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INVERTEBRATE BIODIVERSITY IN AGRICULTURAL AND URBAN HEADWATER STREAMS: IMPLICATIONS FOR CONSERVATION AND MANAGEMENT

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Abstract. The urbanization of agricultural lands is currently one of the dominant patterns of land use change in developed countries. In the United States and parts of Europe, this has led to the implementation of agricultural land preservation programs and riparian protection and replanting efforts along urban streams. The ecological benefits of such programs for the conservation of freshwater biodiversity have yet to be fully explored. We designed a study to investigate the patterns of stream macroinvertebrate community structure along a gradient of agriculture to urban development, and the patterns among urban streams that vary in the amount of intact riparian buffer. In 2001 and 2002, we sampled the 29 small headwater streams comprising the outlying tributaries of four watersheds just north of Washington, D.C., in Montgomery County, Maryland, USA. This region has had dramatic urban development over the last 50 years, yet significant efforts have been made to maintain riparian buffers and promote preservation of agricultural land.

Macroinvertebrate richness was strongly related to land use, with agricultural streams exhibiting the highest macroinvertebrate diversity. Taxa richness was related negatively and linearly (no statistical threshold) to the amount of impervious surface cover. For the urban streams, there was a strong positive relationship between invertebrate diversity and riparian forest cover. Urban streams with high amounts of intact riparian forest exhibited biodiversity levels more comparable to less urban areas despite high amounts of impervious cover in their catchments. The agricultural headwater streams in this study were not only more diverse than the urban headwaters, but their levels of macroinvertebrate diversity were high compared to other published estimates for agricultural streams. These higher richness values may be due to widespread use of ''best management practices'' (BMPs), including no-till farming and the implementation of woody and herbaceous riparian buffers, which may alleviate many acute stressors caused by cultivation. These findings suggest that, if managed properly, the preservation of agricultural land from development may help conserve stream invertebrate biodiversity, and that maintenance of riparian forests even in highly urbanized watersheds may help alleviate ecological disturbances that might otherwise limit macroinvertebrate survival.

Key words: best management practices; development; impervious cover; land use; macroinvertebrates; richness; riparian; urbanization.

INTRODUCTION

The ecological consequences of land use change for the health of rivers and streams is a prominent environmental issue worldwide. Agricultural practices such as livestock grazing and tilling on land adjacent to streams can lead to soil erosion and subsequent runoff of fine sediments, nutrients, and pesticides (e.g., Schulz and Liess 1999, Cuffney et al. 2000, Kang et al. 2001). Urbanization leads to enhanced runoff, channel erosion, and reduced water quality due to inputs of metals, oils, and road salts (Hammer 1972, Booth and Jackson 1997, Paul and Meyer 2001). While changes in stream ecosystem processes may result from such land use changes (Young and Huryn 1999, Gessner and Chauvet 2002), impacts on macroinvertebrate assemblages have been the most extensively studied ecological response (Paul and Meyer 2001). Decreases in diversity and overall abundance (e.g., Whiting and Clifford 1983, Lenat and Crawford 1994), increases in the relative abundance of pollution tolerant taxa (e.g., Hall et al. 2001, Walsh et al. 2001), and changes in the distribution of invertebrate feeding groups (e.g., Lamberti and Berg 1995) have all been associated with human-dominated land use.

In many developed countries, the urbanization of agricultural lands that long ago replaced forests is a major land use concern (Jacobs 1999, USDA 2000). For example, in the mid-Atlantic region of the United States, the amount of forested land in many watersheds is significantly less than the combined area of agriculture and urban development, with the latter currently increasing at the expense of farmlands (Moglen 2000).

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In such areas, agricultural acreage can be viewed as the primary form of remaining ''undeveloped'' land. Agricultural land preservation programs have been implemented to prevent urban sprawl and promote conservation of these agricultural lands in regions throughout the United States and Europe (Alterman 1997, Daniels and Bowers 1997, Jacobs 1999). For example, the states of Pennsylvania, Maryland, and Delaware have had agricultural easement programs for more than a decade, resulting in the preservation of >214 575 ha of farmland (*data available online*).[2]

While many agricultural preservation programs tout their conservation value, ecological research has historically shown that farming practices are detrimental to stream health (e.g., Rothrock et al. 1998, Genito et al. 2002). Only recently have researchers suggested that agricultural lands may support diverse and compositionally different aquatic invertebrate communities when compared to nearby urbanized areas (Lenat and Crawford 1994, Wang et al. 2000, Stepenuck et al. 2002). Demonstrating predictable (and positive) changes in biodiversity along a gradient of urban development to agriculture would support the continued growth of these agricultural preservation programs. Furthermore, an understanding of why these changes occur could bolster conservation and management priorities in areas experiencing urbanization pressure.

Two factors, the amount of impervious surface and of riparian forest cover, are often the focal point of discussions on the link between land use change and stream ecosystem health (e.g., Schueler 1994, Weigel et al. 1999, Stewart et al. 2001). These two variables influence stream hydrology and water quality (Brabec et al. 2002). Furthermore, impervious cover has been shown to be correlated with the diversity of macroinvertebrates (Schueler 1994), and the removal or clearcutting of riparian trees in forested watersheds has been shown to have a strong influence on entire stream invertebrate communities (Wallace et al. 1997). However, whether or not forested buffers can mitigate the impacts of urbanization on stream invertebrates in highly developed watersheds remains an open question.

In portions of the Chesapeake Bay drainage area, agricultural preservation programs are quite active, and there have been aggressive efforts to restore or protect riparian vegetation, even in some highly urbanized watersheds just north of Washington, D.C. We designed a study to take advantage of these regional features by investigating patterns of stream invertebrate community structure along a gradient of agriculture to urban development, and among urban streams that vary in the amount of intact riparian buffer. Here we report on the study of 29 headwater streams showing that invertebrate diversity was extremely high in agricultural headwaters and dramatically declined as urbanization increased; however, the decline in diversity was less among urban streams if the urban streams had intact riparian forest buffers.

[2] ⟨www.agriculture.state.pa.us/agriculture/lib/agriculture/farmlandfiles/2002-2003annualreport.pdf⟩, ⟨www.malpf.info/facts.html⟩, ⟨www.state.de.us/deptagri/aglands/lndpres.shtml⟩

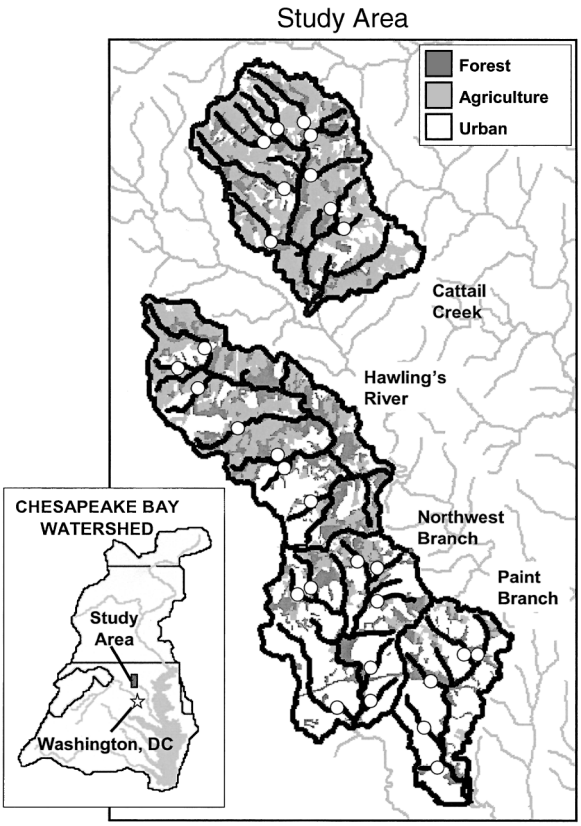

FIG. 1. Location of the four watersheds containing the 29 macroinvertebrate sampling sites (circles) examined in this study. The map shows the position of the general study region within the Chesapeake Bay drainage basin, as well as the general patterns of agriculture, forest, and development in the four study watersheds.

METHODS

Study sites and land use

This study took place in the Piedmont region of Maryland, USA, on the northern outskirts of the Washington, D.C. metropolitan area (Fig. 1). Study streams were located within four watersheds (29–68 km^2) that ultimately drain into the Chesapeake Bay and have historically been dominated by agriculture. Over the last 50 years, the two watersheds closest to Washington, Northwest Branch and Paint Branch, have experienced dramatic development (presently, 53% and 64%, respectively) and a corresponding loss of agricultural land as a result of urban sprawl. The development in these watersheds is a combination of low-density (0.08–0.8 dwellings/ha) and medium-density (0.8–3.2 dwellings/ha) residential development with some high-density and commercial/industrial development. In

TABLE 1. Specific land use attributes of the four groups of subwatersheds draining to each of the 29 headwater streams studied.

Attribute	Land use group			
	Agriculture	Mixed-agriculture	Mixed-urban	Urban
Subwatersheds (no.)	11	6	6	6
Residential development (%)	15.7	19.4	52.6	83.2
Commercial and industrial (%)	0.4	0.8	1.2	5.0
Open developed and institutional (%)	1.9	19.0	5.9	8.2
Crop cover (%)	51.3	22.0	7.2	0.0
Pasture (%)	8.8	6.7	4.0	0.1
Forest (deciduous) (%)	20.4	30.5	26.9	6.8
Impervious surface cover (%)	5.1	11.5	17.6	33.9
Riparian forest (%)	39.3	52.7	55.1	18.5

Notes: Land use values represent the mean percentages of each site in the group. All percentages indicate the mean land use within the catchment area (subwatershed) of each site, except for riparian forest, which represents the percentage of forest within 30 m of the stream channel.

contrast, the watershed of Cattail Creek is still dominated by agricultural land use (56%), while the Hawling's River watershed is a mix of agriculture and residential development (36% and 25%, respectively). Historical and current land use (year 2000) information for these watersheds was obtained from Maryland Department of Planning GIS (geographic information system) coverages. These coverages are available in the ArcView supplement program (GISHydro 2000) and use level II Anderson classifications with 30-m resolution. This is high quality land use data that is based on high altitude aerial photography and satellite imagery, and was refined using digitized parcel-level data (Irwin and Bockstael 2002) from the Maryland Division of Taxation and Assessment database.

We examined the 29 small headwater streams (2.7–9.2 km^2 catchment areas) comprising the outlying tributaries of these four watersheds (Fig. 1). This allowed us to minimize the confounding effects of heterogeneous land use that is typical of larger catchments, as well as the potentially confounding effects that stream size can have on taxa richness. The catchment areas for each of the sites (hereafter, subwatersheds) were delineated using GIS-based digital elevation maps. Land use percentages were then calculated for each subwatershed, and riparian forest cover was determined by examining land use within 30 m of the stream (Appendix A).

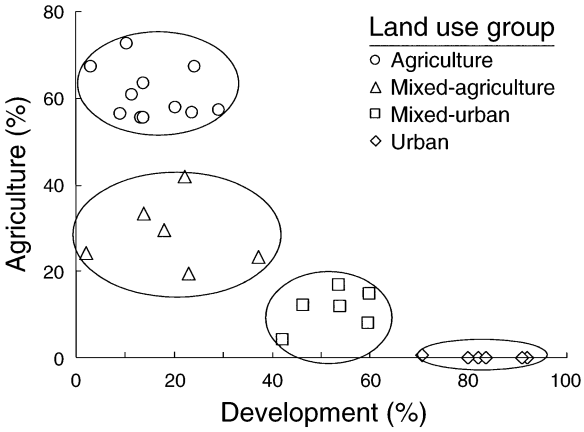

FIG. 2. Results of a cluster analysis performed on headwater stream sites (n = 29), based on the percentages of forest, agriculture, and development land use within each subwatershed. Clusters (ellipses surrounding individual sites) are plotted against percentages of agriculture and development to illustrate their distinct separation along these two variables.

The 29 streams were divided into distinct categorical land use groups based on the overall percentages of agriculture (% crop + % pasture), development (% residential + % industrial/commercial), and forest (primarily deciduous) in their subwatersheds using cluster analysis with Ward's minimum variance method (Proc Cluster, SAS 2001). This analysis revealed four distinct underlying land use groups (Table 1; Fig. 2). The first group of sites was represented by high percentages of agriculture (predominantly corn, soybean, and winter wheat crop cultivation) and relatively low amounts of urbanization. The ''mixed-agriculture'' cluster had moderate amounts of agriculture (predominantly crops), low percentages of development, and a comparatively high percentage of ''open developed and institutional land'' (a category dominated by permeable developed lands such as golf courses, athletic fields, and one small landfill). ''Mixed-urban'' sites had moderately high amounts of development, with little agriculture. The ''urban'' sites were dominated by residential and some commercial development, had comparatively large amounts of impervious surface cover, and no agriculture. All land use groups except for urban had similar amounts of forest, and because of proactive riparian conservation strategies within the state, all groups had a higher amount of riparian forest (percentage of land within 30 m of the stream that is in forest) than subwatershed forest (percentage of entire subwatershed that is in forest).

Macroinvertebrate communities

Macroinvertebrate communities were sampled from 15 March–15 April in both 2001 and 2002 using a 0.04-m^2 Surber sampler (0.25-mm mesh). Three riffles were sampled in each stream by selecting alternating riffles beginning with the second riffle upstream of the tributary confluence, and two samples were randomly collected within each of these three riffles (total sampling area was 0.24 m^2). The six individual samples were pooled together, and one-third of this composite material was subsampled and preserved using a 10% formalin solution. Invertebrates were removed from detritus and sediment debris, identified to the lowest possible taxonomic level (100× magnification), and assigned functional feeding group (FFG) designations using Merritt and Cummins (1996) and Thorp and Covich (2001). Most insect taxa were identified to the genus level, while most non-insect taxa were identified to class or order. Fifty organisms from the family Chironomidae from each 2001 sample were slide-mounted and identified to genus (400× magnification) using Merritt and Cummins (1996) and Epler (2001) to obtain estimated values of richness and density for these taxa.

Total taxa richness, density, and Shannon-Wiener diversity and evenness were determined for both years at all study sites. Richness of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT; a conventional group of indicator taxa) was also determined across both years. FFG richness required genus level identification of Chironomidae, and was calculated for 2001 only.

Data analysis

Strong intercorrelations between the percentages of development, forest, and agriculture prevented the use of traditional multiple regression models for examining the effects of land use on biological responses. By splitting the 29 streams into well-defined land use groups using a multivariate cluster analysis, we were able to eliminate this multicollinearity and simultaneously consider the effects of these three land use variables.

Differences in total community diversity, richness, evenness, and density between the land use groups were examined across both 2001 and 2002 using repeated-measures ANOVAs (Proc Mixed, SAS 2001). Differences in EPT richness between land use groups were similarly tested across sampling years using a repeated-measures ANOVA, while differences in FFG richness were tested in 2001 using simple one-way ANOVAs. Tukey's test for pairwise comparisons was used to test for differences between individual land use groups when a significant overall main effect of land use was found.

We used simple one-way ANOVAs with Tukeys pairwise comparisons to test for differences in the amount of impervious surface between the land use groups; we similarly tested for differences in the amount of riparian forest. A moderate but significant correlation between impervious surface and riparian forest cover ($r^2 = 0.25$, $P = 0.005$) prevented us from including these variables together in multiple regression models. Instead, simple linear regression models were used to examine the strength of the relationships between these variables and macroinvertebrate richness across all streams (Proc Reg, SAS 2001). We also performed separate linear regressions to investigate the relationship between riparian forest cover and stream invertebrate richness within each of the four land use groups. This analysis allowed us to examine the effects of riparian forest cover on taxa richness while removing intercorrelations between riparian cover and subwatershed land use and/or impervious surface cover.

To compile estimates of macroinvertebrate richness (total and EPT) in other agricultural systems to compare to our findings, we used the ISI Web of Science literature database to objectively search for articles from the time period 1990–2003 containing the keywords ''agriculture'' or ''agricultural'' and ''macroinvertebrate'' (*data available online*).[3] This search returned 88 articles; all those reporting on empirical studies of macroinvertebrate richness in agriculturally impacted streams were examined, and richness values were estimated from graphical, tabular, or text values (Appendix B).

Results

There was no significant effect of land use on taxa evenness or invertebrate density (Table 2; $P > 0.05$), although density was significantly higher in 2002 ($P = 0.0002$). There was a significant main effect of land use on the diversity and richness of macroinvertebrate communities (Table 2; $P < 0.0001$), with urban sites having lower richness and diversity than all other land use groups across both sampling years (Fig. 3; richness only is plotted for brevity, though patterns for diversity were nearly identical). There was no effect of sampling year, but there was a significant interaction between year and land use group (Table 2; $P < 0.05$). This was simply due to a change in the magnitude of differences in richness and diversity values between land use groups during the 2002 sampling season, rather than a difference in the way the land use groups were ordered.

Very similar community patterns were found when examining differences in richness between specific macroinvertebrate groups. Mean EPT richness varied between land use groups (Table 2; $P < 0.0001$), with agricultural sites having significantly more taxa than urban sites across both sampling years (Fig. 3; $P < 0.0001$). There was no difference in EPT richness between sampling years, nor was there an interaction between year and land use group ($P > 0.05$). There was a significant effect of land use on the richness of all functional feeding groups except shredders (Table 2). Collector, filterer, predator, and scraper richness were

[3] ⟨http://isi17.isiknowledge.com/portal/.cgi/wos⟩

TABLE 2. Results of ANOVA mixed models to determine if macroinvertebrate variables differ significantly between year and land use (agricultural, $n = 11$; mixed-agricultural, $n = 6$; mixed-urban, $n = 6$; urban, $n = 6$) at 29 headwater streams.

	Land use		Year		Land use × year	
Variable	*F*	*P*	*F*	*P*	*F*	*P*
Total density	0.45	0.72	**19.00**	0.0002	0.27	0.84
Total evenness	2.68	0.07	0.04	0.85	0.63	0.6
Total diversity	**10.74**	0.0001	0.63	0.44	**3.19**	0.04
Total richness	**18.51**	<0.0001	0.62	0.44	**3.99**	0.02
EPT richness†	**13.85**	<0.0001	0.07	0.8	2.67	0.07
Collector richness	**8.18**	0.0006				
Filterer richness	**9.73**	0.0002				
Predator richness	**8.34**	0.0005				
Scraper richness	**19.44**	<0.0001				
Shredder richness	1.97	0.14				

Notes: Models using all taxa and EPT taxa were tested using repeated-measures ANOVA across both the 2001 and 2002 sampling seasons, with chironomid taxa identified to the family level. Functional feeding group models include genus-level chironomid identification and used the 2001 data only. For land use main effects, df = 3, 25; for main effects of year, df = 1, 23; for year × land use interaction terms, df = 3, 23. Bold *F* values are significant at $P < 0.05$.

† Indicator taxa: Ephemeroptera, Plecoptera, and Trichoptera.

all significantly higher in the agricultural streams compared to the urban sites (Fig. 3). The general pattern from all ANOVA models was that the mixed-agriculture and mixed-urban sites were consistently intermediate between the agriculture and urban groups.

There was a highly significant difference in impervious surface cover between land use groups ($P < 0.0001$). Urban sites had higher imperviousness than all other land use groups, while agriculture and mixed-agriculture sites had significantly lower imperviousness than either of the urban groups. We also found a subsequently strong negative linear relationship between impervious surface cover and richness in these headwater streams (Fig. 4; $r^2 = 0.70$, $P < 0.0001$). It has been previously suggested that thresholds may exist in this relationship at ~10–15% imperviousness (Schueler 1994, Stepenuck et al. 2002); however, we found that a quadratic model was no better at explaining the relationship between richness and impervious surface cover ($r^2 = 0.68$) than a simple linear model.

Riparian forest cover was significantly lower at the developed sites ($P = 0.0001$), but all other land use groups had similar amounts of forested buffer. The subsequent linear relationship between richness and riparian forest across all sites was weak but significant (Fig. 5; $r^2 = 0.18$, $P = 0.02$) because the overall model was strongly influenced by the urban data points. When regression models were run separately for each of the land use groups, we found that riparian forest cover explained 78% of the variation in richness at the most urban sites (Fig. 5; $P = 0.02$), and a bit less at the mixed-urban sites ($r^2 = 0.52$ with a $P = 0.10$ due to the small N). There was no significant relationship within the agriculture ($r^2 = 0.05$, $P = 0.52$) or the mixed-agriculture ($r^2 = 0.09$, $P = 0.56$) groups.

The literature survey produced 31 journal articles that allowed us to estimate total and/or EPT richness values in agricultural streams from many areas around the world (Appendix B). For illustration, richness estimates from these studies were grouped by geographical region and level of taxonomic resolution, and compared to richness values from agricultural streams in the current study (Fig. 6). It is evident that the mean richness values from this study are much higher than the mean richness values of all the other published studies combined (Fig. 6). These differences were not statistically tested because the number of sites used to estimate richness varied dramatically among studies, as did the precision of those estimates.

DISCUSSION

Two of our findings have significant implications for watershed management and restoration. First, the fact that invertebrate biodiversity was extremely high in the agricultural headwater streams and progressively declined along a land use gradient toward urbanization suggests that agricultural preservation programs in our region may be important to conservation of freshwater biodiversity. Invertebrate diversity and richness at our agricultural sites was almost twice that of the urban sites, with mixed land use sites falling consistently between these groups (Fig. 3). This decreased diversity could have important ecosystem-wide consequences since invertebrates can influence rates of primary production, decomposition, and resource acquisition (e.g., Lamberti et al. 1987, Jonsson and Malmqvist 2000, Cardinale and Palmer 2002). We suggest below that the very high levels of invertebrate diversity we found in these Piedmont agricultural headwaters compared to other agricultural streams worldwide may be related to the management strategies of agricultural operations surrounding these streams.

The second implication arises from our finding that urban streams with the greatest amount of intact ri-

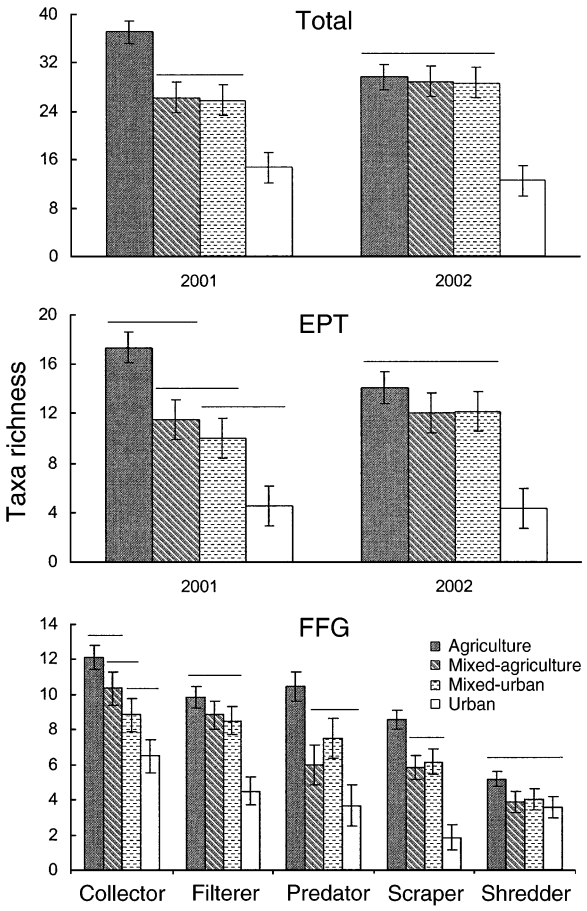

FIG. 3. Invertebrate total, EPT (Ephemeroptera, Plecoptera, Trichoptera), and FFG (functional feeding group) richness (mean ± SE) for the four land use groups into which the 29 headwater streams were clustered. Mean results for total and EPT richness are shown for both the 2001 and 2002 sampling seasons. Taxa richness refers to the number of taxa within each of the specified groups. A significant main effect of land use was found for all richness variables (all $P < 0.001$) except shredder richness. Bars connected by a short horizontal line indicate no significant differences between land use groups (Tukey's $P > 0.05$).

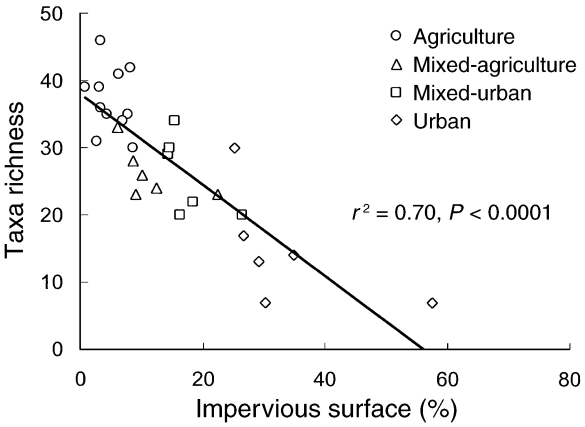

FIG. 4. Relationship between total macroinvertebrate richness and the percentage of impervious surface cover in each subwatershed in 2001, based on simple linear regression models ($n = 29$). Similar patterns were found in 2002, and across other invertebrate groups (i.e., EPT and FFGs).

parian forest buffer had higher levels of biodiversity than other urban streams we studied (Fig. 5). This suggests that efforts to restore or preserve riparian buffers, even when there is a substantial amount of paved surface in urban watersheds, may mitigate some of the impacts on stream biodiversity. Thus, from a biodiversity perspective, headwater streams in areas already highly urbanized should not be viewed as lost causes; a balance between conservation, restoration, and ecologically designed solutions to the problems caused by urbanization may be warranted (Palmer et al. 2005).

We intentionally use the words management and restoration *implications* in describing the significance of our findings because they are based on an analysis of patterns (e.g., correlations between land use and biodiversity), not experiments to identify underlying causation. Furthermore, the extent to which these results can be extrapolated to new areas (e.g., beyond the Piedmont mid-Atlantic) awaits confirmation. Many large cities like Washington, D.C., exhibit strong gradients in urbanization that decrease with distance from the

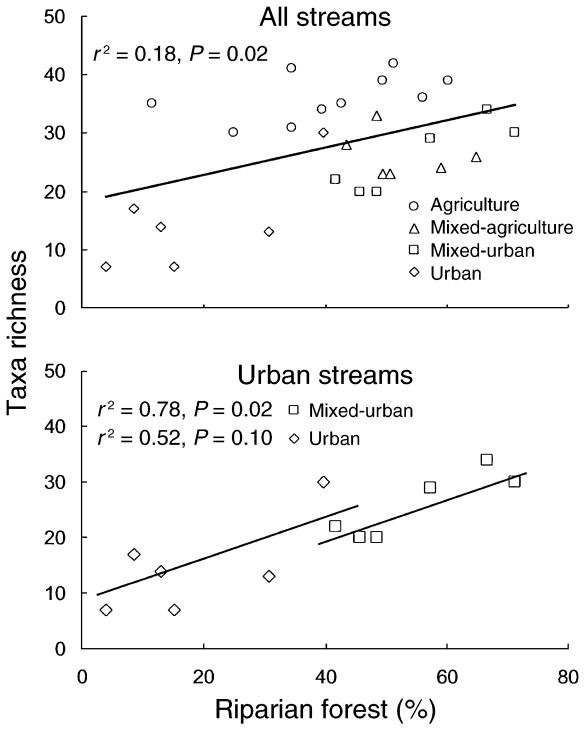

FIG. 5. Linear regression models showing the relationship between total taxa richness and riparian forest cover in 2001. The top panel illustrates the effect of riparian forest across all 29 headwater streams. The bottom panel demonstrates this relationship within the two land use groups characterized by urban ($n = 6$; imperviousness = 25–58%) and mixed-urban ($n = 6$; imperviousness = 14–27%) sites.

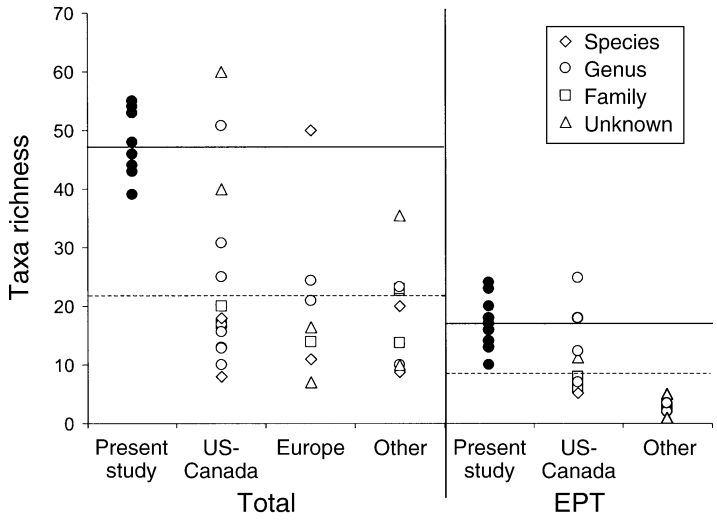

FIG. 6. Comparison of total and EPT richness values in the current study (filled symbols) with richness values from published studies (open symbols) in other agricultural systems throughout the world (Table 2). Symbol shapes indicate the level of taxonomic resolution used in each study. ''Other'' regions include Africa, Asia, Australia, and South America for total richness, and include these continents plus Europe for EPT richness. Horizontal lines indicate mean richness values for the current study (solid line) and all other studies (dashed line).

city center. Therefore, the nature of urban sprawl in our study watersheds is such that most of the agricultural headwater streams are in the two study watersheds furthest from Washington, D.C. (i.e., Hawling's and Cattail; Fig. 1). Thus, spatial autocorrelation between watershed position and biological responses could be responsible for diversity patterns. We believe this unlikely however, because latitudinal difference among sites is miniscule (watersheds are only miles apart), and urban headwaters within the Hawling's watershed follow the same diversity pattern as the more southern urban headwaters in the Northwest Branch and Paint Branch watersheds.

Research to identify the underlying causes of the patterns we have documented remains critical, and there is an extensive literature that can be used to direct such research. Diversity patterns are known to be influenced by ecological factors, as well as large-scale evolutionary and climatic processes (Ricklefs 2004). Primary productivity, habitat heterogeneity, and local environmental disturbances have all been used to explain patterns of species diversity (Huston 1994), including lotic species richness (Voelz and McArthur 2000). We explored the relationship between streambed habitat heterogeneity (a measure commonly assumed to influence stream invertebrate diversity; Downes et al. 1998, Cardinale et al. 2002) and found no evidence that macroinvertebrate richness at our study sites was linked to heterogeneity. We also found no significant relationship between diversity and primary productivity at these sites, estimated using both the accrual of periphyton on artificial substrates and whole-stream metabolism (A. A. Moore and M. A. Palmer, *unpublished data*).

A more likely explanation for decreased diversity at the urban sites is the inability of populations to recover from mortality caused by frequent and intense disturbances (Huston 1994). Impervious surface cover is specifically known to lead to extreme disturbances in stream ecosystems, including increased flood flows (Booth and Jackson 1997, Paul and Meyer 2001). Our urban subwatersheds have more impervious surface cover than the agricultural subwatersheds, and there was a strong negative relationship between invertebrate richness and imperviousness (Fig. 4). Wang et al. (2000) and Stepenuck et al. (2002) examined stream communities across a similar land use gradient in Wisconsin, and likewise found that macroinvertebrate and fish diversity, respectively, decreased with the amount of impervious surface cover. The possibility that flow strongly influenced the patterns we observed is bolstered by recent hydrological work in the Northwest Branch watershed (including eight of our urban study sites) showing that flood discharges have increased by 30% since pre-urbanization levels (Beighley and Moglen 2003). We presume that inputs of chemical pollutants have also dramatically increased since pre-urbanization times (Brabec et al. 2002).

The strong positive relationships between riparian forest and taxa richness for our urban streams (Fig. 5) suggests that the presence of intact riparian zones may mitigate some of the impacts of urbanization on biodiversity loss. Urban development, with subsequent impacts on streams, is only expected to increase globally, and statistical trends in the United States show that this form of land use change is effectively irreversible (Irwin and Bockstael 2005). Thus, the maintenance of riparian buffers in urban areas may become increasingly important. Surprisingly, we did not find a relationship between invertebrate richness and riparian forest in agricultural areas. Given prior research suggesting the importance of riparian buffers in agricultural areas (Watzin and McIntosh 1999, Stewart et al. 2001) and the high level of biodiversity we found at our agricultural sites, we probed the nature of agricultural land use at our sites more deeply. We contacted local county soil conservation district officials and gathered detailed information for agricultural parcels

in our watersheds. We found that of 37 farms being used primarily for crop cultivation, 65% are currently using best management practices (BMPs) involving grassed waterways or vegetative filter strips. Both of these conservation practices involve the use of herbaceous vegetation along streams and drainage areas. Because herbaceous buffers do not show up in the GIS land use database, we had not considered them in our original analyses. There is evidence that herbaceous buffers may be very important for decreasing erosion and sedimentation (Trimble 1997, Lyons et al. 2000); however, in general the role of herbaceous vegetation to stream invertebrate communities has received little attention.

We also found that several other land management practices intended to limit disturbances from cultivation and livestock production were commonly used on these farms: contour farming, nutrient management, manure storage, and rotational grazing. No-till farming, where the previous year's crop residue is left on the ground surface before and during planting operations, was also quite common. Combined with herbaceous buffers, these practices likely help protect headwater streams from much of the sediment and chemical pollution associated with agricultural operations (Watzin and McIntosh 1999) and provide shading and structural heterogeneity to these small streams. It is extremely difficult to obtain detailed and accurate information on farming practices for entire watersheds (thus our use of ''expert knowledge'' from local conservation district officers), and multiple factors typically have confounding effects on stream biodiversity. Because the Maryland agricultural streams we studied had more diverse invertebrate assemblages than we found reported in nearly all other published studies of agricultural streams (Appendix B), it is critical to get a better understanding of how specific farming practices influence stream ecosystems. Carefully designed studies to tease apart the biodiversity conservation value of various best management practices in both agricultural and urban areas should be a research priority.

At this point, we can only say that it is plausible that riparian grass and shrubs combined with progressive farming practices help maintain high levels of macroinvertebrate diversity. Our correlative data suggest that agricultural land preservation could be a useful biodiversity conservation strategy, particularly if the impact of agricultural activities is mitigated through the use of BMPs. In areas that are destined for development, the patterns we document suggest that maintenance of riparian forested buffers is vital, even in the most urbanized areas.

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APPENDIX A

A table showing land use attributes and values of each of the biological response variables for the 29 headwater streams used in the study is available in ESA's Electronic Data Archive: *Ecological Archives* A015-032-A1.

APPENDIX B

A table listing published reports of macroinvertebrate richness in agriculturally impacted streams based on a literature search is available in ESA's Electronic Data Archive: *Ecological Archives* A015-032-A2.