Harper 1992).

Indirect effects of nutrient enrichment are primarily a result of lowered oxygen concentrations following enhanced decomposition and plant respiration. The physiological response to lowered oxygen is typically a reduction in metabolism, as organisms decrease energetically-costly activities to minimize oxygen requirements (Baker and Mann 1992, Harper 1992). A decline in metabolic rates results in decreased growth and lowered predation efficiency (Wannamaker and Rice 2000, Harper 1992). In addition, the production and release of ammonium and sulfide during hypoxic conditions may adversely affect reproduction and survival of fish and invertebrates (Gray et al. 2002, Miltner and Rankin 1998, Hickey et al. 1999). Furthermore, lowered oxygen concentrations can impact distribution of aquatic fauna. Some benthic organisms may rise from the sediment to reach oxygenated water, altering their depth distribution and siphon extension, and often increasing their susceptibility to predation (Gray et al. 2002). In mobile species, there is a general pattern of migration away from areas of low dissolved oxygen, potentially resulting in higher densities and increased competition for resources in oxygenated areas (Wannamaker and Rice 2000). Nonmobile organisms must adapt to conditions of lowered oxygen or perish (Bilyard 1987). Hypoxia/anoxia may also inhibit larval settlement (Baker and Mann 1992).

Changes in Benthic Community Structure with Nutrient Enrichment

Individual organisms respond differently to nutrient enrichment, as a result of variations in feeding habit, mobility, and life cycle (Mucha and Costa 1999). As a result, the composition and abundance of faunal communities exhibit a predictable pattern of change in response to increasing enrichment. The general model presented by Grall and Chauvaud (2002) has been observed in lentic, estuarine, and riverine systems, but has not been well tested in shallow streams, especially those in the coastal plain (Figure 2).

In conditions of fairly low nutrient enrichment, species richness, abundance, and biomass of all faunal groups increase as an elevated food supply enhances secondary production (Harper 1992, Grall and Chauvaud 2002, Cloern 2001). Faunal diversity is often highest at this stage of enrichment, while total abundance is moderate. With increasing enrichment, species richness decreases below background levels, and a gradual shift in community composition occurs (Bonsdorff and Pearson 1999). Replacement of less tolerant plants by opportunistic autotrophs causes shifts in dominance of invertebrate communities toward groups adapted to the prevailing food source (Harper 1992). For example, in typically heterotrophic headwater streams, shredders feeding on allochthonous materials compose about 35% of the invertebrate population, while grazers make up only 5% (Cushing and Allan 2001). When nutrient enrichment increases production of opportunistic algae, headwater streams may become autotrophic, and behave more like mid-stream reaches (Hynes 1970, Harper 1992, Miltner and Rankin 1998), with shredders decreased to 5% of the population while grazers increase to 30% (Cushing and Allan 2001). Several enrichment studies have demonstrated that elevated

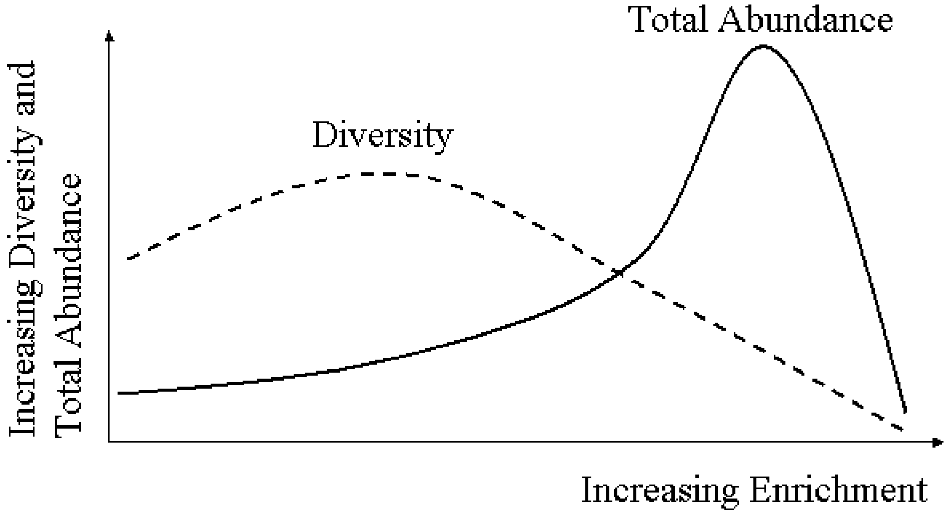

Figure 2. Pattern of response of benthic fauna to increasing enrichment (from Grall and Chauvaud 2002).

algal biomass had a positive effect on growth of grazers (Miltner and Ranking 1998), including herbivorous snails (Rosemond et al. 1993, Elwood et al. 1981), mayflies (Rosemond et al. 1993), and the grazing chironomid *Orthocladius rivulorum* (Hershey et al. 1988).

In addition to changes in functional feeding groups, shifts in dominance by certain taxonomic groups may occur. Organisms that typically take advantage of nutrient enrichment include aquatic worms, certain midges, true flies, and non-insects (Stribling et al. 1998, Harper 1992). Some larval chironomid midges, for example, display adaptations to low oxygen conditions by containing hemoglobin, an oxygen-binding blood pigment (Harper 1992). In addition, the oligochaete *Tubifex tubifex* has been shown to grow and reproduce under continual anoxia for long periods of time (Harper 1992). In contrast, certain larvae of mayflies, stoneflies, and caddisflies are typically sensitive organisms that decline in number with increasing nutrient pollution (Stribling et al. 1998, Harper 1992, Sponseller et al. 2001). In general, communities dominated by non-insects, such as worms, snails, leeches, and others, are more tolerant of pollution than assemblages dominated by insects (US EPA 1997).

Overall, nutrient enrichment results in increased dominance by short-lived, fast-growing, and often tolerant invertebrates, at the expense of longer-lived, less opportunistic species unable to adapt to changed conditions (Voshell 2002, Grall and Chauvaud 2002, Harper 1992, Miltner and Rankin 1998, Delong and Brusven 1998). This leads to greatest total abundance of organisms and lowest diversity at high levels of enrichment, as evidenced by the work of many researchers (Grall and Chauvaud 2002, Miltner and Rankin 1998, Sponseller et al. 2001, Manel et al. 2000, Delong and Brusven

1998). In the final stage of enrichment, even the most opportunistic fauna are unable to adapt to excessive changes in primary production and in the physico-chemical environment, and an afaunal state results (Voshell 2002, Grall and Chauvaud 2002).

Use of Benthic Macroinvertebrates as Bioindicators

Biological monitoring is defined as the use of biological responses to detect changes in the environment that are often due to anthropogenic activities (Rosenberg and Resh 1993). Because the responses of ecosystems to environmental impacts are typically complex and diverse, it has been recognized that chemical and physical measurements are unable to properly assess impacts (Karr 1991). Analysis of biological parameters of aquatic systems improves water quality assessment programs by directly measuring the source at risk, and by detecting problems that solely physical or chemical methods may miss or underestimate (Karr 1991). In addition, physical and chemical analyses are often static measurements, while the resident biotic communities integrate water quality conditions over time, acting as continuous monitors of the water they inhabit (Rosenberg and Resh 1993). Furthermore, living communities respond to the entire range of biogeochemical factors in the environment (Karr and Chu 2000). For these reasons, it is widely accepted that to adequately detect impairment in aquatic systems, it is necessary to monitor the biota (Downes et al. 2002).

The recognition that invertebrates are essential components of freshwater, marine, and terrestrial environments has led to increased demand for conservation of their populations, and for the use of invertebrates as tools in ecological assessment and monitoring (New 1998). Because the response of invertebrates to ecosystem changes varies among different taxa, estimates of diversity and abundance of those known to respond to certain factors are commonly used as indicators of habitat quality in comparing assemblages or sites (New 1998, Wallace and Webster 1996). Biomonitoring of stream invertebrates to assess the impacts of anthropogenic and environmental variables has been performed in many regions both in the United States (Miltner and Rankin 1998, Richards et al. 1997, Harding et al. 1998, Sponseller et al. 2001, Whiles et al. 2000, Lammert and Allan 1999, Delong and Brusven 1998, Allan et al. 1997), and around the world (Smith et al. 1999, Kay et al. 1999, Clenaghan et al. 1998, Ometo et al. 2000, Manel et al. 2000). Many of these studies demonstrated relationships between macroinvertebrate community structure and land use, both present and historic, at a variety of spatial scales from riparian zones to entire watersheds. However, very few studies have assessed impacts of agricultural land use on macroinvertebrate communities in low-gradient nontidal streams, and to our knowledge, none have done so in Virginia's coastal plain.

Invertebrates inhabiting the benthic environment have become particularly valuable in biomonitoring studies. Most benthic invertebrates have short life cycles and respond rapidly to alterations in habitat (New 1998). In addition, many benthic invertebrates are relatively sedentary, so are forced to either adapt to environmental stress or perish (Bilyard 1987). As a result, benthic environments are the most sensitive habitats of aquatic ecosystems to nutrient pollution, and changes in the structure of benthic communities are often extremely sensitive signs of enrichment (Cloern 2001, Gray et al. 2002). Benthic infauna, in particular, are superior to many other benthic groups for use in biomonitoring, as a result of their nonmotile nature and importance to overall ecosystem structure and function (Bilyard 1987).

A variety of metrics of invertebrate community structure have been developed for use in biomonitoring studies. Typical metrics include measures of taxonomic richness, taxonomic composition, dominance by tolerant and sensitive taxa, and abundance of functional feeding groups (i.e., grazer, filterer, etc.). Tolerance values for specific taxa, usually ranging from 0 (very intolerant) to 10 (very tolerant), have been developed by ranking the ability of a taxon to survive during exposure to stresses resulting from pollution, habitat degradation, or modifications in hydrology (Stribling et al. 1998, Maxted et al. 1999, Wallace and Webster 1996). Members of the Ephemeroptera, Plecoptera, and Trichoptera (EPT) orders are known to be sensitive to water quality impairment, and as a result, the number of EPT taxa is commonly used as an indicator of low stress in bioassessment studies (Miltner and Rankin 1998, Sponseller et al. 2001, Whiles et al. 2000, Maxted et al. 1999). Because non-insects, especially aquatic worms, and midge and other true fly larvae are less sensitive to pollution than other groups, high relative abundances of these taxa are typically used as indicators of poor water quality (Barbour et al. 1999, Stribling et al. 1998, US EPA 1997, Harper 1992). In addition, a variety of composite indices have been developed and employed that combine numerous metrics of community structure, allowing measurements of multiple facets of communities to be incorporated into a single value (Kerans and Karr 1994, Stribling et al. 1998, Maxted et al. 1999, Ohio EPA 1988, Barbour et al. 1999, Karr 1999). Table 1 describes the 21 metrics employed in this study and indicates expected direction of change with increase in nutrient enrichment.

Table 1. Metrics of macroinvertebrate community structure and expected response to increasing nutrient enrichment (adapted from Stribling et al. 1998, Barbour et al. 1999, USEPA 1997).

Metric	Definition	Expected Response
Taxonomic Richness		
# Taxa	Indicates overall richness of the community	Increase, then decrease
# EPT Taxa	Number of taxa in the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)	Increase, then decrease
Taxonomic Composition		
Total Abundance	Total number of organisms of all taxa	Increase
% EPT	Percent of total abundance composed of mayflies, stoneflies, and caddisflies	Decrease
% Ephemeroptera	Percent mayflies	Decrease
% Plecoptera	Percent stoneflies	Decrease
% Trichoptera	Percent caddisflies	Decrease
% Diptera	Percent "true" flies	Increase
% Chironomidae	Percent midges	Increase
% Non-insects	Percent non-insects	Increase
% Oligochaeta	Percent aquatic worms	Increase
Simpson's DI	Measure of taxonomic richness and evenness $D = E\, p_i^2$ where p_i = proportion of total number of individuals in the i^{th} taxa	Decrease
Tolerance		
% Dominant Taxon	Percent of the most abundant taxon	Increase
Hilsenhoff BI	Measure of tolerance/intolerance of community $HBI = E\, \frac{(x_i t_i)}{N}$ where x_i = number of individuals within genera i t_i = tolerance value for genera i n = total number of organisms in sample	Increase
Feeding		
% Scrapers	Percent of individuals that graze attached periphyton	Increase, then decrease
% Shredders	Percent that tear and shred CPOM	Decrease
% Gatherers	Percent that gather FPOM on or in sediment	Increase
% Filterers	Percent that filter FPOM in suspension	Increase
Other		
$\delta^{15}N$ amphipods	$\delta^{15}N \text{ (per mil)} = \frac{\text{Rsample} - \text{Rstandard}}{\text{Rstandard}} \times 1000$ where $R = {}^{15}N/{}^{14}N$	Decrease
$\delta^{15}N$ chironomids	same as above	Decrease
$\delta^{15}N$ worms	same as above	Decrease

In addition to measures of faunal community structure, stable isotope analysis has become a useful tool in biomonitoring studies in recent history. Changes in composition and abundance of faunal communities reflect alterations that have already taken place in the biota, while the use of stable isotope signatures may provide early detection of nutrient enrichment before the occurrence of major changes in aquatic communities (McClelland et al. 1997). Though fractionations may cause difficulty in determination of sources, the change in isotopic signature among different locations may provide information on the relative intensity of nutrient loading. For example, a decrease in $\delta^{15}N$ signatures in consumers along a gradient of increasing agricultural land use may indicate an increase in nutrient loading along this gradient, as N signatures are lowered due to preferential uptake of ^{14}N with increased nitrogen availability (McClelland and Valiela 1998). Several studies have used stable isotopic analysis of N in primary producers and consumers in attempts to identify the source of nitrogen and to determine the relative intensity of N loading (McClelland et al. 1997, McClelland and Valiela 1998, Costanzo et al. 2001, Evgenidou and Valiela 2002, Jones et al. 2001).

OBJECTIVES AND HYPOTHESES

The overall objective of this work was to examine the impacts of agricultural land use on aquatic communities in nontidal coastal plain streams by using benthic macroinvertebrates as bioindicators of nutrient pollution. In conjunction with the measurement of a suite of chemical and physical variables, macroinvertebrate community structure was assessed.

<u>Objective 1</u>: To determine composition, diversity, abundance, and dominance of invertebrate communities, and examine changes in these measures with variation in percentage of agricultural land in catchments.

<u>Objective 2:</u> To analyze relationships between macroinvertebrate diversity, composition, abundance, and dominance and water column and sediment variables with variation in percentage of agricultural land in catchments.

<u>Working Hypothesis 1</u>: The structure of macroinvertebrate communities will vary as a function of percentage of agricultural land use in watersheds and associated nutrient enrichment.

a. Watersheds with moderate levels of agricultural land use will have highest diversity.

b. Watersheds with high levels of agricultural land use will have highest total abundance of organisms, reflecting dominance of opportunistic, tolerant taxa.

Working Hypothesis 2: Patterns in the structure of macroinvertebrate communities will be correlated to changes in sediment and water column characteristics and to food availability.

a. Watersheds with low water column dissolved oxygen, and high sediment organic matter, porewater sulfide, and porewater ammonium will have low diversity and high total abundance, with communities dominated by opportunistic, tolerant taxa.

b. Watersheds with moderate levels of water column and sediment chlorophyll will have high food availability, resulting in high diversity and biomass.

SIGNIFICANCE OF RESEARCH

Though many researchers have established that agricultural land use and other anthropogenic activities often increase nutrient inputs to flowing waters, the effects of nutrient pollution in streams and rivers has been considerably understudied. Nontidal coastal plain streams, in particular, are often ignored, despite the fact that intensive agricultural activity and characteristics of topography and soils in the coastal plain make ground and surface waters highly susceptible to nutrient pollution (US EPA 1997, Reay et al. 1992). Furthermore, little is known about the structure of macroinvertebrate communities in low-gradient coastal streams, and even less about their potential response to nutrient enrichment.

In addition, invertebrates play an essential role in a variety of processes in flowing water ecosystems. Macroinvertebrates are important in nutrient and organic matter cycling, primary productivity, and decomposition, and serve as an energy source for fish, birds, bats, predaceous insects, and spiders (Malmqvist 2002, Wallace and Webster 1996). The importance of these organisms to the structure and function of stream ecosystems, along with their invaluable use as bioindicators of impairment, make conservation of aquatic invertebrates a necessity.

Finally, a goal of research at the VCR-LTER is to “address the effects of changing land use on groundwater quality and the impact of these waters on the adjacent marshes and lagoons” (Hayden et al. 2000). The present research will provide information on relationships between land use and physical, chemical, and biological parameters of VCR tidal creeks that link uplands to marshes and lagoons. Furthermore, this dataset should provide future researchers with a foundation for additional studies of VCR stream ecosystems, and enable managers to make informed decisions concerning land use and water quality in coastal plain environments.

METHODS

Site Overview

The Virginia Coast Reserve (VCR) is a complex system of mainland, marshes, lagoons, and barrier islands located on the Eastern Shore of Virginia. The mainland peninsula is composed primarily of deciduous and evergreen forests, fringing salt marshes and mudflats, and agricultural fields drained by tidal creeks that flow through marshes into offshore lagoons (Hayden et al. 2000). The tidal creeks are fed predominantly by groundwater, with little surface runoff due to flat topography, small watershed area, and highly permeable soils.

The VCR mainland consists of 63 sub-watersheds, ranging in size from 19 to 1,970 ha (Porter and Hayden 2001). Each catchment contains one or several streams that flow into the adjacent marshland, and ultimately into the coastal lagoons. The VCR system is unique in that the sub-watersheds are similar in size, shape, climate, and topography, but vary in land use patterns. Cropland agriculture is the dominant form of anthropogenic land use, and ranges from 20-80% of watershed area (Hayden et al. 2000). Major crops are soybeans, tomatoes, corn, cotton, and wheat. The unique structure of the VCR landscape provides the opportunity to analyze effects of agricultural land use on aquatic communities using a multiple watershed approach within a relatively small regional area.

Site Selection and GIS Analyses

The study area consists of nine headwater tidal creeks located on the southern portion of the Delmarva Peninsula on Virginia's Eastern Shore. Watersheds were chosen to represent a range of agricultural land use, and vary from 28 – 91 % of area in agriculture. The percent of land in forest is inversely related to percent agricultural land, and ranges from 9 - 68 %. All catchments have < 5% urban development. Watersheds are similar in location (all are located in Northampton County, along a 22 mile transect), catchment area (63 – 293 ha) and type of agricultural land use (predominantly cropland). In addition, attempts were made to avoid watersheds containing septic lagoons, large dammed ponds, channeled portions, and registered point source inputs.

Watershed boundaries and land use percentages were generated using ArcGIS and ArcVIEW. Watershed boundaries were digitized based on USGS 1:24,000 quadrangles, aerial photographs, and an existing watershed coverage obtained from the VCR-LTER Dataset Catalog (Porter and Hayden 2001, Dataset ID: VCR01076). Watershed boundaries extended down to the 5-foot contour. Land cover for the catchments was obtained from the 1988-1999 NOAA C-CAP dataset, based on landsat thematic mapper scenes from November 1988 and October 1989 (www.csc.noaa.gov/crs/lca/ccap.html). To develop the watershed land use dataset, the C-CAP data was overlaid with the watershed coverage, and resulting polygon areas were tabulated. Polygon areas for cropland and grassland were summed to represent % agriculture, deciduous forest, evergreen forest, mixed forest, mixed shrub/scrub, palustrine forest, and estuarine and palustrine emergent wetland were summed to represent % forest, and high and low intensity development were summed to represent % development. Extent of the riparian zone was determined with aerial photographs, by measuring the distance from the center of the creek to the edge of forest at 0, 50, and 100 m upstream of the sampling location.

One sampling station was established in the non-tidal portion of each creek, near the headwaters. All creeks are transected by Rte. 600, from which each sampling station was accessed. Table 2 and Figure 3 indicate watershed land use patterns and location.

Water Column Analyses

All water column variables were measured in June, August, and October 2003, and in January and March 2004 (Table 3). Samples were collected during baseflow conditions for all months except August. August samples were gathered during a period of relatively high flow following a storm. Triplicate water samples were collected just below the surface in dark 2 L Nalgene bottles for analysis of nutrients, total suspended sediments (TSS), suspended particulate organic matter (SPOM), and chlorophyll a. The bottles were stored in the dark on ice until return to the lab, and all samples were filtered within 24 h of collection.

For nutrient analysis, 15 ml subsamples were filtered (0.45 um pore size) and frozen for later analysis of ammonium, nitrate, total dissolved nitrogen (TDN), and phosphate. Ammonium was analyzed by the phenol hypochlorite method (Grasshoff et al. 1983), nitrate through cadmium reduction with a Perstorp autoanalyzer (Perstorp 1993), and phosphate by the molybdate method (Grasshoff et al. 1983). TDN samples were oxidized by persulfate digestion (Grasshoff et al. 1983), and were subsequently

Table 2. Study creeks and watershed land use.

Creek	Total Watershed Area (ha)	% Agriculture	% Forest	% Development
Greens	237	28	68	4
Mill	127	33	65	2
Indiantown	103	35	64	1
Walls Landing	293	61	35	4
Phillips	114	62	36	2
Taylor	151	66	31	3
Narrow Channel	253	71	29	0
Holt	168	74	25	1
Lakewood	63	91	9	0

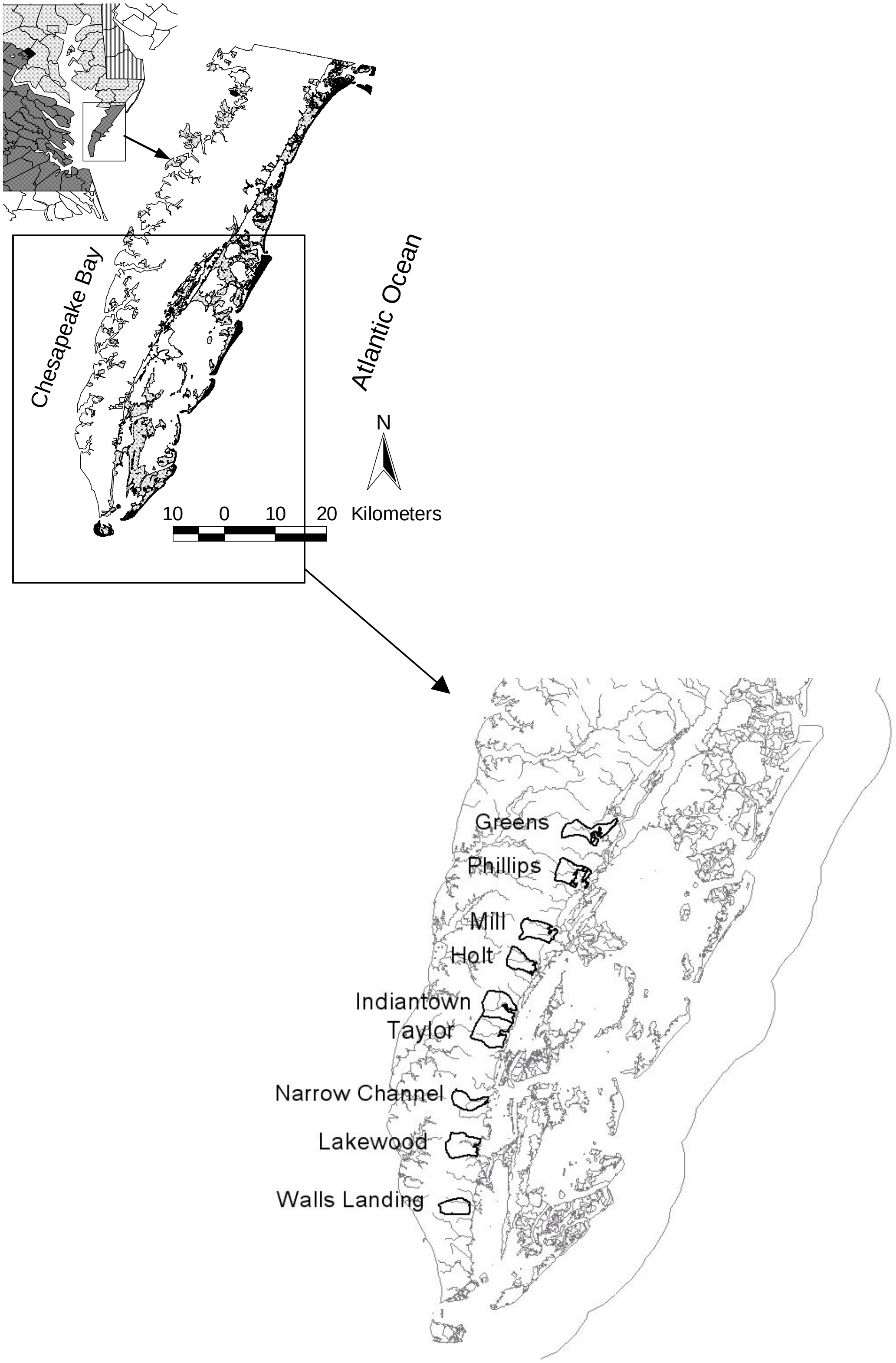

Figure 3. Eastern Shore of Virginia and study watersheds.

analyzed for nitrate as above. DON was determined by subtracting DIN (nitrate + ammonium) from TDN.

For determination of TSS, a minimum of 300 ml of each water sample was filtered through glass fiber filters (0.7 um pore size) using a vacuum pump. Filters were dried for 24 h at 60 C, and TSS was calculated as follows:

$$\text{TSS (mg/L)} = \frac{\text{[final filter weight (mg) – tare filter weight (mg)]}}{\text{volume water filtered (L)}}$$

To determine SPOM, dried filters were ashed at 500 C for 5 hours and SPOM was calculated as follows:

$$\text{SPOM (mg/L)} = \frac{\text{[final filter weight (mg) – AFDW of filter (mg)]}}{\text{volume water filtered (L)}}$$

For determination of water column chlorophyll a, a minimum of 500 ml of each water sample was filtered through glass fiber filters (1.0 um pore size) using a vacuum pump. Filters were frozen, and chlorophyll a retained on the filters was later extracted using acetone and analyzed with a spectrophotometer.

Water column dissolved oxygen (DO) was measured in the field using a YSI Model 57 DO probe just above the sediment surface. Water temperature and salinity was measured using a YSI Model 30 conductivity meter. Creek discharge was measured using a SonTek Flowtracker Handheld Acoustic Doppler Velocimeter (ADV). Light was measured above and below the water surface using a Licor LI-1000 light meter. pH was measured one time, in March, using a pH test kit. Flow was not measured in June, and due to problems with equipment, light was not measured in June or October.

To facilitate comparison between watersheds, all water column concentrations were calculated as fluxes and were converted to area-weighted values. To compute area-weighted fluxes, the monthly concentration was multiplied by the monthly discharge rate and this value was divided by watershed area. Because flow was not measured in June, area-weighted fluxes for June were calculated using the line equation generated by the regression of concentration and area-weighted fluxes for the other months. As a result, standard errors could not be computed for the June flux data. Annual mean fluxes were calculated for each water column variable by averaging monthly fluxes for the year. Because the August sampling occurring during a period of high flow, while all other sampling events were during baseflow, August water column variables were left out of calculations of annual mean values.

Sediment Analyses

Sediments were analyzed for benthic microalgal chlorophyll a, organic matter (OM) content, extractable ammonium, and carbon to nitrogen ratio (C:N), in July and October 2003, and in January and March 2004 (Table 3). Sediment particle size distribution, and porewater ammonium and hydrogen sulfide were analyzed in July 2003 (Table 2). Samples were collected during baseflow conditions for all months except August. August samples were gathered during a period of relatively high flow following a storm.

For determination of benthic microalgal chlorophyll a, three sediment cores to a depth of 1 cm were collected using a 5 cc syringe corer. Each core was placed on ice in the dark until return to the lab, where samples were frozen until extraction with acetone and analysis with a spectrophotometer.

For OM, extractable ammonium, and C:N profiles, three cores (8 cm diameter, 10 cm depth) were collected at each site, and returned intact to the laboratory. Sediment was extruded from the cores to make sections of 0-2 cm, 2-5 cm, and 5-10 cm. To determine OM content, a 2 cc subsample was removed from each core section using a 5 cc syringe corer. The subsample was dried at 60 C for 24 h to obtain dry weight, then ashed at 500 C for 5 h to yield ash-free dry weight (AFDW). OM content was calculated as follows:

$$\text{OM content (\%)} = \frac{\text{dry weight} - \text{AFDW}}{\text{dry weight}} \quad \text{x} \quad 100$$

To determine extractable ammonium, each core section was added to a solution of 1 M KCl (2 x volume of sediment), extracted on a shaker table for 1 h, and centrifuged for 10 min at 2600 rpm. Supernatant was filtered (0.45 um pore size) and frozen for later analysis of ammonium by the phenol hypochlorite method. For determination of sediment C:N, a subsample from each core section was frozen, freeze-dried, ground with a mortar and pestle, and analyzed on a Carlo Erba Elemental Analyzer.

For determination of particle size distribution, three cores (4.5 cm diameter, 10 cm depth) were collected at each site. Sediment was sieved into particle size classes for coarse sand, medium sand, fine sand, very fine sand, and clay + silt. Wet sediments for the sand fractions were dried for 24 h at 60 C. For the clay + silt fraction, three 10 ml subsamples were sampled from a known volume of slurry (clay + silt fraction plus DI water), and the subsamples were dried for 24 h at 60 C. The percentages of each size fraction were calculated as follows:

$$\text{Sand (\%)} = \frac{\text{dry weight of sand fraction}}{\text{dry weight of total sediment}} \quad \text{x} \quad 100$$

Percentage of clay + silt were calculated as follows:

$$\text{Dry weight} = \text{mean subsample conc.(dry wt./subsample vol.)} \times \text{total vol. slurry}$$

$$\text{Clay + silt (\%)} = \frac{\text{dry weight of clay + silt}}{\text{dry weight of total sediment}} \times 100$$

To determine porewater hydrogen sulfide and ammonium, three cores (8 cm diameter, 10 cm depth) were collected from each site and transported intact on ice to the lab. Using a porewater probe, a sample of pore water was removed at depths of 0, 2, 4, 6, and 8 cm. Samples for ammonium were filtered (0.45 um pore size) and frozen for later analysis by the phenol hypochlorite method. Sulfide was analyzed immediately by the Cline method as modified by Otte and Morris.

Benthic Macroinvertebrates

Benthic macroinvertebrates were collected in June, August, and October 2003, and in January and March 2004 (Table 3). Samples were collected during baseflow conditions for all months except August. August samples were gathered during a period of relatively high flow following a storm. At each site, eight cores (8 cm diameter, 10 cm depth) were collected randomly within a 10 m stream reach. Contents of the cores were sieved through a 0.5 mm mesh sieve and organisms retained on the sieve were removed. All individuals collected were identified to the lowest taxonomic level possible (usually genus), counted, and preserved in 90% alcohol. Because August samples were collected for analysis of isotopic signatures, these individuals were kept alive overnight in vials containing deionized water in order to flush the gut contents. These samples were then frozen, freeze-dried, ground, and sent to the stable isotope facility at UC Davis for analysis of $\delta^{15}N$.

References used in identification of organisms included Jessup et al. (2002), Peckarsky (1990), Pennak (1989), Merritt and Cummins (1978), and McCafferty (1981). Because of the difficulty in generic identifications of the Family Chironomidae and Class Oligochaeta, the services of taxonomic specialists were used to obtain identification of these individuals to genus. August samples needed to remain frozen; thus, they were not submitted. In addition, flatworms were identified only to family, due to difficulty in identification and lack of available assistance.

Tolerance values and functional feeding groups for all organisms were obtained from the literature (Stribling et al. 1998, Resh et al. 1996, Merritt and Cummins 1978). Measures of macroinvertebrate community structure were tabulated, including total abundance, metrics of taxonomic richness and composition, tolerance, and relative abundance by organisms of four functional feeding groups (scrapers, gatherers, filterers, and shredders). Because August chironomids and oligochaetes were not identified to genus, number of taxa, Simpson’s DI, percentages of functional feeding groups, percentage of dominant taxon, and the Hilsenhoff BI were not tabulated for this month. In addition, flatworms were left out of taxonomic richness calculations because they were only identified to family.

Data Analysis

Four main statistical analyses were performed on the data, using SAS Version 8.2. To assess impacts of land use on water column and sediment variables, linear regressions were conducted on annual averages, using % agriculture, % agriculture^2, % forest, % development, and riparian zone width as the independent variables. The square of agriculture was used in regressions to identify patterns that were expected to be nonlinear. To gain information on potential relationships among water column and sediment variables, a correlation matrix was performed using annual averages of variables. For analysis of land use impacts on macroinvertebrate communities, linear and repeated measures regressions were used. Linear regressions with land use were conducted on annual averages of macroinvertebrate metrics, while repeated measures analysis was used to determine effects of land use during each of the four seasons. A correlation matrix was utilized to assess relationships between macroinvertebrate measures and water column and sediment variables. Data were transformed as necessary using log, natural log, or square root transformations, and significance was determined at p values < 0.05.

Table 3. Methods and frequency of analyses.

	Variable	Analysis Method	Jun 03	Jul 03	Aug 03	Oct 03	Jan 04	Mar 04
Water Column	Nitrate	Cadmium reduction	X		X	X	X	X
	Ammonium	Phenol hypochlorite method	X		X	X	X	X
	Total Dissolved Nitrogen	Persulfate digestion and cadmium reduction	X		X	X	X	X
	Phosphate	Molybdate method	X		X	X	X	X
	TSS/SPOM	Drying and combustion	X		X	X	X	X
	Chlorophyll a	Acetone extraction	X		X	X	X	X
	Dissolved Oxygen	YSI DO probe	X		X	X	X	X
	Salinity, Temperature	YSI Conductivity meter	X		X	X	X	X
	Light	Licor light meter			X		X	X
	Discharge	SonTek Flowtracker ADV			X	X	X	X
	pH	pH test kit						X
Sediment	Organic Matter	Drying and combustion		X		X	X	X
	C:N	Carlo Erba Elemental Analyzer		X		X	X	X
	Extractable Ammonium	KCl extraction and phenol hypochlorite method		X		X	X	X
	Benthic Microalgal Chlorophyll a	Acetone extraction		X		X	X	X
	Particle Size Distribution	Sieving and drying		X				
	Porewater sulfide and ammonium	Phenol hypochlorite method (ammonium) and Cline method (sulfide)		X				
Macro-invertebrates	Community structure metrics	Sieved through 0.5 mm sieve, identified, and counted	X		X	X	X	X
	$\delta^{15}N$	Analyzed at UC Davis stable isotope facility			X			

RESULTS

Watershed Land Cover and Water Column and Sediment Variables

Annual averages of all variables were used in regression and correlation analyses, and are presented here. Measurements of water column variables were converted to area-weighted flux rates to enable comparisons among sites. Data are organized so that percent agriculture increases from left to right on x-axes of graphs. All raw data are located in Appendix A.

Agricultural land use ranged from 28 – 91% of total watershed area, and was inversely correlated with percent forest ($r = -0.99792$, $p < 0.0001$). Urban development was < 5% of watershed area in all catchments, and was not significantly correlated with either % agriculture ($r = -0.49886$, $p = 0.1716$) or % forest ($r = 0.44195$, $p = 0.2336$). Width of the riparian forest ranged from 0 – 185 m, and was highest at 100 m upstream of the sampling location for the majority of sites. Watershed land cover was not correlated with riparian zone widths at 0, 50, or 100 m upstream.

The nine study reaches were narrow and shallow, with average width ranging from 1.44 – 2.33 m and depth ranging from 0.14 – 0.26 m. Baseflow discharge was generally low for all sites, and monthly values ranged from 0.0027 – 0.038 m^3/s (Figure 4). Greens Creek had the highest annual mean discharge (0.027 m^3/s), while Narrow Channel and Holt creeks had moderate values (0.016 and 0.015 m^3/s, respectively), and all other sites had discharge rates < 0.01 m^3/s. Log-annual mean discharge did not vary significantly with watershed area ($r = 0.60651$, $p = 0.0833$) or percentage of land cover

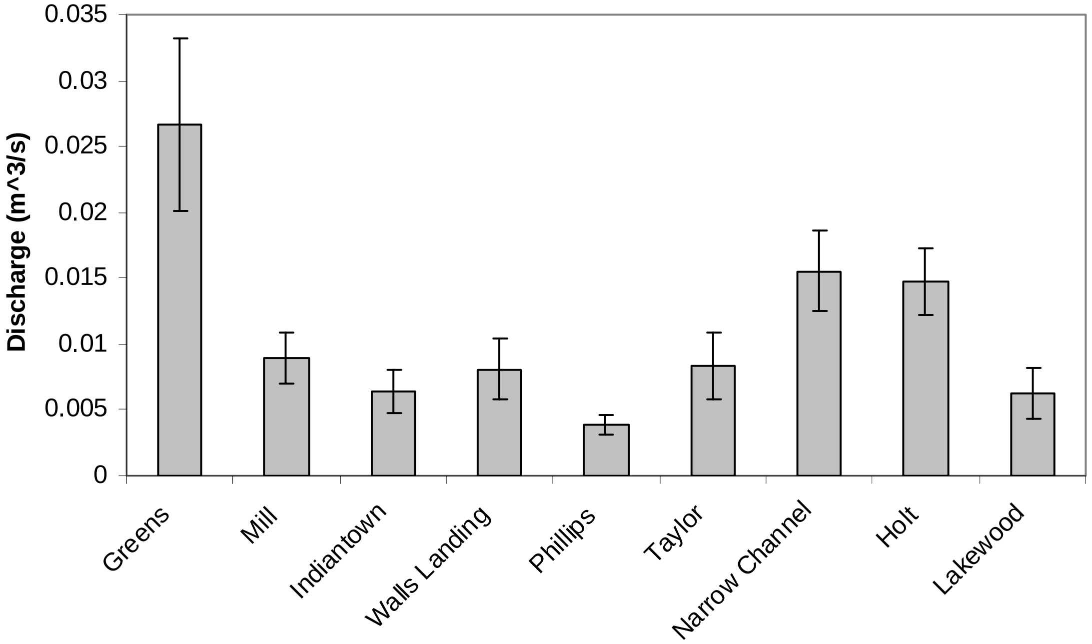

Figure 4. Annual mean discharge and standard error bars.

(% agriculture: r = -0.26364, p = 0.4931, % forest: r = 0.25794, p = 0.5028, % development: r = 0.20079, p = 0.6045).

Stream bed sediments were composed primarily of sand, which varied from 74 – 97 % of total dry weight of sediment (Figure 5). Percentages of sand in the coarse (> 500 um) and medium (250 – 500 um) size classes were higher than those of fine and very fine sand for most sites. The clay + silt fraction was < 13% for all sites but Phillips and Taylor, which had higher % clay + silt values (26 and 28 %, respectively). Stream water salinities were fresh at 0 – 0.2 ppt during all seasons, with a value of 0.1 ppt for 93% of samples. As measured in March, all creeks were slightly acidic, with pH values of either 6.0 or 6.5.

Total dissolved N (TDN) concentrations ranged from 19.10 – 672.03 uM, and fluxes ranged from 0.93 – 103.33 umol/ha/s. Nitrate concentrations ranged from 0.98 – 815.33 uM, and fluxes ranged from 0.04 – 77.54 umol/ha/s. Lakewood Creek had the highest annual mean TDN and nitrate fluxes (57.77 and 69.68 umol/ha/s, respectively) (Figure 6). Creeks with the lowest % agriculture (Greens, Mill, and Indiantown) had the lowest annual mean fluxes of TDN (5.31, 2.75. and 2.33 umol/ha/s, respectively) and nitrate (8.92, 0.71, and 3.43 umol/ha/s, respectively). For many samples, nitrate concentrations were higher than TDN, indicating likely error in measurement of one or both of these variables. Concentrations of ammonium ranged from 0 – 24.84 uM, and fluxes ranged from 0 – 0.74 umol/ha/s. Annual mean ammonium fluxes were highest in Lakewood Creek (0.50 umol/ha/s) and lowest in Narrow Channel (0.12 umol/ha/s) and Holt Creek (0.11 umol/ha/s) (Figure 7). Because nitrate was often higher than TDN, calculations of dissolved organic N (DON) (from TDN – (NO_3^- + NH_4^+)) resulted in

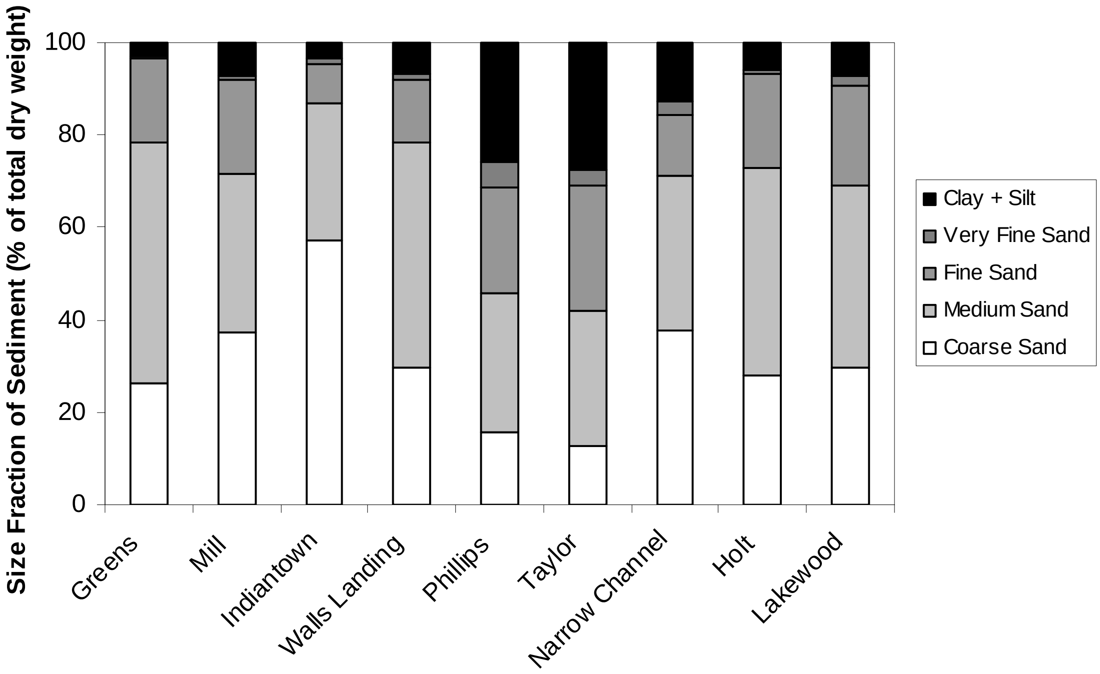

Figure 5. Sediment particle size composition.

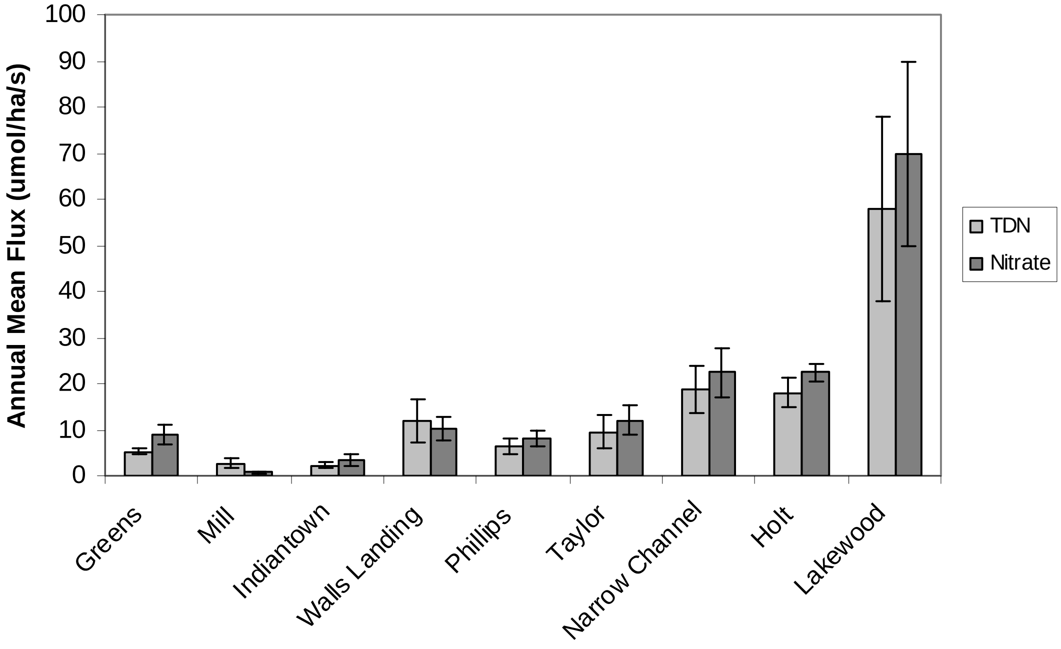

Figure 6. Annual mean fluxes and standard error bars for TDN and nitrate.

negative values for many samples, and these were assigned concentrations of zero. As a result, standard errors were very high for most annual mean DON fluxes (Figure 7). DON concentrations ranged from 0 – 19.50 uM, and fluxes ranged from 0.00 – 4.49 umol/ha/s. Mill was the only creek that had positive DON values for all months. Phosphate concentrations and fluxes were low, with values ranging from 0.013 – 1.78 uM, and 0.0016 – 0.11 umol/ha/s, respectively. Annual mean phosphate flux was < 0.05 umol/ha/s for all sites (Figure 8).

Sediment C:N ranged from 14.99 – 37.49 and no consistent trend with depth was noticeable. Greens and Holt creeks had highest annual mean C:N values (25.09 – 28.85, and 26.49 – 27.87, respectively), while Lakewood had the lowest (16.81 – 19.11) (Figure 9). Extractable ammonium ranged from 0 – 0.70 umol/g dry sediment, and concentrations increased with depth in most streams. Phillips Creek had the highest annual mean extractable ammonium values (range: 0.30 – 0.43 umol/g dry sediment), while concentrations in Greens and Holt were <0.06 umol/g dry sediment (Figure 10). Porewater ammonium was measured in July, and varied from 4.72 – 193.73 uM, increasing with depth below the sediment surface. Highest mean annual porewater ammonium concentrations were found in Walls Landing (16.12 – 193.73 uM) and Lakewood (24.29 – 213.22 uM), while values in Greens Creek were <10 uM for all depths (Figure 11). Porewater sulfide ranged from 0.48 – 25.49 uM, and again increased with depth. Annual mean porewater sulfide was highest in Walls Landing (1.37 – 25.49 uM), and was generally < 10 uM for all other streams (Figure 12).

Water column chl a concentrations and fluxes were low, and varied from 0.21 – 16.27 ug/L, and 0.007 – 1.43 ug/ha/s, respectively. Taylor Creek had the highest annual

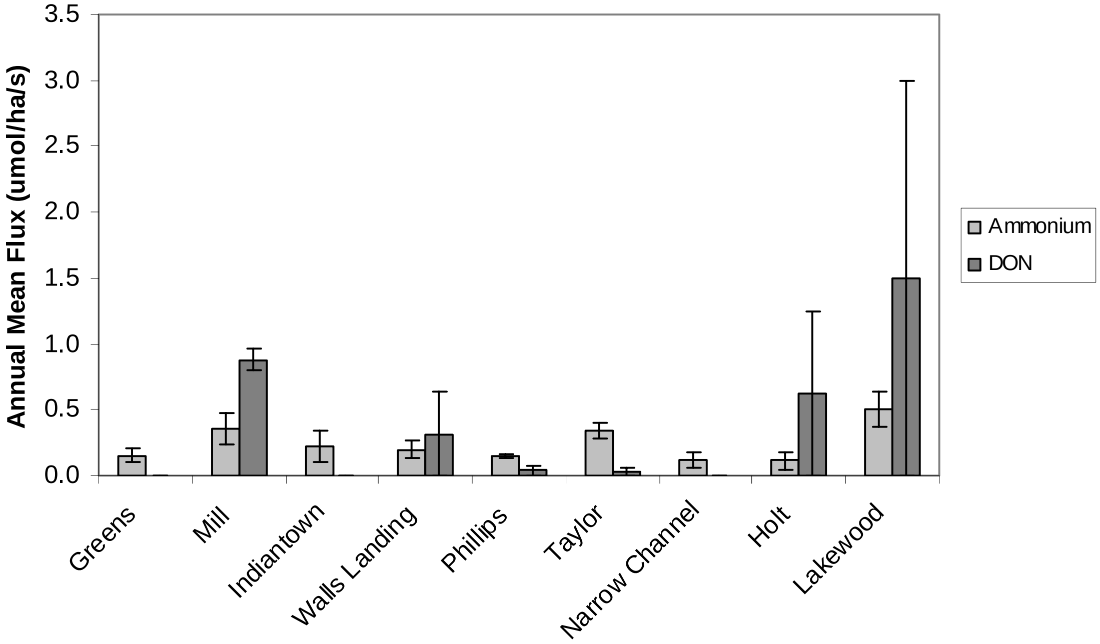

Figure 7. Annual mean fluxes and standard error bars for ammonium and DON.

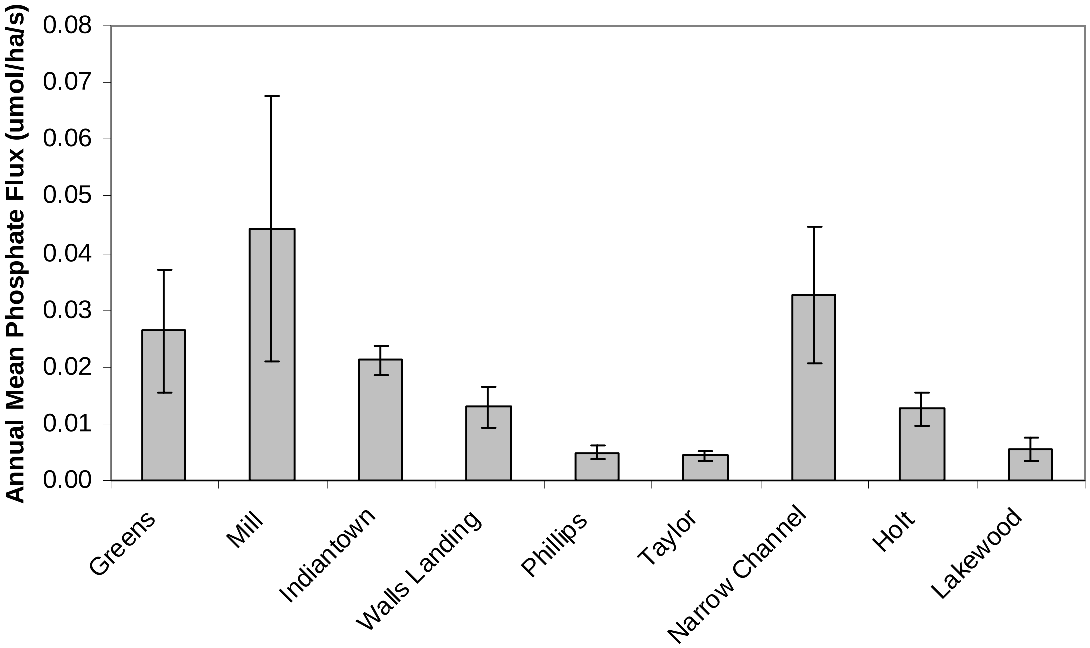

Figure 8. Annual mean phosphate flux and standard error bars.

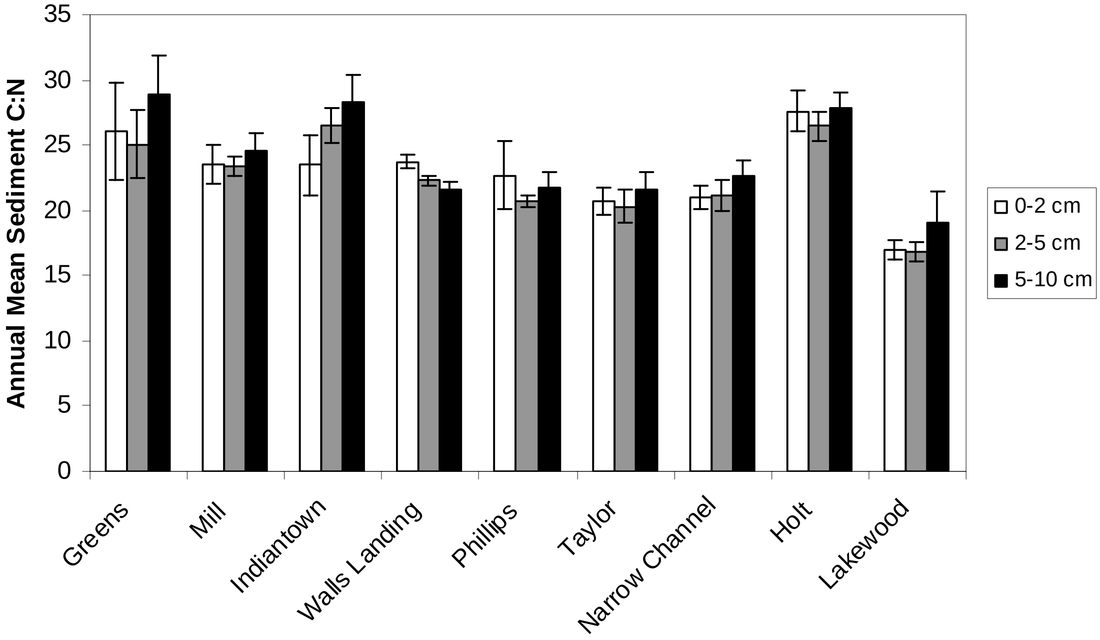

Figure 9. Annual mean sediment C:N profiles and standard error bars.

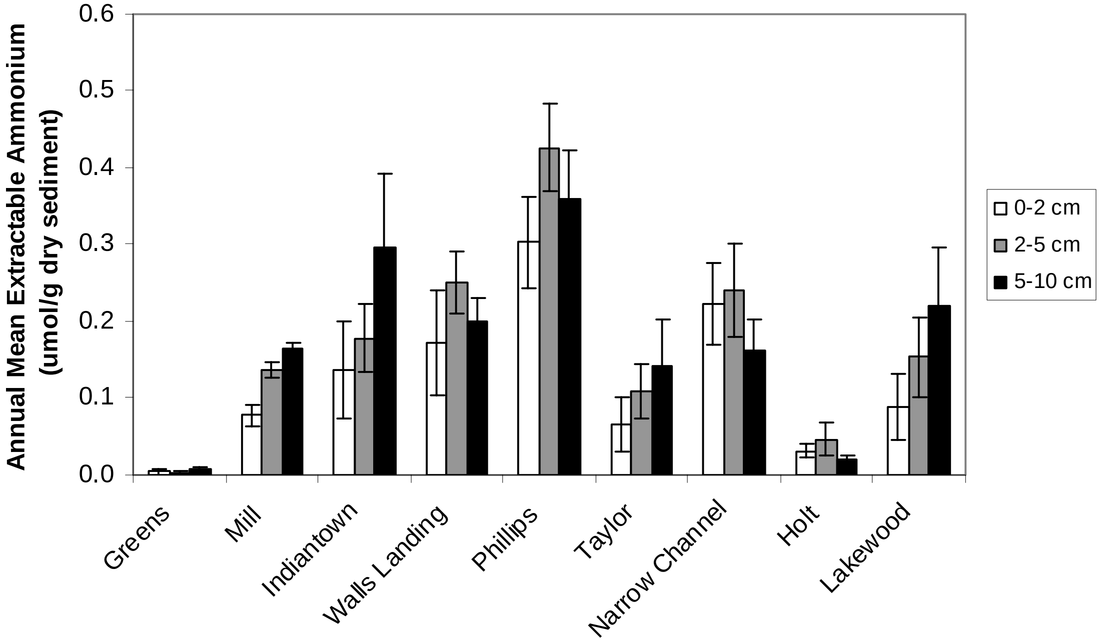

Figure 10. Annual mean profiles and standard error bars for extractable NH_4^+.

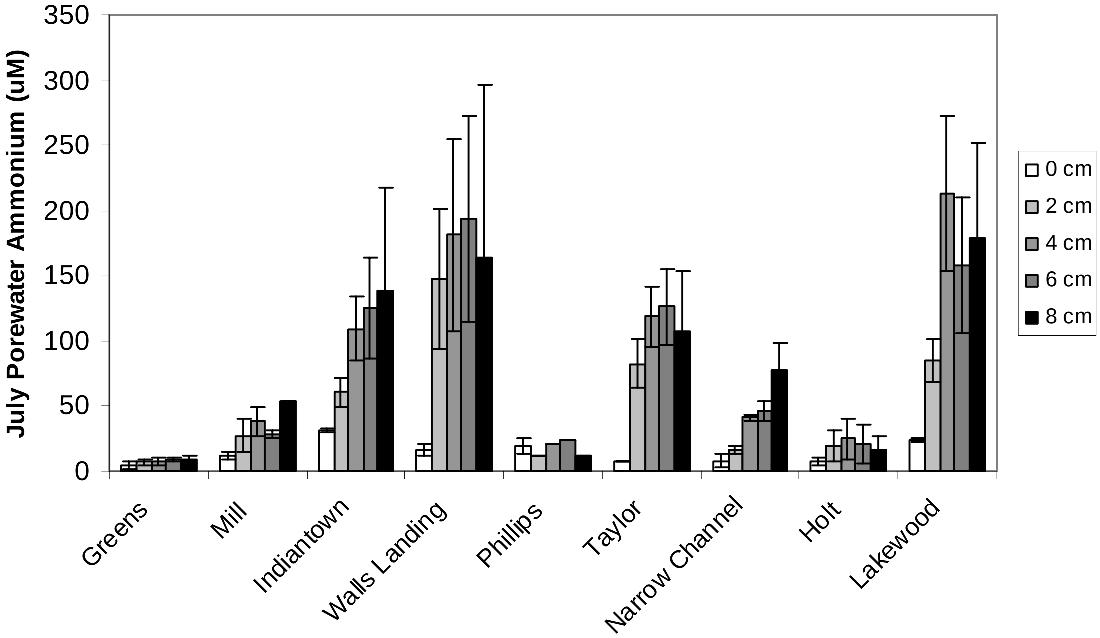

Figure 11. July porewater ammonium profiles and standard error bars.

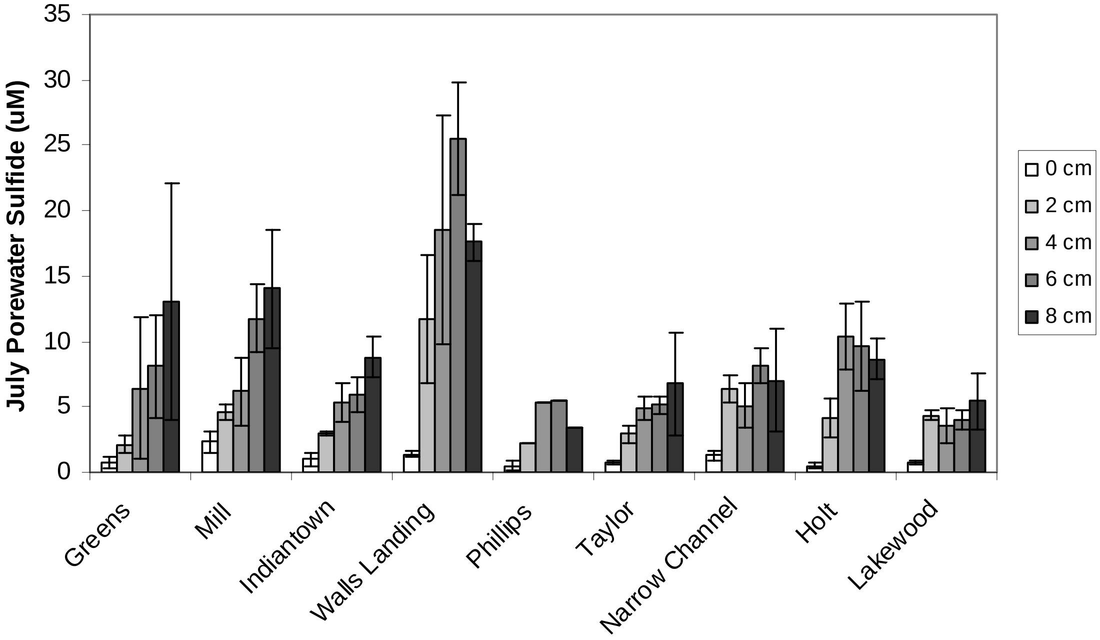

Figure 12. July porewater sulfide profiles and standard error bars.

mean water column chl a flux (0.478 ug/ha/s), while values were less than 0.25 ug/ha/s for all other streams (Figure 13). Benthic chl a was higher than that of the water column, and ranged from 2.68 – 49.39 mg/m^2. Greens Creek had higher annual mean benthic chl a (34.76 mg/m^2) than all other streams (<20 mg/m^2) (Figure 14).

Total suspended sediment (TSS) concentrations and fluxes ranged from 3.9 – 38.1 mg/L and 0.1 – 3.1 mg/ha/s, respectively. Annual TSS fluxes were similar (1.0 – 1.5 mg/ha/s) for all sites except Walls Landing and Phillips (0.4 and 0.2 mg/ha/s, respectively) (Figure 15). Suspended particulate organic matter (SPOM) concentrations varied from 1.6 – 23.3 mg/L, and fluxes ranged from 0.1 – 3.5 mg/ha/s. Annual SPOM fluxes were more variable than TSS fluxes, with highest concentration in Lakewood (1.3 mg/ha/s) and lowest values in Walls Landing (0.2 mg/ha/s) and Phillips (0.1 mg/ha/s) (Figure 15).

Depth of the detrital layer on the surface of stream bed sediments ranged from 0 – 2.8 cm, with an annual average of >1 cm for five of the nine creeks. Taylor, Holt, and Lakewood had average detrital layer depths ranging from 0.46 – 0.67, and no detrital layer was present in Greens Creek (Figure 16). Sediment organic matter (OM) ranged from 0.26 – 22.47 %, and no depth trend was evident. Annual mean OM was highest in Phillips Creek (15.76 – 20.35 %) and lowest in Greens (0.62 – 2.54 %) and Lakewood (2.88 – 3.84 %) (Figure 17).

Dissolved oxygen (DO) concentrations were variable among sites, with values ranging from 2.6 – 11.4 ppm. DO fluxes ranged from 0.04 – 1.8 mg/ha/s, and annual mean fluxes were highest in Greens and Lakewood (0.8 mg/ha/s) and lowest in Walls Landing (0.2 mg/ha/s) (Figure 18). Incident light was only measured in summer, winter,

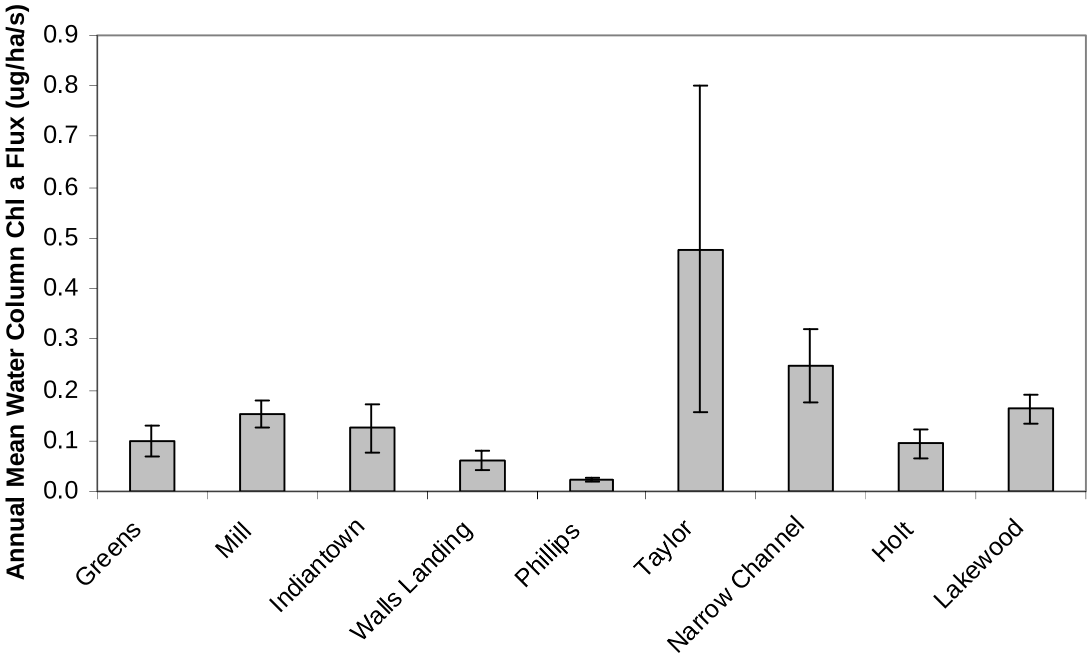

Figure 13. Annual mean water column chl a flux and standard error bars.

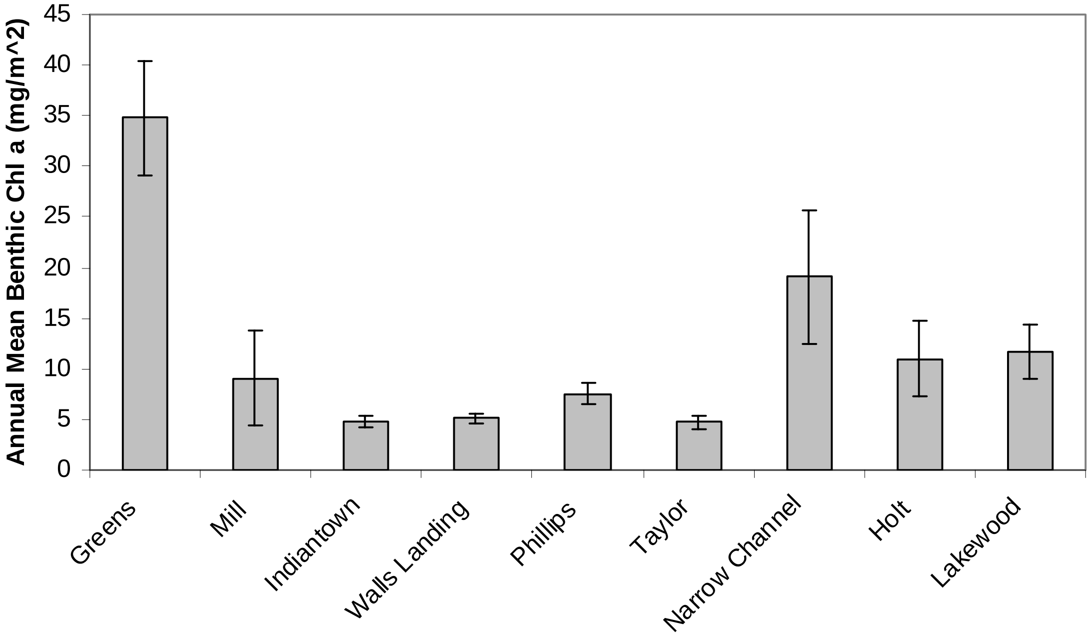

Figure 14. Annual mean benthic chl a concentrations and standard error bars.

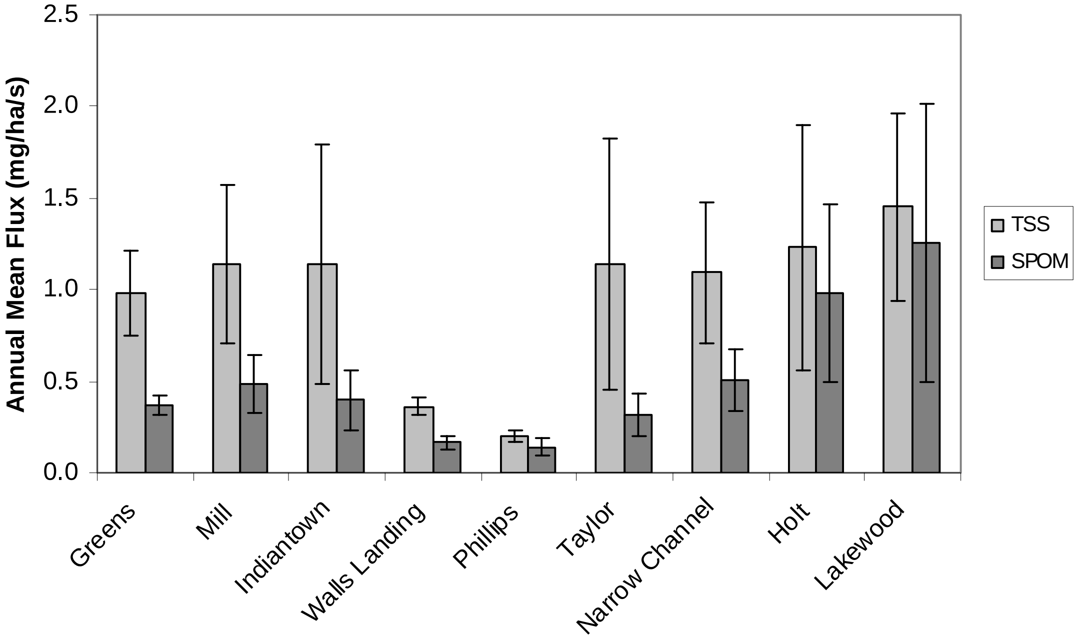

Figure 15. Annual mean fluxes and standard error bars for TSS and SPOM.

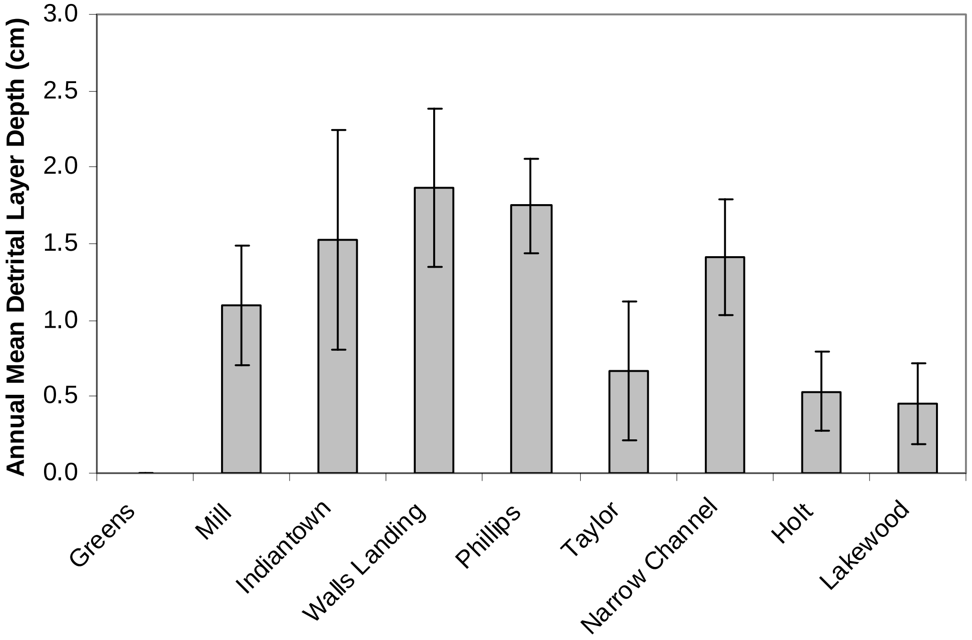

Figure 16. Annual mean depth of detrital layer and standard error bars.

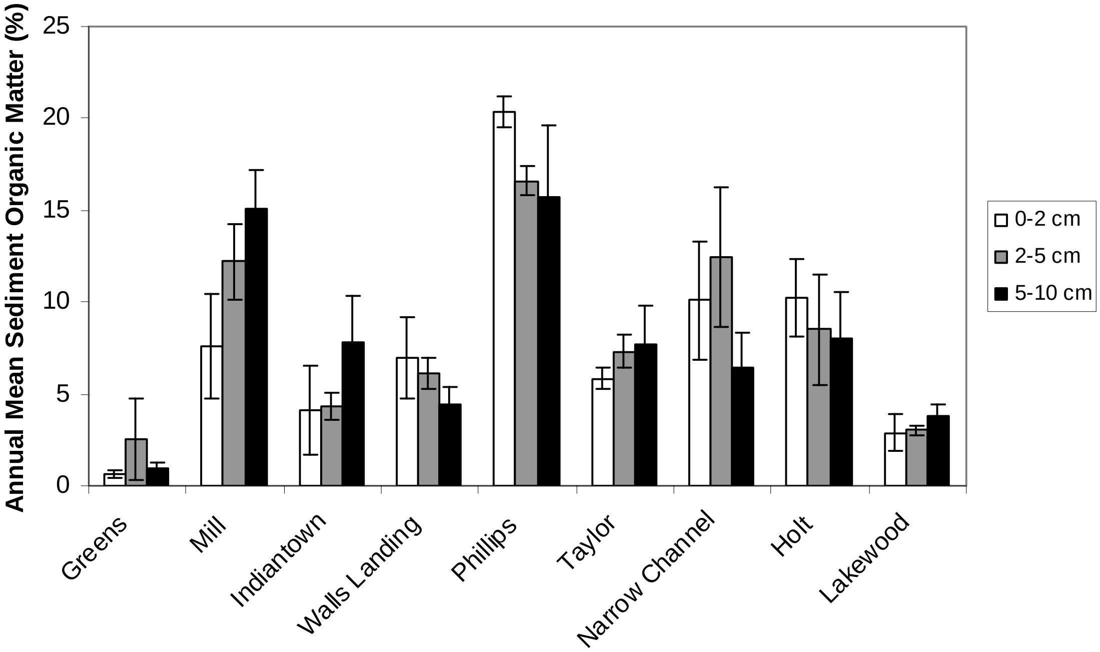

Figure 17. Annual mean sediment organic matter profiles and standard error bars.

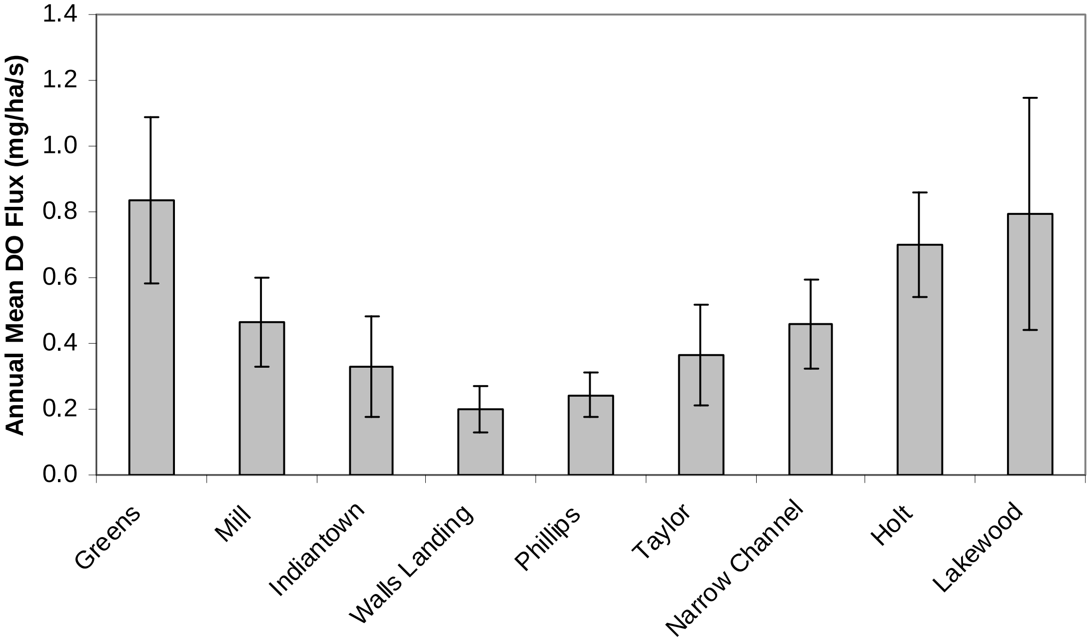

Figure 18. Annual mean DO flux and standard error bars.

and spring, and ranged from 26.2 – 1320 uE/m^2/s. Annual mean light was highest in watersheds with lowest % agriculture (Greens: 738.17, Mill: 866.40, Indiantown: 1035.90 uE/m^2/s) (Figure 19). Water temperature ranged from 4.2 C in the winter to 24.6 C in the summer, with annual averages of 12.75 – 14.95 C, and generally highest values in creeks with highest % agriculture (Figure 20).

Regressions of Land Use and Water Column and Sediment Variables

Regressions of water column and sediment variables versus % agriculture, % agriculture^2, % forest, % development, and riparian zone width were conducted on annual averages of all variables. Four variables were significantly related to agricultural land use ($p<0.05$). Log-TDN flux and log-NO_3^- flux had strong positive relationships with % agriculture (Figure 21, log-TDN flux: $r^2 = 0.8414$, $p = 0.0005$; log-NO_3^- flux: $r^2 = 0.6836$, $p = 0.006$). Water temperature also increased with agriculture ($r^2 = 0.5116$, $p = 0.0303$, Figure 22). A weaker, but significant negative relationship was evident between % agriculture and sediment C:N at 5-10 cm ($r^2 = 0.4656$, $p = 0.0429$, Figure 23). Regressing variables with % agriculture^2 did not generate any additional significant relationships with land use.

As expected due to the negative relationship between % forest and % agriculture, % forest was inversely related to log-TDN flux ($r^2 = 0.8459$, $p = 0.0004$), log-NO_3^- flux ($r^2 = 0.6905$, $p = 0.0055$), and water T ($r^2 = 0.5210$, $p = 0.0281$), and positively related to sediment C:N at 5-10 cm depth ($r^2 = 0.69443$, $p = 0.0379$). The regression between % forest and light was also significant, with light increasing as forest increases ($r^2 = 0.4644$, $p = 0.0432$, Figure 24). Regressions with % development were not significant for any

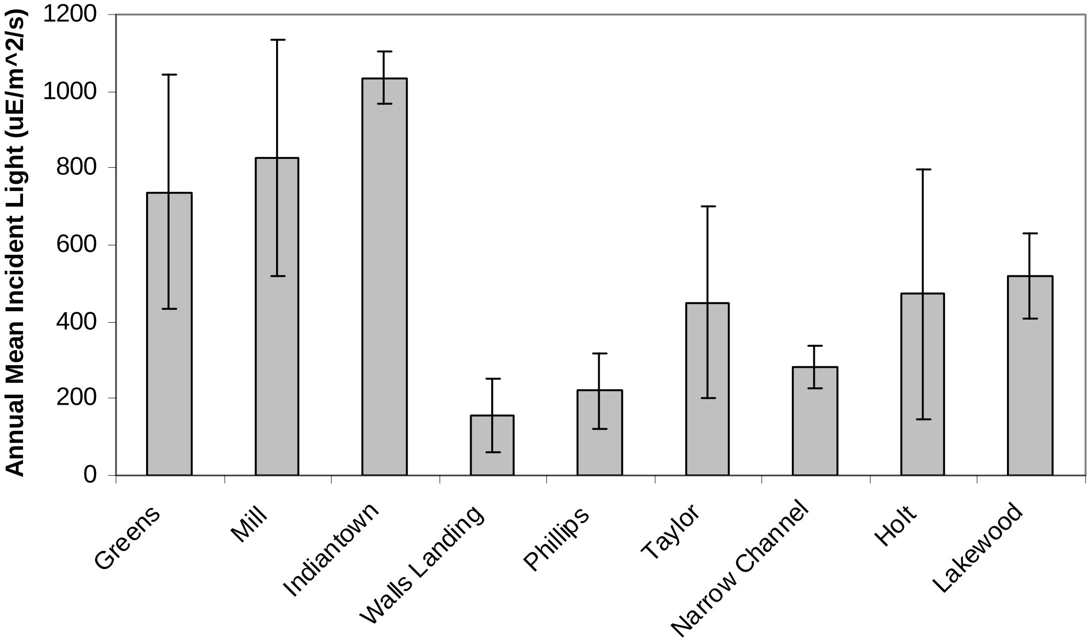

Figure 19. Annual mean light and standard error bars.

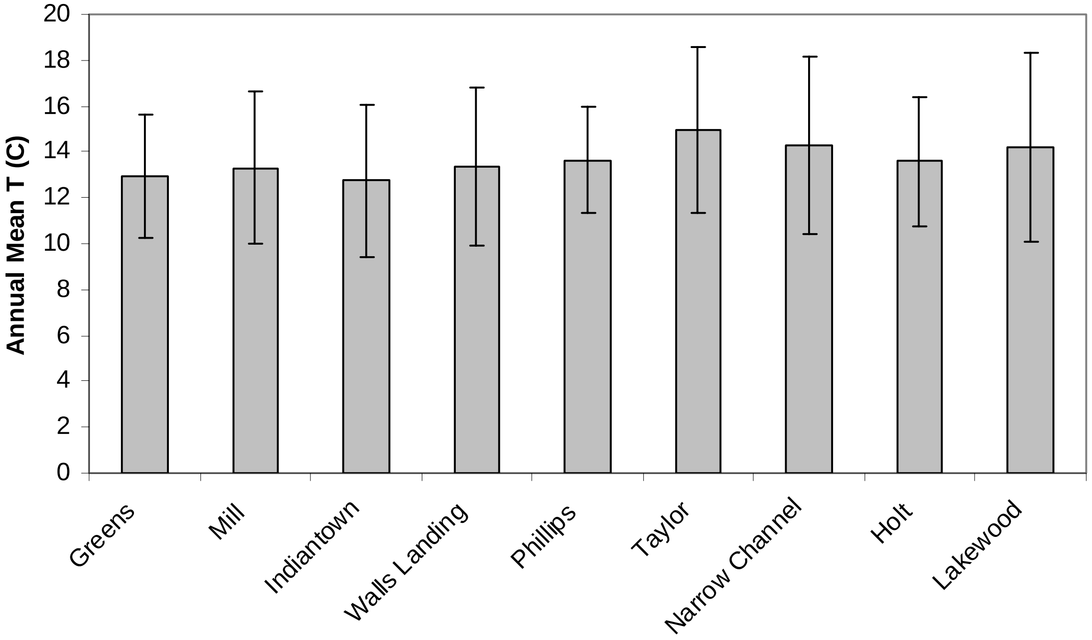

Figure 20. Annual mean temperature and standard error bars.

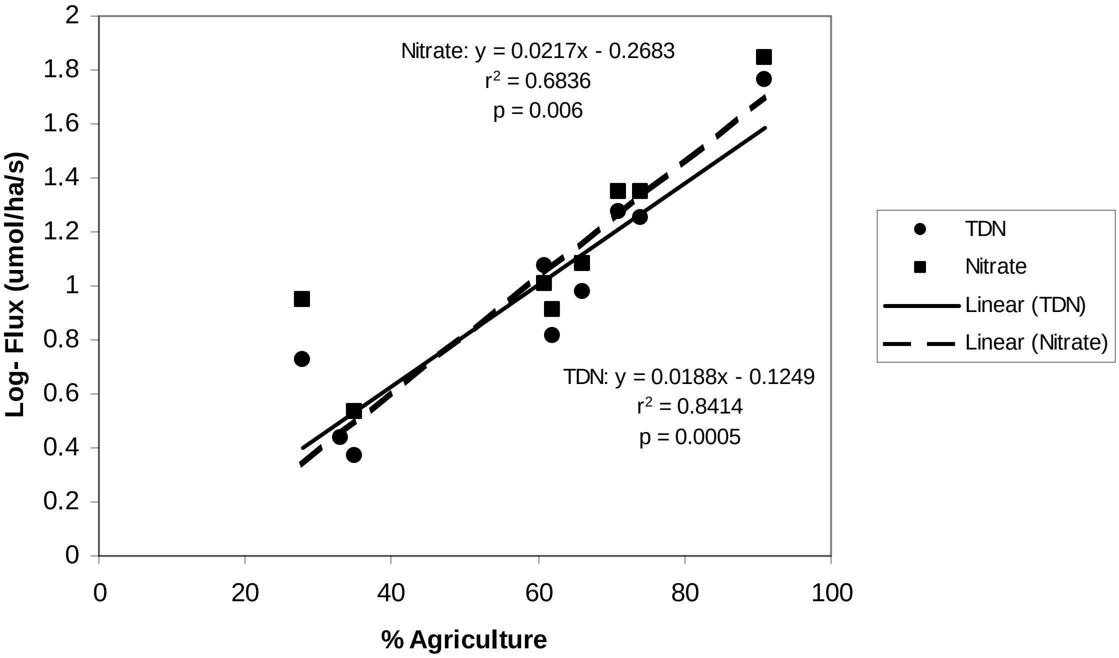

Figure 21. Regressions of % agriculture vs. log-TDN flux and log-NO3 flux.

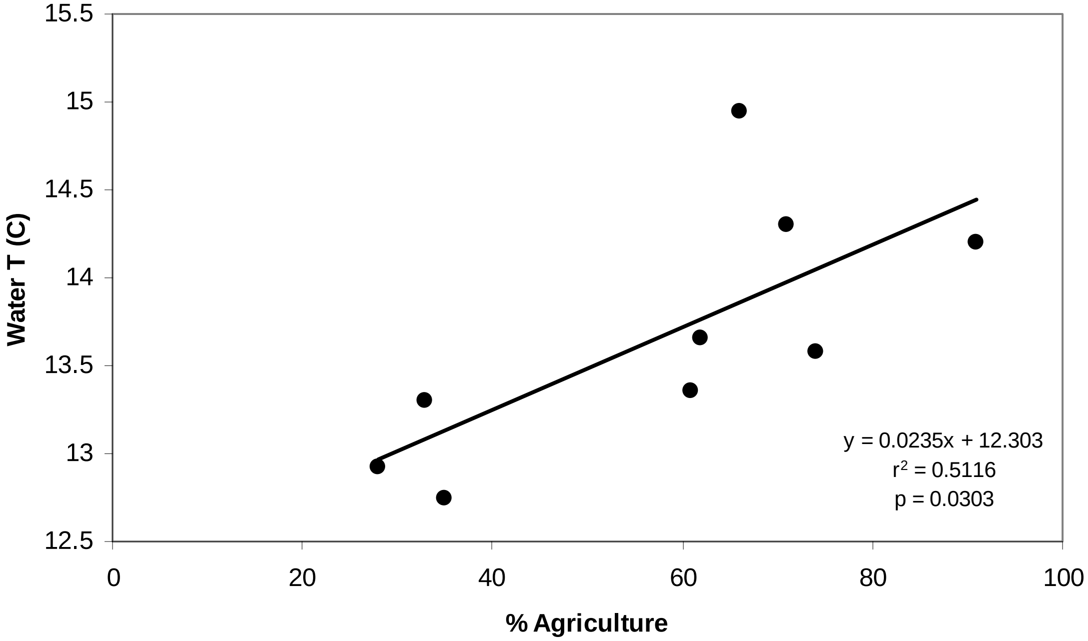

Figure 22. Regression of % agriculture vs. water temperature.

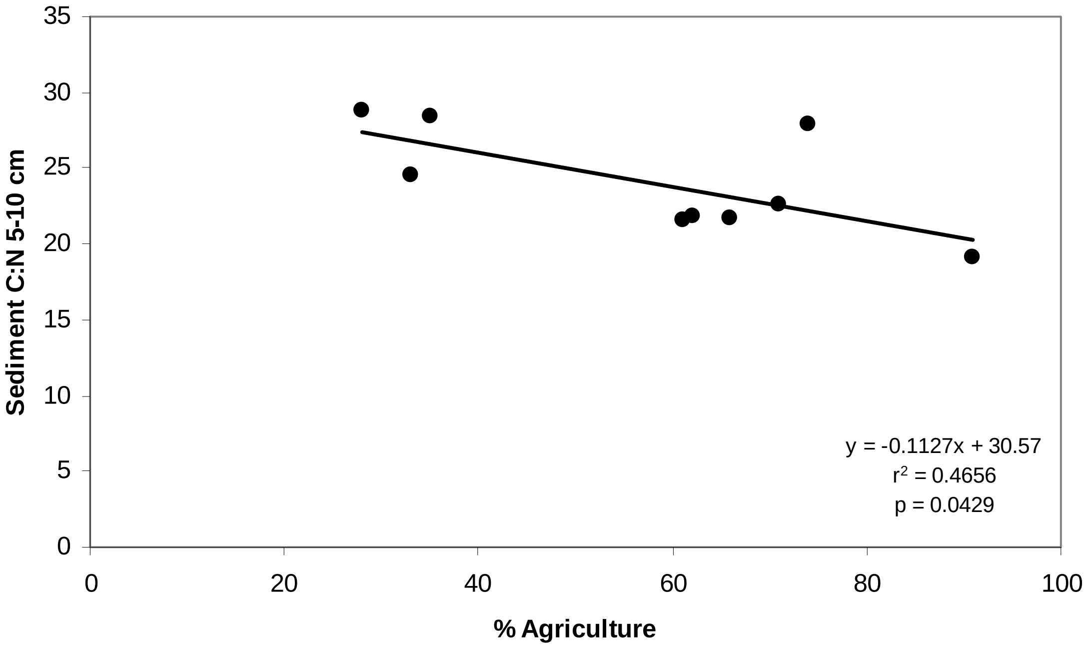

Figure 23. Regression of % agriculture vs. sediment C:N at 5-10 cm depth.

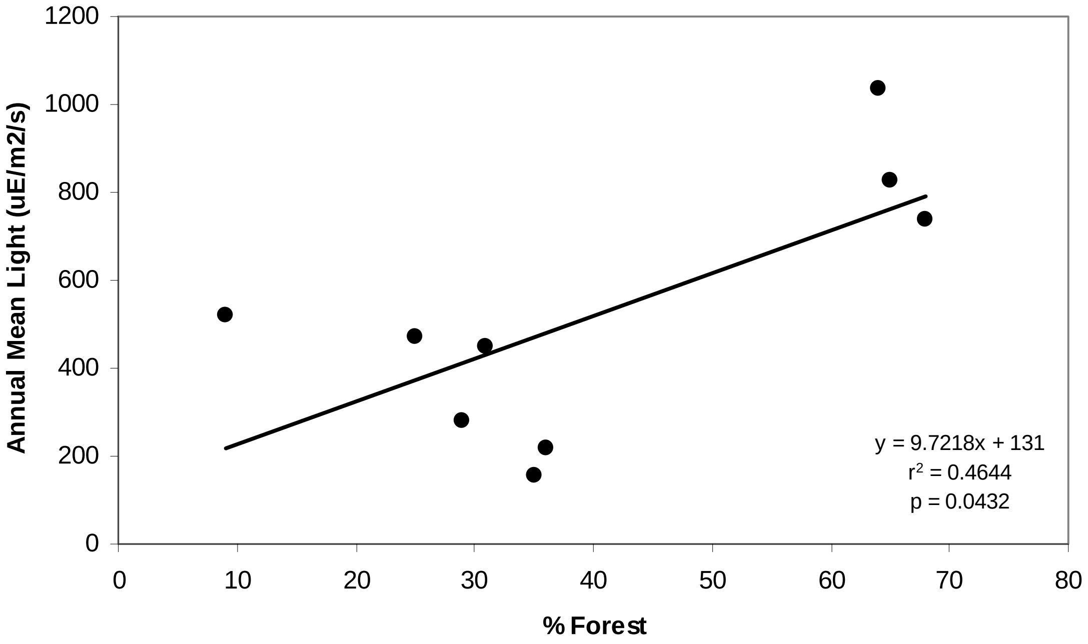

Figure 24. Regression of % forest vs. annual mean light flux.

variables. Riparian zone width at the sampling location was inversely related to phosphate flux ($r^2 = 0.4662$, $p = 0.0427$, Figure 25).

Correlations of Water Column and Sediment Variables

To explore potential relationships among water column and sediment variables, a correlation matrix was conducted on annual averages of all values. A total of 41 variables were included in the matrix. Significant correlations are listed in Table 4 ($p<0.05$).

Water column nutrients were correlated with relatively few of the other measured variables. Not surprisingly, a positive correlation was found between log-TDN flux and log-nitrate flux. Ammonium flux was positively correlated with DON flux, and negatively with sediment C:N at 0-2 cm. In addition, DON flux was directly correlated with SPOM flux, and water column phosphate flux was positively related to log-porewater sulfide at the sediment surface. Significant relationships between chlorophyll and water column and sediment variables were also fairly small in number. Log–water column chl a flux was directly related to TSS flux, and log-benthic chl a was positively correlated with log–discharge and DO flux. Finally, SPOM fluxes were positively correlated to TSS and DO fluxes.

In contrast to the relatively few relationships found between water column nutrients, benthic and pelagic chlorophyll a, and other physico-chemical variables, the number of significant correlations among sediment variables was high. As expected, values measured within sediment profiles of organic matter, ammonium, and sulfide were positively correlated with one another, and extractable and porewater ammonium were

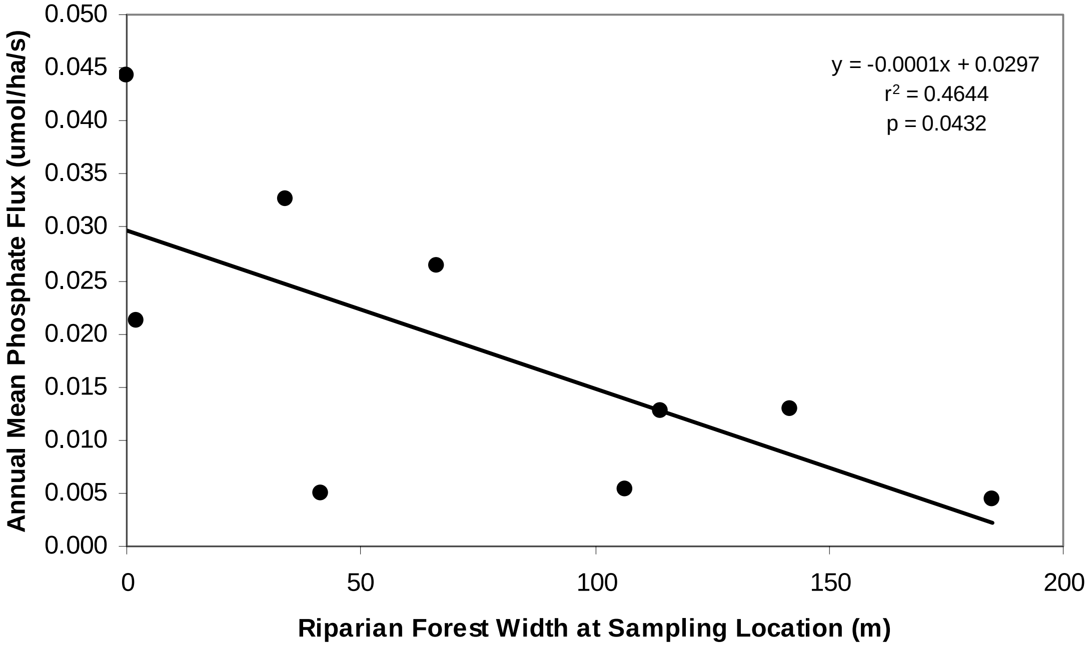

Figure 25. Regression of riparian zone width at sampling location vs. annual mean phosphate flux.

Table 4. Results of correlation matrix with water column and sediment variables. All correlations listed are significant at $p<0.05$.

Variable 1	Variable 2	r	p
Log-TDN Flux	Log-Nitrate Flux	0.91104	0.0006
Ammonium Flux	DON Flux	0.7151	0.0303
	Sediment C:N 0-2 cm	-0.72298	0.0277
DON Flux	SPOM Flux	0.79594	0.0103
Phosphate Flux	Log-Porewater Sulfide 0 cm	0.74168	0.0222
Log-Water Column Chl a Flux	TSS Flux	0.75391	0.019
Log-Benthic Chl a	Log-Discharge	0.75025	0.0199
	DO Flux	0.75224	0.0194
SPOM Flux	TSS Flux	0.74866	0.0203
	DO Flux	0.7319	0.025
Detrital Layer Depth	Extractable Ammonium 0-2 cm	0.8477	0.0039
	Extractable Ammonium 2-5 cm	0.84088	0.0045
	Extractable Ammonium 5-10 cm	0.7442	0.0215
	DO Flux	-0.89652	0.0011
Sqrt-Organic Matter 0-2 cm	Extractable Ammonium 0-2 cm	0.73256	0.0248
	Extractable Ammonium 2-5 cm	0.75204	0.0194
	Organic Matter 2-5 cm	0.90671	0.0007
	Organic Matter 5-10 cm	0.76419	0.0165
Organic Matter 2-5 cm	Extractable Ammonium 0-2 cm	0.67055	0.0481
	Extractable Ammonium 2-5 cm	0.67155	0.0476
	Organic Matter 5-10 cm	0.83161	0.0054
Extractable Ammonium 0-2 cm	DO Flux	-0.71	0.0321
	Extractable Ammonium 2-5 cm	0.97684	<0.0001
	Extractable Ammonium 5-10 cm	0.79877	0.0098
Extractable Ammonium 2-5 cm	DO Flux	-0.729	0.0259
	TSS Flux	-0.71238	0.0313
	Log-Discharge	-0.72542	0.027
	Extractable Ammonium 5-10 cm	0.84973	0.0037
Extractable Ammonium 5-10 cm	DO Flux	-0.66707	0.0497
	Log–Discharge	-0.90044	0.0009
	Porewater Ammonium 0 cm	0.78933	0.0114
Porewater Ammonium 0 cm	Log-Discharge	-0.73539	0.0239
Porewater Ammonium 2 cm	Porewater Ammonium 4 cm	0.89591	0.0011
	Porewater Ammonium 6 cm	0.95741	<0.0001
	Porewater Ammonium 8 cm	0.85703	0.0032
	Salinity	0.78769	0.0117
Porewater Ammonium 4 cm	Porewater Ammonium 6 cm	0.96038	<0.0001
	Porewater Ammonium 8 cm	0.95682	<0.0001
	Salinity	0.85422	0.0034
Porewater Ammonium 6 cm	Porewater Ammonium 8 cm	0.95326	<0.0001
	Salinity	0.77213	0.0148
Porewater Ammonium 8 cm	Salinity	0.74693	0.0207
	pH	0.66759	0.0494
Log-Porewater Sulfide 2 cm	Log-Porewater Sulfide 6 cm	0.71277	0.0311
Log-Porewater Sulfide 6 cm	Porewater Sulfide 8 cm	0.86915	0.0023

Table 4 (cont).

Sediment C:N 0-2 cm	Sediment C:N 2-5 cm	0.89569	0.0011
	Sediment C:N 5-10 cm	0.81784	0.0071
Sediment C:N 2-5 cm	Sediment C:N 5-10 cm	0.94515	0.0001
	Water T	-0.71432	0.0306
Sediment C:N 5-10 cm	Water T	-0.6975	0.0367
Water T	% Sand	-0.67251	0.0472

also positively correlated at some depths. Detrital layer depth and/or sediment organic matter were directly related to extractable ammonium at most depths. Increased thickness of the detrital layer and elevated extractable ammonium concentrations were also related to lowered DO fluxes. In addition, extractable ammonium at most depths and porewater ammonium at 0 cm were inversely related to log-discharge. Porewater ammonium at all other depths showed a positive correlation with salinity. Lastly, sediment C:N was negatively related to water temperature.

Macroinvertebrate Community Structure and Calculated Metrics

Annual averages of all measures of macroinvertebrate community structure were calculated, and are presented here. Data are organized so that percent agriculture increases from left to right on x-axes of graphs and in column headings of tables. Monthly values of community structure measures are located in Appendix B.

A total of nearly 5,000 macroinvertebrates representing 70 different taxa were collected over the study period. The ten groups that composed the majority of macroinvertebrate communities were: bivalves, amphipods, dipterans, oligochaetes, isopods, trichopterans, gastropods, odonates, turbellarians, and ephemeropterans. Bivalves and oligochaetes were the only taxonomic groups present in all creeks at least once throughout the year. Bivalves, amphipods, and dipterans were the most abundant organisms found, and each group was dominant in 3 of the 9 study streams. Communities in Indiantown, Walls Landing, and Narrow Channel were dominated by bivalves; Greens, Taylor, and Lakewood were dominated by amphipods; and Mill, Phillips, and Holt were dominated by dipterans (Figure 26). Average annual abundance

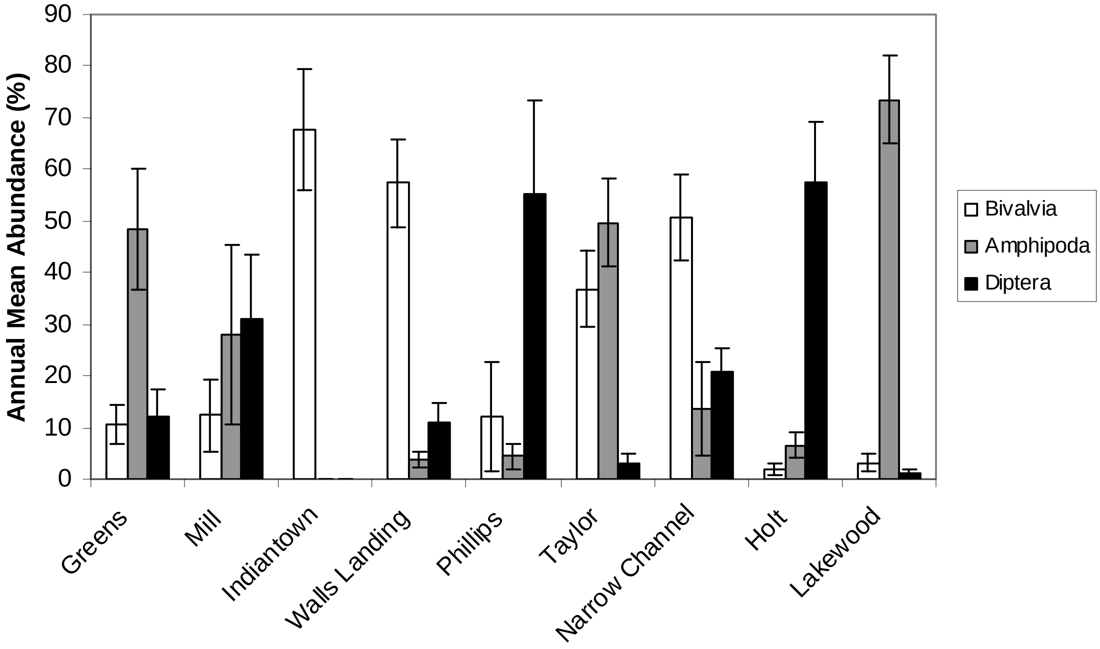

Figure 26. Annual mean percent abundance and standard error bars for bivalves, amphipods, and dipterans.

of bivalves ranged from 1.8 – 67.6, amphipods ranged from 0 – 73.5, and dipterans ranged from 0 – 55.1% of total macroinvertebrate abundance. Annual mean abundances of oligochaetes, isopods, trichopterans, and gastropods were less than 30% of the population in each site (Figure 27). Odonates, turbellarians, and ephemeropterans generally composed < 5% of total abundance (Figure 28). Coleopterans, decapods, and megalopterans were present in some sites, but had abundances of typically less than 1%. No stoneflies (Plecoptera) were found in any site during any season.

Taxonomic classification (mostly to genus), tolerance values (TV), and functional feeding groups (FFG) of organisms found are listed in Table 5. Tolerance values are on a scale of 1 -10, with values of 0 assigned to those taxa with the greatest sensitivity (least tolerance) to stressors and values of 10 allocated to taxa with the least sensitivity (most tolerance) (Stribling et al. 1998, US EPA 1997). Organisms with TVs of 7 – 10 are generally considered to be tolerant, and those with values of 0 – 3 are usually deemed intolerant (Stribling et al. 1998). The FFG classification of organisms denotes mode of feeding, and includes scrapers (graze attached periphyton), shredders (shred and tear CPOM), gatherers (gather FPOM in or on sediment), filterers (filter FPOM in suspension), and predators (prey on other animals) (Cummins 1978, Voshell 2002, Wallace and Webster 1996).

Metrics of macroinvertebrate taxonomic richness, composition, tolerance, and mode of feeding varied among the creeks (Table 6). Number of taxa was highest in Narrow Channel, Phillips, and Walls Landing, and ranged from 5 – 15.25. The lowest numbers of taxa were found in Indiantown (5) and Taylor (6.25) (Figure 29). The

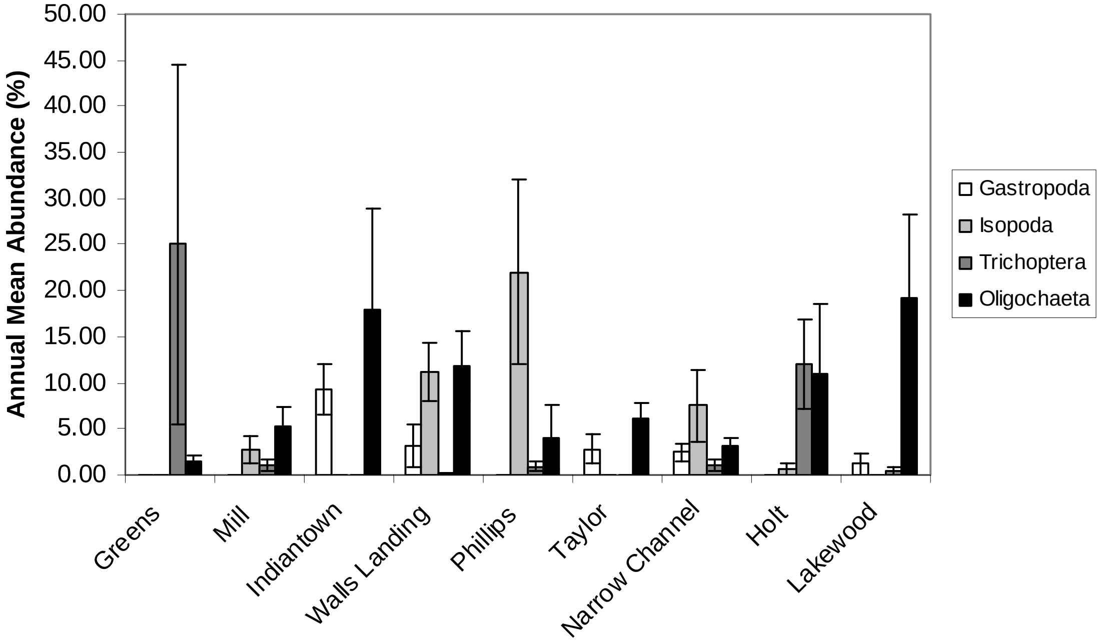

Figure 27. Annual mean percent abundance and standard error bars for gastropods, isopods, trichopterans, and oligochaetes.

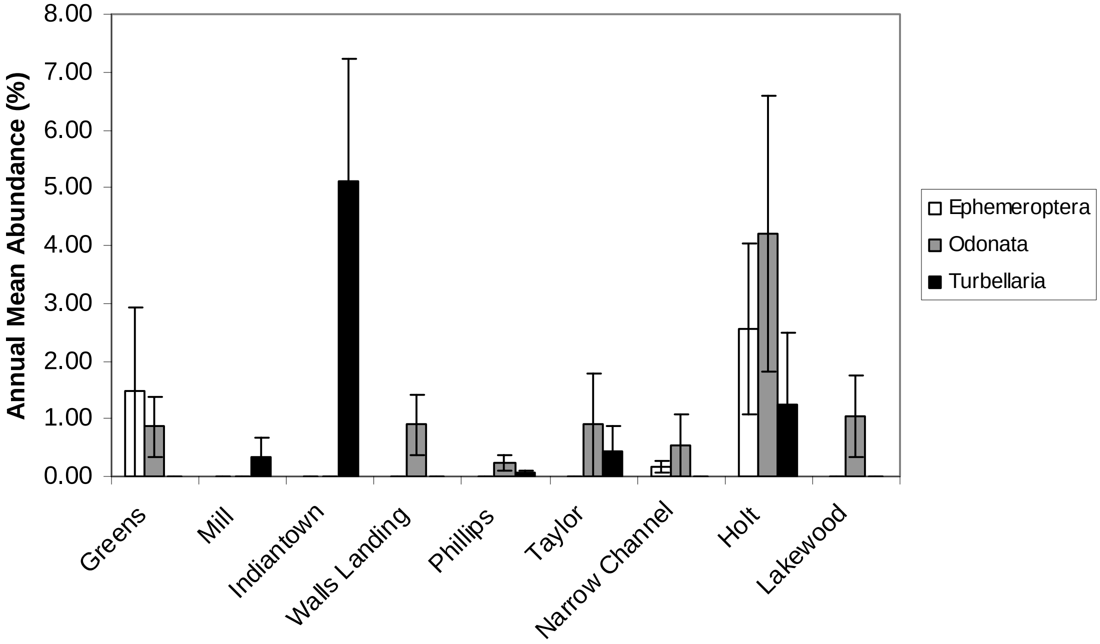

Figure 28. Annual mean percent abundance and standard error bars for ephemeropterans, odonates, and turbellarians.

Table 5. Macroinvertebrate taxa found. TV = tolerance value (0-10); FFG = functional feeding group (G = gatherer, P = predator, Sc = scraper, F = filterer, Sh = shredder).

Class	Order	Family	Genus	TV	FFG
Oligochaeta	Haplotaxidae	Lumbricidae	(unidentified)	10	G
	Lumbriculida	Lumbriculidae	Eclipidrilus	8	G
			Lumbriculus	8	G
			(unidentified)	10	G
	Tubificida	Tubificidae	Limnodrilus	10	G
			Psammoryctides	8	G
			Spirosperma	10	G
			(unidentified)	10	G
Phylum Nematoda (unidentified)	(unidentified)	(unidentified)	(unidentified)		
Phylum Nemertea (unidentified)	(unidentified)	(unidentified)	(unidentified)		P
Turbellaria	Tricladida	Planariidae	(unidentified)	1	P
Gastropoda	Basommatophora	Lymnaeidae	Pseudosuccinea	6	G
		Physidae	Physella	8	Sc
		Planorbidae	Menetus	8	Sc
			Promenetus	7	Sc
Bivalvia	Veneroida	Sphaeriidae	Pisidium	8	F
			Musculium	8	F
			Sphaerium	8	F
Malacostraca	Amphipoda	Crangonyctidae	Crangonyx	4	G
			Stygonectes	6	Sh
		Gammaridae	Gammarus	6	Sh
	Decapoda	Cambaridae	Cambarus	6	G
	Isopoda	Asellidae	Caecidotea	8	G
Insecta	Coleoptera	Dytiscidae	Deronectes	5	P
		Elmidae	Stenelmis	6	Sc
		Scirtidae	Cyphon	7	Sc
	Diptera	Ceratopogonidae	Probezzia	6	P
		Chironomidae	Chironomus	10	G
			Clinotanypus	8	P
			Cricotopus/ Orthocladius	7	Sh
			Cryptochironomus	8	P
			Hydrobaenus	8	Sc
			Microspectra	7	G

Table 5 (cont.)

		Microtendipes	6	F
		Natarsia	8	P
		Paracladopelma	7	G
		Parametriocnemus	5	G
		Paratanytarsus	6	G
		Polypedilum	6	Sh
		Procladius	9	P
		Stictochironomus	9	G
		Tanytarsus	6	F
		Thienemannimyia	6	G
		Tribelos	5	G
		Zavrelimyia	8	P
		(unidentified)		
	Empididae	Hemerodromia	6	P
	Simuliidae	Simulium	7	F
		Simulium/ Prosimulium	7	F
	Tabanidae	Chrysops	7	P
		(unidentified)	8	P
	Tipulidae	Dicranota	4	P
		Hexatoma	4	P
		Pseudolimnophila	2	P
Ephemeroptera	Heptageniidae	Stenonema	4	Sc
	Leptophlebiidae	Leptophlebia	4	G
		Paraleptophlebia	2	G
Megaloptera	Sialidae	Sialis	4	P
Odonata	Aeshnidae	Boyeria	2	P
	Calopterygidae	Calopteryx	6	P
	Cordulegastridae	Cordulegaster	3	P
	Corduliidae	Epicordulia	5	P
		Tetragoneuria	5	P
	Libellulidae	Plathemis	9	P
Trichoptera	Beraeidae	Beraea	unk	G
	Calamoceratidae	Heteroplectron	3	Sh
	Hydropsychidae	Cheumatopsyche	5	F
		Diplectrona	2	F
	Limnephilidae	Ironoquia	3	Sh
	Molannidae	Molanna	6	Sc

Table 5 (cont.)

	Odontoceridae	Marilia	0	Sh
		Psilotreta	0	Sc
	Phryganeidae	Oligostomis	4	P, Sh

Table 6. Annual mean values of macroinvertebrate metrics.

Metric	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
# Taxa	7.50	8.00	5.00	11.75	13.25	6.25	15.25	9.50	7.00
# EPT Taxa	1.50	0.50	0.00	0.50	1.75	0.00	1.50	2.00	0.25
Total Abundance	40.50	69.75	120.75	215.00	287.00	45.75	248.50	33.00	92.00
% EPT	26.48	1.10	0.00	0.14	0.91	0.00	1.23	14.61	0.38
% Plecoptera	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
% Trichoptera	25.01	1.10	0.00	0.14	0.91	0.00	1.05	12.04	0.38
% Diptera	12.21	30.89	0.00	11.08	55.05	3.10	20.63	57.57	1.23
% Chironomidae	8.47	27.08	0.00	11.08	54.99	3.10	20.33	39.04	0.95
% Non-Insect	60.46	67.68	75.00	87.82	43.24	96.00	72.83	22.37	97.35
% Oligochaeta	1.53	5.19	18.00	11.82	4.06	6.18	3.10	10.90	19.16
Simpson's DI	0.52	0.55	0.45	0.61	0.56	0.58	0.78	0.70	0.40
% Dominant Taxon	65.04	61.10	67.59	56.75	61.61	57.32	37.01	44.52	73.53
Hilsenhoff BI	4.71	5.95	7.72	8.11	6.72	6.93	7.26	6.74	6.72
% Scrapers	25.31	0.33	8.96	3.27	0.67	2.81	1.93	3.54	1.20
% Shredders	50.64	2.70	0.00	0.85	0.13	49.57	17.61	0.00	73.81
% Gatherers	3.00	60.22	18.35	37.45	39.71	7.35	15.25	67.31	19.54
% Filterers	16.44	32.94	67.59	57.33	55.56	36.87	63.29	4.60	3.55
$\delta^{15}N$ amphipods	7.02	10.14	NA	NA	NA	9.83	9.03	NA	11.13
$\delta^{15}N$ chironomids	8.01	5.89	NA	NA	6.69	NA	9.76	7.47	8.83
$\delta^{15}N$ worms	8.06	5.40	6.31	NA	6.28	11.36	7.42	NA	8.34

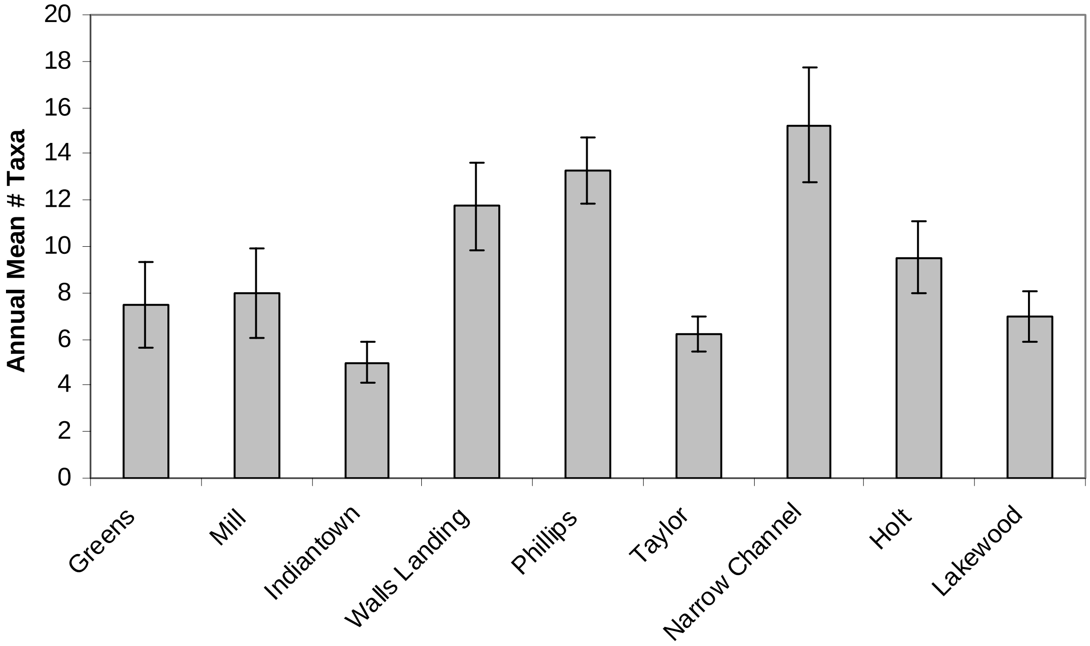

Figure 29. Annual mean # taxa and standard error bars.

number of EPT taxa found was low for all sites, ranging from 0 – 2, and no EPT taxa were found in Indiantown or Taylor creeks (Figure 30).

Annual mean total abundance of organisms ranged from 33 – 287, and was highest in Phillips Creek, Narrow Channel, and Walls Landing (Figure 31). Total abundance was lowest in Holt, Greens, and Taylor (33, 40, and 46, respectively). Relative abundance of EPT taxa was much higher in Greens (25%) than in other streams (Figure 32). Holt had an annual mean % EPT abundance of 14.6, and in all other sites, EPT taxa composed <1.5% of total macroinvertebrate abundance. Relative abundance of chironomids was highest in Phillips (55.0%), and was lowest in Lakewood and Indiantown (<1%) (Figure 33). Percent non-insect taxa was highest in Lakewood Creek (97.4%), and comprised more than 40% of total abundance for all sites but Holt (22.4%) (Figure 34). Percent oligochaete abundance was also highest in Lakewood (19.2%), and was lowest in Greens (1.5%) (Figure 27). Simpson's diversity index (DI), a measure of community richness and evenness, varied from 0.4 – 0.8, with the lowest value in Lakewood, and highest values in Holt (0.70) and Narrow Channel (0.78) (Figure 35).

Differences in community sensitivity were also observed among creeks. The percentage of the most abundant taxon ranged from 37.0 in Narrow Channel to 73.5 % in Lakewood (Figure 36). Values of the Hilsenhoff biotic index (Hilsenhoff BI), a measure of overall tolerance/intolerance of the assemblage, ranged from a low of 4.7 in Greens to a high of 8.1 in Walls Landing (Figure 37).

Functional feeding groups that composed the majority of the population in each creek included shredders, gatherers, and filterers. Shredder abundance was highest in

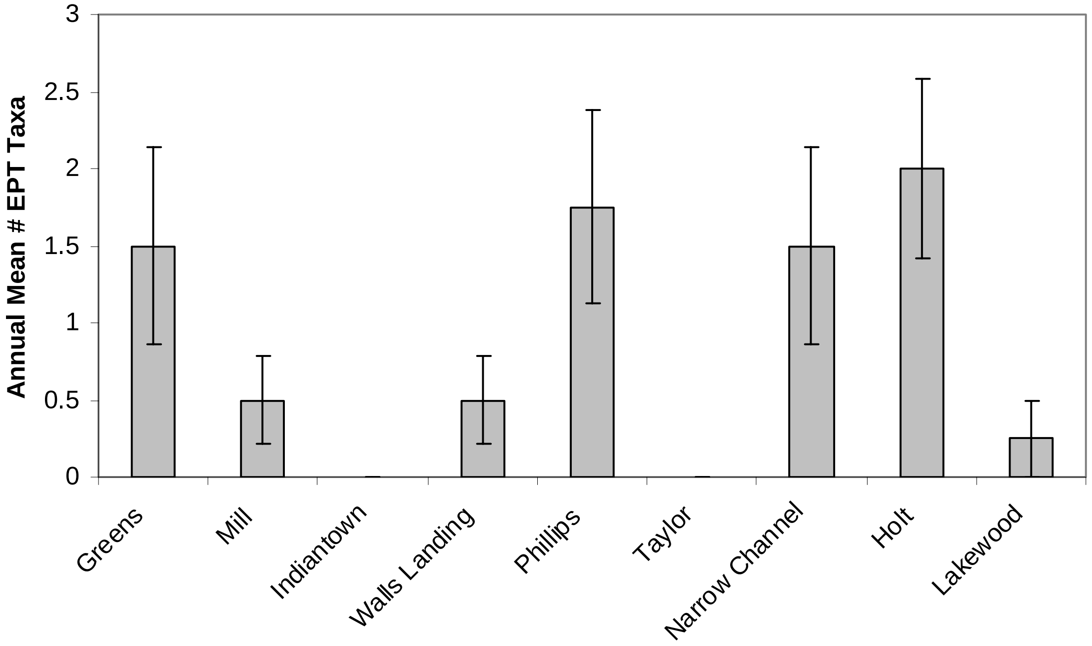

Figure 30. Annual mean # EPT taxa and standard error bars.

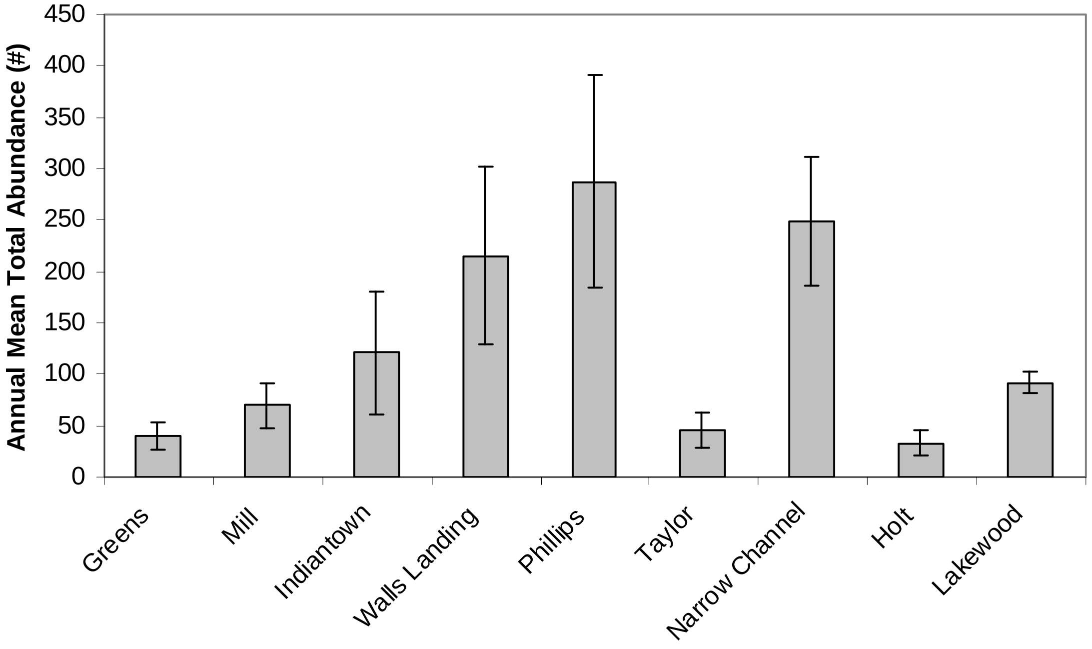

Figure 31. Annual mean total macroinvertebrate abundance and standard error bars.

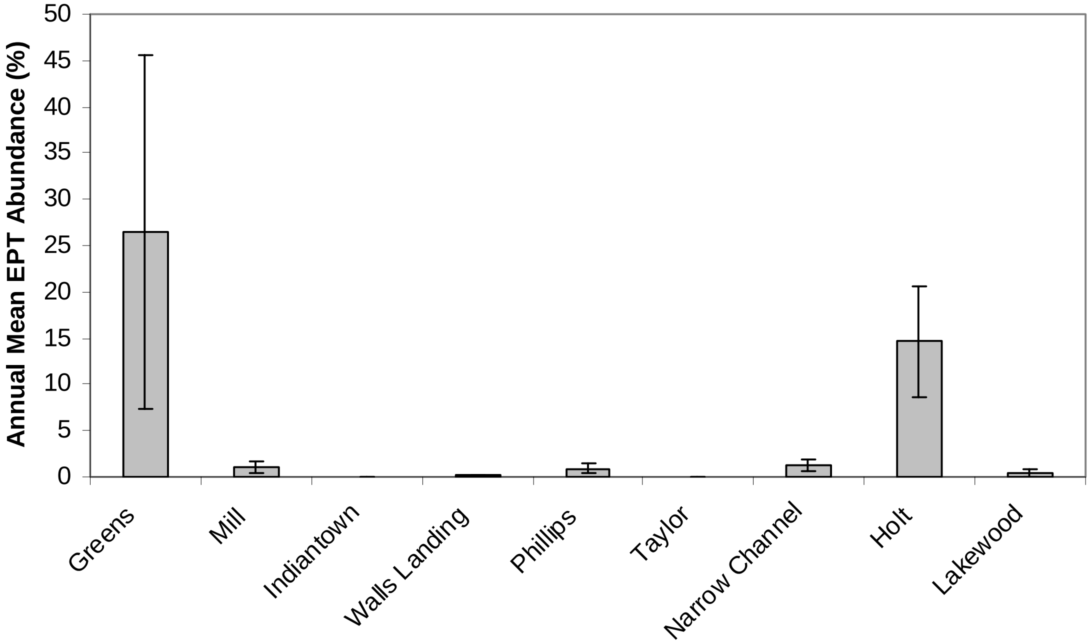

Figure 32. Annual mean percent EPT abundance and standard error bars.

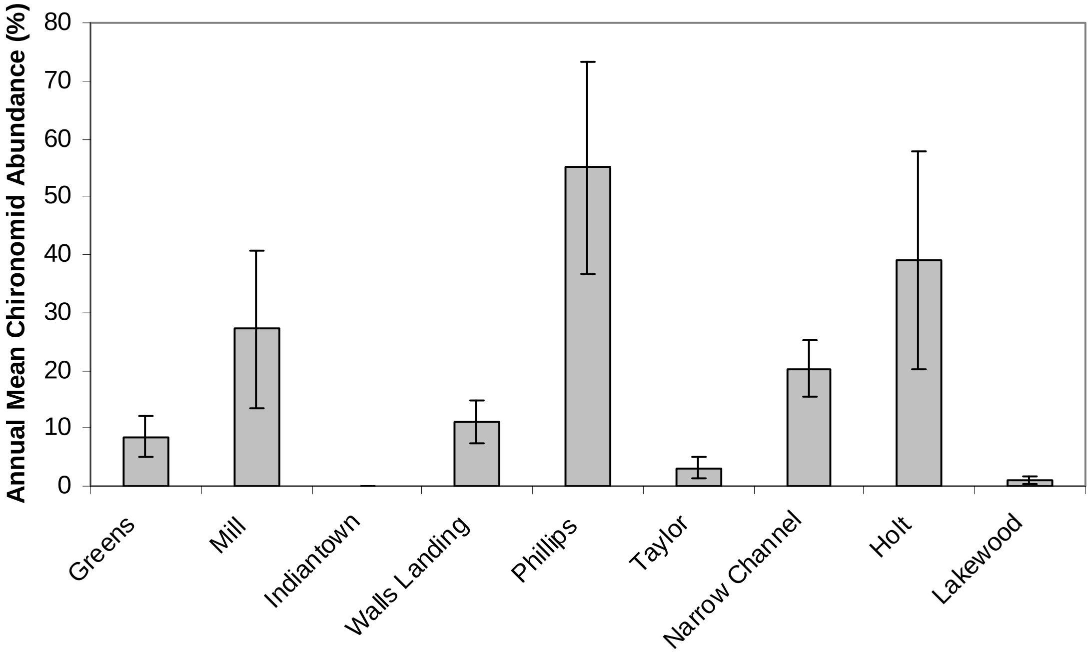

Figure 33. Annual mean percent chironomid abundance and standard error bars.

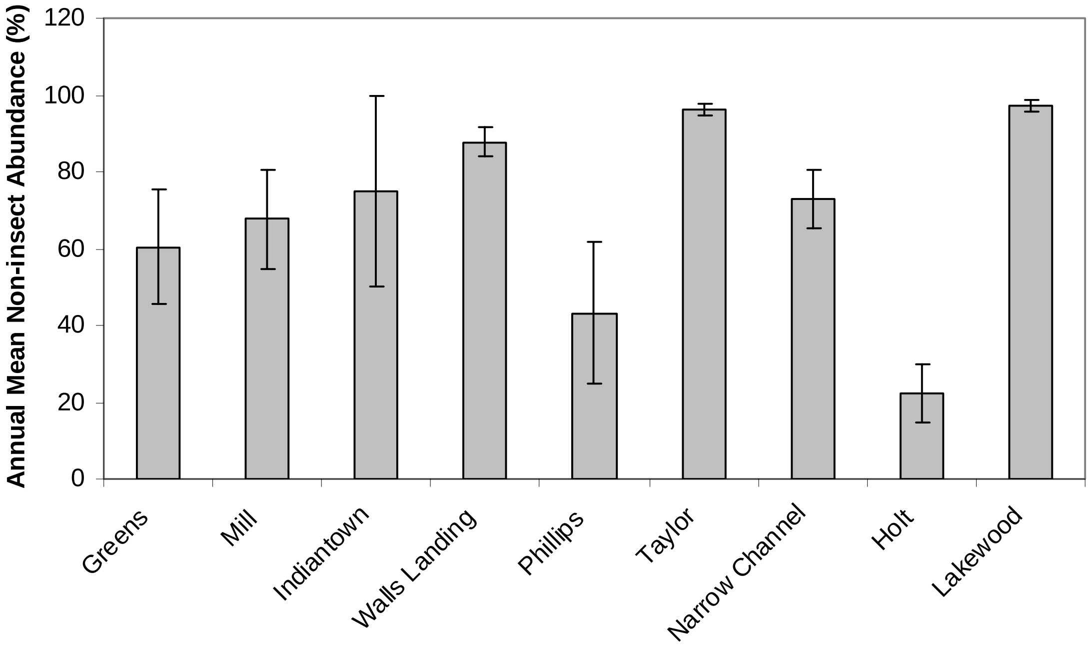

Figure 34. Annual mean percent non-insect abundance and standard error bars.

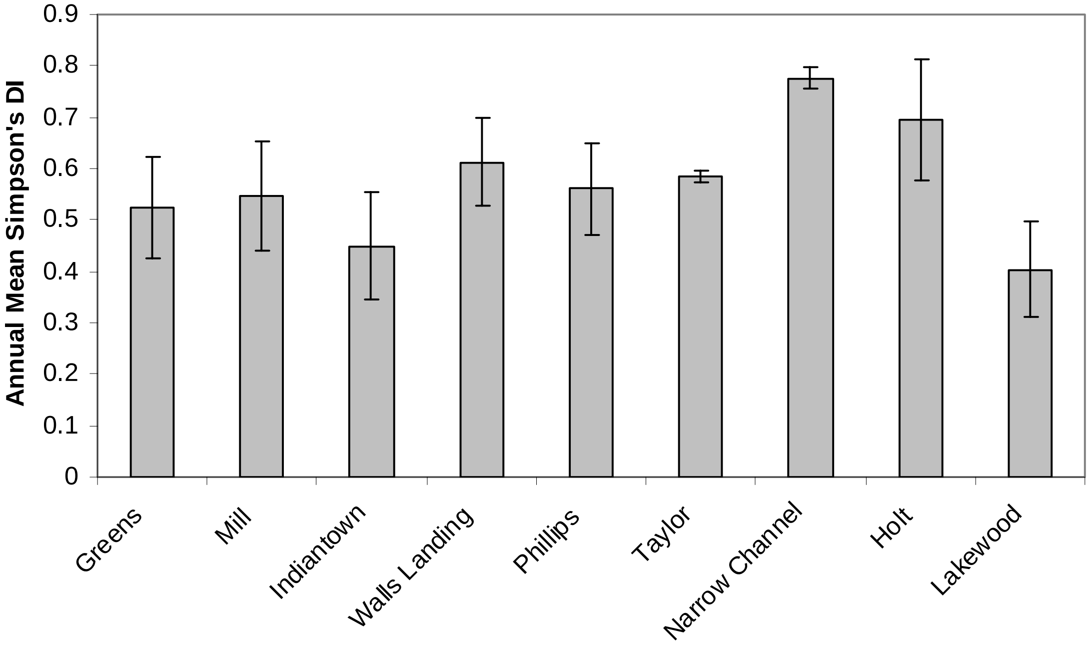

Figure 35. Annual mean values and standard error bars for Simpson’s diversity index.

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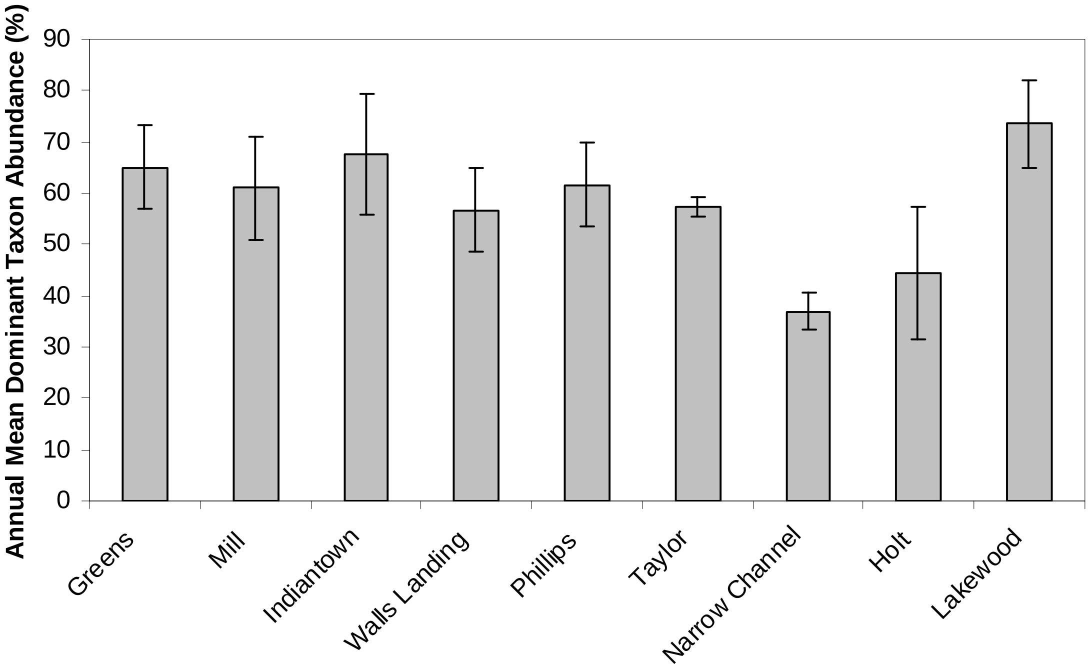

Figure 36. Annual means and standard error bars for % dominant taxon abundance.

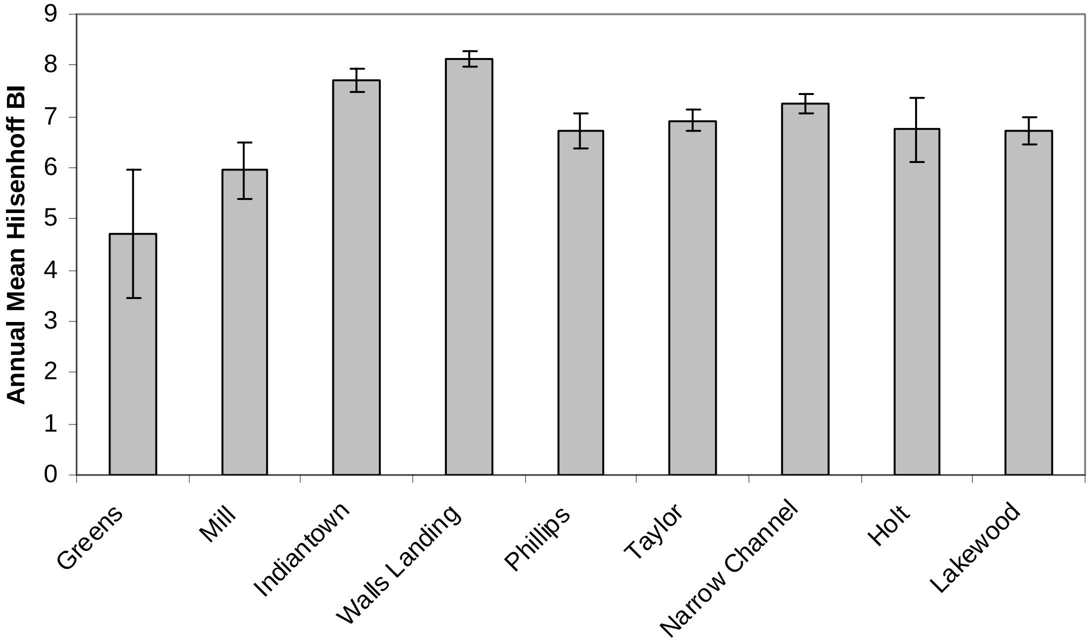

Figure 37. Annual mean values and standard error bars for the Hilsenhoff biotic index.

Lakewood Creek (73.81%), while gatherers were highest in Holt (67.31%), and filterers were highest in Indiantown (67.59%) (Figure 38). Relative abundance of scrapers was less than 10% for all creeks but Greens (25.31%). Predators were generally less that 5% of total abundance, with the exception of a relatively high value of 24.16% in Holt.

Isotopic N signatures for amphipods, chironomids, and worms ranged from 5.4 – 11.4 per mil. Values for chironomids and worms were lowest in Mill Creek, and values for amphipods were lowest in Greens (Figure 39). Signatures were highest for amphipods, chironomids, and worms in Lakewood, Narrow Channel, and Taylor, respectively. Because of low abundances of these three taxonomic groups, gaps are present in the isotopic data for Indiantown, Walls Landing, Phillips, Taylor, and Holt.

Regressions of Land Use and Macroinvertebrate Metrics

Regressions of macroinvertebrate metrics versus % agriculture, % agriculture^2, % forest, % development, and riparian zone width were conducted on annual means and on seasonal values. The only metric that demonstrated a significant relationship with watershed land use was the relative abundance of scrapers in January. Percent scrapers was negatively correlated to % agriculture ($r^2 = 0.4913$, $p = 0.0354$, Figure 40), and as expected due to the inverse relationship between agricultural and forest land use, was positively related to % forest ($r^2 = 0.5046$, $p = 0.0032$). Regressions with development were not significant for any macroinvertebrate metrics. However, riparian zone width was significantly correlated with one metric. The $\delta^{15}N$ of worms increased significantly with riparian zone widths at 0 ($r^2 = 0.9253$, $p = 0.0005$), 50 ($r^2 = 0.7390$, $p = 0.0131$), and

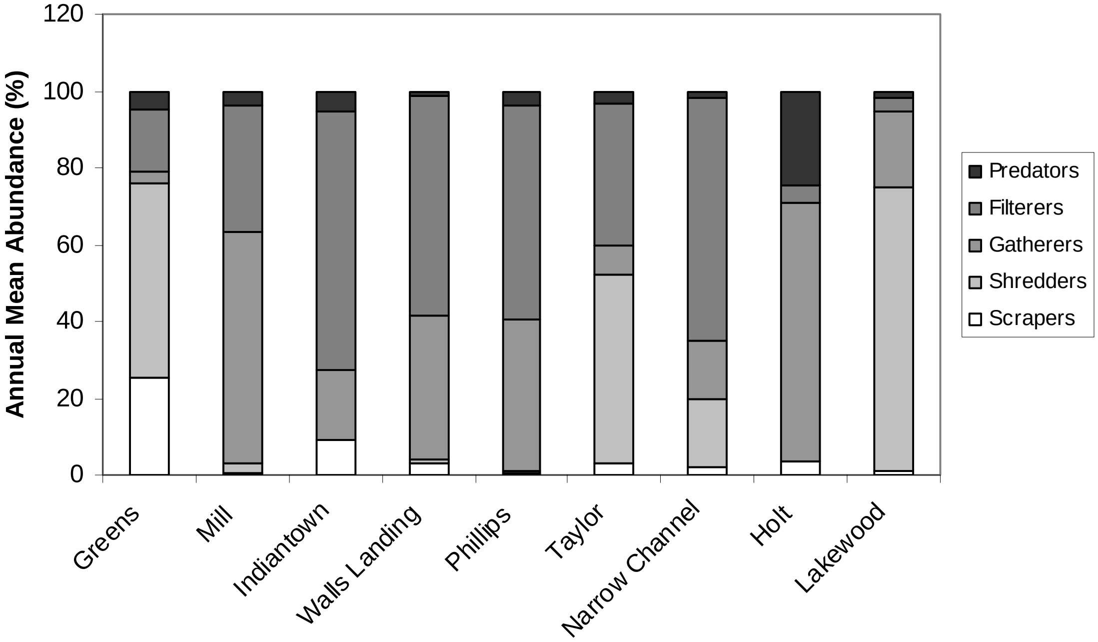

Figure 38. Annual mean abundance of functional feeding groups.

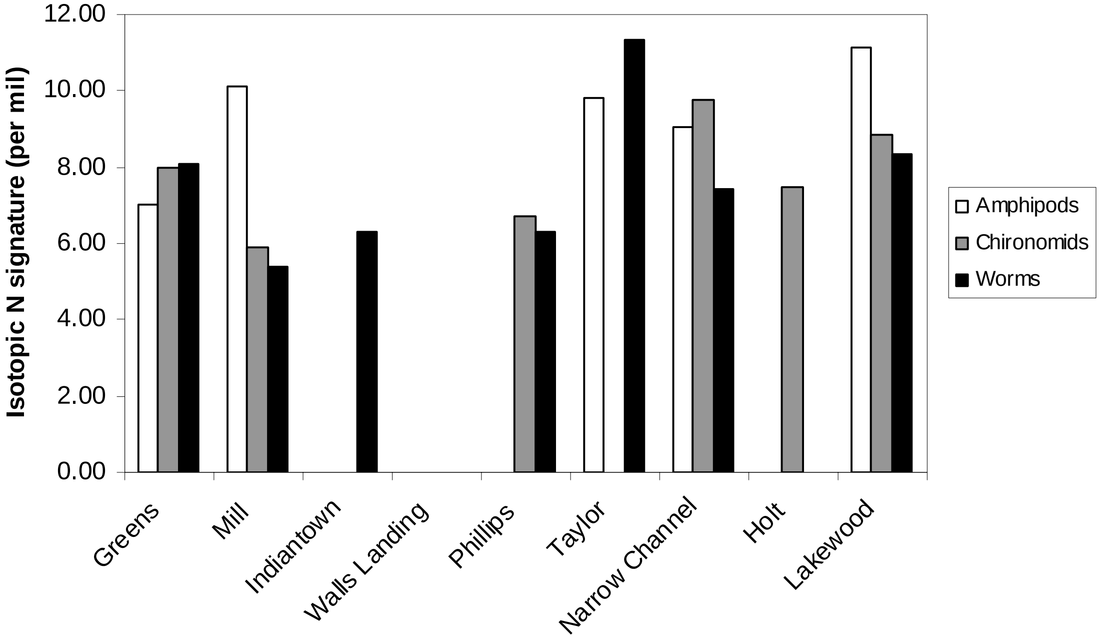

Figure 39. Isotopic N signatures of amphipods, chironomids, and worms. Indiantown, Walls Landing, Phillips, Taylor, and Holt had missing values due to low abundances.

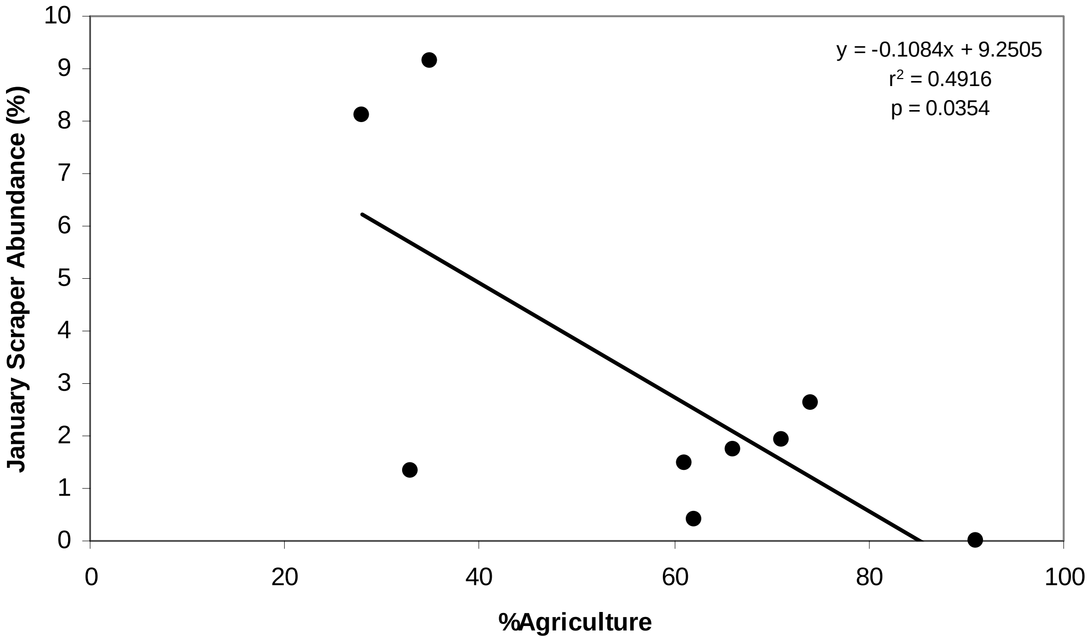

Figure 40. Regression of January scraper abundance vs. % agriculture.

100 m ($r^2 = 0.7299$, $p = 0.0146$) upstream of sampling location, and with average width along the 100 m upstream reach ($r^2 = 0.8967$, $p = 0.0012$, Figure 41).

Correlations Between Macroinvertebrate Metrics and Water Column and Sediment Variables

To explore potential relationships between macroinvertebrate metrics and characteristics of water column and sediment, correlations were performed on annual averages of all values. Significant correlations are listed in Table 7 ($p<0.05$).

Many of the macroinvertebrate metrics were correlated with sediment and water column variables. Depth of detrital layer and water column and sediment ammonium were the variables most commonly correlated with metrics. Depth of the detrital layer was positively related to total abundance, Hilsenhoff BI, and % filterer abundances. Percent trichopteran and shredder abundances were negatively correlated with detrital layer depth. Extractable ammonium was positively correlated with total abundance and total # taxa, and negatively with ephemeropteran, trichopteran, log-EPT, and filterer abundances. Porewater ammonium was inversely related to # EPT taxa and dipteran, chironomid, and log-EPT abundances, and positively correlated with percent oligochaete and non-insect abundances, and with the Hilsenhoff BI. Water column ammonium flux was positively correlated to percent non-insect abundance, and negatively to Simpson’s DI and # EPT taxa.

Several metrics were positively correlated with sediment organic matter, including # taxa and abundances of dipterans and chironomids, while log- % scrapers had

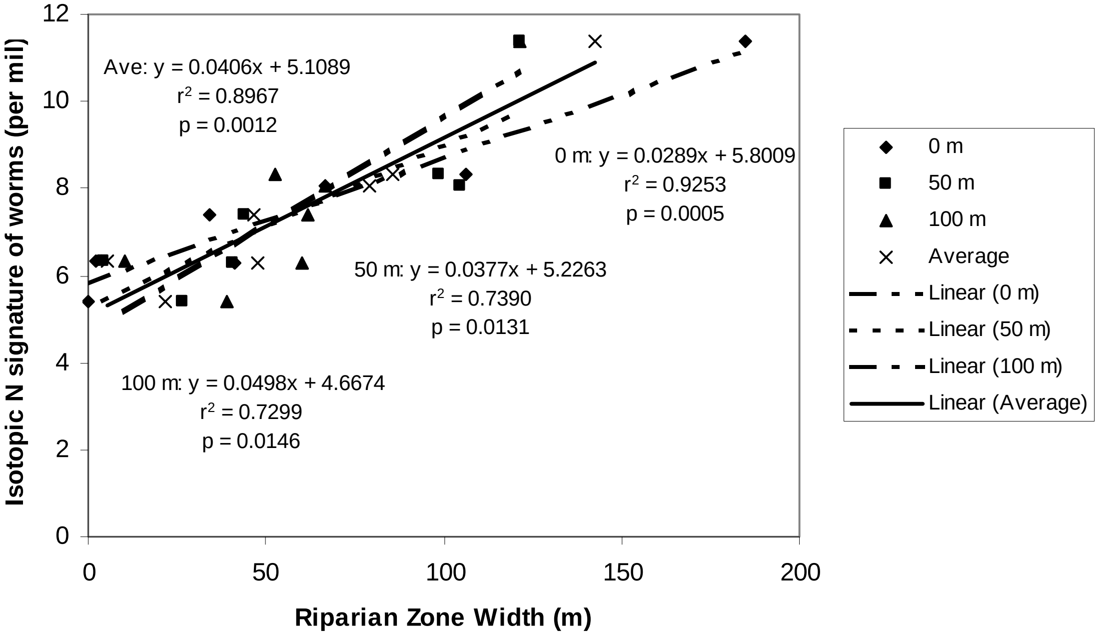

Figure 41. Regressions of isotopic N signature of worms and riparian zone width at 0, 50, and 100 m, and with average width along 100 m upstream reach.

Table 7. Results of correlation matrix for macroinvertebrate metrics and water column and sediment variables. All correlations listed are significant at $p<0.05$.

Variable 1	Variable 2	r	p
# Taxa	Sqrt-Organic Matter 0-2 cm	0.66725	0.0496
	Extractable Ammonium 0-2 cm	0.68996	0.0397
	Light	-0.77604	0.014
	Organic Matter 2-5 cm	0.67928	0.0442
# EPT Taxa	Porewater Ammonium 2 cm	-0.6812	0.0434
	Porewater Ammonium 4 cm	-0.73185	0.025
	Porewater Ammonium 6 cm	-0.75515	0.0186
	Porewater Ammonium 8 cm	-0.79943	0.0097
	Water Column Ammonium Flux	-0.73302	0.0246
Total Abundance	Extractable Ammonium 0-2 cm	0.96363	<0.0001
	Extractable Ammonium 2-5 cm	0.92166	0.0004
	Extractable Ammonium 5-10 cm	0.6683	0.0491
	Detrital Layer Depth	0.81849	0.007
	TSS Flux	-0.71331	0.031
Log-% EPT Abundance	Extractable Ammonium 5-10 cm	-0.75576	0.0494
	Porewater Ammonium 0 cm	-0.75927	0.0477
	Porewater Ammonium 6 cm	-0.77933	0.0389
	Porewater Ammonium 8 cm	-0.78644	0.0359
	Sediment C:N 5-10 cm	0.91113	0.0043
% Ephemeroptera Abundance	Extractable Ammonium 5-10 cm	-0.76255	0.0169
	Sediment C:N 0-2 cm	0.72143	0.0282
% Trichoptera Abundance	Log-Discharge	0.78064	0.013
	DO Flux	0.68001	0.0439
	Extractable Ammonium 5-10 cm	-0.74196	0.0221
	Detrital Layer Depth	-0.69254	0.0387
% Diptera Abundance	Sqrt-Organic Matter 0-2 cm	0.73548	0.0239
	Porewater Ammonium 6 cm	-0.68032	0.0437
	Porewater Ammonium 8 cm	-0.74832	0.0204
	Organic Matter 2-5 cm	0.71339	0.0309
% Chironomidae Abundance	Sqrt-Organic Matter 0-2 cm	0.82503	0.0062
	Porewater Ammonium 8 cm	-0.71174	0.0315
	Organic Matter 2-5 cm	0.83069	0.0055
	Organic Matter 5-10 cm	0.69855	0.0363
% Non-Insect Abundance	Porewater Ammonium 2 cm	0.69442	0.0379
	Porewater Ammonium 4 cm	0.77671	0.0138
	Porewater Ammonium 6 cm	0.7732	0.0145
	Porewater Ammonium 8 cm	0.82702	0.006
	Sediment C:N 0-2 cm	-0.74599	0.021
	Water Column Ammonium Flux	0.68386	0.0422
% Oligochaeta Abundance	Porewater Ammonium 0 cm	0.77784	0.0136
	Porewater Ammonium 4 cm	0.75111	0.0197
	Porewater Ammonium 6 cm	0.69335	0.0383
	Porewater Ammonium 8 cm	0.74688	0.0208
Simpson's DI	Water Column Ammonium Flux	-0.66815	0.0492
Hilsenhoff BI	Log-Benthic Chl a	-0.72496	0.0271

Table 7 (cont).

	DO Flux	-0.68136	0.0433
	Detrital Layer Depth	0.73559	0.0239
	Porewater Ammonium 6 cm	0.68027	0.0438
	pH	0.73194	0.025
Log-% Scraper Abundance	Organic Matter 2-5 cm	-0.70062	0.0355
	Organic Matter 5-10 cm	-0.75227	0.0194
% Shredder Abundance	Detrital Layer Depth	-0.70751	0.033
% Filterer Abundance	DO Flux	-0.85226	0.0035
	Extractable Ammonium 0-2 cm	-0.74225	0.022
	Detrital Layer Depth	0.86163	0.0028
	SPOM Flux	-0.73443	0.0242

a negative relationship with organic matter. Sediment C:N was positively correlated with ephemeropteran and log-EPT abundances, and negatively with % non-insect abundance. DO flux was related positively to % trichopterans and negatively to filterer abundances and the Hilsenhoff BI. Filterer abundances were also negatively correlated with SPOM flux, and TSS flux was negatively correlated with total abundance. Log-benthic chl a was negatively correlated with the Hilsenhoff BI. Additionally, positive correlations were found between percent trichopterans and log-discharge, and between the Hilsenhoff BI and pH. Finally, total # taxa was negatively related to light.

DISCUSSION

Evidence for Nutrient Enrichment

Overall, the observed patterns in water column, sediment, and macroinvertebrate data suggest that agricultural land use is not causing high levels of nutrient enrichment in the study streams. Watersheds with high percentages of land in agriculture were hypothesized to have elevated nutrient fluxes, enhanced autotrophic biomass, and increased sedimentation and decomposition of organic matter. These changes in food availability and water column and sediment chemistry were expected to impact macroinvertebrate community structure. Watersheds with moderate percentages of agriculture were anticipated to have highest diversity, and watersheds with high levels of agricultural land use were expected to have highest total abundance of organisms, reflecting dominance of opportunistic, tolerant taxa.

Nitrate was the only nutrient significantly related to agricultural land use. The positive correlation between nitrate flux and % agriculture is likely a result of higher fertilizer application in watersheds with more cropland, and/or increased nitrate retention and removal in watersheds with greater percent forest (Reay et al. 1992, Norton and Fisher 2000). Nitrate is very soluble and is transported from agricultural fields to streams primarily through infiltration and groundwater delivery (Biggar and Corey 1970, Mainstone and Parr 2002, Summers et al. 1999). The study region is especially susceptible to nitrate contamination and subsequent transport, as a result of a shallow water table and highly permeable aquifer (Reay et al. 1992). Other studies on the Delmarva Peninsula have demonstrated similar relationships between land use and nitrate in ground and surface waters. Reay et al. (1992) found higher concentrations of inorganic N in waters underlying agricultural fields when compared to waters near forested areas, and Stanhope (2003) and Norton and Fisher (2000) observed significant inverse relationships between streamwater nitrate concentrations and % forest in catchments. The link between agricultural land use and nitrate concentrations in ground and streamwaters has also been established in other coastal plain systems (Correll et al. 1992, Lowrance et al. 1985, Hall et al. 2001, Stone et al. 2004).

Ammonium and phosphate fluxes on the other hand, were low and were not correlated to agricultural land use. Anthropogenic additions of ammonium to ground and surface waters are primarily associated with human and animal wastes emanating from septic systems, municipal discharges, concentrated animal operations, and manure applications. Because the primary form of anthropogenic land use in the study streams is cropland agriculture, ammonium inputs associated with human and animal wastes are not likely to be significant. In addition, low ammonium concentrations may have been due to alteration and retention during transport to surface waters, through volatilization, nitrification, or adsorption to surfaces of sediment particles (McClelland and Valiela 1998, Valiela et al. 1992). However, while cropland agriculture is dominant in the southern portion of the Delmarva Peninsula, commercial poultry production is significant in the northern peninsula, potentially leading to higher ammonium concentrations in this region than in the southern streams of the present study. Phosphate concentrations were also depleted, and demonstrated no relationship to agricultural land use. In contrast to nitrate, phosphate is extremely insoluble and tends to adsorb to particulates, so is primarily delivered to streams attached to soil particles in surface runoff (Mainstone and

Parr 2002, Summers et al. 1999, Correll 1998). The relatively flat topography and high aquifer permeability in the region serve to reduce runoff (Reay et al. 1992, Lowrance et al. 1997, Robinson and Reay 2002), potentially resulting in the low phosphate concentrations observed in the study streams. Alternatively, the significant inverse relationship between phosphate flux and riparian zone width may indicate that retention of groundwater-transported soluble P in riparian forests decreased PO_4^- concentrations. The presence of vegetation along stream channels has been demonstrated to reduce transport of N and P in both groundwater and surface runoff (Peterjohn and Correll 1984, Lowrance et al. 1997, Reed and Carpenter 2002, Perry et al. 1999). Though runoff is typically more significant than groundwater in phosphate transport, groundwater may be a significant pathway if sediments have accumulated large amounts of P or in sandy soils with low P-retention capacities (Mainstone and Parr 2002, Summers et al. 1999, Lowrance et al. 2000). The combination of many years of agricultural land use, an aquifer composed of predominantly sand and gravel sediments, and low surface runoff may make phosphate delivery in groundwater an important transport pathway to the streams in this study.

Though significantly higher concentrations of nitrate were observed in streams with higher percent agriculture, autotrophic communities did not increase biomass in response to elevated concentrations. The lack of a relationship between nutrient flux and water column chlorophyll was not surprising, as enrichment in flowing waters is likely to lead to enhanced growth of periphyton or macrophytes rather than phytoplankton (Harper 1992, Mainstone and Parr 2002, Cushing and Allan 2001). Several studies have demonstrated increased production of benthic algae associated with elevated inputs of nutrients in streams and rivers (Elwood et al. 1981, Rosemond et al. 1993, Peterson et al. 1983, Peterson et al. 1985). The absence of a correlation between nutrients and benthic chlorophyll in this study may be due to the fact that P is usually considered to be the limiting nutrient in freshwater systems, and concentrations of P were low. On the other hand, algal biomass may have been limited by a variety of factors other than nutrients, including light, temperature, discharge, substrate, flooding, or grazing (Young et al. 1999, Smith et al. 1999, Miltner and Rankin 1998). In headwater reaches and small streams, light and grazing are major factors potentially limiting benthic algal production (Bernhardt and Likens 2004, Cushing and Allan 2001, Harper 1992). In the present study, light was not correlated to algal biomass, and the lack of a relationship between scraper abundance and benthic chlorophyll suggests that grazing pressure also does not appear to be a limiting factor. In small streams, limitation by light often varies in importance with time of year and location in the channel (Cushing and Allan 2001). Due to problems with equipment, measurements of light availability were not made in the fall, when shading may have caused light limitation to be greatest (Cushing and Allan 2001).

In addition to the lack of evidence for nutrient enrichment in the chlorophyll data, the predicted relationships between agricultural land use and enhanced OM deposition and decomposition were not observed. Anticipated relationships with increasing agriculture included elevated sedimentation of organic matter, lower DO, and release of sulfide and ammonium. The absence of these patterns suggests that elevated nitrate loadings are not leading to the changes in water column and sediment chemistry that were expected to occur with increased agricultural land use.

Furthermore, expected patterns in macroinvertebrate communities were lacking, as metrics of diversity, total abundance and overall tolerance were not correlated with agricultural land use. This is probably because the majority of the measured variables known to impact the structure of macroinvertebrate communities were found to have no relationship to % agriculture. Of the many physical, chemical, and biological factors that affect macroinvertebrate community structure, food availability and habitat composition are likely to be the most important (Hawkins et al. 1982, Harper 1992, Hershey et al. 1988, Lammert and Allan 1999). Food availability was expected to be highest in watersheds with moderate levels of nutrient enrichment, as a result of enhanced abundance and taxonomic richness of the food supply. Further enrichment was anticipated to result in decreases in the diversity of autotrophic communities and associated reductions in the availability of certain species. Though measures of diversity, total abundance, and tolerance were not related to percent agriculture, two other biotic metrics – January scraper abundance and δ^{15}N of worms – were related to land use. The significant decrease in January scraper abundance with increasing agriculture was surprising, as scrapers, who feed predominantly on autotrophic stream production, were expected to increase in abundance with initial increases in % agriculture, and decline with further enrichment. Though benthic algal biomass was not related to agriculture, the taxonomic composition of the algal community may have varied in a manner that impacted scrapers by limiting the quantity of a specific food source. Scrapers are considered to be specialist feeders, so are thought to be sensitive to changes in availability of particular algal taxa (Rawer-Jost et al. 2000, Hart and Fuller 1974). Alternatively, the relationship between scraper abundance and % agriculture may not have been real, as the regression seemed to be driven by two high scraper abundances (8.11 and 9.13 %) in watersheds with low agriculture, while all other creeks had abundances less than 3 %. In addition, the positive relationship between $\delta^{15}N$ of worms and riparian zone width was unexpected. Most aquatic worms are collector-gatherers, and feed by burrowing through sediment and digesting fine particulate organic matter (FPOM) (Voshell 2002). Because isotopic signatures of consumers typically reflect the signatures of their food sources (Zah et al. 2001, Fry 1991, Peterson and Fry 1987), the increase in $\delta^{15}N$ with increasing riparian forest width is likely associated with increased N signatures in FPOM ingested by worms. FPOM in headwater streams typically originates as coarse particulate organic matter (CPOM) contributed from the overhanging forest canopy (Cushing and Allan 2001). Elevated N signatures in CPOM entering streams may be the result of enhanced denitrification in riparian corridors with greater forested widths. Isotopic fractionations during denitrification can be extensive, and result in loss of the isotopically light ^{14}N and an increase in $\delta^{15}N$ of the remaining nitrate pool (Fourqurean et al. 1997). Thus, enhanced denitrification could cause higher signatures in riparian vegetation due to uptake from an enriched nitrate pool, and elevated N signatures in worms feeding on the riparian material.

Overall, the results of this research suggest that the study streams are on the low end of the nutrient enrichment spectrum (Figure 42). The first component in the conceptual model of nutrient enrichment is that increased nutrient loading causes enhanced autotrophic production. Elevated production then leads to changes in food availability and OM decomposition that may impact macroinvertebrates. Though nitrate fluxes were higher in watersheds with higher % agriculture, elevated loadings were not

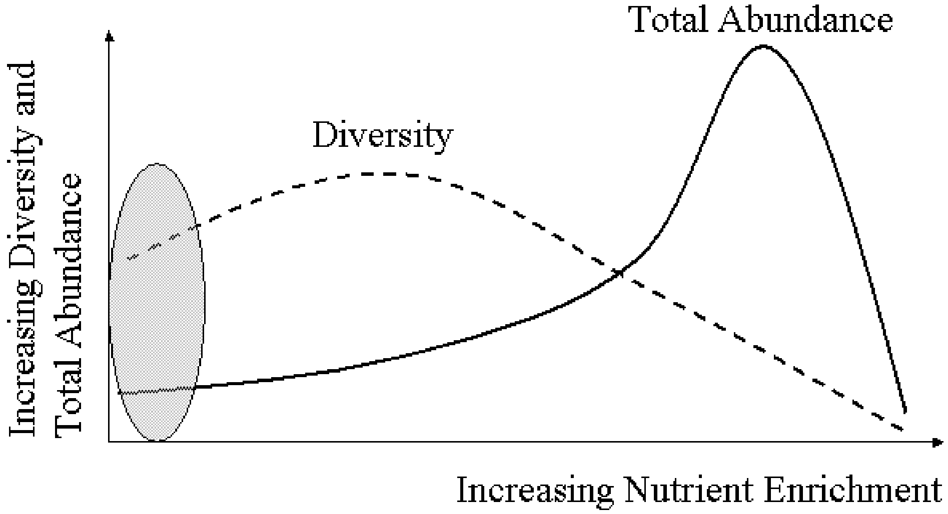

Figure 42. Probable location of study streams (shaded circle) on scale of increasing nutrient enrichment.

linked to changes in chlorophyll, decomposition, or macroinvertebrate community structure that were predicted to occur with increasing nutrient enrichment. This may be because P, rather than N, is likely to be the limiting nutrient in freshwaters, and agricultural land use in the study watersheds did not correlate to increased fluxes of P. The depleted phosphate concentrations may be related to low surface runoff and/or retention by riparian vegetation. Therefore, these data suggest that non-point source nutrient pollution in coastal plain streams with low gradients and forested riparian buffers is not sufficient to generate changes in primary production, OM decomposition, and macroinvertebrate community structure often associated with nutrient enrichment.

Importance of Local Factors

While agricultural land use was not a determinant of macroinvertebrate community structure in this study, local factors played a significant role. Patterns in the structure of macroinvertebrate communities were expected to be correlated to changes in sediment and water column characteristics associated with nutrient enrichment. Creeks with low dissolved oxygen, and high sediment organic matter, ammonium, and porewater sulfide were expected to have low diversity and high total abundance, with communities dominated by opportunistic, tolerant taxa. The basis for the anticipation of these patterns was that enhanced autotrophic production resulting from nutrient enrichment in highly agricultural watersheds would lead to elevated deposition of organic matter. Consequently, increased decomposition was expected to lower DO and cause release of ammonium and sulfide.

Though changes in organic matter, ammonium, and DO were not linked to enhanced autotrophic production, these variables were related to the quantity of allochthonous organic material. Streams with substantial amounts of allochthonous material, primarily in the form of leaf detritus, and/or high sediment organic matter contents had increased sediment ammonium concentrations and lowered water column DO fluxes, likely due to enhanced decomposition of the organic matter. Allochthonous organic matter from surrounding riparian vegetation is often the most important source of energy in forested headwater streams (Rowe et al. 1996, Huryn et al. 2002, Hutchens and Wallace 2002). The quantity and quality of detritus can impact abundance, diversity, and taxonomic and functional feeding group composition of macroinvertebrate communities (Shieh et al. 1999). Decomposition of this material may influence the chemistry of sediments and the water column in the same manner as nutrient or organic enrichment, with potentially detrimental effects on aquatic fauna (Figure 43).

As a result, depth of the detrital layer, sediment organic matter, and sediment and water column NH_4^+ were correlated with many metrics of macroinvertebrate community structure. Of the 21 biotic metrics employed in this study, 15 varied with allochthonous organic enrichment in the same manner as was expected due to nutrient enrichment (Table 8). Metrics of diversity (# taxa and # EPT taxa) were expected to decline with increased enrichment (Barbour et al. 1999, Stribling et al. 1998, USEPA 1997). As a result, the increase in total # of taxa was surprising, but is probably attributed to the high number of typically opportunistic taxa of Chironomidae that were found in streams with thicker detrital layers. Decreases in the number of EPT taxa were expected because of their known sensitivity to nutrient and organic enrichment. In addition, most of the

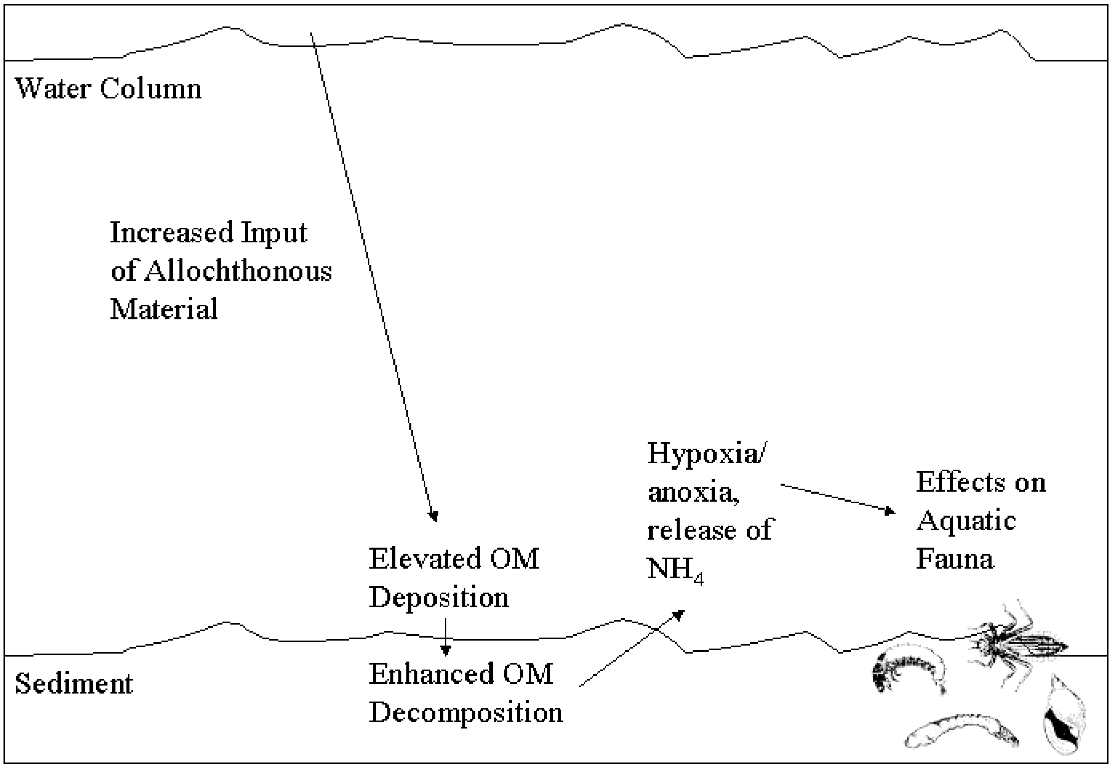

Figure 43. Model of allochthonous organic enrichment in study streams (adapted from Gray et al. 2002).

Table 8. Expected response of biotic metrics to increasing nutrient enrichment, and observed response due to enrichment with allochthonous organic matter (adapted from Stribling et al. 1998 and USEPA 1997).

Metric	Expected Response with Increased Nutrient Enrichment	Observed Response with Increased Detritus/Sediment OM/NH_4^+?
Taxonomic Richness		
# Taxa	Increase, then decrease	Y
# EPT Taxa	Increase, then decrease	Y
Taxonomic Composition		
Total Abundance	Increase	Y
% EPT	Decrease	Y
% Ephemeroptera	Decrease	Y
% Trichoptera	Decrease	Y
% Diptera	Increase	Y
% Chironomidae	Increase	Y
% Non-insects	Increase	Y
% Oligochaeta	Increase	Y
Simpson's DI	Decrease	Y
Tolerance		
Hilsenhoff BI	Increase	Y
Feeding		
% Scrapers	Increase, then decrease	Y
% Shredders	Decrease	Y
% Filterers	Increase	Y

metrics of taxonomic composition varied as expected with increased enrichment. Enhanced enrichment was expected to cause increases in total abundance and shifts in community composition toward dominance by opportunistic, tolerant taxa. Total abundance, and percent abundance of dipterans, chironomids, oligochaetes, and non-insects increased with detrital layer depth, sediment OM and/or ammonium. Dipterans, especially chironomids, are known to respond positively to conditions of enrichment (Harper 1992, Stribling et al. 1998), and assemblages dominated by organisms such as worms, snails, and others, are generally more tolerant of pollution than communities dominated by insects (US EPA 1997). Decreases in relative abundance of typically sensitive taxa, including EPT, ephemeropterans, and trichopterans, and declines in Simpson’s DI, were all expected as increased enrichment causes reductions in the number of intolerant organisms relative to opportunistic taxa (Harper 1992, Miltner and Rankin 1998, Sponseller et al. 2001, Whiles et al. 2000, Maxted et al. 1999). The Hilsenhoff BI, a measure of overall tolerance of the assemblage, increased as expected with increasing enrichment, as higher values represent greater tolerance of the community (Stribling et al. 1998, Barbour et al. 1999). Metrics of feeding were also useful as indicators of elevated enrichment. The increased relative abundance of filterers and lowered abundance of scrapers and shredders was likely due to changes in food availability as organic matter increased. Scrapers and shredders are specialist feeders, and are more sensitive to changes in availability of certain foods than are generalists (i.e. gatherers and filterers); thus their abundance decreases with enrichment relative to that of gatherers and filterers (Rawer-Jost et al. 2000).

The data suggest that while enrichment due to increased nutrient inputs and enhanced in-stream production is not significant in the study streams, enrichment with allochthonous organic matter does appear to be occurring. Though the creeks are likely on the low end of the enrichment scale with regards to nutrient enrichment (Figure 42), the streams probably fall towards the high end of the spectrum if enrichment is defined as allochthonous organic enrichment (Figure 44). Overall, these data indicate that local factors, especially changes in the chemical environment associated with decomposition of material from the riparian zone, are more important than watershed land use in dictating macroinvertebrate community structure.

Regional Comparisons

Though cropland agriculture did correlate with increased nitrate in the study streams, the effects of nutrient enrichment are likely to be greater in watersheds in which other agricultural activities, such as commercial animal production, lead to higher nutrient and organic loading rates. Over the past 50 years, production of poultry, beef, and hogs has increased in intensity as producers have concentrated larger numbers of animals in smaller areas (Howarth et al. 2000, Carpenter et al. 1998). Intensive animal production can lead to high levels of enrichment by concentrating nutrients and organic matter and increasing fluxes to rivers and offshore waters. Increased nutrient transport may result from accumulation of N and P in soils underlying feedlots and in fields subject to manure applications (Howarth et al. 2000, Smith et al. 1999). N inputs may also be increased through release of ammonia and subsequent volatilization and deposition (Howarth et al. 2000, Smith et al. 1999). In addition, increased transport of organic

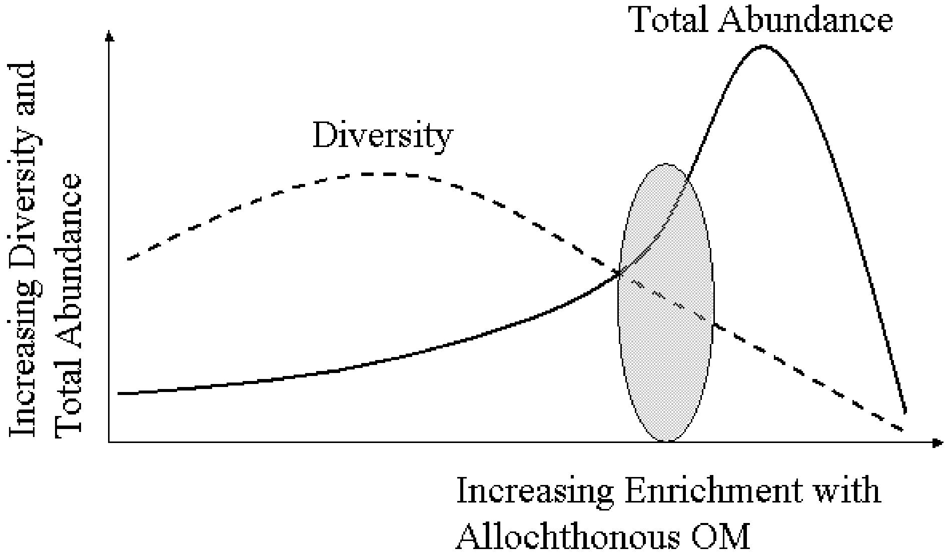

Figure 44. Probable location of study streams (shaded circle) on scale of increasing allochthonous organic enrichment.

wastes associated with animal production may lead to hypoxia/anoxia in aquatic systems, as natural waters typically have a biochemical oxygen demand (BOD) of 0.5 – 7 mg/l, whereas poultry wastes, for example, have a BOD of 24,000 – 67,000 mg/l (Cooper 1993).

While cropland agriculture is the dominant form of anthropogenic land use in the southern portion of the peninsula, intensive poultry production is dominant in the northern peninsula. Stanhope (2003) found significantly higher nitrate and DON concentrations in northern streams of the Delmarva Peninsula than in southern streams. These results suggest that commercial poultry production in the study region increases transport of nutrients to ground and surface waters more so than does fertilizer application. As a result, relationships between watershed land use and macroinvertebrate community structure may be more pronounced in streams in northern watersheds than was evident in the present study of southern catchments.

In addition, though elevated nitrate loadings associated with agricultural land use were not linked to adverse effects of nutrient enrichment in the headwater coastal streams, transport of nutrients downstream may have more serious impacts. Excessive input of nutrients from coastal watersheds has been linked to nutrient enrichment in many coastal water bodies, often with severe consequences for aquatic flora and fauna (Lapointe et al. 1992, McClelland et al. 1997, Correll et al. 1992, Valiela et al. 1992, Sfriso et al. 1992, Bachelet et al. 2000). In the VCR, tidal creeks are the primary linkage between terrestrial watersheds and coastal waters, and as a result, they serve as a conduit for transport of nutrients and organic matter from the mainland to the lagoons (Lerberg et al. 2000). In addition, groundwater has recently been recognized as an important source of nutrients to coastal waters, especially in regions where anthropogenic activities increase nutrient inputs to groundwaters (Lapointe and Clark 1992, Paerl 1997).

Increased inputs of N may be more likely to lead to excessive production of organic matter in VCR coastal lagoons than in the study streams, primarily because N is typically the limiting nutrient in temperate coastal waters, while P is generally more limiting in freshwater systems (Howarth et al. 2000, Jordan et al. 1991, Paerl 1997, Smith et al. 1999). The VCR lagoons may be especially susceptible to nutrient enrichment because restricted exchange with the ocean and low freshwater input reduces flushing of nutrients, shallow depths allow accumulation of benthic autotrophs, and a high surface area to volume ratio enables close interaction between sediments and the water column (McComb 1995, Boynton et al. 1996). Because of the shallow nature of coastal lagoons, these systems tend to be dominated by benthic autotrophs, such as seagrasses, macroalgae and benthic microalgae, rather than phytoplankton (Tyler et al. 2001, McGlathery et al. 2001). In conditions of increased nutrient loading, enhanced production by fast-growing, opportunistic macroalgae may lead to development of macroalgal blooms and declines in distribution of slower-growing rooted macrophytes (Tyler et al. 2001, Hauxwell et al. 2001, Valiela et al. 1997, Havens et al. 2001, Flindt et al. 1999). The importance of seagrasses in providing shelter and food for a variety of organisms, removing toxins from the water column, and in stabilizing sediments is well understood, and loss of this vegetation has had serious consequences for many systems (Davis 1985, Flindt et al. 1999, Howarth et al. 2000).

A central hypothesis of VCR-LTER research is that groundwater delivery of nutrients is a major factor influencing primary and secondary production and determining nutrient and organic matter processing in the lagoons (Hayden et al. 2000). The dominant primary producers in VCR coastal lagoons are benthic macroalgae and microalgae, while seagrasses disappeared in the 1930s and have only recently returned in small, isolated distributions (Tyler et al. 2001, Havens et al. 2001, McGlathery et al. 2001). The high production by benthic macroalgae may be supported by nutrient inputs from the mainland aquifer, the primary source of nutrients to the lagoon (McGlathery et al. 2001). Hog Island Bay (HIB) is a shallow lagoon in the VCR, and is typical of other coastal lagoons along the Atlantic Coast (McGlathery 2001). Evidence for transport of terrestrially-derived nitrogen to VCR coastal lagoons is apparent in the gradient in nutrient loading across HIB, with highest water column nutrient concentrations, highest sediment N and organic contents, and highest macroalgal tissue N near the mainland (McGlathery et al. 2001). The importance of this nitrogen to macroalgal growth in VCR lagoons has been demonstrated by Stanhope (2003), who found that baseflow TDN loads from watersheds of HIB were sufficient to support annual mean macroalgal N demand in the lagoon. Thus, increases in nutrient inputs associated with agricultural activity in VCR terrestrial watersheds, while not affecting headwater segments, may enhance production of coastal autotrophs, with potentially detrimental effects on coastal ecosystems.

In addition, the impact of elevated N loadings on downstream coastal systems may be different along an east-west transect of the southern Eastern Shore. The present study focused on the eastern, or seaside of the peninsula, where tidal creeks are relatively deep and narrow, and drain through extensive salt marshes prior to entering shallow coastal lagoons (MacMillin et al. 1992, Hayden et al. 2000). On the western shore (bayside), marshes are less extensive, and creeks are wider and shallower, and drain into Chesapeake Bay (MacMillin et al. 1992). MacMillan et al. (1992) found higher inorganic nutrient concentrations, lower bacterial productivity, and lower water column chlorophyll a concentrations in saline reaches of tidal creeks on the seaside when compared to creeks on the bayside. They suggest that the higher dissolved organic carbon turnover times on the bayside may be due to differences in sources and quality of organic matter between the two locations (MacMillan et al. 1992). Seaside OM is derived primarily from marsh grasses, while bayside OM originates mostly from phytoplankton, which is typically thought to be higher in quality than macrophytic material (MacMillan et al. 1992, Gray et al. 2002). Regardless of higher concentrations of nutrients, seaside creeks had lower standing stocks of chlorophyll a and smaller and less productive bacterial populations, indicating that the potential for increased nutrient loading to cause enrichment may be lesser on the seaside. The combination of enhanced production of high quality organic matter (phytoplankton) and higher bacterial productivity would likely result in more rapid decomposition (Gray et al. 2002), potentially leading to more serious conditions of hypoxia/anoxia on the bayside. In contrast, differences in freshwater reaches of tidal creeks in the southern peninsula are likely to be small, as baseflow from the same unconfined aquifer is the major component of stream flow for both sides, and land uses are similar (Robinson and Reay 2002, Reay et al. 1992, Lowrance et al. 1997).

Conclusions and Future Directions for Research

The results of this research suggest that non-point source nutrient pollution in the study streams is insufficient to produce significant changes in primary production, decomposition, and macroinvertebrate community structure often associated with nutrient enrichment. As a result, the sampling reaches are likely on the low end of the nutrient enrichment spectrum, perhaps because P is typically the limiting nutrient in freshwaters, and P concentrations were low. Therefore, in coastal plain streams with flat topography and forested riparian buffers, low surface runoff and/or retention by vegetation along stream corridors appear to be important in reducing nutrient transport and associated enrichment.

Local factors, primarily input of detritus from the riparian forest, were more important than catchment land use in dictating macroinvertebrate community structure, even in watersheds with high % agriculture. Overall, increased inputs of allochthonous organic matter caused elevated decomposition, increases in total abundance of macroinvertebrate communities, and a shift in community structure toward dominance by tolerant taxa, all patterns which were hypothesized to occur in response to nutrient enrichment. Therefore, while enrichment due to increased nutrient inputs and enhanced in-stream production is not significant in the study streams, enrichment with allochthonous organic matter does appear to be occurring. Though the creeks are probably on the low end of the nutrient enrichment scale, the streams are likely towards the high end of the spectrum with respect to allochthonous organic enrichment.

In addition, this study highlights the important role of riparian forests in organic matter cycling and decomposition in coastal streams. The potential for riparian forests in the study area to act as sinks for both N and P may also be significant. As a result, future studies assessing relationships between characteristics of riparian zones and macroinvertebrate community structure may provide further insight into the effects of changes in land use adjacent to stream channels.

The results of this study also indicate that caution should be used when interpreting biomonitoring data. Changes detected through monitoring of macroinvertebrate communities may not necessarily indicate impairment due to human impacts, as observed patterns in this study appeared to be unrelated to anthropogenic activities.

On the other hand, the biotic metrics employed in this study were generally successful in detecting changes in sediment and water column chemistry that are often associated with anthropogenically-driven nutrient and organic enrichment. Their effectiveness indicates that macroinvertebrates may be a useful tool in monitoring of human-induced pollution in other streams of this region. Metrics of community richness, composition, tolerance, and trophic abundance were all effective indicators of enhanced organic matter deposition, lowered DO, and ammonium sensitivity. It is likely then, that these communities would exhibit similar, if not stronger, responses to pollution in watersheds subject to discharges of animal wastes. As a result, benthic macroinvertebrates may be very useful tools for assessing impacts of land use in streams in the northern portion of the Delmarva Peninsula, where commercial poultry production leads to higher nutrient and organic matter loadings. Inclusion of northern streams in future studies would provide larger gradients in nutrient and organic loading rates, potentially resulting in more pronounced relationships between watershed land use and macroinvertebrate community structure.

Future studies might also encompass analysis of effects of increased nutrient loadings along the transect from headwater tidal creek to coastal lagoon. Due to differences in nutrient limitation, light availability, and flushing rates, the response of aquatic systems to increased loadings is likely to vary along this continuum. Coastal lagoons may be more susceptible to nutrient enrichment in conditions of elevated N loadings than are headwater stream reaches, as a result of the likelihood of N limitation in coastal waters and P limitation in freshwater segments.

The use of isotopes in assessing nutrient pollution is relatively new, but may prove quite useful in detecting impacts prior to the occurrence of changes in macroinvertebrate community structure. Further studies should attempt to incorporate a wider range of individuals occupying different trophic roles to assess impacts due to changing N signatures in various food sources. In addition, studies of isotopic signatures of riparian vegetation, aquatic plants, and ground and surface waters will potentially enable better assessment of the impacts of nutrient enrichment on the availability of a variety of foods.

Furthermore, inclusion of episodic events and experimental enrichment of the study streams may provide information on changes in the short term and in response to pulses of nutrients that may occur at different times of the year. Future studies should also involve more intensive sampling during summer months, when autotrophic, microbial, and macroinvertebrate activities are likely to be most pronounced. Finally, a multiyear study to assess interannual variability and stability of macroinvertebrate communities in response to changes in local and catchment land use would provide additional and useful information.

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APPENDIX A:

MONTHLY VALUES OF WATER COLUMN AND SEDIMENT VARIABLES

Water column TDN (uM)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	19.10	55.49	39.69	496.95	189.22	138.30	358.44	200.12	550.72
8/03	21.71	64.19	48.91	155.58	276.07	146.91	345.89	194.77	604.22
10/03	67.24	21.99	28.13	176.35	106.92	83.97	141.00	166.81	163.24
1/04	38.70	28.57	33.78	308.17	220.03	221.19	334.24	227.64	672.03
3/04	40.22	23.69	24.18	274.53	177.12	194.73	222.31	218.64	514.41
Mean[1]	41.31	32.43	31.44	314.00	173.32	159.55	264.00	203.30	475.10

Water column nitrate (uM)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	27.30	13.68	43.42	506.59	219.59	167.55	386.30	220.56	770.91
8/03	74.42	4.92	83.86	135.28	337.78	137.71	330.35	166.42	588.03
10/03	92.07	0.98	43.03	167.26	134.20	124.27	169.12	230.62	232.62
1/04	81.69	16.22	75.63	287.26	230.54	228.78	370.26	216.11	650.03
3/04	69.24	6.02	64.64	451.68	340.83	330.97	369.96	368.11	815.33
Mean[1]	67.57	9.23	56.68	353.20	231.29	212.89	323.91	288.85	617.22

Water column ammonium (uM)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	1.44	4.02	10.43	1.42	5.01	4.42	1.29	0.09	5.10
8/03	0.62	4.54	5.84	1.94	4.24	4.91	1.99	6.77	2.32
10/03	2.93	3.73	13.27	24.84	8.24	12.49	7.37	3.91	2.58
1/04	0.15	5.45	0.01	4.64	3.15	4.60	1.30	0.00	4.84
3/04	2.43	6.99	1.01	4.13	3.66	5.05	0.40	0.97	5.85
Mean[1]	1.74	5.05	6.18	8.76	5.02	6.64	2.59	1.24	4.59

Water column DON (uM)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.00	37.79	0.00	40.40	0.00	0.00	0.00	0.00	28.48
8/03	0.00	54.74	0.00	18.36	0.00	4.89	15.02	21.58	14.60
10/03	0.00	17.28	0.00	0.08	0.00	0.00	0.00	0.00	0.00
1/04	0.00	12.24	0.00	22.59	2.47	0.98	0.00	15.86	29.18
3/04	0.00	10.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mean[1]	0.00	19.50	0.00	15.77	0.62	0.25	0.00	3.97	14.41

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

Water column phosphate (uM)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.44	1.14	1.20	1.12	0.15	0.20	1.78	0.34	0.24
8/03	0.27	1.16	1.63	2.47	0.19	2.17	1.11	1.01	0.22
10/03	0.04	0.19	0.56	1.30	0.07	0.07	1.09	0.16	0.15
1/04	0.14	0.38	0.17	0.19	0.14	0.04	0.14	0.06	0.01
3/04	0.23	0.29	0.36	0.24	0.20	0.12	0.24	0.17	0.02
Mean[1]	0.21	0.50	0.57	0.71	0.14	0.11	0.81	0.18	0.11

Water column TSS (mg/L)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	14.6	38.1	28.1	30.7	22.8	25.4	24.8	28.3	15.1
8/03	23.4	16.8	13.4	20.1	12.8	14.3	8.8	26.6	8.2
10/03	10.3	12.6	3.9	18.9	6.9	7.5	5.8	5.2	4.4
1/04	8.9	6.7	11.3	10.5	5.6	10.4	18.8	21.8	15.8
3/04	4.5	12.3	5.9	14.2	5.5	5.7	10.4	10.7	11.1
Mean[1]	9.6	17.4	12.3	18.6	10.2	12.2	14.9	16.5	11.6

Water column SPOM (mg/L)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	5.7	23.3	15.8	9.2	15.0	10.7	15.0	17.5	10.0
8/03	13.2	9.8	6.8	11.9	6.7	7.2	5.1	13.4	3.8
10/03	7.8	9.6	4.9	17.3	5.7	5.4	6.1	3.7	3.4
1/04	2.1	1.9	4.6	2.2	1.6	4.6	7.4	9.0	5.7
3/04	2.1	5.2	2.4	5.6	2.5	1.7	3.7	3.7	5.3
Mean[1]	4.4	10.0	6.9	8.6	6.2	5.6	8.0	8.5	6.1

Water column chlorophyll a (ug/L)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	1.60	2.97	3.03	1.48	0.78	1.98	3.31	1.56	3.10
8/03	1.16	6.17	3.60	6.00	2.19	5.91	10.83	3.38	5.53
10/03	0.53	3.34	0.21	1.93	0.92	1.28	4.63	1.82	4.25
1/04	0.51	1.22	0.88	1.40	0.33	16.27	2.14	0.66	0.52
3/04	0.93	2.28	3.49	4.66	1.07	6.27	6.54	2.03	1.99
Mean[1]	0.89	2.45	1.90	2.37	0.77	6.45	4.15	1.52	2.47

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

Water column DO (ppm)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	6.7	5.2	2.9	5.7	6.6	5.1	5.8	7.1	4.9
8/03	6.5	4.1	4.1	4.1	5.8	5.1	6.3	5.8	5.1
10/03	6.3	3.6	2.6	2.6	3.2	4.1	3.8	6.5	2.6
1/04	9.5	8.6	8.1	8.8	8.7	9.2	9.6	10.0	11.4
3/04	8.2	7.8	6.4	8.8	7.8	7.0	8.8	8.8	9.0
Mean[1]	7.7	6.3	5.0	6.5	6.6	6.4	7.0	8.1	7.0

Water column temperature (C)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	19.9	21.2	20.3	20.6	19.5	23.8	22.9	19.8	24.3
8/03	20.5	21.5	21.3	22.7	19.9	23.3	22.3	21.5	24.6
10/03	13.7	15.4	14.3	15.8	14.7	16.0	16.8	16.1	15.8
1/04	7.0	5.5	4.3	4.2	8.4	6.1	4.4	6.6	4.2
3/04	11.1	11.1	12.1	12.8	12.0	13.9	13.1	11.8	12.5
Mean[1]	12.9	13.3	12.8	13.4	13.7	15.0	14.3	13.6	14.2

Incident light (uE/m^2/s)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
8/03	1220.0	1320.0	982.3	52.3	26.2	26.6	246.8	111.5	329.3
1/04	820.5	259.0	1170.0	64.8	286.5	440.0	206.0	184.4	518.0
3/04	174.0	900.2	955.4	350.4	346.4	885.8	389.2	1121.0	708.2
Mean	738.2	826.4	1035.9	155.8	219.7	450.8	280.7	472.3	518.5

Discharge (m^3/s)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
8/03	0.0155	0.0136	0.0084	0.0101	0.0096	0.0188	0.0083	0.0296	0.0098
10/03	0.0151	0.0054	0.0036	0.0045	0.0027	0.0054	0.0095	0.0127	0.0030
1/04	0.0378	0.0093	0.0093	0.0124	0.0053	0.0133	0.0191	0.0198	0.0097
3/04	0.0269	0.0121	0.0064	0.0073	0.0036	0.0063	0.0180	0.0117	0.0060
Mean[1]	0.0266	0.0089	0.0064	0.0081	0.0039	0.0083	0.0155	0.0147	0.0062

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

Salinity (ppt)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.1	0.1	0.1	0.1	0.2	0.1	0.1	0.1	0.2
8/03	0.1	0	0.1	0.1	0.1	0.1	0.1	0.1	0.1
10/03	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
1/04	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
3/04	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Mean[1]	0.1	0.1	0.1	0.1	0.13	0.1	0.1	0.1	0.13

pH

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
3/04	6	6	6.5	6.5	6	6.5	6.5	6.5	6.5

Porewater ammonium (uM)

Depth (cm)	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
7/03									
0	4.72	11.62	30.96	16.12	19.17	7.45	7.91	7.48	24.29
2	6.85	27.28	60.57	147.20	11.64	82.31	16.77	19.66	85.21
4	7.32	38.18	109.42	181.24	21.13	118.60	41.39	25.00	213.22
6	8.91	27.69	125.32	193.73	24.25	125.96	45.79	21.05	158.33
8	9.49	53.35	138.73	164.18	12.58	107.46	76.90	16.30	179.34
Mean	7.46	31.62	93.00	140.49	17.76	88.35	37.75	17.90	132.08

Porewater sulfide (uM) (7/03)

Depth (cm)	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
7/03									
0	0.73	2.35	0.98	1.37	0.48	0.76	1.29	0.51	0.81
2	2.15	4.64	2.97	11.74	2.18	2.91	6.37	4.09	4.37
4	6.45	6.16	5.34	18.51	5.31	4.96	5.06	10.39	3.57
6	8.14	11.75	5.93	25.49	5.46	5.12	8.13	9.65	4.00
8	13.08	14.06	8.80	17.62	3.46	6.76	6.99	8.63	5.42
Mean	6.11	7.79	4.80	14.95	3.38	4.10	5.57	6.66	3.63

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

Extractable ammonium (umol/g dry sediment)

Depth (cm)	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
7/03									
0-2	0.02	0.05	0.17	0.35	0.20	0.17	0.17	0.04	0.18
2-5	0.01	0.11	0.24	0.35	0.33	0.21	0.20	0.11	0.23
5-10	0.02	0.11	0.18	0.16	0.28	0.27	0.16	0.03	0.22
Mean	0.02	0.09	0.20	0.29	0.27	0.22	0.18	0.06	0.21
10/03									
0-2	0.00	0.11	0.30	0.20	0.32	0.03	0.36	0.04	0.13
2-5	0.00	0.15	0.24	0.27	0.57	0.05	0.39	0.04	0.25
5-10	0.01	0.14	0.17	0.19	0.40	0.01	0.17	0.02	0.41
Mean	0.00	0.13	0.24	0.22	0.43	0.03	0.31	0.03	0.27
1/04									
0-2	0.00	0.05	0.03	0.08	0.47	0.04	0.24	0.01	0.02
2-5	0.00	0.14	0.05	0.17	0.47	0.11	0.26	0.01	0.05
5-10	0.00	0.13	0.14	0.25	0.46	0.20	0.25	0.01	0.11
Mean	0.00	0.11	0.07	0.17	0.47	0.12	0.25	0.01	0.06
3/04									
0-2	0.00	0.09	0.05	0.05	0.23	0.02	0.12	0.03	0.01
2-5	0.00	0.14	0.19	0.20	0.34	0.07	0.11	0.03	0.08
5-10	0.00	0.28	0.70	0.20	0.29	0.09	0.08	0.02	0.14
Mean	0.00	0.17	0.31	0.15	0.29	0.06	0.10	0.03	0.08
0-2 Mean	0.00	0.08	0.14	0.17	0.30	0.07	0.22	0.03	0.09
2-5 Mean	0.00	0.14	0.18	0.25	0.43	0.11	0.24	0.05	0.15
5-10 Mean	0.01	0.17	0.30	0.20	0.36	0.14	0.16	0.02	0.22

Sediment C:N

Depth (cm)	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
7/03									
0-2	21.10	24.62	21.45	22.84	19.99	22.08	22.65	24.09	17.56
2-5	18.91	24.83	28.69	22.62	21.14	22.78	22.70	25.28	16.24
5-10	28.42	26.86	27.12	20.59	21.51	23.12	23.86	25.23	16.20
Mean	22.81	25.44	25.75	22.02	20.88	22.66	23.07	24.87	16.67
10/03									
0-2	19.79	27.04	17.96	23.17	30.47	17.76	18.71	31.73	18.34
2-5	30.33	23.88	28.49	22.41	21.79	16.86	17.76	24.12	18.45
5-10	26.33	25.73	34.10	20.97	25.04	17.91	18.94	26.51	26.05
Mean	25.48	25.55	26.85	22.18	25.77	17.51	18.47	27.45	20.95
1/04									
0-2	27.23	20.70	25.81	25.35	19.63	21.02	22.32	27.35	17.04
2-5	23.00	21.34	25.82	22.80	19.55	21.05	21.98	29.31	17.39
5-10	23.16	24.94	27.11	22.03	21.24	22.50	24.57	29.85	18.50
Mean	24.46	22.32	26.25	23.39	20.14	21.52	22.96	28.84	17.64
3/04									
0-2	36.26	21.76	28.66	23.59	20.72	22.11	20.34	27.19	14.99
2-5	28.14	23.64	22.91	21.25	20.31	20.53	22.15	27.23	15.14
5-10	37.49	20.68	25.14	22.92	19.41	23.02	22.93	29.89	15.69
Mean	33.97	22.03	25.57	22.59	20.15	21.89	21.81	28.10	15.27
0-2 Mean	26.10	23.53	23.47	23.74	22.70	20.74	21.01	27.59	16.98
2-5 Mean	25.09	23.42	26.48	22.27	20.70	20.31	21.15	26.49	16.81
5-10 Mean	28.85	24.55	28.37	21.63	21.80	21.64	22.58	27.87	19.11

Sediment organic matter (%)

Depth (cm)	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
7/03									
0-2	1.15	1.17	3.22	5.42	21.65	6.14	4.82	7.69	4.18
2-5	9.08	8.35	3.46	8.03	18.42	5.47	7.02	2.36	3.17
5-10	0.97	6.79	3.09	4.34	18.58	7.52	10.35	7.88	4.09
Mean	3.73	5.43	3.26	5.93	19.55	6.38	7.40	5.98	3.81
10/03									
0-2	0.75	13.34	11.30	4.02	18.13	6.42	18.22	7.94	4.93
2-5	0.31	15.87	6.55	4.05	17.48	8.60	22.47	5.33	3.56
5-10	0.35	16.36	2.98	2.43	16.89	10.60	3.84	4.38	4.17
Mean	0.47	15.19	6.94	3.50	17.50	8.54	14.84	5.88	4.22
1/04									
0-2	0.26	11.16	1.03	13.49	21.72	6.75	12.39	8.65	1.19
2-5	0.47	15.71	3.51	5.94	15.32	9.08	13.98	10.22	2.80
5-10	1.78	14.91	7.28	6.04	15.40	9.39	7.91	14.92	3.25
Mean	0.84	13.93	3.94	8.49	17.48	8.41	11.43	11.26	2.41
3/04									
0-2	0.33	4.58	1.03	4.94	19.89	4.09	4.90	16.61	1.24
2-5	0.29	8.85	3.74	6.56	15.21	6.07	6.25	16.06	2.51
5-10	0.88	22.12	18.01	4.88	12.17	3.34	3.45	4.86	3.83
Mean	0.50	11.85	7.59	5.46	15.76	4.50	4.87	12.51	2.53
0-2 Mean	0.62	7.56	4.14	6.97	20.35	5.85	10.08	10.22	2.88
2-5 Mean	2.54	12.19	4.31	6.14	16.61	7.31	12.43	8.49	3.01
5-10 Mean	1.00	15.04	7.84	4.42	15.76	7.71	6.39	8.01	3.84

Benthic chlorophyll a (mg/m^2)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
7/03	29.80	7.25	5.10	4.29	7.65	3.76	17.18	7.25	8.59
10/03	49.39	2.68	5.64	4.43	9.93	5.37	7.78	6.98	19.33
1/04	23.08	3.49	2.95	6.17	7.52	3.22	13.42	7.52	7.52
3/04	36.77	22.82	5.37	5.64	4.83	6.44	38.12	22.28	11.01
Mean	34.76	9.06	4.76	5.13	7.48	4.70	19.13	11.01	11.61

Sediment particle size (% of total dry weight of sediment)

Sieve Size (um)	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
500	26.22	37.41	57.37	29.52	15.51	12.66	37.84	28.10	29.81
250	52.05	34.06	29.57	48.86	30.08	29.49	33.32	44.86	39.05
125	18.21	20.49	8.53	13.72	23.08	27.12	13.07	20.08	21.83
63	0.36	1.00	1.05	1.20	5.55	3.06	3.05	1.13	2.10
<63	3.16	7.03	3.48	6.70	25.77	27.67	12.73	5.83	7.20

APPENDIX B:

MONTHLY VALUES OF MACROINVERTEBRATE COMMUNITY COMPOSITION AND CALCULATED METRICS

June macroinvertebrate community composition (values are # organisms)

Genus	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
Family Lumbricidae (unidentified)		1	1		1				
Eclipidrilus			6	11	1	1	2	2	
Family Lumbriculidae (unidentified)							2		
Limnodrilus			2	2	4			1	1
Pseudosuccinea							1		
Physella			3	1					5
Promenetus				9					
Pisidium			6	40	18	7	8		2
Musculium							13		1
Sphaerium									5
Crangonyx		8		7	5			1	
Gammarus	1					3	28		90
Cambarus				2	1				
Caecidotea				17	7				
Chironomus				6					
Cryptochironomus						1			
Microtendipes							14		
Thienemannimyia					2				
Tribelos				1	2		1		
Dicranota								3	
Hexatoma		1							
Sialis							1		
Calopteryx								1	
Epicordulia				1					
Beraea								1	
Psilotreta	5								

August macroinvertebrate community composition (values are # organisms) * non-baseflow conditions

Genus	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
Class Oligochaeta (unidentified)	3	2	19		5	1	10		9
Family Planariidae (unidentified)			45						
Physella			2				1		
Menetus			2						
Promenetus			3						
Pisidium			12	2	1		2		
Musculium							1		
Sphaerium							1		
Crangonyx	1	4							
Gammarus						9	26		42
Cambarus				1					
Caecidotea				1	1	3			
Family Chironomidae (unidentified)	4	7			68		33	2	5
Dicranota	1								
Order Diptera (unidentified)								1	
Sialis							1		
Order Trichoptera (unidentified)							2		

October macroinvertebrate community composition (values are # organisms)

Genus	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
Eclipidrilus				1		1	6		2
Lumbriculus		1							
Limnodrilus	1				1				4
Psammoryctides				2					
Spirosperma		1							
Phylum Nemertea? (unidentified)								2	
Family Planariidae (unidentified)			2						
Physella			1			1			
Menetus						1	6		
Promenetus							7		
Pisidium	8	89	18	34		6	92	3	
Musculium				1		1	89		
Sphaerium							2		
Crangonyx		13			4				
Gammarus	30					17	28		56
Caecidotea		1		1	101		7		
Deronectes					1				
Stenelmis								1	
Chironomus				3					1
Clinotanypus					17		1		
Microtendipes	3				44		16	4	
Parametriocnemus					2				
Stictochironomus								52	
Tanytarsus					1				
Thienemannimyia					9				
Tribelos					6		6		
Zavrelimyia		9			1				
Hexatoma	2								
Stenonema	3								
Paraleptophlebia							1		
Sialis					2		1		
Calopteryx						1	5	1	1
Cordulegaster	1							1	
Tetragoneuria					1				
Plathemis				1			1		1
Heteroplectron		2			1				
Cheumatopsyche							6		1
Molanna					4				
Marilia	1								
Psilotreta	2								
Oligostomis							1	1	

January macroinvertebrate community composition (values are # organisms)

Genus	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
Family Lumbricidae (unidentified)									1
Eclipidrilus			21	6	2	2	8	2	17
Limnodrilus				12			2		9
Psammoryctides				1					
Spirosperma					1				
Family Tubificidae (unidentified)	1	1		1					
Family Planariidae (unidentified)		1	15		1	1			
Pseudosuccinea			2						
Physella			16				1		
Menetus			1			1	1		
Promenetus			1	5			5		
Pisidium	6	21	141	225	7	18	164	1	3
Musculium							32		
Sphaerium							25		
Crangonyx		8		9	8	1		2	
Stygonectes				9					
Gammarus	17			1		33	3		80
Caecidotea		3		44	55		38	1	
Deronectes				1					
Cyphon		1							
Probezzia	1						3		
Chironomus				83					
Clinotanypus					4		1		
Hydrobaenus				1					
Microspectra				1					
Microtendipes	5	1			357		49		
Natarsia							1		
Polypedilum	1								
Procladius					2			1	
Stictochironomus								17	
Thienemannimyia					1		3	1	
Tribelos		35		5	51	1	18		
Hemerodromia								1	
Simulium/Prosimulium		1							
Chrysops		1							
Tabanidae (unidentified)	1								
Dicranota								5	
Hexatoma	1								
Paraleptophlebia								2	
Boyeria								1	
Cordulegaster							1		
Plathemis				1	2				
Beraea								3	

Heteroplectron		2						
Cheumatopsyche							5	
Molanna					1			
Psilotreta	3				1			1
Oligostomis				1	1		1	

March macroinvertebrate community composition (values are # organisms)

Genus	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
Family Lumbricidae (unidentified)				1					
Eclipidrilus		3	28	12	1	7	5		14
Lumbriculus		3							
Limnodrilus				53	1	1		1	23
Family Tubificidae (unidentified)	1								
Planariidae (unidentified)			8					1	
Pseudosuccinea			1						
Physella			12						
Menetus			1	2		2	1		
Promenetus				2			2		
Pisidium	7	16	197	161	13	28	84		1
Musculium							19		
Sphaerium							25		1
Crangonyx		8		9	7	1		2	
Stygonectes				1					
Gammarus	49					45	10		44
Cambarus									
Caecidotea		5		44	34		49		
Deronectes					1				
Stenelmis								1	
Chironomus				22					
Clinotanypus					3				
Cricotopus/Orthocladius				1		2	46		1
Cryptochironomus					1				
Microtendipes	1	3			328		23		
Natarsia							1		
Paracladopelma	3								
Paratanytarsus							1		
Polypedilum	3	5							
Procladius							1		1
Stictochironomus								4	
Thienemannimyia	1	1					5		
Tribelos		32		1	24		14		
Zavrelimyia									
(unidentified)				1					
Simulium								1	
Chrysops		1					1	2	
Dicranota									1
Hexatoma	2								
Pseudolimnophila		1			1			1	
Leptophlebia							1	1	
Sialis					2				
Cordulegaster	1								
Plathemis									1

Beraea								4	
Ironoquia				1					
Molanna					1				
Psilotreta								1	
Oligostomis					1				

Taxa

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	2	3	4	11	9	4	8	6	6
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	9	7	3	7	15	7	17	8	7
1/04	10	11	7	16	15	7	19	13	5
3/04	9	11	6	13	14	7	17	11	10
Mean[1]	7.5	8	5	11.75	13.25	6.25	15.25	9.5	7

EPT Taxa

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	1	0	0	0	0	0	0	1	0
8/03	0	0	0	0	0	0	1	0	0
10/03	3	1	0	0	2	0	3	1	1
1/04	2	1	0	1	3	0	2	3	0
3/04	0	0	0	1	2	0	1	3	0
Mean	1.5	0.5	0	0.5	1.75	0	1.5	2	0.25

Total Abundance (#)

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	6	10	18	97	41	12	70	9	104
8/03	9	13	83	4	75	13	77	3	56
10/03	51	116	21	43	195	28	275	65	66
1/04	37	75	197	406	494	57	361	38	110
3/04	68	78	247	314	418	86	288	20	88
Mean	40.5	69.8	120.75	215	287	45.75	248.5	33	92

% EPT

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	83.3	0.0	0.0	0.0	0.0	0.0	0.0	11.1	0.0
8/03	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0
10/03	11.8	1.7	0.0	0.0	2.6	0.0	2.9	1.5	1.5
1/04	10.8	2.7	0.0	0.2	0.6	0.0	1.7	15.8	0.0
3/04	0.0	0.0	0.0	0.3	0.5	0.0	0.3	30.0	0.0
Mean	26.5	1.1	0.0	0.1	0.9	0.0	1.2	14.6	0.4

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

% Ephemeroptera

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8/03	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
10/03	5.9	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
1/04	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.3	0.0
3/04	0.0	0.0	0.0	0.0	0.0	0.0	0.3	5.0	0.0
Mean	1.5	0.0	0.0	0.0	0.0	0.0	0.2	2.6	0.0

% Trichoptera

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	83.3	0.0	0.0	0.0	0.0	0.0	0.0	11.1	0.0
8/03	0.0	0.0	0.0	0.0	0.0	0.0	2.6	0.0	0.0
10/03	5.9	1.7	0.0	0.0	2.6	0.0	2.5	1.5	1.5
1/04	10.8	2.7	0.0	0.2	0.6	0.0	1.7	10.5	0.0
3/04	0.0	0.0	0.0	0.3	0.5	0.0	0.0	25.0	0.0
Mean	25.0	1.1	0.0	0.1	0.9	0.0	1.1	12.0	0.4

% Diptera

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.0	10.0	0.0	7.2	9.8	8.3	21.4	33.3	0.0
8/03	55.6	53.8	0.0	0.0	90.7	0.0	42.9	100.0	8.9
10/03	9.8	7.8	0.0	7.0	41.0	0.0	8.4	86.2	1.5
1/04	24.3	50.7	0.0	22.2	84.0	1.8	20.8	65.8	0.0
3/04	14.7	55.1	0.0	8.0	85.4	2.3	31.9	45.0	3.4
Mean	12.2	30.9	0.0	11.1	55.0	3.1	20.6	57.6	1.2

% Chironomidae

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.0	0.0	0.0	7.2	9.8	8.3	21.4	0.0	0.0
8/03	44.4	53.8	0.0	0.0	90.7	0.0	42.9	66.7	8.9
10/03	5.9	7.8	0.0	7.0	41.0	0.0	8.4	86.2	1.5
1/04	16.2	48.0	0.0	22.2	84.0	1.8	19.9	50.0	0.0
3/04	11.8	52.6	0.0	8.0	85.2	2.3	31.6	20.0	2.3
Mean	8.5	27.1	0.0	11.1	55.0	3.1	20.3	39.0	0.9

% Non-insects

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	16.7	90.0	100.0	91.8	90.2	91.7	77.1	44.4	100.0
8/03	44.4	46.2	100.0	100.0	9.3	100.0	58.4	0.0	91.1
10/03	76.5	90.5	100.0	90.7	54.4	96.4	86.2	9.2	93.9
1/04	64.9	45.3	0.0	77.1	15.0	98.2	77.3	15.8	100.0
3/04	83.8	44.9	100.0	91.7	13.4	97.7	50.7	20.0	95.5
Mean	60.5	67.7	75.0	87.8	43.2	96.0	72.8	22.4	97.3

% Oligochaeta

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.0	10.0	50.0	13.4	14.6	8.3	5.7	33.3	1.0
8/03	33.3	15.4	22.9	0.0	6.7	7.7	13.0	0.0	16.1
10/03	2.0	1.7	0.0	7.0	0.5	3.6	2.2	0.0	9.1
1/04	2.7	1.3	10.7	4.9	0.6	3.5	2.8	5.3	24.5
3/04	1.5	7.7	11.3	22.0	0.5	9.3	1.7	5.0	42.0
Mean	1.5	5.2	18.0	11.8	4.1	6.2	3.1	10.9	19.2

Simpson’s Diversity Index

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.3	0.3	0.7	0.8	0.7	0.6	0.8	0.8	0.2
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	0.6	0.4	0.3	0.4	0.7	0.6	0.8	0.4	0.3
1/04	0.7	0.7	0.5	0.6	0.5	0.6	0.7	0.8	0.4
3/04	0.5	0.8	0.3	0.7	0.4	0.6	0.8	0.9	0.7
Mean[1]	0.5	0.5	0.4	0.6	0.6	0.6	0.8	0.7	0.4

% Dominant Taxon

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	83.3	80.0	33.3	41.2	43.9	58.3	40.0	33.3	86.5
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	58.8	76.7	85.7	79.1	51.8	60.7	33.5	80.0	84.8
1/04	45.9	46.7	71.6	55.4	72.3	57.9	45.4	44.7	72.7
3/04	72.1	41.0	79.8	51.3	78.5	52.3	29.2	20.0	50.0
Mean[1]	65.0	61.1	67.6	56.7	61.6	57.3	37.0	44.5	73.5

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

Hilsenhoff Biotic Index

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	1.0	4.6	8.3	7.7	7.5	7.5	6.7	6.0	6.3
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	5.8	7.3	7.3	8.2	7.1	6.7	7.4	8.5	6.4
1/04	5.8	5.9	7.4	8.3	6.1	6.6	7.5	6.9	6.7
3/04	6.2	6.0	7.8	8.3	6.1	6.9	7.4	5.6	7.5
Mean	4.7	5.9	7.7	8.1	6.7	6.9	7.3	6.7	6.7

% Scrapers

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	83.3	0.0	16.7	10.3	0.0	0.0	0.0	0.0	4.8
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	9.8	0.0	4.8	0.0	2.1	7.1	4.7	1.5	0.0
1/04	8.1	1.3	9.1	1.5	0.4	1.8	1.9	2.6	0.0
3/04	0.0	0.0	5.3	1.3	0.2	2.3	1.0	10.0	0.0
Mean	25.3	0.3	9.0	3.3	0.7	2.8	1.9	3.5	1.2

% Shredders

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	16.7	0.0	0.0	0.0	0.0	25.0	40.0	0.0	86.5
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	60.8	1.7	0.0	0.0	0.5	60.7	10.2	0.0	84.8
1/04	48.6	2.7	0.0	2.5	0.0	57.9	0.8	0.0	72.7
3/04	76.5	6.4	0.0	1.0	0.0	54.7	19.4	0.0	51.1
Mean	16.7	0.0	0.0	0.0	0.0	25.0	40.0	0.0	86.5

% Gatherers

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.0	90.0	50.0	47.4	56.1	8.3	8.6	55.6	1.0
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	2.0	21.6	0.0	16.3	63.1	3.6	7.3	80.0	10.6
1/04	2.7	62.7	11.7	39.9	23.9	7.0	19.1	73.7	24.5
3/04	7.4	66.7	11.7	46.2	15.8	10.5	26.0	60.0	42.0
Mean	0.0	90.0	50.0	47.4	56.1	8.3	8.6	55.6	1.0

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.

% Filterers

Date	Greens	Mill	Indiantown	Walls Landing	Phillips	Taylor	Narrow Channel	Holt	Lakewood
6/03	0.0	0.0	33.3	41.2	43.9	58.3	50.0	0.0	7.7
8/03	NA	NA	NA	NA	NA	NA	NA	NA	NA
10/03	21.6	76.7	85.7	81.4	23.1	25.0	74.5	10.8	1.5
1/04	32.4	30.7	71.6	55.4	73.7	31.6	76.2	2.6	2.7
3/04	11.8	24.4	79.8	51.3	81.6	32.6	52.4	5.0	2.3
Mean[1]	0.0	0.0	33.3	41.2	43.9	58.3	50.0	0.0	7.7

[1] August was not included in mean calculations due to non-baseflow conditions on the sampling date.