# Environmental Factors Structuring Benthic Macroinvertebrate Communities of Agricultural Ditches in Maryland

ALAN W. LESLIE,[1,2] ROBERT F. SMITH,[1] DAVID E. RUPPERT,[3] KRESHNIK BEJLERI,[3] JOSHUA M. MCGRATH,[3] BRIAN A. NEEDELMAN,[3] AND WILLIAM O. LAMP[1]

Environ. Entomol. 41(4): 802–812 (2012); DOI: http://dx.doi.org/10.1603/EN12049

**ABSTRACT** Agricultural drainage ditches are artificial structures used to optimize soil hydrology for crop production and secondarily have been co-opted as a tool to manage the quality of water draining from agricultural lands. We investigated the relationship between the aquatic macroinvertebrate community and environmental variables associated with physical and biogeochemical processes that affect water quality. Aquatic macroinvertebrates were sampled along with physical and chemical measures of the soil and water from 29 agricultural drainage ditches on the Eastern Shore of Maryland. Cluster analysis and multivariate ordination showed that ditches that had higher flow velocities supported communities of lotic invertebrates (i.e., *Stenelmis*, *Prosimulium*) versus those that had properties of linear wetlands, which supported communities of lentic invertebrates (i.e., Oligochaeta, *Caecidotea*). Taxon richness varied from four to 31 taxa per ditch, and was higher within ditches that had higher flow velocities. Small ditches had low diversity, but may have provided refugia from fish predators. Macroinvertebrate communities did not show a significant linear relationship with water quality or with nutrient concentrations within the soil or water. The addition of flow-control structures designed to improve the quality of water draining from agricultural lands may decrease the quality of ditches as habitat for certain aquatic macroinvertebrates. Management decisions for drainage ditches may consider tradeoffs between the benefits of ditches as a source of biodiversity and as a tool for improving water quality.

**KEY WORDS** drainage ditch, macroinvertebrate, nutrients, water quality, biodiversity

Drainage is essential for maintaining productive agricultural areas around the globe. The International Commission on Irrigation and Drainage estimates that globally, 190 million ha of agricultural lands are drained artificially, with most of that land in the Americas (65 Mha), Asia (58 Mha), and Europe (47 Mha) (ICID 2010). In the United States, agricultural drainage networks often comprise subsurface drains that empty into small in-field ditches, which feed into larger collection ditches running between individual properties (Pavelis 1987). Drainage ditches are at the interface between agriculture and aquatic ecosystems, and replace the natural headwaters of regional watersheds.

These engineered waterways provide habitat for species of plants, fish, and macroinvertebrates and contribute to landscape scale biodiversity (Armitage et al. 2003, Davies et al. 2008, Herzon and Helenius 2008). Ditches may have lower diversity than other types of aquatic habitats, but they can provide habitat for species not found in larger, perennial bodies of water (Williams et al. 2003). Drainage ditches may increase taxonomic richness of invertebrates in stream networks by increasing habitat heterogeneity relative to natural stream networks without ditches (Simon and Travis 2011). Studies of invertebrate communities in drainage ditches in fens of England have discovered invertebrate species of conservation value (Painter 1999). Native fish species use intermittently flowing drainage ditches as a refuge during seasonally stressful times and also as sheltered breeding grounds (Colvin et al. 2009).

Modern surface ditches are designed to minimize transport of sediments and chemicals from agricultural land into surface waters. Agriculture represents a widespread nonpoint source of pollution to aquatic ecosystems, but drainage ditches represent a specific location where loads of sediment, nutrients, and pesticides can be targeted for mitigation (Cooper 1993, Skaggs et al. 1994). Biogeochemical and physical processes occurring within drainage ditches are being used as a tool for improving the quality of water leaving agricultural fields and entering local watersheds (Needelman et al. 2007b). Managing the flow regimes of drainage ditches can promote physical processes that prevent erosion of sediments and support oxidation-reduction (redox) reactions that may decrease nutrients draining from agricultural ditches (Needelman et al. 2007a, Strock et al. 2007). Installation of

[1] Department of Entomology, University of Maryland, College Park, MD 20742.
[2] Corresponding author, e-mail: aleslie@umd.edu.
[3] Department of Environmental Science and Technology, University of Maryland, College Park, MD 20742.

structures, such as weirs or flashboard risers, that control discharge from ditches limits the export of nutrients in the form of nitrogen and phosphorus to downstream habitats (Skaggs et al. 1994, Thomas et al. 1995). These flow-control structures also increase residence time of nutrient-rich waters within ditches, which promotes increased rates of denitrification (Penn et al. 2010). The United States Department of Agriculture - Natural Resources Conservation Service (USDA-NRCS), in conjunction with state agencies, provides funding for ditch management aimed at reducing the load of nutrients exported from agricultural fields to streams and rivers (http://www.nrcs.usda.gov/programs/, http://www.mda.state.md.us/pdf/pda.pdf). This has led to widespread installation of flow-control structures through much of the Ohio and Mississippi River valleys and the Chesapeake Bay watershed that reduce nutrient export from agricultural fields, while at the same time increasing yields for farmers (Fouss and Sullivan 2009, Penn et al. 2010).

Management of drainage ditches to increase the quality of water draining from agricultural lands involves manipulation of ditch habitat characteristics that could increase or decrease quality of ditches as habitat for aquatic organisms. Riparian habitat (Moore and Palmer 2005) and benthic habitat (including hydrology) (Verdonschot and Higler 1989, Painter 1999, Davis et al. 2003, Stone et al. 2005, Stephens et al. 2008) are the factors most closely associated with patterns in aquatic invertebrate community composition of agriculturally impacted streams and ditches. Increased sedimentation behind flow-control structures could change the physical structure of substrate encountered by benthic macroinvertebrates. Changing vegetation structure by mowing or dredging within ditches alters substrates and food resources used by macroinvertebrates. Aquatic macroinvertebrate species richness is positively related to dissolved oxygen concentration, and species composition is related to pH in ditches (Verdonschot and Higler 1989, Werner et al. 2010). Controlling drainage to promote microbially mediated redox transformations of nutrients will decrease flow velocities and promote development of anaerobic sediments in ditch habitats.

We sought to determine what communities of invertebrates are present in drainage ditches, and investigated patterns between aquatic macroinvertebrate communities and the physical and chemical parameters that are managed to promote biogeochemical processing of nutrients in agricultural drainage ditches. Our hypothesis is that physical, benthic habitat characteristics and water chemistry parameters determine the composition of aquatic macroinvertebrate communities of drainage ditches. Physical alterations to ditches related to management for drainage and nutrient mitigation will affect the suitability of ditches as habitat for different invertebrate communities. The results will help to determine what invertebrate taxa colonize ditches, and whether management for nutrient removal might conflict with management for a diverse aquatic community within ditches.

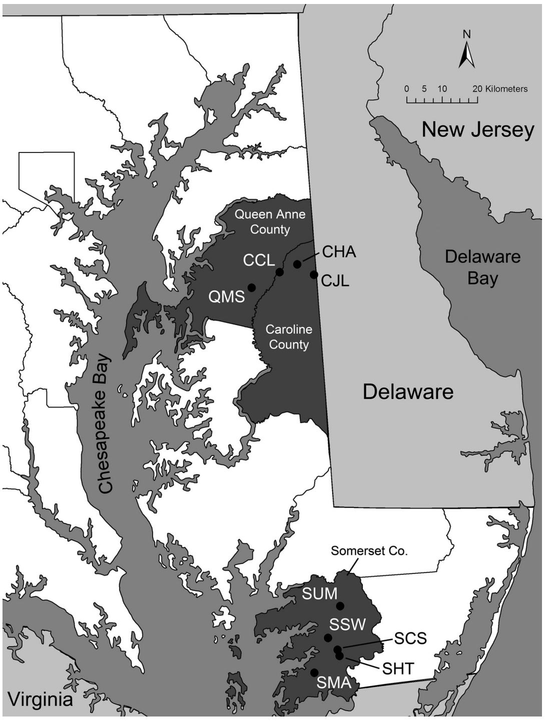


**Fig. 1.** Map of the Eastern Shore of Maryland showing Maryland counties and bordering states. The dark shaded areas indicate the three counties that were sampled. Individual farms are indicated by black circles and are labeled with respective site codes.

## Materials and Methods

**Site Description.** The study was conducted on drainage ditches located on the Eastern Shore of Maryland, which is the portion of Maryland east of the Chesapeake Bay (Fig. 1). Agriculture makes up the highest percentage of land-use of the area (48%), with the majority of that land being used for crops such as corn and soybeans (Denver et al. 2004). The middle and lower regions of the Eastern Shore are within the Coastal Plain physiographic province, and are characterized by flat topography and poorly drained soils. Most fields throughout the middle and lower regions are arable only because of drainage structures that lower the field water table below the zone of crop roots (Bell and Favero 2000, Denver et al. 2004). Most of the farms in this area lie within the watershed of the Chesapeake Bay, which suffers from negative impacts of eutrophication caused in part by nutrient losses from agriculture (Phillips 2007). For this reason, some drainage ditches on the Eastern Shore of Maryland are designed for mitigation of nutrient inputs to receiving waters.

In total, 29 ditches were sampled between 23 February and 15 March 2008. Selected farms were located in three counties, representing the middle (Queen Anne and Caroline Counties) and lower (Somerset County) Eastern Shore. Ditches were chosen to represent a wide range of environmental conditions, including ditch size, flow velocity, substrate type, and presence of flow-control structures. Twenty-three ditches were located within actively farmed fields; 18 ditches were within fields that were either planted with cover crops (winter wheat, *Triticum aestivum* L.) or retained crop residues (corn, *Zea mays* L.); whereas five were adjacent to tilled fields. Of the ditches not within actively farmed fields, three ditches were located within fallow fields (CCL-1, 5, 6); and three ditches were adjacent to a site recently converted from agricultural lands to wetlands (CJL-1, 2, 3).

We collected invertebrate and environmental samples concurrently to reflect each ditch as a single sample unit in our analysis. At each ditch, a 50-m reach was selected just upstream (≈10 m) of either the confluence with its receiving body, or the flow-control structure if present. Invertebrate collection, soil and water sampling, and ditch physical characteristics were all done within this reach.

**Environmental Measurements.** Water chemistry and flow were measured at a single point at the downstream end of the sampling reach. A single 500-ml water sample was taken from each ditch before water was disturbed with other sampling, transported on ice, and then frozen until analyzed for total nitrogen, total phosphorus, and total solids (QuikChem 8500, Lachat Instruments, Loveland, CO). Dissolved oxygen, pH, and specific conductivity were measured on site by using handheld meters (YSI 55/63, YSI Inc., Yellow Springs, OH). Flow velocity was measured at the center of the downstream end of the reach by using a Flow Mate model 2000 portable flowmeter (Marsh McBirney Inc., Frederick, MD).

Ditch physical structure and soil chemistry were measured at three points along the reach by dividing the reach into three equal strata, and choosing a point among 1-m increments within each stratum by using a random numbers table. Ditch geometry was assessed by measuring ditch depth from bankfull height to ditch bottom along transects across the three sampling points. Cross sectional area and maximum depth, as well as measures of the average wetted width and maximum water depth within the ditch were calculated from these measurements. Percentages of vegetative and detrital cover were estimated visually across the wetted width of the ditch, along 1-m length of the ditch. Soil cores were extracted using a metal ring 15.2 cm in diameter and 5 cm in height pushed into the soil. Samples were separated by horizon in the field, transported on ice to the laboratory where they were stored at 4°C until analyzed for bulk density as well as total carbon and nitrogen (Tru Spec, Leco, MI). Measurements of soil pH and redox potential were made at the soil surface and 2.5 cm below the surface by using six platinum-tipped electrodes and a Calomel (Hg/HgCl) reference electrode connected to a handheld voltmeter modified according to Rabenhorst (2009). A correction factor of 251 mV was added to the measured voltages so that reported redox potentials are relative to a standard hydrogen electrode (Vepraskas and Faulkner 2001). We plotted mean redox potentials against soil pH from each ditch along with a technical standard for soil redox potential, which is an empirically determined line that divides conditions where redox reactions are predominantly oxidizing from conditions where redox reactions would be predominantly reducing, adjusted for pH (USDA 2010).

**Macroinvertebrate Collection.** Invertebrates were collected from ten points selected by dividing the sampling reach into 1-m increments and choosing points from a table of random integers 0–50. At each point, two successive 1-m-long sweeps were made with a D-frame net (0.05-$m^2$ opening, 500-$\mu$m mesh) to collect invertebrates within the substrate and water column. The D-frame net was chosen for its ability to capture a diverse assemblage of aquatic invertebrate species (Turner and Trexler 1997). Individual sweeps were combined into a composite sample for the entire ditch, and preserved using 80% ethyl alcohol. In the laboratory, samples were rinsed in stacked 4-mm and 500-$\mu$m sieves to remove coarse plant material while retaining macroinvertebrates and fine debris. The remaining sample material retained by the 500-$\mu$m sieve was subsampled by spreading the material onto a numbered 7 by 7 mesh grid of 4- by 4-cm squares and randomly selecting squares for sorting. Macroinvertebrates were removed from sample material under a dissecting microscope. Subsequent subsamples were taken until a minimum of 300 macroinvertebrates were recovered, or until the entire sample had been sorted. Total abundance of macroinvertebrates within samples was extrapolated based on the amount of sample material sorted.

All insects and crustaceans were identified to genus, except for some Diptera larvae of the suborder Brachycera, and early-instar insect larvae, which were identified to family. Samples containing large numbers of larvae in the family Chironomidae (order Diptera) were further subsampled so that 20% of each morphotype were slide-mounted and identified to genus. Mollusks were identified to the family level, and aquatic Oligochaeta were not identified beyond the subclass level. Merritt et al. (2008) and Covich and Thorp (2001) were used to assign taxa to functional feeding groups (FFG) and groups based on habits of that taxon (e.g., swimming versus burrowing) to determine functional roles of invertebrates within ditch habitats.

**Data Analyses.** Hierarchical cluster analysis was used to determine whether ditch invertebrate communities could be divided into distinct groups based upon differences in invertebrate community composition. A Bray–Curtis distance matrix was calculated from $\log(x + 1)$ taxon counts, and clusters were formed using Ward's agglomerative method. Taxa associated with groups formed by cluster analysis were determined using IndVal scores, which are calculated based on fidelity and relative abundance of taxa within groups created by cluster analysis. The number of groups that resulted in the maximum sum of significant IndVal scores was used as the stopping point for forming groups of sites from cluster analysis (Dufrene and Legendre 1997). Taxon richness, Shannon diversity, Simpson's dominance, total abundance of organisms per sample, and Shannon diversity of FFG were calculated for groups formed by cluster analysis to further describe differences between groups.

Site groupings based on cluster analysis were compared with a plot of sites in multivariate space. Nonmetric multidimensional scaling (NMDS) was used to create a multivariate ordination by using the same distance matrix calculated for cluster analysis. Ordinations were constructed using multiple runs with random starting configurations. All benthic habitat and water quality variables (Table 1) were fitted to the final NMDS ordination to determine patterns between clusters of ditch macroinvertebrate communities and environmental parameters. Environmental variables were checked for colinearity by using linear regressions, and when explanatory variables were found to be collinear, only the variable with the higher $r^2$ value was retained. Significance of the relationship between environmental vectors and the ordination of sites was determined by Monte Carlo permutation ($\alpha = 0.05$). Significant environmental variables then were used in linear regressions with FFG diversity to determine changes in the diversity of feeding habits along environmental gradients.

Statistical analyses all were performed using R v. 2.11.0 (R Foundation for Statistical Computing, Vienna, Austria). Cluster analysis and NMDS were performed using the package vegan v. 1.17–2, and IndVal scores were calculated using the package labdsv v. 1.4–1.

## Results

**Environmental Measurements.** The physical and chemical attributes of the ditches are summarized in Table 1. Ditch size varied from small field drains with a ditch depth from bankfull height of 0.38 m and a maximum cross-sectional area of 0.53 $m^2$ to larger collection ditches with a depth of 1.64 m and maximum cross-sectional area of 6.71 $m^2$. Most ditches contained an abundance of detritus in the form of coarse particulate organic matter within the channel, with greater than half of sites having coverage >90%. Cover by rooted vegetation showed greater variation and ranged between 0 and 100% between ditches. All ditches were periodically dry during the previous summer, but contained water at the time of sampling, with a maximum depth ranging from 0.03 to 0.37 m. Flow velocity ranged from undetectable in stagnant ditches to 0.20 m $s^{-1}$ in ditches more typical of channelized streams. All ditch water was acidic, with a mean pH of 5.4 (range, 4.6–6.4). Mean specific conductivity was 217.4 $\mu$S $cm^{-1}$ (range, 34.5–448.1 $\mu$S $cm^{-1}$) and total solids had a mean of 2.5 g $L^{-1}$ (range, 0.03–11.6 g $L^{-1}$). Dissolved oxygen also varied between ditches, but most ditches had relatively high DO, with 12 of 29 ditches being near or even beyond saturation.

Six ditches had redox states below the technical standard (anaerobic) for both depths, three ditches were below the technical standard at a single depth, and 20 were above (aerobic) at the time of measurement (Fig. 2). Ditch soils often exhibit aerobic conditions at the soil-water interface and anaerobic conditions at depth (Needelman et al. 2007a). The predominance of aerobic conditions at the 0 and 2.5 cm depths at these sites indicate that we likely took our measurements within the soil zone influenced by the soil-water interface.

Mean total nitrogen concentration of ditch water was 6.1 mg $L^{-1}$ (range, 0.81–27.7 mg $L^{-1}$) and average total phosphorus of ditch water was 0.48 mg $L^{-1}$ (range, 0.08–3.09 mg $L^{-1}$). In comparison, the average nutrient concentrations of surface waters of the area in early spring are 3.5 mg $L^{-1}$ for total nitrogen and 0.1 mg $L^{-1}$ for total phosphorus (Denver et al. 2004). The mean total carbon concentration of the upper 5 cm of soil was 67 g C $kg^{-1}$ soil (range, 2.6–206 g C $kg^{-1}$ soil), while mean total soil nitrogen was 4.6 g N $kg^{-1}$ soil (range, 0.29–13.8 g N $kg^{-1}$ soil).

**Macroinvertebrate Community.** In total, 9,081 individual organisms were identified from subsamples of all benthic invertebrate samples, representing 85 invertebrate taxa (Table 2). Six of the seven Odonata, Ephemeroptera, Plecoptera, and Trichoptera taxa identified to family were represented by a single individual, or multiple individuals recovered from a single ditch, and therefore the use of family level identifications should not have underestimated overall taxa richness for these groups compared with those taxa identified to genus. Abundance of organisms per sample was estimated by extrapolation of the proportion that was subsampled, and varied from 58 to 10,780 individuals per sample. Taxon richness varied from four to 31 taxa with a mean of 13.9 taxa identified per ditch. Insects represented the majority of the invertebrate taxa, with 75 taxa from seven different orders. Diptera was the most diverse insect order, with 38 different taxa recovered from the ditches. Noninsect taxa had a greater total abundance than insects, representing three quarters of all individuals recovered. Among the noninsect taxa, isopods of the genus *Caecidotea* and aquatic oligochaete worms were the most numerous and represented the dominant taxon in 21 ditches. The majority (80.7%) of individuals collected across all ditches were detritivores (collector-gatherer feeding group). Other feeding groups represented were predators (7.6%), filter feeders (5.9%), scrapers (3.7%), shredders (2.1%), and herbivores (0.1%). The dominant habits of ditch invertebrates were sprawling on the sediment surface (54.0%) and burrowing within sediments (38.5%). Other habits represented included swimming within the water column (5.2%), clinging to hard surfaces (2.2%), and climbing emergent vegetation (0.2%).

Four groups (I–IV) resulted in the greatest sum of IndVal scores, and were chosen to represent distinct invertebrate communities from cluster analysis (Fig. 3). Significant indicator taxa are listed below their respected groups in Fig. 3. Group I showed the greatest difference among all the groups constructed from cluster analysis and had the greatest abundance of the isopod *Caecidotea.* Sites in group I also show the lowest

**Table 1. Benthic habitat and chemical measurements of 29 sampled ditches**

| Ditch[a] | Dissol. oxygen | Spec. cond. | Water pH | Total solids | Water Total N | Water Total P | Redox 0 cm | Soil pH 0 cm | Redox 2.5 cm | Soil pH 2.5 cm | Soil Total C | Soil Total N | Bulk density | Mean plant cover | Mean detritus cover | Flow velocity | Mean cross section | Mean ditch depth | Mean water depth | Flow-control structure |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | % sat. | $\mu S\ cm^{-1}$ | | $g\ L^{-1}$ | $mg\ L^{-1}$ | $mg\ L^{-1}$ | mV | | mV | | g C $kg^{-1}$ soil | g N $kg^{-1}$ soil | $g\ cm^{-3}$ | % | % | $m\ s^{-1}$ | $m^2$ | m | m | Y/N |
| CCL-1 | 87 | 165 | 5.7 | 0.26 | 10.4 | 0.26 | 467 | 6.2 | 370 | 5.5 | 206 | 13.8 | 0.17 | 17 | 100 | 0 | 3.1 | 1 | 0.08 | Y |
| CCL-2 | 142 | 266 | 6.2 | 0.05 | 13.4 | 0.47 | 499 | 5.5 | 439 | 6 | 77 | 4.9 | 0.45 | 60 | 100 | 0.02 | 4.7 | 0.9 | 0.17 | N |
| CCL-3 | 34 | 213 | 5.5 | 0.1 | 1.73 | 0.26 | 499 | 5.5 | 529 | 4.7 | 74 | 5.2 | 0.16 | 23 | 100 | 0.02 | 4.9 | 1.6 | 0.28 | Y |
| CCL-4 | 8 | 249 | 5.9 | 0.12 | 10.3 | 0.08 | 230 | 5.4 | 232 | 5.3 | 71 | 4.7 | 0.2 | 2 | 67 | 0 | 3.2 | 1.3 | 0.37 | Y |
| CCL-5 | 88 | 234 | 4.8 | 0.12 | 10.3 | 0.1 | 483 | 5.4 | 475 | 5.7 | 14 | 1 | 0.48 | 0 | 100 | 0.02 | 4.6 | 1.3 | 0.09 | Y |
| CCL-6 | 74 | 136 | 5.5 | 0.06 | 1.12 | 0.18 | 31 | 6.1 | 48 | 6.3 | 4 | 0.3 | 1.12 | 1 | 100 | 0 | 5.2 | 1.6 | 0.11 | Y |
| CHA-1 | 57 | 115 | 4.6 | 0.07 | 5.62 | 0.16 | 378 | 5.4 | 400 | 5.4 | 49 | 3.4 | 0.52 | 40 | 100 | 0.03 | 1.8 | 0.7 | 0.13 | N |
| CHA-2 | 48 | 129 | 5.3 | 0.06 | 5.43 | 0.48 | 378 | 5.3 | 265 | 5.6 | 162 | 11.9 | 0.17 | 17 | 100 | 0.02 | 3.7 | 1 | 0.28 | N |
| CJL-1 | 44 | 49 | 4.7 | 0.11 | 7.62 | 0.75 | 455 | 5.1 | 293 | 5.9 | 185 | 11.1 | 0.16 | 15 | 100 | 0 | 3.3 | 0.9 | 0.18 | Y |
| CJL-2 | 101 | 170 | 5.6 | 0.18 | 9.37 | 1.61 | 336 | 5.9 | 194 | 5 | 198 | 11.4 | 0.28 | 100 | 63 | 0.03 | 6.7 | 1.6 | 0.29 | N |
| CJL-3 | 58 | 75 | 5.3 | 0.03 | 0.81 | 0.08 | 271 | 5.3 | 129 | 5.1 | 164 | 9.1 | 0.34 | 5 | 80 | 0 | 2.2 | 0.8 | 0.17 | N |
| QMS-1 | 148 | 163 | 5.5 | 0.08 | 9.05 | 0.13 | 533 | 6.2 | 398 | 4.7 | 51 | 3.9 | 0.43 | 59 | 100 | 0.05 | 4.8 | 1.6 | 0.11 | Y |
| QMS-2 | 108 | 207 | 6.4 | 0.1 | 3.52 | 0.1 | 620 | 5.7 | 652 | 5.6 | 4 | 0.4 | 1.62 | 2 | 38 | 0.2 | 3.4 | 1.1 | 0.21 | N |
| QMS-3 | 121 | 234 | 5.6 | 0.31 | 6.9 | 0.1 | 567 | 5.5 | 481 | 5.2 | 3 | 0.3 | 1.9 | 13 | 35 | 0.06 | 2.5 | 0.9 | 0.09 | N |
| SCS-1 | 110 | 254 | 5.3 | 0.16 | 1.75 | 0.26 | 130 | 5.3 | 139 | 4.9 | 30 | 2.3 | 0.7 | 33 | 95 | 0.02 | 0.5 | 0.4 | 0.08 | Y |
| SCS-2 | 29 | 243 | 4.9 | 0.16 | 0.97 | 0.12 | 49 | 6 | 198 | 5.1 | 30 | 2.7 | 1.08 | 50 | 100 | 0.02 | 0.9 | 0.6 | 0.19 | N |
| SCS-3 | 50 | 35 | 5.4 | 0.2 | 10.3 | 0.76 | 367 | 5.6 | 328 | 5.3 | 72 | 5.1 | 0.48 | 33 | 100 | 0 | 0.8 | 0.4 | 0.06 | N |
| SHT-1 | 44 | 229 | 5.5 | 0.15 | 5.55 | 0.08 | 293 | 5.7 | 227 | 5.4 | 23 | 1.8 | 0.85 | 5 | 67 | 0 | 0.7 | 0.4 | 0.08 | Y |
| SHT-2 | 106 | 309 | 4.7 | 1.92 | 8.34 | 3.09 | 531 | 5 | 445 | 5.3 | 29 | 2.2 | 0.46 | 37 | 97 | 0.03 | 1.6 | 0.8 | 0.06 | N |
| SHT-3 | 114 | 342 | 5.8 | 0.21 | 4.94 | 0.4 | 266 | 5.9 | 291 | 5.7 | 99 | 7.5 | 0.39 | 27 | 100 | 0.03 | 1.5 | 0.6 | 0.14 | N |
| SMA-1 | 94 | 304 | 5.5 | 11.36 | 2.66 | 0.69 | 321 | 5.6 | 337 | 5.3 | 42 | 3.4 | 0.56 | 10 | 93 | 0 | 1.4 | 0.7 | 0.22 | Y |
| SMA-2 | 34 | 208 | 5.2 | 11.6 | 1.5 | 0.23 | 397 | 5.5 | 367 | 5.2 | 45 | 3.2 | 0.79 | 62 | 48 | 0 | 1.6 | 0.8 | 0.04 | N |
| SMA-3 | 71 | 268 | 5.7 | 10.67 | 1.79 | 0.23 | 439 | 5.8 | 413 | 5.4 | 21 | 1.5 | 0.8 | 83 | 77 | 0 | 1.5 | 0.7 | 0.03 | N |
| SSW-1 | 16 | 156 | 5.4 | 0.53 | 4.68 | 0.84 | 519 | 5.2 | 450 | 5.2 | 25 | 2.1 | 0.97 | 22 | 83 | 0 | 1.6 | 0.7 | 0.05 | Y |
| SSW-2 | 60 | 143 | 5.7 | 0.27 | 2.75 | 0.77 | 240 | 5.1 | 289 | 5.5 | 33 | 2.8 | 0.68 | 0 | 80 | 0 | 1.2 | 0.6 | 0.11 | N |
| SSW-3 | 100 | 167 | 5.6 | 0.1 | 1.08 | 0.19 | 182 | 4.3 | 196 | 5.5 | 27 | 2.3 | 0.81 | 5 | 83 | 0 | 2 | 0.9 | 0.22 | N |
| SUM-1 | 138 | 369 | 5.5 | 11.46 | 4.66 | 0.09 | 514 | 4.8 | 587 | 5.1 | 28 | 2.1 | 1.36 | 33 | 40 | 0.02 | 2.3 | 1 | 0.13 | Y |
| SUM-2 | 70 | 448 | 5.1 | 10.85 | 3.91 | 0.79 | 423 | 5.6 | 348 | 5.8 | 132 | 8.9 | 0.39 | 5 | 87 | 0.02 | 0.7 | 0.5 | 0.08 | Y |
| SUM-3 | 99 | 424 | 5.6 | 11.22 | 27.7 | 0.87 | 483 | 5.8 | 371 | 5.2 | 52 | 3.9 | 0.57 | 63 | 87 | 0 | 4.9 | 1.4 | 0.14 | Y |
| Mean (SD) | 78 (38) | 217 (101) | 5.4 (0.4) | 2.5 (4.5) | 6.1 (5.5) | 0.48 (0.62) | 376 (153) | 5.5 (0.4) | 341 (140) | 5.4 (0.4) | 67 (61) | 4.6 (3.8) | 0.7 (0.4) | 28 (27) | 83 (21) | 0.02 (0.01) | 2.7 (1.7) | 0.9 (0.4) | 0.14 (0.1) | N/A |

[a] The first letter of ditch names refers to the county in which the ditch was located (C, Caroline; Q, Queen Anne's; S, Somerset). The second two letters refer to specific farm sites, and the number refers to the individual ditch sampled at that site.

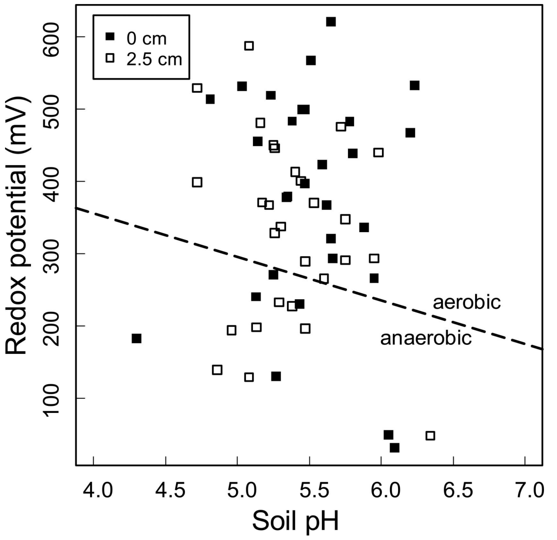


**Fig. 2.** Plot of average redox potential measures and soil pH measured at each ditch. Dark squares are measures taken at the soil surface and open squares are measurements from a depth of 2.5 cm. The line (Redox = 595–60 × pH) represents the division between soils where iron would be oxidized (points above the line) and reduced (points below the line) as determined by National Technical Committee for Hydric Soils.

values for taxon richness and values for Shannon diversity and Simpson's dominance (Table 3). Sites in group II were distinguished by an abundance of larval Dolichopodidae and aquatic oligochaete worms. This group also contains five of the six ditches that are not within actively farmed fields (two within fallow fields and three within wetland restoration site). The two sites in group III contained the most indicator taxa, including the beetle *Stenelmis* (Coleoptera: Elmidae), five dipteran genera and snails in the family Ancylidae. Sites in group III also had the highest values of taxon richness (Table 3). Sites in group IV had the highest abundance of the caddisfly *Ironoquia* (Trichoptera: Limnephilidae), along with two dipteran taxa.

**Community–Environment Relationships.** The NMDS ordination produced a stable solution in two dimensions after five runs with a final stress of 17.2 (Fig. 4). Overlaying cluster identities onto the NMDS biplot shows that groups II, III, and IV are arranged along axis 2, whereas there is incomplete separation of group I from groups II and IV along axis one (Fig. 4). Six environmental variables showed a significant relationship with the NMDS ordination: flow velocity ($r^2 = 0.58, P = 0.001$); cross sectional area ($r^2 = 0.56, P = 0.001$); ditch depth ($r^2 = 0.52, P = 0.001$); redox potential at the sediment surface ($r^2 = 0.29, P = 0.012$) and at 2.5 cm ($r^2 = 0.26, P = 0.023$); and percent saturation of dissolved oxygen ($r^2 = 0.22, P = 0.035$) (Table 4). Flow velocity was linearly related to both redox measures and percent saturation of dissolved oxygen, and cross sectional area was linearly related to ditch depth. Flow velocity and cross sectional area were the two environmental variables with the highest $r^2$ values and were added to the NMDS ordination (Fig. 4). The vector for cross sectional area was correlated with the gradient separating group I from groups II, and IV along axis 1. The vector for flow velocity was correlated with the gradient separating groups II, III, and IV along axis 2. Linear regressions then were fit between the two environmental variables (flow velocity and cross sectional area), and FFG diversity. FFG diversity showed a significant positive linear relationship with flow velocity ($r^2 = 0.426, P < 0.001$) and cross sectional area ($r^2 = 0.336, P = 0.001$).

Flow control structures were present in 14 of the 29 ditches sampled (Table 1). Presence of flow control structures did not explain any patterns between invertebrate communities of groups of ditches formed by cluster analysis. Group I comprised five ditches with and four ditches without flow control structures. Group II comprised four ditches with and six without flow control structures. Group IV comprised four ditches with and four without flow control structures. Group III comprised two ditches, neither of which had flow control structures.

## Discussion

Our two main objectives were to investigate patterns of invertebrate community composition among agricultural ditch types and to determine if community composition was related to physical and chemical characteristics of ditches managed to promote improved water quality. We found that the agricultural ditches did not contain homogenous invertebrate communities, and the invertebrate community composition differed between ditch groupings. Although invertebrate community composition was related to certain physical and chemical variables managed for water quality, basic physical properties of ditches (flow and size) were primarily responsible for differences in community composition.

The composition of indicator taxa within groups II, IV, and III suggests that ditches varied from forms that are characteristic of long wetlands (temporary lentic habitats) to channelized streams (permanent lotic habitats) (Verdonschot and Higler 1989). Ditches within group II primarily contain taxa that are semiaquatic, which suggests these sites may have dried recently. Ditches in group IV contain larvae of the caddisfly genus *Ironoquia*, which are strictly aquatic, but are adapted to develop within small, temporary pools and streams (Flint 1958). Ditches in group III have indicator taxa that include *Stenelmis* (Coleoptera: Elmidae) and *Prosimulium* (Diptera: Simuliidae), which are adapted to lotic, erosional habitats (Merritt et al. 2008). Fitting the vector for flow velocity onto the NMDS ordination shows that a gradient in flow velocity underlies the pattern in invertebrate composition. Flow velocity increases from little to no flow in group II to the highest flow in group III.

Sites in group III had greater taxon richness than groups II and IV. This suggests that ditches with flow

**Table 2. Macroinvertebrate taxa sampled from ditches**

| Taxon | | | | | Frequency | Total |
|---|---|---|---|---|---|---|
| Phylum | Class | Order | Family | Genus | (No. of ditches) | (No. of individuals) |
| Arthropoda | Hexapoda | Odonata | Gomphidae | *Dromogomphus* | 1 | 2 |
| | | | Libellulidae | *Libellula* | 1 | 1 |
| | | | Coenagrionidae | – | 2 | 4 |
| | | | | *Amphiagrion* | 1 | 1 |
| | | | | *Argia* | 2 | 2 |
| | | Ephemeroptera | Baetidae | – | 1 | 3 |
| | | | Ephemerellidae | – | 1 | 14 |
| | | | Caenidae | *Caenis* | 1 | 3 |
| | | | Heptageniidae | – | 1 | 1 |
| | | Plecoptera | Leuctridae | *Leuctra* | 1 | 1 |
| | | | Taeniopterigidae | *Taeniopteryx* | 1 | 4 |
| | | | Chloroperlidae | – | 1 | 3 |
| | | Trichoptera | Limnephilidae | *Ironoquia* | 11 | 110 |
| | | | | *Limnephilus* | 5 | 7 |
| | | | Polycentropodidae | – | 1 | 1 |
| | | | Hydropsychidae | *Hydropsyche* | 1 | 2 |
| | | | | *Cheumatopsyche* | 1 | 2 |
| | | Megaloptera | Corydalidae | *Chauliodes* | 1 | 1 |
| | | Coleoptera | Haliplidae | *Peltodytes* | 2 | 2 |
| | | | Elmidae | *Dubiraphia* | 1 | 7 |
| | | | | *Microcylloepus* | 1 | 1 |
| | | | | *Stenelmis* | 2 | 19 |
| | | | Scirtidae | *Cyphon* | 1 | 1 |
| | | | Hydrophilidae | *Hydrochus* | 2 | 3 |
| | | | | *Tropisternus* | 2 | 2 |
| | | | | *Berosus* | 2 | 2 |
| | | | | *Paracymus* | 3 | 4 |
| | | | | *Enochrus* | 1 | 2 |
| | | | Dytiscidae | *Copelatus* | 3 | 6 |
| | | | | *Hydroporus* | 3 | 3 |
| | | | | *Neoporus* | 10 | 43 |
| | | | | *Uvarus* | 1 | 1 |
| | | | | *Hydrovatus* | 1 | 1 |
| | | | | *Agabus* | 15 | 61 |
| | | | Noteridae | *Hydrocanthus* | 1 | 4 |
| | | | | *Suphisellus* | 1 | 1 |
| | | Diptera | Chironomidae | *Thienemannimyia* | 3 | 82 |
| | | | | *Larsia* | 3 | 15 |
| | | | | *Zavrelimyia* | 2 | 15 |
| | | | | *Limnophyes* | 14 | 281 |
| | | | | *Tvetenia* | 11 | 237 |
| | | | | *Smittia* | 10 | 159 |
| | | | | *Paraphenocladius* | 5 | 49 |
| | | | | *Psilometriocnemus* | 1 | 54 |
| | | | | *Cricotopus/Orthocladius* | 13 | 272 |
| | | | | *Diplocladius* | 4 | 31 |
| | | | | *Eukiferellia* | 2 | 11 |
| | | | | *Zalutschia* | 1 | 5 |
| | | | | *Tanytarsus* | 2 | 10 |
| | | | | *Rheotanytarsus* | 1 | 5 |
| | | | | *Polypedilum* | 11 | 299 |
| | | | | *Tribelos* | 1 | 10 |
| | | | | *Endotribelos* | 1 | 5 |
| | | | | *Dicrotendipes* | 1 | 5 |
| | | | | *Chironomus* | 1 | 10 |
| | | | Culicidae | *Aedes* | 1 | 10 |
| | | | Chaoboridae | *Mochlonyx* | 2 | 6 |
| | | | Tipulidae | *Tipula* | 16 | 65 |
| | | | | *Ormosia* | 2 | 5 |
| | | | Ceratopogonidae | *Forcipomyia* | 1 | 1 |
| | | | | *Probezzia* | 4 | 21 |
| | | | | *Bezzia/Palpomyia* | 5 | 14 |
| | | | | *Culicoides* | 4 | 38 |
| | | | | *Ceratopogon* | 3 | 8 |
| | | | | *Alluaudomyia* | 2 | 4 |
| | | | | *Dasyhelea* | 1 | 1 |
| | | | Psychodidae | *Pericoma* | 3 | 6 |
| | | | | *Psychoda* | 1 | 1 |
| | | | Simuliidae | *Prosimulium* | 2 | 148 |
| | | | Stratiomyidae | *Odontomyia* | 1 | 2 |
| | | | Tabanidae | *Tabanus* | 4 | 10 |

Continued on following page

**Table 2. Continued**

| Taxon | | | | | Frequency | Total |
|---|---|---|---|---|---|---|
| Phylum | Class | Order | Family | Genus | (No. of ditches) | (No. of individuals) |
| | | | Dolichopodidae | – | 13 | 23 |
| | | | Empididae | – | 12 | 25 |
| | | | Phoridae | – | 4 | 20 |
| | | | Muscidae | – | 9 | 19 |
| | Crustacea | Isopoda | Asellidae | *Caecidotea* | 23 | 3492 |
| | | Amphipoda | Crangonyctidae | *Crangonyx* | 19 | 405 |
| | | Decapoda | Cambaridae | – | 2 | 9 |
| | Arachnida | Acari | – | – | 16 | 314 |
| Annelida | Oligochaeta | – | – | – | 28 | 1838 |
| Mollusca | Gastropoda | Pulmonata | Planorbidae | – | 14 | 166 |
| | | | Physidae | – | 13 | 88 |
| | | | Lymnaeidae | – | 12 | 75 |
| | | | Ancylidae | – | 2 | 2 |
| | Bivalvia | Veneroida | Sphaeriidae | – | 13 | 384 |

characteristics of streams may harbor significantly higher diversity than ditches that have no or reduced flow. Sites with greater flow velocities also tended to have greater FFG diversity, which suggests that ditches with flowing water contain niches not present in stagnant ditches. Although flow-control structures alter hydrology of ditches and decrease flow rates to promote lower redox potentials, we found that flow control structures were distributed among the sites in all taxon groups except group III, which had the highest values for flow velocity. In practice, slowing drainage from a channelized stream to reduce nutrient export could result in the replacement of a community of lotic invertebrates with a lentic community, and subsequently could alter the FFG diversity (McDowell and Naiman 1986). Groups I, II, and IV all were characterized by species adapted to lentic environments, but we did not find that flow control structures were particularly associated with any one group. Interactions between the effects of flow control struc-

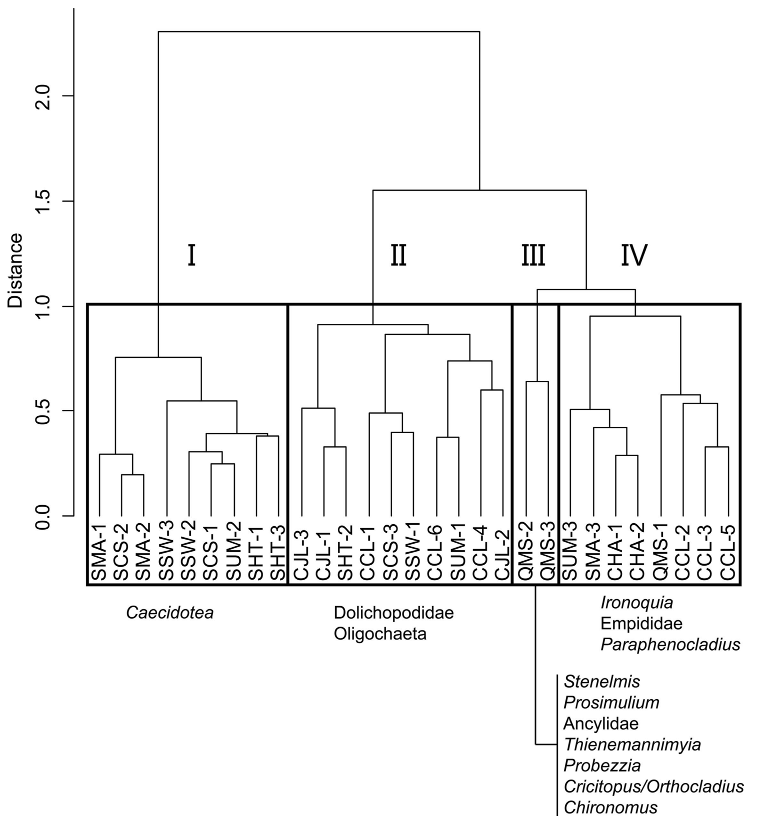


**Fig. 3.** Dendrogram showing clusters formed using Ward's method and a Bray–Curtis distance matrix calculated from $\log(x + 1)$ taxon counts. Boxes are drawn around distinct clusters, with indicator taxa listed below each cluster.

**Table 3. Community metrics across groups formed by cluster analysis**

| Community metric | Group mean ± SD | | | |
|---|---|---|---|---|
| | I | II | III | IV |
| Taxon richness | 7.0 ± 2.0 | 15.2 ± 3.6 | 28.0 ± 2.8 | 16.6 ± 6.6 |
| Shannon diversity | 0.65 ± 0.27 | 1.74 ± 0.20 | 2.46 ± 0.54 | 1.60 ± 0.62 |
| Organisms/sample | 4284 ± 2760 | 1141 ± 1287 | 4855 ± 6266 | 4227 ± 3122 |
| Simpson's dominance | 0.32 ± 0.16 | 0.73 ± 0.06 | 0.85 ± 0.09 | 0.63 ± 0.21 |
| % Detritivore | 95.47 ± 4.26 | 82.08 ± 12.20 | 55.18 ± 7.51 | 80.32 ± 17.68 |
| % Predator | 0.52 ± 0.76 | 8.67 ± 9.66 | 18.25 ± 14.24 | 4.41 ± 4.10 |
| % Scraper | 0.00 ± 0.00 | 0.00 ± 0.00 | 0.44 ± 0.13 | 0.00 ± 0.00 |
| % Filterer | 0.03 ± 0.10 | 0.32 ± 0.79 | 24.60 ± 21.93 | 10.87 ± 18.97 |
| % Shredder | 0.32 ± 0.57 | 0.56 ± 0.85 | 0.70 ± 0.50 | 4.03 ± 5.83 |
| % Herbivore | 0.04 ± 0.11 | 0.10 ± 0.24 | 0.00 ± 0.00 | 0.00 ± 0.00 |

tures and other chemical and physical habitat characteristics may have had unique effects on community composition that did not result in one homogenous community across all sites with flow control structures.

Sites in group I had on average a smaller cross sectional area than other ditches, and also supported fewer taxa than other groups. The only indicator taxon for this group was the aquatic isopod genus *Caecidotea. Caecidotea* was present in 23 of 29 ditches but had the highest proportional abundances in ditches in group I resulting in low values for the Simpson's dominance metric (Table 3). These crustaceans are strictly aquatic, and may be exploiting small ditches as a temporary refuge from predators present in perennial waters (Covich and Thorp 2001). Ditches in group I also had the lowest mean proportional abundance of predatory taxa of all groups (Table 3). These small primary drains do not support a large number of species, but may play an important role for species seeking refuge from fish predators in perennial waters (Colvin et al. 2009).

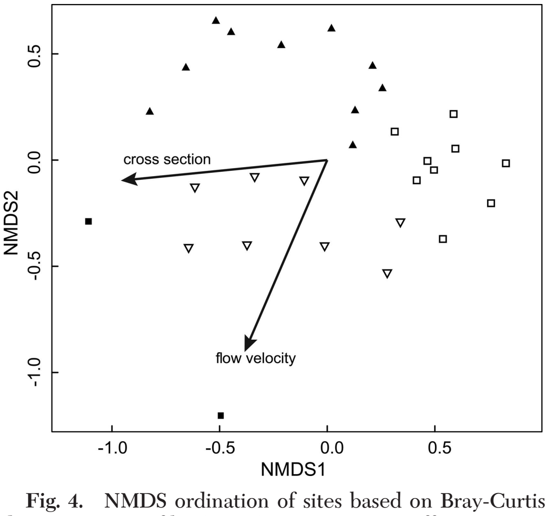


**Fig. 4.** NMDS ordination of sites based on Bray-Curtis distance matrix of $\log(x + 1)$ taxon counts. Different points represent individual ditches belonging to the same group based on cluster analysis (open square, group I; closed triangle, group II; closed square, group III; open triangle, group IV). Arrows indicate direction of change in environmental variables fitted to ordination space.

Soil redox potential and dissolved oxygen concentration were the only chemical variables with a significant relationship to invertebrate community structure. Surface soil redox potential may increase as a function of the availability of dissolved oxygen and decrease as a function of the availability of labile organic matter to respiring microbes at this interface (Vepraskas and Faulkner 2001). Therefore, the relationship between redox potential and invertebrate community composition may be mediated by availability of oxygen and organic matter. Most of the invertebrate taxa collected live on or within the benthic substrate (sprawlers and burrowers) and belong to functional feeding groups that process detrital organic matter (collector/gatherer and shredder). As we sampled exclusively with a D-frame net, we may have underestimated the relative abundance of burrowing organisms. Future studies should explore the role that organic matter processing and bioturbation by this community of invertebrates has on the redox chemistry responsible for nutrient transformations within ditch soils. Finding that none of the other water chemistry variables significantly explained patterns of invertebrate community composition is likely the result of the abundance of taxa adapted to stressful

**Table 4. Summary of fitting environmental vectors to NMDS plot**

| Environmental variable | $r^2$ | *P* |
|---|---|---|
| Dissol. oxygen | 0.220 | 0.040* |
| Spec. cond. | 0.004 | 0.958 |
| Water pH | 0.142 | 0.148 |
| Total solids | 0.071 | 0.403 |
| Water total N | 0.146 | 0.134 |
| Water total P | 0.022 | 0.799 |
| Redox 0 cm | 0.294 | 0.010* |
| Soil pH 0 cm | 0.035 | 0.627 |
| Redox 2.5 cm | 0.262 | 0.021* |
| Soil pH 2.5 cm | 0.003 | 0.963 |
| Soil total C | 0.110 | 0.234 |
| Soil total N | 0.086 | 0.318 |
| Bulk density | 0.073 | 0.379 |
| Plant cover | 0.009 | 0.906 |
| Detritus cover | 0.110 | 0.203 |
| Flow velocity | 0.584 | 0.001* |
| Cross section | 0.562 | 0.001* |
| Ditch depth | 0.520 | 0.001* |
| Max. water depth | 0.087 | 0.316 |
| Flow-control structure | 0.070 | 0.126 |

*$P < 0.05$.

environments found in ditches. The lack of a significant relationship between patterns of invertebrate community composition and nutrient concentrations in either soil or water also suggests that benthic invertebrate communities may not differ between ditches exporting high amounts of nutrients to local watersheds versus those that do not. Therefore, any decrease in nutrient loads will not directly impact any single group of invertebrates, but physical alterations to flow, area, and oxygen concentration related to all aspects of ditch management may alter invertebrate communities.

These results suggest that alterations to water flow in ditches might alter the suitability of ditches as habitat for specific assemblages of aquatic invertebrate species adapted to lotic environments. Although not the focus of our study, this result suggests that flow control devices that decrease flow may lower diversity, or alter community composition. We did not find evidence that sites with flow control structures contained a specific level of taxa richness or type of community. If sites in group III, however, were to have flow control devices installed and their habitats changed from stream-like to more like a wetland, we would expect a drastic change in community composition and richness. As stated, we cannot rule out the possibility that flow control structures interacted with other local features of the ditch such as ground water level, area, slope, detritus, soil composition, nutrients, in-channel vegetation, precipitation, or temperature. The effects of flow control structures also may not have manifested themselves in the time period for which sampling was done for this project. Flow control structures may allow water to remain in the ditch longer and delay drying. This could lead to aquatic communities being present in ditches with flow control structures for longer than those without, and communities at these later times may differ than the community of initial colonizers (Welborn et al. 1996, Brooks 2000).

Decisions regarding management of in-stream processes of ditches to improve the quality of water draining agricultural lands may impact habitat quality within ditches. Pollution mitigation strategies that are not implemented in the channel such as grass buffer filter strips adjacent to ditches (Cooper et al. 2004); below-ground biocurtains (Strock et al. 2007); or phosphorus-sorbing soil amendments (Penn et al. 2007, Leader et al. 2008) may be a way to effectively manage ditches for both habitat quality and nutrient pollution. Our results show that flow, ditch size, and chemical measures related to oxygen consumption were the only factors that explained community composition. These characteristics are likely unaffected by activities outside the stream channel. Alterations to flow and ditch area will likely have the greatest impact on community composition and richness, although the greatest impacts are likely to occur when changing stream-like ditches to ditches more characteristics of wetlands. Finding differences in community composition between ditches in this study indicated that maximizing regional diversity of aquatic invertebrates within ditch habitats may depend on maintaining a diversity of physical characteristics of ditches across the agricultural landscape.

## Acknowledgments

We thank Jason Keppler, John Rhoderick, Arthur Allen, Owen McDonough, and Megan Lang for their help in locating sites, and Ymene Fouli for help collecting and processing samples. We also thank landowners for allowing access to ditches on their farms and property. Funding was provided in part by the Maryland Agricultural Experiment Station Internal Competitive Grant Program and by the Biotechnology Risk Assessment Program Competitive Grant 2009-40002-05821 from the USDA National Institute of Food and Agriculture. We would also like to thank our reviewers for their help in preparing this manuscript.

## References Cited

**Armitage, P. D., K. Szoszkiewicz, J. H. Blackburn, and I. Nesbitt. 2003.** Ditch communities: a major contributor to floodplain biodiversity. Aquat. Conserv. 13: 165–185.

**Bell, W. H., and P. Favero. 2000.** Moving water: a report to the Chesapeake Bay Cabinet by the Public Drainage Task Force, Maryland Department of Natural Resources.

**Brooks, R. T. 2000.** Annual and seasonal variation and the effects of hydroperiod on benthic macroinvertebrates of seasonal forest ("vernal") ponds in central Massachusetts, USA. Wetlands 20: 707–715.

**Colvin, R., G. R. Giannico, J. Li, K. L. Boyer, and W. J. Gerth. 2009.** Fish use of intermittent watercourses draining agricultural lands in the Upper Willamette River Valley, Oregon. Trans. Am. Fish. Soc. 138: 1302–1313.

**Cooper, C. M. 1993.** Biological effects of agriculturally derived surface water pollutants on aquatic systems–a review. J. Environ. Qual. 22: 402–408.

**Cooper, C. M., M. T. Moore, E. R. Bennett, S. Smith, J. L. Farris, C. D. Milam, and F. D. Shields. 2004.** Innovative uses of vegetated drainage ditches for reducing agricultural runoff. Water Sci. Technol. 49: 117–123.

**Covich, A. P., and J. H. Thorp. 2001.** Introduction to the subphylum *Crustacea*, pp. 777–810. *In* J. H. Thorp and A. P. Covich (eds.), Ecology and classification of North American freshwater invertebrates, 2nd ed. Academic, New York.

**Davis, S., S. W. Golladay, G. Vellidis, and C. M. Pringle. 2003.** Macroinvertebrate biomonitoring in intermittent coastal plain streams impacted by animal agriculture. J. Environ. Qual. 32: 1036–1043.

**Davies, B., J. Biggs, P. Williams, M. Whitfield, P. Nicolet, D. Sear, S. Bray, and S. Maund. 2008.** Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agric. Ecosyst. Environ. 125: 1–8.

**Denver, J. M., S. W. Ator, L. M. Debrewer, M. J. Ferrari, J. R. Barbaro, T. C. Hancock, M. J. Brayton, and M. R. Nardi. 2004.** Water quality in the Delmarva Peninsula, Delaware, Maryland, and Virginia, 1999–2001, U.S. Geological Survey Circular, Reston, VA.

**Dufrene, M., and P. Legendre. 1997.** Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67: 345–366.

**Flint, O. 1958.** The larva and terrestrial pupa of Ironoquia parvula (Trichoptera, Limnephilidae). J.N.Y. Entomol. Soc. 66: 59–62.

**Fouss, J. L., and M. Sullivan. 2009.** Agricultural drainage management systems task force (ADMSTF). pp. 4068–

4077. *In* S. Starrett (ed.), Proceedings, World Environmental and Water Resources Congress, 17–21 May 2009, Kansas City, MO. ASCE, Reston, VA.

**Herzon, I., and J. Helenius. 2008.** Agricultural drainage ditches, their biological importance and functioning. Biol. Conserv. 141: 1171–1183.

**ICID. 2010.** Important data of ICID member countries. International Commission of Irrigation and Drainage, New Delhi, India.

**Leader, J. W., E. J. Dunne, and K. R. Reddy. 2008.** Phosphorus sorbing materials: sorption dynamics and physicochemical characteristics. J. Environ. Qual. 37: 174–181.

**McDowell, D. M., and R. J. Naiman. 1986.** Structure and function of a benthic invertebrate stream community as influenced by beaver (*Castor canadensis*). Oecologia 68: 481–489.

**Merritt, R. W., K. W. Cummins, and M. B. Berg (eds.). 2008.** An introduction to the aquatic insects of North America. Kendall/Hunt Publishing Company, Dubuque, Iowa.

**Moore, A. A., and M. A. Palmer. 2005.** Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. Ecol. Appl. 15: 1169–1177.

**Needelman, B. A., D. E. Ruppert, and R. E. Vaughan. 2007a.** The role of soil formation and redox biogeochemistry in mitigating nutrient and pollutant losses from agriculture. J. Soil Water Conserv. 62: 207–215.

**Needelman, B. A., P.J.A. Kleinman, J. S. Strock, and A. L. Allen. 2007b.** Improved management of agricultural drainage ditches for water quality protection: an overview. J. Soil Water Conserv. 62: 171–178.

**Painter, D. 1999.** Macroinvertebrate distributions and the conservation value of aquatic Coleoptera, Mollusca and Odonata in the ditches of traditionally managed and grazing fen at Wicken Fen, UK. J. Appl. Ecol. 36: 33–48.

**Pavelis, G. A. (ed.). 1987.** Farm drainage in the United States. History, status, and prospects. Economic Research Service, U.S. Department of Agriculture.

**Penn, C. J., R. B. Bryant, P.J.A. Kleinman, and A. L. Allen. 2007.** Removing dissolved phosphorus from drainage ditch water with phosphorus sorbing materials. J. Soil Water Conserv. 62: 269–276.

**Penn, C. J., J. M. McGrath, and R. B. Bryant. 2010.** Ditch drainage management for water quality improvement: ditch drainage treatment structures. *In* M. T. Moore and R. Kröger (eds.), Agricultural drainage ditches: mitigation wetlands for the 21st century. Research Signpost, Kerala, India.

**Phillips, S. W. (ed.). 2007.** Synthesis of U.S. Geological Survey science for the Chesapeake Bay ecosystem and implications for environmental management. U.S. Geological Survey Circular.

**Rabenhorst, M. C. 2009.** Making oxidation-reduction potential measurements using multimeters. Soil Sci. Soc. Am. J. 73: 2198–2201.

**Simon, T. N., and J. Travis. 2011.** The contribution of man-made ditches to the regional stream biodiversity of the new river watershed in the Florida panhandle. Hydrobiologia 661: 163–177.

**Skaggs, R. W., M. A. Brev, and J. W. Gilliam. 1994.** Hydrologic and water quality impacts of agricultural drainage. Crit. Rev. Environ. Sci. Technol. 24: 1–32.

**Stephens, W. W., M. T. Moore, J. L. Farris, J. L. Bouldin, and C. M. Cooper. 2008.** Considerations for assessments of wadable drainage systems in the agriculturally dominated deltas of Arkansas and Mississippi. Arch. Environ. Contam. Toxicol. 55: 432–441.

**Stone, M. L., M. R. Whiles, J. A. Webber, K.W.J. Williard, and J. D. Reeve. 2005.** Macroinvertebrate communities in agriculturally impacted Southern Illinois streams: patterns with riparian vegetation, water quality, and instream habitat quality. J. Environ. Qual. 34: 907–917.

**Strock, J. S., C. J. Dell, and J. P. Schmidt. 2007.** Managing natural processes in drainage ditches for nonpoint source nitrogen control. J. Soil Water Conserv. 64: 188–196.

**Thomas, D. L., C. D. Perry, R. O. Evans, F. T. Izuno, K. C. Stone, and J. W. Gilliam. 1995.** Agricultural drainage effects on water quality in the Southeastern U.S. J. Irrig. Drain. E.-ASCE. 121: 277–282.

**Turner, A. M., and J. C. Trexler. 1997.** Sampling aquatic invertebrates from marshes: evaluating the options. J. N. Am. Benthol. Soc. 16: 694–709.

**USDA-NRCS. 2010.** Field indicators of hydric soils in the United States. *In* L. M. Vasilas, G. W. Hurt, and C. V. Noble (eds.), U.S. Dep. Agric., NRCS, in cooperation with the National Technical Committee for Hydric Soils.

**Vepraskas, M. J., and S. P. Faulkner. 2001.** Redox chemistry of hydric soils, pp. 85–105. *In* J. L. Richardson and M. J. Vepraskas (eds.), Wetland soils: genesis, hydrology, landscapes, and classification. Lewis Publishers, Washington, DC.

**Verdonschot, P.F.M., and L.W.G. Higler. 1989.** Macroinvertebrates in Dutch ditches: a typological characterization and the status of the Demmerik ditches. Hydrobiol. Bull. 23: 135–142.

**Welborn, G. A., D. K. Skelly, and E. E. Werner. 1996.** Mechanisms creating community structure across a freshwater habitat gradient. Annu. Rev. Ecol. Syst. 27: 337–363.

**Werner, I., D. A. Markiewicz, K. Goding, and K. Reece. 2010.** Benthic macroinvertebrate communities in ephemeral agricultural drainage ditches of California's Central Valley, pp. 1–15. *In* M. T. Moore and R. Kroger (eds.), Agricultural drainage ditches: mitigation wetlands for the 21st century. Research Signpost, Kerala, India.

**Williams, P., M. Whitfield, J. Biggs, S. Bray, G. Fox, P. Nicolet, and D. Sear. 2003.** Comparative biodiversity of rivers, streams, ditches, and ponds in an agricultural landscape in Southern England. Biol. Conserv. 115: 329–341.

*Received 15 February 2012; accepted 26 April 2012.*