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Benthic Macroinvertebrates of the University of the Philippines Diliman Campus Waterways and Their Variation Across Land Use in an Urban, Academic Landscape

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ABSTRACT

Urban development impacts stream ecosystems primarily via changes in hydrological regime, geomorphology, and in water quality. These changes in turn have biological effects. The University of the Philippines Diliman campus, located at the heart of the highly urbanized Quezon City, has gone through numerous developments in terms of landscape and infrastructure. Unlike the terrestrial environment, the extent to which these developments have impacted the campus waterways is unknown. Hence, our research aims to assess the overall condition of the waterways in the campus based on the benthic macroinvertebrate assemblages. A total of 19 stream reaches were sampled in November 2015 and 2016 in the following land use categories: academic/academic support units (six sites), campus core (eight sites), and parks and open spaces (five sites). One-way analysis of variance (ANOVA) detected significant spatial difference in several macroinvertebrate-based metrics, stream physicochemistry, and in-stream habitat condition elements. Our study reveals that all sampled stream reaches, regardless of their land use categories, are under poor to severe pollution conditions. All macroinvertebrate-based metrics and indices indicate degraded water quality and stream health. Our results are consistent with urban stream studies elsewhere, which suggest that land-based activities can be stressful for some aquatic organisms, and at times, result in reduced abundance and even reduction in species composition.

Keywords: Biomonitoring, biotic indices, stream habitat assessment, urban land use, water quality

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INTRODUCTION

Urbanization affects the natural environment worldwide (Pickett et al. 2001; Grimm et al. 2008). In particular, urban development impacts stream ecosystems primarily via changes in hydrological regime through increased magnitude and frequency of high flows or through reduced base flow due to increase in impervious surfaces; changes in geomorphology through channel alteration; and changes in water quality through contaminated runoff and from direct point source discharges (Walsh et al. 2005; Moggridge et al. 2014). As a consequence, these physical and chemical changes have biological effects.

While urban areas, such as the University of the Philippines Diliman campus, can support a wide range of terrestrial biota (Ong et al. 1999; Vallejo et al. 2009), we do not know whether the same is true for streams flowing through the urban landscape particularly in developing and emerging economies (but see Freitag (2013) wherein he described a new species of hydraenid beetle found in headwater creeks inside the Ateneo de Manila University campus). This is due to the fact that, for over the past 10 years, the observed marked increase in research on urban aquatic ecosystems is biased towards temperate regions and in developed countries (Francis 2012). A recent study has documented that tropical streams are naturally flashy due to high precipitation and watershed features, and thus, do not significantly differ with urban streams (Ramirez et al. 2009). Moreover, Roy et al. (2009) reported that biological responses to urbanization range from broadly consistent to highly variable or understudied. Consequently, there is a need for further research to understand mechanisms of response to urbanization in other regions, such as the tropics, where cities are larger and growing rapidly.

The University of the Philippines Diliman (UPD) campus, located at the heart of the highly urbanized Quezon City, has gone through numerous developments in terms of landscape and infrastructure. However, unlike the terrestrial environment (Vallejo and Aloy 2014), the extent to which these developments have impacted the waterways in the campus is unknown as no baseline study was conducted to compare the current conditions. Meanwhile, evidence that many freshwater species are being threatened with extinction by urban development are being discovered elsewhere (Paul and Meyer 2001; Walsh et al. 2005, 2007; Brown et al. 2009; Ramirez et al. 2012).

In 2012, UPD formulated the Master Site Development Plan that serves as a framework for the university's physical growth for the next 13 years and as a set of guidelines for all improvements in the campus, including, among others, land use allocation, and building and landscape designs (Espina and Espina 2013). In this master plan, eight land uses have been recognized: campus core, academic/academic support units, science and technology park, resource generation zone, residential, community services, parks and open spaces, and protected forest area. Nonetheless, we do not know whether the waterways, if any, in these areas are in good condition to support aquatic biota.

To address this knowledge gap, we investigated the stream macroinvertebrate biodiversity in UPD campus. Specifically, we assessed the overall condition of the waterways based on the benthic macroinvertebrates, water quality, and physical instream habitats along stream reaches in the following land uses: campus core, academic/academic support units, and parks and open spaces.

MATERIALS AND METHODS

Study Site

The University of the Philippines Diliman campus located in Quezon City (14° 38' N, 121° 2' E) is the flagship and one of the constituent units of the University of the Philippines System. With an area of 493 ha, the campus is a fully functional community and a government unit as it hosts an array of facilities, such as academic units, parks, and residential and commercial areas. Daytime population peaks at around 40,000 individuals, which are mainly composed of students, faculty, employees, and some informal settlers (Ong et al. 1999; Vallejo et al. 2008). Quezon City climate is classified as tropical monsoonal with a pronounced dry season from November to April and wet season from May to October (Figure 1).

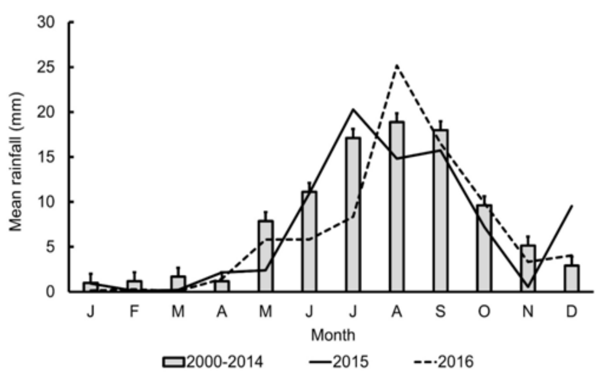

Figure 1. Mean rainfall (± standard error) values in Science Garden, Quezon City for the period 2000-2014, and for years, 2015 and 2016. Data are from the Climatology and Agrometeorology Division of the Philippine Atmospheric Geophysical and Astronomical Services Administration (PAGASA).

Nineteen sampling sites within the campus were selected and sampled in November 2015 and 2016 (Figure 2). These sites were located in the following land use categories: academic units (AU; 6 sites), campus core (CC; 8 sites), and parks and open spaces (PO; 5 sites). Because of a strong dry spell prevailing in the country in November 2015 (mean rainfall ± standard error = 0.54 ± 0.27 mm; Figure 1), several sites ran dry, and hence, were not sampled. These include preselected waterways located in other land use categories (e.g., science and technology park). Nonetheless, the average (± standard error) rainfall in November 2016 was 3.34 ±1.26 mm (Figure 1).

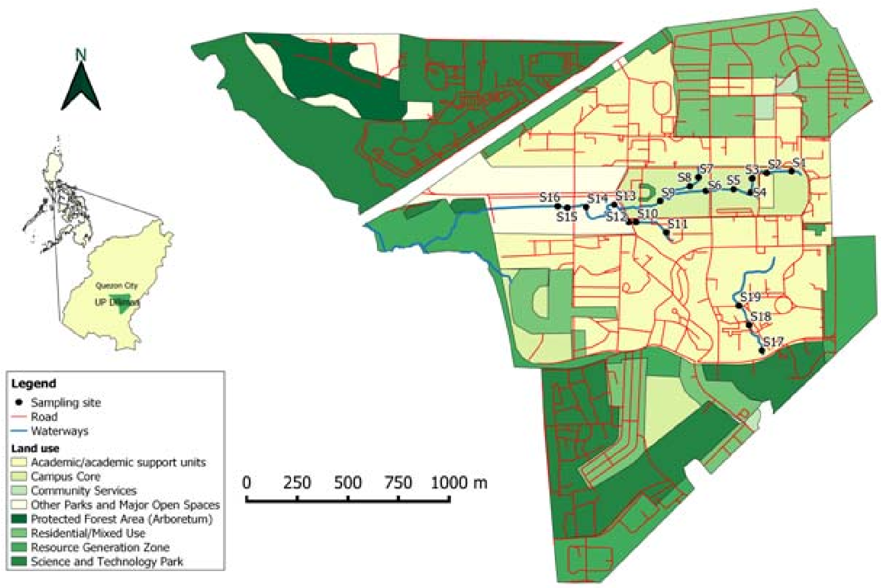

Figure 2. Map of the University of the Philippines Diliman campus in Quezon City showing land uses and the location of sampling sites.

Benthic Macroinvertebrates

A 50-m sampling reach was established within each land use. Following the method of de Jesus-Crespo and Ramirez (2011), three collectors handpicked for 15 minutes all macroinvertebrates from each of the four major habitats (leaf packs, margin vegetation, pools, and riffles) within the 50-m reach. This procedure was continued until three replicate samples per habitat (one from each collector) had been collected. For comparison among sites with different proportions of stream habitat, an overall habitat-weighted value per taxon per site was calculated (de Jesus-Crespo and Ramirez 2011).

All samples were preserved in 95% ethanol and were brought to the Aquatic Biology Research Laboratory, Institute of Biology, UPD for sorting and identification. In the laboratory, samples were washed and elutriated using a 250-μm sieve to separate macroinvertebrates from plants, sediment, and other inorganic materials. Macroinvertebrates were counted and identified to genus level under a stereo microscope. Identification was performed using the keys of Dudgeon (1999), Yule and Yong (2004), and the Mekong River Commission (2006).

Using the macroinvertebrate-habitat weighted value, the following macroinvertebrate metrics were calculated: total invertebrate density (the number of individual organisms collected per m^2); taxon richness (the number of taxa counted in a sample); richness of the pollution-sensitive insect orders Ephemeroptera-Plecoptera-Trichoptera (EPT) and Ephemeroptera-Plecoptera-Trichoptera-Coleoptera (EPTC); Simpson's index of diversity (D); and Simpson's measure of evenness (E). Moreover, biotic indices used in stream bioassessment and biomonitoring were calculated to determine the current condition of the UPD waterways: Hilsenhoff's family biotic index, a biotic index for assessing organic and nutrient pollution using tolerance values of arthropod families (Hilsenhoff 1988); Biological Monitoring Working Party (BMWP), a standardized score system based on tolerance scores of macroinvertebrate families to organic pollution (Mustow 2002); Average Score per Taxa (ASPT), a biotic index which measures river status using the calculated BMWP score divided by number of taxa (Mustow 2002); Stream Invertebrate Grade Number – Average Level version 2 (SIGNAL 2), a biotic index for Australian river macroinvertebrates (Chessman 1995, 2003); Singapore's stream biotic index score (SingScore), a newly developed biotic index for measuring the health of Singapore's streams using benthic macroinvertebrates (Blakely et al. 2014); and Average Tolerance Score per Taxon (ATSPT), a biotic index for evaluating stream health integrity using site disturbance scores and benthic macroinvertebrate abundance (Chessman and Giap 2010).

Physicochemical and Habitat Parameters

In the same stream reach where macroinvertebrates were sampled, various physicochemical parameters were measured on site at three randomly selected locations within the 50-m reach: water temperature (°C) and dissolved oxygen (DO; mg L^{-1}) were obtained using a DO meter (YSI EcoSense DO200A; Yellow Spring Instruments, Ohio, USA), and conductivity (μS/cm) and total dissolved solids (TDS; mg L^{-1}) with a hand-held meter (YSI EcoSense300A; Yellow Spring Instruments, Ohio, USA). In addition, stream width (m), depth (cm), flow rate (m s^{-1}), and water discharge (m^3 s^{-1}) were measured within each reach. These physicochemical parameters were considered in this study because they have been shown to influence the abundance and distribution of benthic macroinvertebrates (Narangarvuu et al. 2014; Yazdian et al. 2014).

To evaluate the riparian zones and instream habitats, the modified stream visual assessment protocol (Magbanua et al. 2013) was used. The protocol is composed of 15 items describing stream environmental condition in relation to channel flow; depth regime; bank stability; vegetative protection and zone; canopy cover; water appearance; nutrient enrichment; streambed characteristics, such as sediment deposition, habitats, habitat complexity, and barriers to movement; and aquatic macroinvertebrate community. Each item is scored from 1 to 20, and the sum of all items scored was divided by the number of items scored to assess a site's habitat condition. Hence, a site having a score of ≤ 5 is considered poor, 5-10 is marginal, 10-15 is suboptimal, and 16-20 optimal.

Data Analyses

Differences in macroinvertebrate assemblage across land uses were evaluated using non-metric multidimensional scaling (NMDS) ordination technique through Bray-Curtis similarity matrix after fourth-root transformation of assemblage data, followed by a confirmatory analysis of similarity (ANOSIM). Global *R* values less than 0.25 indicate similarity in macroinvertebrate communities (refer to Maroneze et al. (2011) and Novais et al. (2012)). All analyses were performed using the software PRIMER 6.0 (Primer-E Ltd, Plymouth, UK).

Moreover, we tested differences for the various macroinvertebrate metrics, biotic indices, and physicochemical and habitat parameters among waterways under different land uses using analysis of variance (ANOVA) in IBM SPSS Statistics 20.0 (IBM Corp., New York USA). In the model, land use (academic units, campus core, and parks and open spaces) was the fixed main (between-subjects effects) factor. If analyses of the fixed main factor showed significance, we performed pairwise comparisons using *post hoc* tests (Tukey's HSD). For all significant findings, effect sizes (ES = partial η2 values, range 0-1; refer to Garson (2012)) were reported to compare the magnitudes of effects detected (Nakagawa and Cuthill 2007). Where necessary, data were $\log_{10}$ (x)- or $\log_{10}$ (x + 1)-transformed prior to analyses to improve normality and homoscedasticity (Quinn and Keough 2002).

RESULTS AND DISCUSSION

Stream Physicochemistry, Riparian Zone and In-stream Habitats

Our results showed that, except for water temperature, all measured physicochemical parameters had significant differences across different land use (P d ≤ 0.048 in all cases; Table 1). Other than DO, all parameters were highest in the parks and open spaces land use categories. By contrast, among measured riparian and in-stream habitat parameters, only canopy cover, water appearance, sediment deposition, and aquatic macroinvertebrate community differed across land uses, with academic units obtaining the highest score in all four parameters ($P \leq 0.039$ in all cases; Table 1).

Table 1. Summary of the one-way ANOVAs comparing physicochemistry, habitat parameters, biological response metrics, and biotic indices across different land uses. Rankings for post hoc tests or specific contrasts in cases with significant effects are given. P-values < 0.05 are in bold print. Effect sizes (ES = partial η2 values; range 0-1; categories: weak > 0.1, moderate > 0.3, strong > 0.5; Nakagawa and Cuthill 2007) are given for all significant findings (in bold). AU = Academic units; CC = Campus Core; PO = Parks and Open Spaces; HFBI = Hilsenhoff Family Biotic Index; SingScore = Singapore Score; BMWP^{THAI} = Biological Monitoring Working Party THAI version; ASPT^{THAI} = Average Score per Taxon THAI version; SIGNAL 2 = Stream Invertebrate Grade Number Average Level version 2; ATSPT = Average Tolerance Score per Taxon

Parameter		AU	CC	PO	P-value	ES	Ranking
Physicochemistry	Water temperature	26.74 (0.22)	27.09 (0.16)	27.45 (0.24)	0.064	0.048	
	Dissolved oxygen	2.22 (0.26)	1.36 (0.14)	1.52 (0.21)	**0.005**	**0.090**	**AU > (CC = PO)**
	TDS	160.45 (11.40)	184.55 (10.89)	209.83 (10.47)	**0.008**	**0.083**	**PO > AU**
	Conductivity	338.26 (10.22)	360.37 (16.95)	481.00 (27.64)	**0.048**	**0.053**	**PO > CC**
	Stream width	1.36 (0.01)	1.84 (0.17)	2.26 (0.22)	**0.001**	**0.124**	**PO > AU**
	Water depth	8.36 (0.77)	9.81 (0.78)	13.35 (0.88)	**<0.001**	**0.133**	**PO > (CC = AU)**
	Flow rate	0.09 (0.02)	0.10 (0.02)	0.19 (0.03)	**0.003**	**0.100**	**PO > (CC = AU)**
	Stream discharge	0.01 (0.002)	0.03 (0.01)	0.05 (0.01)	**<0.001**	**0.131**	**PO > (CC = AU)**
Riparian and instream habitat	Channel flow	7.72 (0.90)	7.58 (0.87)	8.67 (0.91)	0.357	0.044	
	Channel alteration	10.06 (0.93)	8.65 (0.80)	9.40 (0.98)	0.102	0.082	
	Depth regime	6.06 (0.77)	6.76 (0.83)	8.57 (0.95)	0.185	0.066	
	Bank stability	7.67 (0.88)	8.50 (0.86)	8.90 (0.93)	0.129	0.075	
	Bank vegetative protection	9.89 (0.88)	9.33 (0.84)	9.87 (1.11)	0.465	0.035	
	Riparian vegetative zone	9.53 (0.96)	9.50 (0.92)	9.93 (1.05)	0.726	0.018	
	Canopy cover	8.75 (0.97)	8.58 (0.88)	5.63 (1.03)	**0.039**	**0.109**	**(AU = CC) > PO**
	Water appearance	7.69 (0.84)	4.77 (0.71)	4.80 (0.74)	**0.006**	**0.156**	**AU > (CC = PO)**
	Nutrient enrichment	6.92 (0.80)	6.69 (0.77)	5.97 (0.80)	0.725	0.018	
	Sediment deposition	6.58 (0.69)	4.90 (0.67)	6.50 (0.71)	**0.018**	**0.129**	**(AU = PO) > CC**
	Riffle embeddedness	6.61 (0.73)	4.93 (0.61)	6.30 (0.74)	0.161	0.073	
	Barriers to species movement	9.92 (0.98)	7.83 (0.85)	9.57 (1.10)	0.109	0.080	
	Fish habitat complexity	6.50 (0.68)	6.33 (0.65)	6.30 (0.82)	0.872	0.010	
	Aquatic macro-invertebrate habitat	8.22 (0.81)	8.00 (0.75)	7.90 (0.95)	0.947	0.005	
	Aquatic macro-invertebrate community	3.72 (0.13)	2.29 (0.10)	1.80 (0.10)	<0.001	0.527	**AU > CC > PO**
	Overall habitat score	7.72 (0.61)	6.99 (0.56)	7.34 (0.70)	0.818	0.004	

Table 1. Summary of the one-way ANOVAs comparing physicochemistry, habitat parameters, biological response metrics, and biotic indices across different land uses. Rankings for post hoc tests or specific contrasts in cases with significant effects are given. *P*-values < 0.05 are in bold print. Effect sizes (ES = partial η2 values; range 0-1; categories: weak > 0.1, moderate > 0.3, strong > 0.5; Nakagawa and Cuthill 2007) are given for all significant findings (in bold). AU = Academic units; CC = Campus Core; PO = Parks and Open Spaces; HFBI = Hilsenhoff Family Biotic Index; SingScore = Singapore Score; BMWP^{THAI} = Biological Monitoring Working Party THAI version; ASPT^{THAI} = Average Score per Taxon THAI version; SIGNAL 2 = Stream Invertebrate Grade Number Average Level version 2; ATSPT = Average Tolerance Score per Taxon (Cont'n.)

Parameter		AU	CC	PO	*P*-value	ES	Ranking
Biological response metrics	Macroinvertebrate density	306.36 (99.46)	121.47 (24.30)	215.77 (34.41)	0.154	0.033	
	Taxon richness	10.47 (0.89)	8.12 (0.60)	8.50 (0.95)	0.344	0.019	
	EPT taxa richness	0.86 (0.14)	0.27 (0.07)	0.20 (0.07)	**<0.001**	**0.160**	**AU > (CC = PO)**
	EPTC taxa richness	1.44 (0.17)	0.83 (0.11)	0.87 (0.15)	**0.020**	**0.068**	**AU > (CC = PO)**
	Simpson's diversity index	3.02 (0.31)	3.26 (0.63)	2.12 (0.23)	0.352	0.019	
	Simpson's evenness	0.36 (0.05)	0.43 (0.08)	0.34 (0.06)	0.803	0.004	
Biotic indices	HFBI	7.76 (0.13)	7.97 (0.13)	8.14 (0.12)	0.095	0.044	
	SingScore	67.68 (3.29)	62.02 (2.07)	61.26 (2.76)	0.409	0.016	
	BMWP^{THAI}	2.68 (0.29)	2.30 (0.29)	2.83 (0.36)	0.197	0.030	
	ASPT^{THAI}	4.12 (0.13)	4.10 (0.11)	3.89 (0.17)	0.097	0.043	
	SIGNAL 2	2.84 (0.07)	2.66 (0.05)	2.66 (0.06)	0.241	0.026	
	ATSPT	57.43 (0.22)	58.69 (0.28)	58.61 (0.39)	**0.001**	**0.114**	**(CC = PO) > AU**

These findings are consistent with most urban stream studies done in the past (e.g., Couceiro et al. 2007; de Jesus-Crespo and Ramirez 2011; Baltazar et al. 2016; Docile et al. 2016). Increasing loads of organic and inorganic carbon in urban stream decrease the amount of DO (Daniel et al. 2002; Butman et al. 2015; Tromboni and Dodds 2017). Moreover, high dissolved solid concentrations had been observed in UPD streams. Studies conducted by Horn et al. (2017), Taka et al. (2017), and Toor et al. (2017) all noted that dissolved solids are known to accumulate in areas with higher rates of inorganic runoff (e.g., industrial sites, residential sites) and contribute to an increased ion concentration annually. Lastly, changes in land use and hydrological gradients altered stream channels, depth, flow rate, and discharge in the campus waterways due to continued habitat degradation, land cover modification, and subsurface drainage, which through time, may negatively affect local stream habitat and biodiversity (Allan 2004; Potter et al. 2014; Walsh and Webb 2016; Baumgartner and Robinson 2017).

In habitat assessment, only canopy cover, water appearance, sediment deposition and aquatic macroinvertebrate community exhibited marked differences across different land use types (Table 1). Changes in riverine spatial gradients has been tagged as major driver in declines of stream biota. Canopy cover is essential for maintaining lower stream temperature and for increasing the allochthonous source of energy which in turn promotes diverse stream biotic assemblages (Sponseller et al. 2001; Kominoski et al. 2011). In urban streams, the amount of detritus breakdown is lower, leading to a much poorer biotic assemblages (Roy et al. 2005; Martins et al. 2015). Furthermore, Uriarte et al. (2011) noted that in water appearance the increasing load of organic and inorganic materials in streams brought by continued urban runoff and riparian habitat degradation leads to its much poorer state. Likewise, Extence et al. (2013) reported that sediment deposition also increases in streams with low flow, modified habitat, and excessive sediment output from the catchment. The diversity of aquatic macroinvertebrate community heavily depends on the condition of its habitat, which determines the community that it can support (Weijters et al. 2009). Modified habitats (e.g., high conductivity, eutrophic streams) tend to support pollution tolerant taxa, while undisturbed habitats (e.g., high DO, low water temperature) support diverse benthic communities comprised mainly of pollution-sensitive taxa (Miserendino et al. 2011).

Benthic Macroinvertebrate Assemblages

A total of 42,663 macroinvertebrates belonging to 45 families and 56 genera were collected in 19 stream reaches within the UPD campus. Of these 56 genera identified, 10 comprised 93.6% of the total: the non-biting midge *Chironomus* spp. (67.2%), the segmented worm Oligochaeta (Genus 1) (11.2%), the non-biting midge *Cricotopus* spp. (4.5%), the moth fly *Psychoda* spp. (3.3%), the lymnaeid snail *Radix quadrasi* (1.7%), the shore fly *Brachydentera* spp. (1.7%), the dragonfly *Brechmorhoga* spp. (1.2%), the freshwater leech *Helobdella* spp. (1.0%), the mayfly *Labiobaetis* spp. (1.0%), and the non-biting midge *Thienamannimyia* spp. (0.9%) (Figure 3).

The results of the ordination analysis reveal weak clustering (2D Stress=0.24) across different land uses (Figure 4). This was further supported by the global R of ANOSIM for land uses (Global R = 0.070, P = 0.1), indicating no observable variation in the macroinvertebrate community. However, among the different biological metrics analyzed in this study, the richness of the pollution-sensitive insect orders EPT and EPTC exhibited significant differences across different land uses (Table 1).

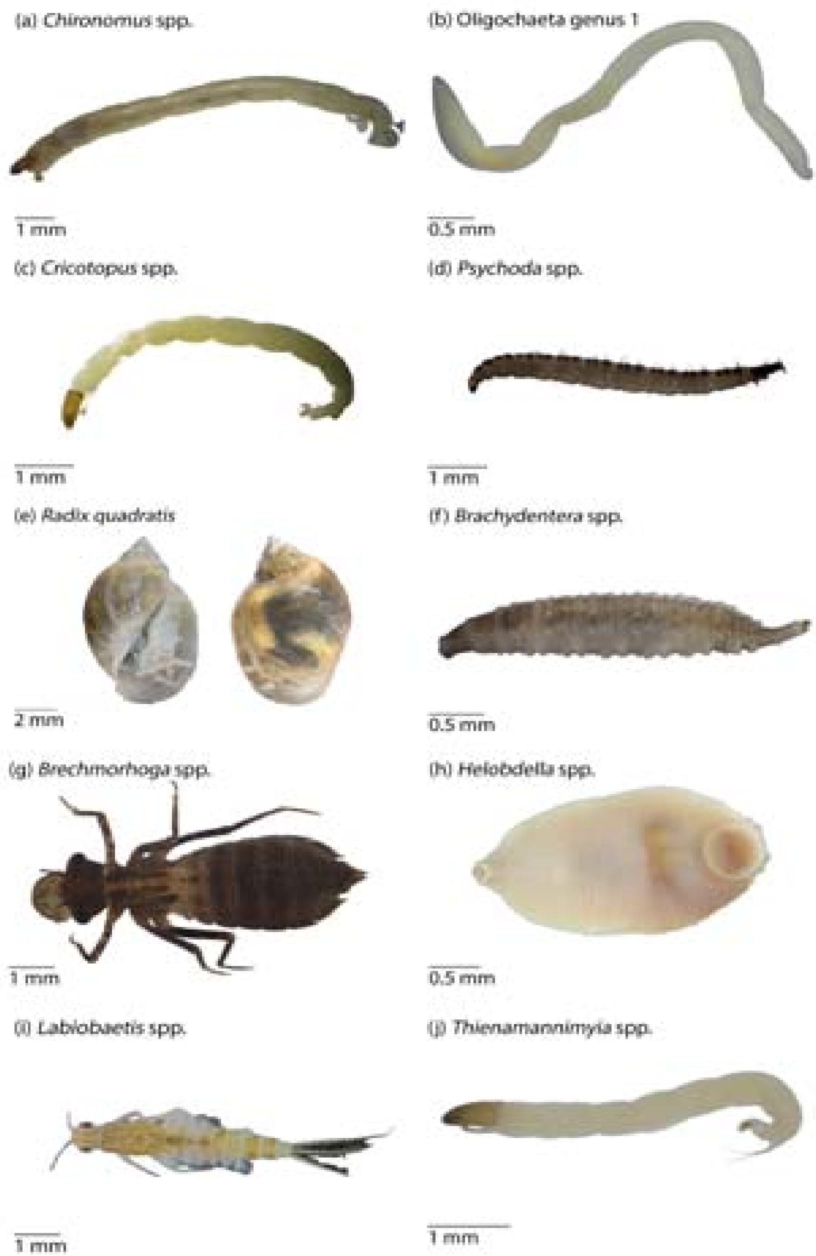

Figure 3. Ten most dominant macroinvertebrates across land use types in the University of the Philippines Diliman campus waterways.

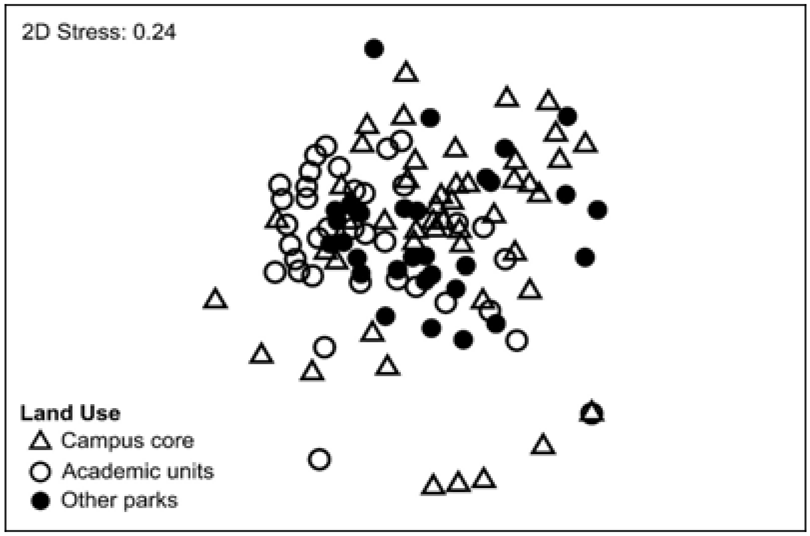

Figure 4. Two-dimensional non-metric multidimensional scaling (NMDS) plot of the stream macroinvertebrate community structure across different land uses in the University of the Philippines Diliman, Quezon City, Philippines.

The resulting similarity of identified macroinvertebrate community across different land uses is a general trend in urban streams due to higher rates of organic and inorganic solute contamination along altered riparian and stream reaches (Cuffney et al. 2010; Ricart et al. 2010). For instance, the presence of pollution-tolerant taxa (e.g., Chironomidae, Oligochaeta) and absence of pollution-sensitive taxa (e.g., Perlidae, Psephenidae) may contribute to the similar aquatic invertebrate communities across different land uses, since the former can thrive in urban streams due to higher rates of water stress, while the latter requires pristine environmental conditions (de Paiva Silva et al. 2010; Chang et al. 2014; Mehring et al. 2017). UPD streams are dominated by several members of the family Chironomidae (e.g., *Chironomus* spp., *Cricotopus* spp., *Thienamannimyia* spp.) and Oligochaeta, both of which are pollution-tolerant. These organisms can tolerate a wide range of environmental conditions (e.g., low DO concentration, high dissolved solids); thus, allowing them to thrive in all types of habitat, ranging from pristine to heavily degraded streams (Cortelezzi et al. 2011; Frizzera and Alves 2012; Rosa et al. 2014).

However, the marked differences in EPT and EPTC taxa richness across different land uses indicated the capacity of UPD streams to be inhabited by these taxa. Similar findings had been observed in the studies of Lenat and Crawford (1994) and Violin et al. (2011), wherein these authors observed EPT and EPTC taxa in urban sites. Nonetheless, it should be noted that these identified taxa (e.g., Baetidae, Hydrophilidae) are considered mildly tolerant to pollution by others (e.g., Rizo-Patron et al. 2013; Chang et al. 2014), similar to the EPT and EPTC taxa identified in UPD streams. Furthermore, higher rates of organic runoff in urban streams can significantly increase the density of pollution-tolerant invertebrates and prevent possible colonization of pollution-sensitive taxa (Roy et al. 2003; Niyogi et al. 2007; Shin et al. 2011).

Stream Condition Based on Macroinvertebrate Biotic Indices

All measured biotic indices reveal that, across different land uses, only ATSPT (P = 0.001) showed marked difference (Table 1). In addition, the high pollution tolerance score of collected and identified invertebrates in UPD waterways led to poor stream condition ratings in all biotic indices across different land uses (Table 2).

Table 2. Mean (± standard error) values of computed biotic indices and the corresponding condition ratings across different land uses. HFBI = Hilsenhoff Family Biotic Index; SingScore = Singapore Score; BMWP^{THAI} = Biological Monitoring Working Party THAI version; ASPT^{THAI} = Average Score per Taxon THAI version; SIGNAL 2 = Stream Invertebrate Grade Number Average Level version 2; ATSPT = Average Tolerance Score per Taxon

	Academic units		Campus core		Parks and open spaces	
Biotic Index	Index score	Condition rating	Index score	Condition rating	Index score	Condition rating
HFBI	7.76 (0.13)	Very poor	7.97 (0.13)	Very poor	8.14 (0.11)	Very poor
SingScore	67.68 (3.29)	Poor	62.02 (2.07)	Poor	61.26 (2.76)	Poor
BMWP^{THAI}	2.68 (0.29)	Very bad	2.30 (0.29)	Very bad	2.83 (0.36)	Very bad
ASPT^{THAI}	4.12 (0.13)	Bad	4.10 (0.11)	Bad	3.89 (0.17)	Bad
SIGNAL2	2.84 (0.07)	Probable severe pollution	2.66 (0.05)	Probable severe pollution	2.66 (0.06)	Probable severe pollution
ATSPT	57.43 (0.22)	Unhealthy	58.69 (0.28)	Unhealthy	58.61 (0.39)	Unhealthy

Changed ATSPT values depict the quality of UPD streams across different land uses, indicating the importance of riparian habitats in supporting diverse biotic communities (Poff and Zimmerman 2010). Nonetheless, our results underscored the poor water quality condition of UPD streams, regardless of land use (Table 2). Biotic indices assign numerical value to a specific taxon with a corresponding tolerance score based on its tolerance to pollution (Zimmerman 1993). In the case of UPD's macroinvertebrate assemblage, the abundance of tolerant taxa resulted in the streams' poor condition ratings. Lastly, the biotic indices used in this study are all derived from other countries and have failed to consider local taxon that has no pre-assigned tolerance value, and thus, may not provide a true picture of the streams in regions outside its origin (Zeybek et al. 2014).

CONCLUSION AND RECOMMENDATION

Globally, urban streams generally have higher loads of organic and inorganic pollution, compromised stream and riparian areas, abundant pollution-tolerant taxa, and poor water and habitat quality. Our results reveal poor to severe stream conditions across land uses. Marginal habitat assessment scores and sub-optimal physicochemical parameters in all streams supported these findings, reflecting the intensity of riparian and stream modification. Similarly, water quality based on considered variables also indicated poor quality, which is consistent with the stream biota dominated by pollution-tolerant taxa. These resulted in lower biotic index scores, providing further support for the severity of the conditions of UPD streams.

Our findings reflect similar patterns observed in urban streams, which may persist if UPD streams and riparian habitats are not protected and restored. Therefore, we recommend a campus-wide restoration of streams and waterways, as well as improvement of the wastewater treatment facility in the campus. We also suggest monitoring the streams and waterways during wet and dry seasons to provide a complete picture of the conditions of these waterways. This bioassessment may provide additional knowledge on the benthic macroinvertebrate community structure and the possible effects of environmental flow on these urban communities.

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REFERENCES

Allan JD. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology, Evolution and Systematics. 35:257-284.

Baltazar DES, Magcale-Macandog D, Tan MFO, Zafaralla MT, Cadiz, NM. 2016. A river health status model based on water quality, macroinvertebrates and land use for Niyugan River, Cabuyao City, Laguna, Philippines. Journal of Environmental Science and Management. 19(2):38-53.

Baumgartner SD, Robinson CT. 2017. Changes in macroinvertebrate trophic structure along a land-use gradient within a lowland stream network. Aquatic Sciences. 79(2):407-418.

Blakely TJ, Eikaas HS, Harding JS. 2014. The Singscore: A macroinvertebrate biotic index for assessing the health of Singapore's streams and canals. Raffles Bulletin of Zoology. 62:540-548.

Brown LR, Cuffney TF, Coles JF, Fitzpatrick F, McMahon G, Steuer J, Bell AH, May JT. 2009. Urban streams across the USA: Lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society. 28:1051-1069.

Butman DE, Wilson HF, Barnes RT, Xenopoulos MA, Raymond, PA. 2015. Increased mobilization of aged carbon to rivers by human disturbance. Nature Geoscience. 8(2):112-116.

Chang FH, Lawrence JE, Rios-Touma B, Resh VH. 2014. Tolerance values of benthic macroinvertebrates for stream biomonitoring: Assessment of assumptions underlying scoring systems worldwide. Environmental Monitoring and Assessment. 186(4):2135-2149.

Chessman BC. 1995. Rapid assessment of rivers using macroinvertebrates: A procedure based on habitat specific sampling, family level identification and a biotic index. Australian Journal of Ecology. 20(1):122-129.

Chessman BC. 2003. New sensitivity grades for Australian river macroinvertebrates. Marine and Freshwater Research. 54:95-103.

Chessman B, Giap DH. Biological metrics calculation. In: Resh VH, Giap DH, editors. Biomonitoring methods for the lower Mekong Basin. Vientiane: Mekong River Commission; c2010. p. 57-60.

Cortelezzi A, Paggi AC, Rodríguez M, Capítulo AR. 2011. Taxonomic and nontaxonomic responses to ecological changes in an urban lowland stream through the use of Chironomidae (Diptera) larvae. Science of the Total Environment. 409(7):1344-1350.

Couceiro SRM, Hamada N, Luz SLB, Forsberg BR, Pimentel TP. 2007. Deforestation and sewage effects on aquatic macroinvertebrates in urban streams in Manaus, Amazonas, Brazil. Hydrobiologia. 575:271-284.

Cuffney TF, Brightbill RA, May JT, Waite IR. 2010. Responses of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. Ecological Applications. 20(5):1384-1401.

Daniel MH, Montebelo AA, Bernardes MC, Ometto JP, De Camargo PB, Krusche AV, Ballester MV, Victoria RL, Martinelli LA. 2002. Effects of urban sewage on dissolved oxygen, dissolved inorganic and organic carbon, and electrical conductivity of small streams along a gradient of urbanization in the Piracicaba river basin. Water, Air, and Soil Pollution. 136(1-4):189-206.

de Jesus-Crespo R, Ramirez A. 2011. Effects of urbanization on stream physicochemistry and macroinvertebrate assemblages in a tropical urban watershed in Puerto Rico. Journal of the North American Benthological Society. 30:739-750.

de Paiva Silva D, De Marco P, Resende DC. 2010. Adult odonate abundance and community assemblage measures as indicators of stream ecological integrity: A case study. Ecological Indicators. 10(3):744-752.

Docile TN, Figueiro R, Portela C, Nessimian, J L. 2016. Macroinvertebrate diversity loss in urban streams from tropical forests. Environmental Monitoring and Assessment. 188(4):237.

Dudgeon D. 1999. Tropical Asian Streams: Zoobenthos, Ecology and Conservation. Hong Kong: Hong Kong University Press.

Espina MAA, Espina CSP. 2013. Quezon City: Principles of a Sustainable UP Diliman Campus; [cited 2015 January 14]. Available from http://www.ovcrd.upd.edu.ph/wp-content/uploads/2013/08/S2_4-ESPINA-NEW-PRINCIPLES-2.pdf.

Extence A, Chadd RP, England J, Dunbar MJ, Wood, PJ, Taylor, ED. 2013. The assessment of fine sediment accumulation in rivers using macroinvertebrate community response. River Research and Applications. 29(1):17-55.

Francis R. 2012. Positioning urban rivers within urban ecology. Urban Ecosystems. 15:285-291.

Freitag H. 2013. *Hydraena (Hydraenopsis) ateneo*, new species (Coleoptera, Hydraenidae) and other aquatic Polyphaga from a small habitat patch in a highly urbanized landscape of Metro Manila, Philippines. ZooKeys. 329:9-21.

Frizzera GL, Alves RDG. 2012. The influence of taxonomic resolution of Oligochaeta on the evaluation of water quality in an urban stream in Minas Gerais, Brazil. Acta Limnologica Brasiliensia. 24(4):408-416.

Garson GD. 2012. Testing Statistical Assumptions. Asheboro, NC: Statistical Associates Publishing.

Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu J, Bai X, Briggs JM. 2008. Global change and the ecology of cities. Science. 319:756-760.

Hilsenhoff WL. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society. 7(1):65-68.

Horn AH, Torres IC, Ribeiro EV, Junior APM. 2017. Relationship between metal water concentration and anthropogenic pressures in a tropical watershed, Brazil. Geochimica Brasiliensis. 30(2):158.

Kominoski JS, Marczak LB, Richardson JS. 2011. Riparian forest composition affects stream litter decomposition despite similar microbial and invertebrate communities. Ecology. 92(1):151-159.

Lenat DR, Crawford JK. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia. 294(3):185-199.

Magbanua FS, Mendoza NYB, Fontanilla AM, Ong PS. 2013. Modified Stream Visual Assessment Protocol: A Field Guide. UP Biology-EDC Biodiversity Field Guide Series No. 1. Quezon City and Pasig City: Institute of Biology, University of the Philippines Diliman and Energy Development Corporation.

Maroneze DM, Tupinambás TH, Alves CB, Vieira F, Pompeu PS, Callisto M. 2011. Fish as ecological tools to complement biodiversity inventories of benthic macroinvertebrates. Hydrobiologia. 673:29-40.

Martins RT, Melo AS, Gonçalves Jr JF, Hamada N. 2015. Leaf-litter breakdown in urban streams of Central Amazonia: Direct and indirect effects of physical, chemical, and biological factors. Freshwater Science. 34(2):716-726.

Mehring AS, Cook PL, Evrard V, Grant SB, Levin LA. 2017. Pollutiontolerant invertebrates enhance greenhouse gas flux in urban wetlands. Ecological Applications. 27(6):1852-1861.

Mekong River Commission. 2006. Identification of Freshwater Invertebrates of the Mekong River and its Tributaries. Vientiane: Mekong River Commission. p. 274.

Miserendino ML, Casaux R, Archangelsky M, Di Prinzio CY, Brand C, Kutschker AM. 2011. Assessing land-use effects on water quality, in-stream habitat, riparian ecosystems and biodiversity in Patagonian northwest streams. Science of the Total Environment. 409(3):612-624.

Moggridge HL, Hill MJ, Wood PJ. 2014. Urban aquatic ecosystems: The good, the bad and the ugly. Fundamental and Applied Limnology. 185:1-6.

Mustow SE. 2002. Biological monitoring of rivers in Thailand: Use and adaptation of the BMWP score. Hydrobiologia. 479(1):191-229.

Nakagawa S, Cuthill IC. 2007. Effect size, confidence interval and statistical significance: A practical guide for biologists. Biological Reviews. 82:591-605.

Narangarvuu D, Hsu CB, Shieh SH, Wu FC, Yang PS. 2014. Macroinvertebrate assemblage patterns as indicators of water quality in the Xindian watershed, Taiwan. Journal of Asia-Pacific Entomology. 17:505-513.

Niyogi DK, Koren M, Arbuckle CJ, Townsend CR. 2007. Stream communities along a catchment land-use gradient: Subsidy-stress responses to pastoral development. Environmental Management. 39(2):213-225.

Novais MH, Blanco S, Delgado C, Morais M, Hoffmann L, Ector L. 2012. Ecological assessment of Portuguese reservoirs based on littoral epilithic diatoms. Hydrobiologia. 695:265-279.

Ong PS, Pedregosa M, de Guia M. 1999. Wildlife inventory of the UP Diliman and Ateneo de Manila University campuses, Diliman, Quezon City, Luzon, Philippines. Science Diliman. 11:6-20.

Paul MJ, Meyer JL. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics. 32:333-365.

Pickett ST, Cadenasso ML, Grove JM, Nilon CH, Pouyat R, Zipperer WC, Costanza R. 2001. Urban ecological systems: Linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. Annual Review of Ecology and Systematics. 32:127-157.

Poff NL, Zimmerman JK. 2010. Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. Freshwater Biology. 55(1):194-205.

Potter JD, McDowell WH, Helton AM, Daley ML. 2014. Incorporating urban infrastructure into biogeochemical assessment of urban tropical streams in Puerto Rico. Biogeochemistry. 121(1):271-286.

Quinn GP, Keough MJ. 2002. Experimental Design and Data Analysis for Biologists. Cambridge: Cambridge University Press.

Ramirez A, De Jesús-Crespo R, Martinó-Cardona DM, Martínez-Rivera N, Burgos-Caraballo S. 2009. Urban streams in Puerto Rico: What can we learn from the tropics? Journal of the North American Benthological Society. 28(4):1070-1079.

Ramirez A, Engman A, Rosas KG, Perez-Reyes O, Martino-Cardona DM. 2012. Urban impacts on tropical island streams: some key aspects influencing ecosystem response. Urban Ecosystems. 15:315-325.

Ricart M, Guasch H, Barceló D, Brix R, Conceição MH, Geiszinger A, de Alda MJL, Lopez-Doval JC, Muñoz I, Postigo C, Romaní AM, Villagrasa M, Sabater S. 2010. Primary and complex stressors in polluted Mediterranean rivers: Pesticide effects on biological communities. Journal of Hydrology. 383(1):52-61.

Rizo-Patron FV, Kumar K, Colton MBM, Springer M, Trama FA. 2013. Macroinvertebrate communities as bioindicators of water quality in conventional and organic irrigated rice fields in Guanacaste, Costa Rica. Ecological Indicators. 29:68-78.

Rosa BJFV, Rodrigues LFT, de Oliveira, GS, da Gama Alves R. 2014. Chironomidae and Oligochaeta for water quality evaluation in an urban river in southeastern Brazil. Environmental Monitoring and Assessment. 186(11):7771-7779.

Roy AH, Rosemond AD, Paul MJ, Leigh DS, Wallace JB. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). Freshwater Biology. 48(2):329-346.

Roy AH, Faust CL, Freeman MC, Meyer JL. 2005. Reach-scale effects of riparian forest cover on urban stream ecosystems. Canadian Journal of Fisheries and Aquatic Sciences. 62(10):2312-2329.

Roy AH, Purcell AH, Walsh CJ, Wenger SJ. 2009. Urbanization and stream ecology: five years later. Journal of the North American Benthological Society. 28:908-910.

Shin IK, Yi HB, Bae YJ. 2011. Colonization and community changes in benthic macroinvertebrates in Cheonggye Stream, a restored downtown stream in Seoul, Korea. Journal of Ecology and Environment. 34(2):175-191.

Sponseller RA, Benfield EF, Valett HM. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology. 46(10):1409-1424.

Taka M, Kokkonen T, Kuoppamäki K, Niemi T, Sillanpää N, Valtanen M, Warsta L, Setälä H. 2017. Spatiotemporal patterns of major ions in urban stormwater under cold climate. Hydrological Processes. 31(8):1564-1577.

Toor GS, Occhipinti ML, Yang YY, Majcherek T, Haver D, Oki L. 2017. Managing urban runoff in residential neighborhoods: Nitrogen and phosphorus in lawn irrigation driven runoff. PLoS One. 12(6):e0179151.

Tromboni F, Dodds WK. 2017. Relationships between land use and stream nutrient concentrations in a highly urbanized tropical region of Brazil: Thresholds and riparian zones. Environmental Management. 60(1):30-40.

Uriarte M, Yackulic CB, Lim Y, Arce-Nazario JA. 2011. Influence of land use on water quality in a tropical landscape: a multi-scale analysis. Landscape Ecology. 26(8).1151.

Vallejo BMV JR, Aloy AB, Ong PS, Tamino A, Villasper J. 2008. Spatial patterns of bird diversity and abundance in an urban tropical landscape: The University of the Philippines (UP) Diliman Campus. Science Diliman. 20(1):1-10

Vallejo Jr B, Aloy AB, Ong PS. 2009. The distribution, abundance and diversity of birds in Manila's last greenspaces. Landscape and Urban Planning. 89:75-85.

Vallejo BMV Jr, Aloy, AB. 2014. Responses of the bird community in the University of the Philippines Diliman after campus redevelopment and the decline of two common urban bird species. Philippine Science Letters. 7(1):55-61.

Violin CR, Cada P, Sudduth EB, Hassett BA, Penrose DL, Bernhardt ES. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications. 21(6):1932-1949.

Walsh CJ, Roy AH, Feminella, JW, Cottingham PD, Groffman PM, Morgan RP. 2005. The urban stream syndrome: Current knowledge and the search for a cure. Journal of the North American Benthological Society. 24:706-723.

Walsh CJ, Waller KA, Gehling J Mac Nally R. 2007. Riverine invertebrate assemblages are degraded more by catchment urbanisation than by riparian deforestation. Freshwater Biology. 52:574-587.

Walsh CJ, Webb JA. 2016. Interactive effects of urban stormwater drainage, land clearance, and flow regime on stream macroinvertebrate assemblages across a large metropolitan region. Freshwater Science. 35(1):324-339.

Weijters MJ, Janse JH, Alkemade R, Verhoeven JTA. 2009. Quantifying the effect of catchment land use and water nutrient concentrations on freshwater river and stream biodiversity. Aquatic Conservation: Marine and Freshwater Ecosystems. 19(1):104-112.

Yazdian H, Jaafarzadeh N, Zahraie B. 2014. Relationship between benthic macroinvertebrate bio-indices and physicochemical parameters of water: A tool for water resources managers. Journal of Environmental Health Science and Engineering. 12:30-30.

Yule CM, Yong HS. 2004. Freshwater Invertebrates of the Malaysian Region. Kuala Lumpur: Akademi Sains Malaysia.

Zeybek M, Kalyoncu H, Karakas B, Özgul S. 2014. The use of BMWP and ASPT indices for evaluation of water quality according to macroinvertebrates in Deðirmendere Stream (Isparta, Turkey). Turkish Journal of Zoology. 38:603-613.

Zimmerman MC. 1993. The use of the biotic index as an indication of water quality. Tested Studies for Laboratory Teaching. 5:85-98.

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