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PRIMARY RESEARCH PAPER

Impacts of urbanization on stream habitats and macroinvertebrate communities in the tributaries of Qiangtang River, China

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Abstract The impacts of watershed urbanization on streams have been studied worldwide, but are rare in China. We examined relationships among watershed land uses and stream physicochemical and biological attributes, impacts of urbanization on overall stream conditions, and the response pattern of macroinvertebrate assemblage metrics to the percent of impervious area (PIA) of watersheds in the middle section of the Qiantang River, Zhejiang Province, China. Environmental variables and benthic macroinvertebrates of 60 stream sites with varied levels of watershed urban land use were sampled in April, 2010. Spearman correlation analysis showed watershed urbanization levels significantly correlated with increased stream depth, width, and values of conductivity, total nitrogen, ammonia, phosphate, calcium, magnesium, and chemical oxygen demand for the study streams. There was significant difference in total taxa richness, Empheroptera, Plecoptera, and Trichoptera (EPT) taxa richness, and Diptera taxa richness, percentages of individual abundances of EPT, Chironomidae, shredders, filterers, and scrapers, and Shannon–Wiener diversity index between reference streams and urban impacted streams. In contrast, percentages of individual abundances for collectors, oligochaeta, and tolerant taxa, and biotic index were significantly higher in urban impacted than reference streams. All the above metrics were significantly correlated with PIA. The response patterns of total taxa richness, EPT taxa richness, and Shannon–Wiener diversity index followed a drastic decrease at thresholds of 3.6, 3.7, and 5.5% of PIA, respectively. Our findings indicate that stream benthic macroinvertebrate metrics are effective indicators of impacts of watershed urban development, and the PIA-imperviousness thresholds we identified could potentially be used for setting benchmarks for watershed development planning and for prioritizing high valued stream systems for protection and rehabilitation.

Keywords Bioassessment · Stream · Urbanization · Benthic macroinvertebrate · Total impervious area · Qiantang River

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Introduction

Landscape urban development is a global phenomenon. Since the late 1970s, China has been undergoing a rapid and widespread urbanization process as a result of its political reform and rapid economic growth (Zhao et al., 2006). However, compared with urbanization in the United States (80.1% urban), Europe (72.2%), Latin America and the Caribbean (77.4%), and South American (81.6%), the urban population in China (40.4%) is still very low (McDonald, 2008). Urbanization in China is expected to grow more rapidly as a result of increased per capita housing demand, population growth, and work-force migration from rural areas to cities. Such trends are expected to continue in the coming decades. Urbanization converts naturally vegetated or agricultural lands to urban environments that increase storm water runoff to streams, which in turn increases the frequency and severity of flooding, accelerated channel erosion, and altered stream channel forms and bed composition (Klein, 1979). Increased runoff and reduced infiltration in stream watersheds lower stream base flows, alter water temperature regimes and energy inputs, and increase loadings of nutrients and toxic substances (Klein, 1979; Booth, 1991; Galli, 1991). These impacts lead to major changes in aquatic biological communities (Weaver & Garman, 1994; Allan, 2004; Zhao et al., 2006; Wang et al., 2007).

Rivers in China are facing very serious water pollution. Water quality of half of regularly monitored stations in major rivers are worse than the Ministry of Environmental Protection standard of Grade III (suitable for the concentrated drinking water source, swimming are and aquaculture), including sites along the Yangtze River, Yellow River, Pearl River, Haihe River, Huaihe River, Liaohe River, and the Songhuajiang River (Men, 2009). Annual discharge of industrial wastes and domestic sewage into Yangtze River is over 20 billion t, accounting for over 42% of that for the whole country (Chen et al., 2009), while 200 serious pollution events have been recorded in the Huaihe River drainage since 1989 (Zhang et al., 2010b).

Water pollution is one of the major threats to freshwater biodiversity (Dudgeon, 2000). In Yangtze River, water pollution associated with habitat fragmentation, dam construction and overexploitation had severe impacts on fish diversity. For instance, in the Lower Yangtze, there were 230 odd fish species in the period from 1973 to 1975, while only 74 fish species have been found since 2000 (Duan et al., 2007). An accident of chemical wastes spill in upper Yangtze branch resulted in 481 tons of fish death (Chen et al., 2009). However, research on impacts of human disturbance on aquatic organisms, and relations between water pollution and freshwater biodiversity in China is rather insufficient (Chen et al., 2009). Thereby, the impacts of urbanization and impervious area in watersheds on stream benthic macroinvertebrates have far received comparatively little study in China, while many such studies are found in North American (Wang et al., 1997, 2003; Carter et al., 2009; Cuffney et al., 2010; King et al., 2011), Australian (Walsh et al., 2001; Davies et al., 2010), and South American (Ometo et al., 2000; Miserendina et al., 2008). Zhang et al. (2010a) found that urban and rural land use had significant negative impact on water quality and macroinvertebrate diversity in East River in southern China. Although the impacts of urban land use on stream water quality, physical habitat, and biological communities are well recognized in other countries, such impacts have not been thoroughly documented in China. Walsh et al. (2005) termed "urban stream syndrome" to describe the consistently physical, chemical, and biological degradation of streams draining urban landscapes based on the reports from developed countries. Whether the urbanization impacted streams in the Qiangtang River watershed in China have the similar urban syndromes as Walsh et al. (2005) described and characteristic as Zhang et al. (2010a) found in East River is still unknown.

The quantification of relationships among urban land use and water quality, physical habitat, and biological communities and the establishment of watershed urban development thresholds beyond which stream system will be degraded consistently are still in their infancy stage in China. The response patterns of macroinvertebrate assemblage metrics to intensity of urbanization has been described as linear (Moore & Palmer, 2005; Cuffney et al., 2010, King et al., 2011), wedge-shaped (Booth, 2005; Paul et al., 2009), or sharply linear-to-gradual (Walsh et al., 2005). The percent of impervious area (PIA) has been widely used as a measure of intensity of urbanization (Wang et al., 2003; Walsh et al., 2005; Purcell et al., 2009). Studies have found different response patterns of macroinvertebrate assemblage metrics, such as total taxa richness (Morse et al., 2003; Ourso & Frenzel, 2003), Empheroptera, Plecoptera, and Trichoptera (EPT) taxa richness (Morse et al., 2003; Ourso & Frenzel, 2003; Walsh et al., 2007; Cuffney et al., 2010), Shannon–Wiener diversity index (Stepenuck et al., 2002) declined when the PIA reached thresholds from 4 to 12%. Although PIA would be a useful predictor of water quality and river health in East river (Zhang et al., 2010a) and the threshold of PIA of watersheds has been recommended as an important criterion of limiting landscape development for the protection of high priority stream ecosystems (Wang et al., 2001; Cuffney et al., 2010), there is no reported quantitative threshold of PIA as a protective benchmark for stream systems in China.

In this study, we examined relationships among watershed urban land use and instream physicochemical and benthic macroinvertebrate assemblage metrics, described the characteristics of stream condition using the response pattern of stream macroinvertebrate metrics along a rural–urban landscape gradient, and identified the thresholds of macroinvertebrate metrics in responding to PIA gradients in the middle section of the Qiantang River basin, Zhejiang Province, China. The overall goal of the study was to evaluate how physicochemical and biological properties are influenced by landscape natural and anthropogenic factors so that adequate management policy and practices can be implemented to minimize the impacts of urbanization on stream systems.

Materials and methods

Study area

The study area spanned 20,000 km^2 (28.242°–29.683°N, 118.021°–120.726°E) of the middle section of the Qiantang River basin in southwestern Zhejiang Province of China (Fig. 1). The study area is characterized by subtropical monsoon climate, with a mean annual temperature of 17°C and annual precipitation of 1,600 mm that mostly occurs between May and June. The discharge of Qiantang River peaks from May to June and again in September, which is closely associated with the precipitation pattern (http://www.hangzhou.gov.cn/). Land uses in the study area were dominated by forest, but much of the upper Qiantang River region has been modified by aggregated mining, agricultural practice, and urban development during the last several decades (Lin, 2001; Yao, 2007). The impacts of anthropogenic activities are minimal for the majority of the northwestern region, and much of the remaining regions are dominated by agricultural and urban land uses.

We measured physical habitat and sampled water quality and macroinvertebrates from 60 second- to third-order stream sites with watershed areas ranging from 1 to 70 km^2 (average of 15 km^2). The watersheds associated with the sampling sites were relatively similar in geomorphology and climate, but differed primarily in watershed-scale land uses and associated variation in stream physical habitat and water quality conditions.

Physical habitat and water quality

For each site, 16 physical and chemical variables were measured along with the collection of benthic macroinvertebrates. One water sample was collected from a riffle of each site before collecting macroinvertebrate samples, from which measurements of chemical oxygen demand (COD) and concentrations of calcium (Ca^{2+}), magnesium (Mg^{2+}), total nitrogen (TN), ammonia nitrogen (NH_4-N), total phosphorus (TP), and phosphate (PO_4-P) were measured in the laboratory according to the Standard Methods for the Analysis of Water and Wastewater (2002). For each site, we also collected a sestonic chlorophyll *a* sample at the same time of collecting water quality samples by preserving 500 ml water sample in a sealed glass bottle that was kept in a dark on ice during transportation. In the laboratory, water samples were filtered onto GF/C glass fiber filters (Waltman, UK) for measurements of chlorophyll *a* using a spectrofluorophotometer (RF-5301PC, Shimadzu Corporation, Japan). Isolation and measurement of photosynthetic pigments followed the methods developed by Yan et al. (2004).

At each site, water temperature, pH, and conductivity were measured using a portable meter HI98129 (HANNA, Italy). Dissolved oxygen (DO) concentrations were measured using HI 9147 (HANNA, Italy). Water current velocity, mean depth, and wetted width were quantified on transects with equal distance interval across channel sections (Song et al., 2009). Current velocity was measured using a flow probe (FP101, Global Water).

Additionally, we measured the substrate particle size distribution at each site based on methods

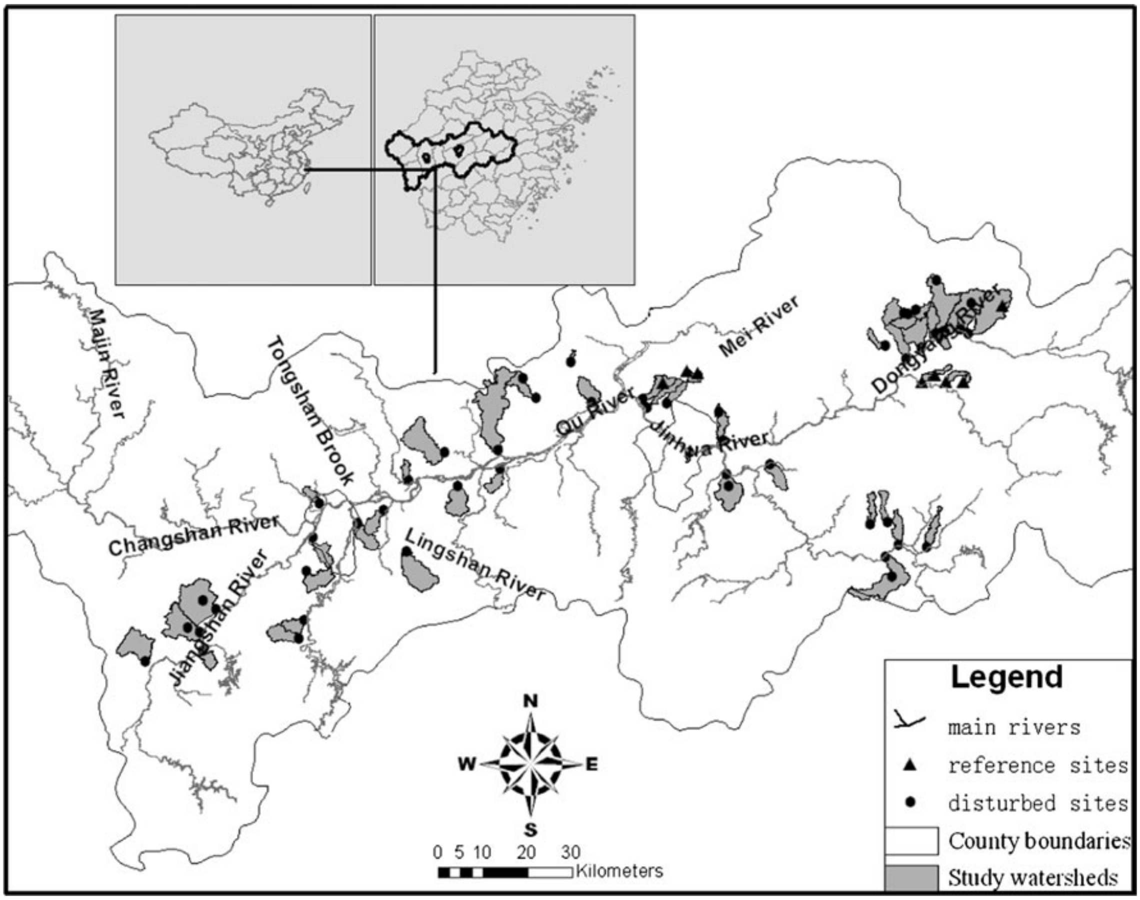

Fig. 1 Map of the 60 sampling stream sites within the Qiangtang River basin, Zhejiang Province of China

described by Wolman (1954) and Kondolf (1997). We recorded the number of each category of substrate ranging from clay to boulder (clay, silt, sand <2 mm, gravel 2–4 mm, pebble 4–64 mm, cobble 64–256 mm, boulder >256 mm) based on the improved Wentworth scale. From the substrate particle size distribution data, we calculated the mean substratum composition (MSUBST) in PHI units for all the 60 sampling stream sites.

Benthic macroinvertebrate

Macroinvertebrate data were collected using a quantitative method, taken with a Surber net (0.09 m^2, 250 μm mesh size). Samples were collected in a 100 m reach for each site in April 2010. At each stream reach, three samples were taken from riffles and two from pools with a total sampling area of 0.45 m^2. The riffle samples were collected by placing the net on the stream bottom and disturbing an area of 0.09 m^2 immediately upstream of the net to dislodge and wash macroinvertebrates into the net. Individual rocks in the sampled area were picked up and attached bugs removed. This process was repeated in three locations within the same riffle or different riffles. The two pool samples were collected by placing a net in the bottom of the water column to remove the top 3–5 cm of bottom sediment which was then washed through a 500-μm sieve using the processes that were similar to riffle sampling. If riffles were not present, all five samples were taken from pools. All the five samples either from riffles and/or pools were combined in the field.

Materials collected in the sampling net were rinsed in the stream to remove fine sediments; all remaining materials in the net were placed into a plastic sampling bag with 10% formaldehyde. In the laboratory, all the collected organisms were hand picked with the aid of a 10× dissecting microscope. All the picked organisms were counted and identified to species or to the lowest taxonomic level possible according to the reference, except for midges (Diptera: Chironomidae) that were mounted on slides in an appropriate medium and identified to subfamily or genus level (Morse et al., 1994).

Watershed land-use delineation and PIA calculation

We delineated watershed boundaries upstream of each sampling site using ARC/INFO 9.3.1 (ESRI, 2008) automated procedures based on a Digital Elevation Model with a 25-m resolution. These sub-watershed boundaries were manually verified and corrected by referencing 1:50,000 digital topographic maps. We quantified the land use within each watershed upstream of the sampling site by overlaying watershed boundaries on the top of a regional land-use vector layer. This land-use layer is a land cover map that was originated in 2007 for Zhejiang Province, and was collected by the Landsat Thematic Mapper with a 30-m resolution using ERDAS Imagine software, version 9.1 (ESRI, 2006). Additionally, we digitized the land uses of doubtful watersheds with the aid of Google Maps (2009). The Satellite Maps of the study region were produced from 2005, 2009, and 2010, with a ground resolution 2.5 m or less. By the combination of ARC/INFO and Google Earth processes, we digitized and refined all the watershed land uses for the study area.

From this land-use data, we summarized watershed land uses into forest, farmland, urban, and imperviousness and expressed them as percentages of the watershed area. Forest included evergreen, deciduous, and mixed forests. Farmland contained all types of croplands and exposed land. Urban land included low, medium, and high density residential, industrial, commercial, parking lot, public utility, green space, parks, and associated transportation systems. Impervious area was calculated from urban land and PIA was a sum of percentages of impervious land from different urban land-use categories. The percentage of imperviousness of each urban land-use category was determined from published data (Walsh et al., 2005; Zhang et al., 2010a) and our field measurements.

Data analyses

From the macroinvertebrate data, we calculated 16 assemblage metrics (Table 4). The total taxa richness and that of individual taxa groups were calculated based on the taxonomic classification; pollution tolerance and feeding groups were calculated following Morse et al. (1994) and Wang & Yang (2004); the Shannon–Wiener diversity index was based on Magurran (1988); and the biotic index was based on Hilsenhoff (1987).

Spearman correlation analyses (SPSS Inc., 1998) were used to explore the relationship among stream physicochemical variables and the percentages of forest, farmland, and urban land uses. We divided our sampling sites into reference streams and non-reference streams. We identified eight reference streams that had no mining activities, no more than five scattered houses, and less than 0.5% farmland in their watersheds. The 52 streams were non-reference streams (hereafter refers as disturbed streams). We used Kruskal–Wallace test to examine the differences between reference and disturbed streams in responses of macroinvertebrate metric values to the levels of urban land use. The response thresholds of total taxa richness, EPT taxa richness, and Shannon–Wiener diversity index to PIA were identified using LOWESS smoothing (R Development Core Team, 2008).

Results

Land use

Land use varied considerably between reference and urbanization disturbed sites (Table 1). Reference sites had high forest cover with an average of 99.9% and low level of urban development (<0.01%). For the disturbed sites, forest ranged from 7.1 to 95%, and urban land varied from 0.2 to 81.6%. The mean PIA for reference sites was less than 0.01%, and disturbed sites had mean PIA of 19.0%.

Relationship among land uses and physicochemical variables

The study sites had varied levels anthropogenic land uses (Table 1) and physical habitat and water quality values (Table 2). Among the 60 sites, land uses ranged

Table 1 Summary statistics of watershed area and land use for reference and disturbed streams

Variables	Reference ($n = 8$)			Disturbed ($n = 52$)		
	Min.	Max.	Mean	Min.	Max.	Mean
Watershed (km^2)	1.3	5.0	2.9	1.0	70.2	16.8
Forest (%)	99.7	100.0	99.9	7.1	95.0	52.5
Farmland (%)	0.0	0.2	0.1	2.6	87.0	27.7
Urban (%)	0	0.1	<0.01	0.2	81.6	19.0
PIA (%)	0.0	0.0	0.0	0.2	81.6	19.0

Table 2 Summary statistics of physical–chemical variables and their Pearson correlation coefficients with forest, farmland, and urban land uses for the 60 sampling sites

Variables	Min.	Max.	Mean	Forest (%)	Farmland (%)	Urban (%)
Temperature (°C)	11.3	27.3	18.3	−0.204	0.197	0.117
pH	5.1	8.6	7.3	0.112	−0.119	−0.028
DO (mg/l)	1.8	10.5	7.8	0.726**	−0.642**	−0.629**
Substrate (PHI units)				−0.591**	0.415**	0.55**
Width (m)	1.0	23.3	5.1	−0.431**	0.335**	0.446**
Depth (m)	0.07	2.50	0.48	−0.390**	0.265*	0.394**
Average velocity (m/s)	0.00	1.11	0.43	0.213	−0.103	−0.175
Conductivity (μs/cm)	30	1,668	243	−0.634**	0.376**	0.7**
TN (mg/l)	0.22	26.50	3.80	−0.722**	0.514**	0.69**
NH_4-N (mg/l)	ND	19.00	1.56	−0.782**	0.562**	0.77**
TP (mg/l)	0.004	2.350	0.380	0.1	−0.14	−0.073
PO_4-P (mg/l)	0.003	0.097	0.015	−0.668**	0.465**	0.698**
COD (mg/l)	0.90	14.65	4.01	−0.839**	0.635**	0.755**
Chlorophyll *a* (mg/m^3)	ND	17.62	2.62	−0.710**	0.592**	0.638**
Mg (mg/l)	0	21	5.2	−0.343**	0.335**	0.281**
Ca (mg/l)	22	221	113.62	−0.615**	0.429**	0.611**

ND Not detectable

* $P < 0.05$; ** $P < 0.01$

from 7 to 100% for forest (average = 59%), from 0 to 87% for farm lands (average = 24%), and from 0 to 80% for urban lands (average = 16%). DO varied from less than 2 mg/l to greater than 10 mg/l; conductivity ranged from 30 to 1,668 μs/cm; and COD varied from 0.9 to 4.0 mg/l. As our reference selection criteria determined, reference streams sites had the highest forest (>99%) and least farmland and urban land uses (<0.2%).

Spearman correlation analysis implied (Table 2) that the stream depth, width, conductivity, TN, NH_4-N, PO_4-P, Ca^{2+}, Mg^{2+}, COD, and chlorophyll *a* positively correlated with farmland and urban land ($P < 0.05$) and negatively correlated with forest land ($P < 0.05$). In contrast, substrate size and DO positively correlated with forest ($P < 0.05$), and negatively correlated with farmland and urban land uses ($P < 0.05$). Water temperature, pH, TP, and current velocity had no significant relationship with the measured land uses.

Kruskal–Wallace test (Table 3) showed that conductivity, TN, NH4-N, PO_4-P, COD, Ca^{2+}, and Mg^{2+} were higher in disturbed streams than in reference streams ($P < 0.05$). Substrate size was smaller and DO concentration was lower in disturbed streams than in reference streams ($P < 0.05$). However,

Table 3 Mean and range of physical–chemical variables for disturbed and reference streams

	Disturbed ($n = 52$)	Reference ($n = 8$)		Disturbed ($n = 52$)	Reference ($n = 8$)
Average depth (m)	0.52 (0.07–2.5)	0.23 (0.15–0.36)	TN (mg/l)[a]	4.28 (0.41–26.50)	0.65 (0.22–1.28)
Average width (m)	5.6 (1.0–23.3)	2.4 (1.4–3.7)	NH_4-N (mg/l)	1.80 (0.00–19.00)	ND
Average velocity (m/s)	0.43 (0.00–1.11)	0.45 (0.17–0.96)	TP (mg/l)	0.380 (0.000–2.350)	0.360 (0.100–0.930)
MSUBST (PHI units)	−0.3 (−7.8–8.0)	−6.1 (−7.8 to −3.6)	PO_4-P (mg/l)[a]	0.017 (0.004–0.097)	0.004 (0.003–0.004)
Temperature (°C)	18.5 (11.3–27.3)	17.1 (15.1–19.6)	Ca (mg/l)[a]	122.8 (41.0–221.0)	52.9 (22.0–89.0)
pH	7.3 (5.1–8.6)	7.1 (6.4–7.9)	Mg (mg/l)[a]	5.7 (0.0–21.0)	1.5 (0.0–5.0)
DO (mg/l)[a]	7.5 (1.8–10.5)	9.5 (9.2–9.9)	Chlorophyll *a* (mg/l)[a]	3.00 (0.001–17.60)	0.110 (0.000–0.410)
Conductivity (µs/cm)[a]	273 (30–1,668)	48 (31–90)	COD (mg/l)[a]	4.41 (1.06–14.65)	1.41 (0.90–2.99)

[a] Variable values were significantly different between disturbed and reference streams using Kruskal–Wallace test ($P < 0.01$)

temperature, pH, TP, and current velocity were no significant different between reference and disturbed streams.

Macroinvertebrate assemblages and their responses to urban land use

A total of 235 taxa were collected. The highest taxa diversity was found for orders of Diptera (62), followed by Ephemeroptera (53), Coleoptera (32), Trichoptera (32), and Odonata (18). Total taxa richness, EPT taxa richness, and Diptera taxa richness of reference streams were significantly higher than that of disturbed streams (Fig. 2). Percentages in abundance of shredders, filterers, and scrapers for disturbed streams were significantly lower than that for the reference streams, while the relative abundance of collectors was significantly higher in disturbed than in reference streams. Percentages in abundance of EPT, tolerant taxa, Chironomidae, and oligochaeta differed significantly between the reference and disturbed streams. Shannon–Wiener diversity index and BI also differed significantly between the reference and disturbed streams. The relative abundance of intolerant taxa and the density of all taxa showed no significant difference between reference and disturbed sites.

All the tested macroinvertebrate metrics, except for the density of all taxa, were significantly ($P < 0.05$) correlated with the percentages of the three types of watershed land uses (Table 4). Among the 15 significantly correlated macroinvertebrate metrics, 11 were negatively correlated with watershed farmland and urban land uses, while the relative abundances of tolerant taxa, oligochaeta, and collector individuals, and BI positively correlated with farmland and urban land uses. In contrast, only relative abundances of tolerant taxa, oligochaeta, and collector individuals, and BI significantly negatively correlated with watershed forest land use, while the rest of the 11 metrics were positively correlated with forest land use.

The relationships among watershed PIA and the three macroinvertebrate metrics showed clear threshold patterns (Fig. 3). The response patterns of total taxa richness, EPT taxa richness, and Shannon–Wiener diversity index identified by LOWESS smoothing fell into the form of sharply linear-to-gradual as PIA increased from the lower to upper thresholds. Such patterns imply that before urban imperviousness reached its lower threshold, macroinvertebrate condition was mainly influenced by non-urban factors; between the lower and upper impervious thresholds, macroinvertebrate metrics values declined drastically as imperviousness values increase; and beyond the upper threshold of urban imperviousness, macroinvertebrate metrics values were consistently low. The lower PIA thresholds identified for total taxa richness (3.6%) and EPT taxa richness (3.6%) were smaller than Shannon diversity index (5.5%). In contrast, the upper thresholds for total taxa richness (16.2%) and EPT taxa richness (15.1%) were higher than for Shannon diversity index (10.8%).

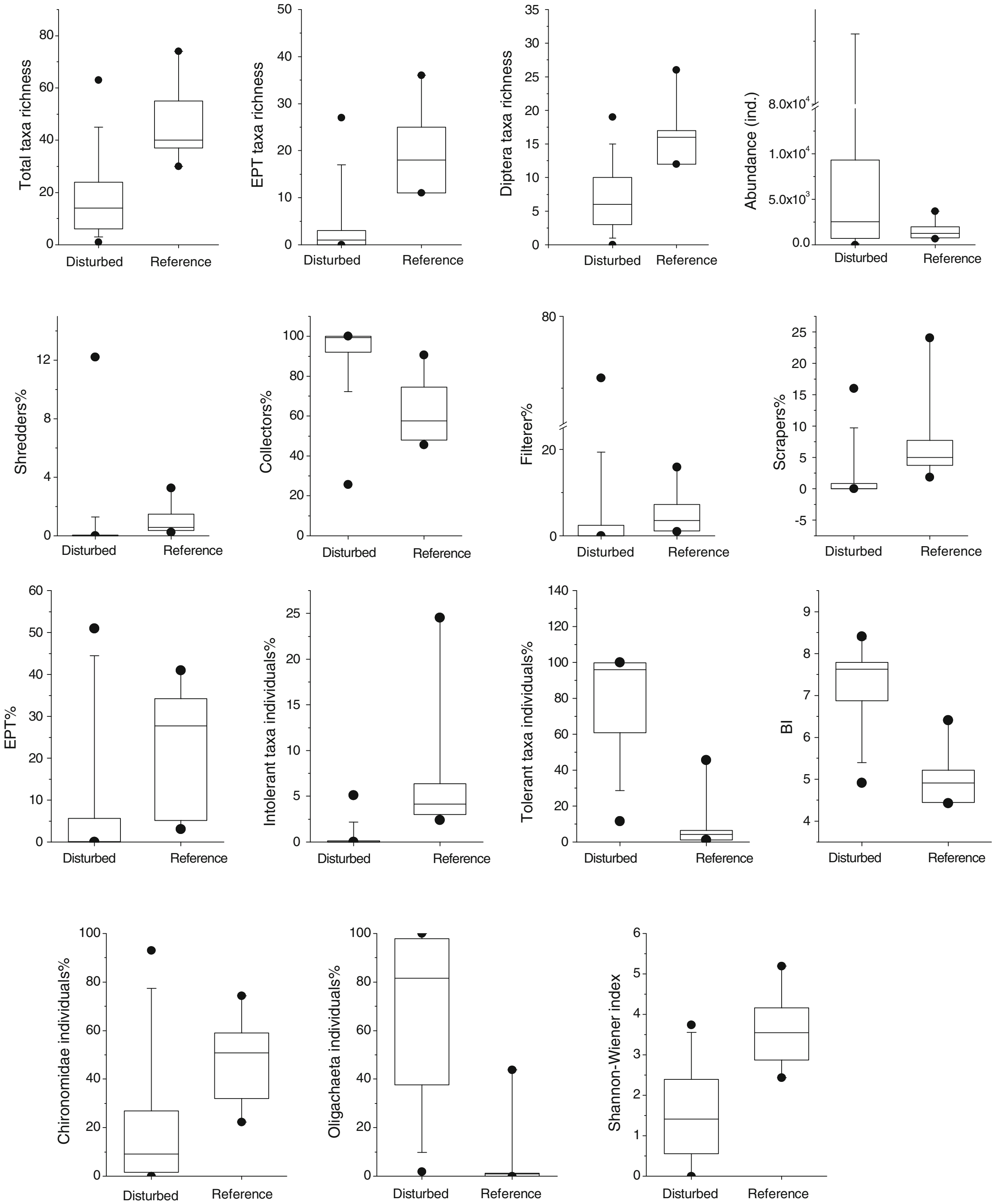

Fig. 2 Box plots of the 15 macroinvertebrate metrics for the 52 disturbed and 8 reference streams. *Ranges of bars* show 5 and 95% percentile values; *boxes* are interquartile ranges (25–75% percentiles); *lines within the boxes* are medians; and *dots* are maximum and minimum values. All metrics, but total abundance, were significantly different between the disturbed and reference streams based on Kruskal–Wallace test ($P < 0.05$)

Table 4 Coefficients of Spearman correlation among percentages of land uses and macroinvertebrate assemblage metrics

	Forest (%)	Farmland (%)	Urban (%)		Forest (%)	Farmland (%)	Urban (%)
Total taxa richness	0.807**	−0.485**	−0.874**	Diptera individuals (%)	0.608**	−0.296*	−0.694**
EPT taxa richness	0.818**	−0.549**	−0.818**	Total density (#/m^2)	−0.125	0.12	0.142
Diptera taxa richness	0.812**	−0.490**	−0.885**	Shredder individuals (%)	0.750**	−0.584**	−0.756**
Intolerant taxa individuals (%)	0.738**	−0.589**	−0.794**	Collector individuals (%)	−0.715**	0.481**	0.811**
Tolerant taxa individuals (%)	−0.741**	0.496**	0.803**	Filterer individuals (%)	0.612**	−0.404**	−0.691**
EPT individuals (%)	0.820**	−0.523**	−0.802**	Scraper individuals (%)	0.682**	−0.463**	−0.768**
Chironomidae individuals (%)	0.588**	−0.267*	−0.682**	BI	−0.661**	0.472**	0.713**
Oligochaeta individuals (%)	−0.678**	0.422**	0.738**	Shannon–Wiener index	0.759**	−0.458**	−0.824**

* $P < 0.05$; ** $P < 0.01$

Discussion

Effects of land uses on stream chemical and physical properties

Our results demonstrated that watershed land uses had strong impacts on stream chemical and physical characteristics. These impacts include the negative effects of both farmland and urban land on concentrations of various forms of nitrogen and phosphorus, COD, DO concentration, stream substrate, and channel morphology. Our results are consistent with the "urban stream syndrome" described by Walsh et al. (2005) in that many water quality measures at disturbed streams were significantly higher than reference streams, and similar to what have found in East River that urban and farm land use had significantly negative relation to DO and positive relation to TN (Zhang et al., 2010a). However, the TP concentrations were not significantly correlated with percentages of any of the three watershed land uses, and the TP concentrations for the reference (0.36 mg/l) and disturbed streams (0.38 mg/l) were similar. This result differs from the previous findings in the other countries (e.g., Meybeck, 1998; Brett et al., 2005; Wang et al., 2007), implying that our study streams have high background TP concentrations naturally. Higher nutrient concentrations in the disturbed streams increase the growth of algae as indicated by significantly higher chlorophyll *a* concentrations in the disturbed relative to reference streams. This was also indicated by the lower DO levels in the disturbed than in the reference streams as a result of high oxygen demand from high algae biomass during non-photosynthesis and algal died off periods. Magnesium and calcium concentrations were positively correlated with farmland and urban land and negatively correlated with forest land, implying that anthropogenic activities are sources of magnesium and calcium in our study streams.

Our results that urban land use associates strongly with wider stream channels and smaller substrate sizes are consistent with the urban stream syndromes reported elsewhere (Walsh et al., 2005). The positive association of channel width and negative association of substrate size with urban land use in our study and East River (Zhang et al., 2010a) are commonly reported in other studies (Paul & Meyer, 2001; Roy et al., 2003; Wang et al., 2003). The strong associations of channel width and substrate size with urban land use suggest that increased stream bank erosion and sediment distribution resulted from riparian vegetation removal and alteration of stream hydrology from urban land use are main contributors to stream physical habitat alteration.

Macroinvertebrate assemblage

Our results showed that the total taxa richness, EPT taxa richness, and Diptera taxa richness, relative abundances of EPT, Chironomidae, shredders, filterers, and scrapers, and Shannon–Wiener diversity index were significantly higher in reference streams than in disturbed streams. The relative abundances of collectors, oligochaeta, and tolerant taxa and biotic index were significantly lower in reference than in

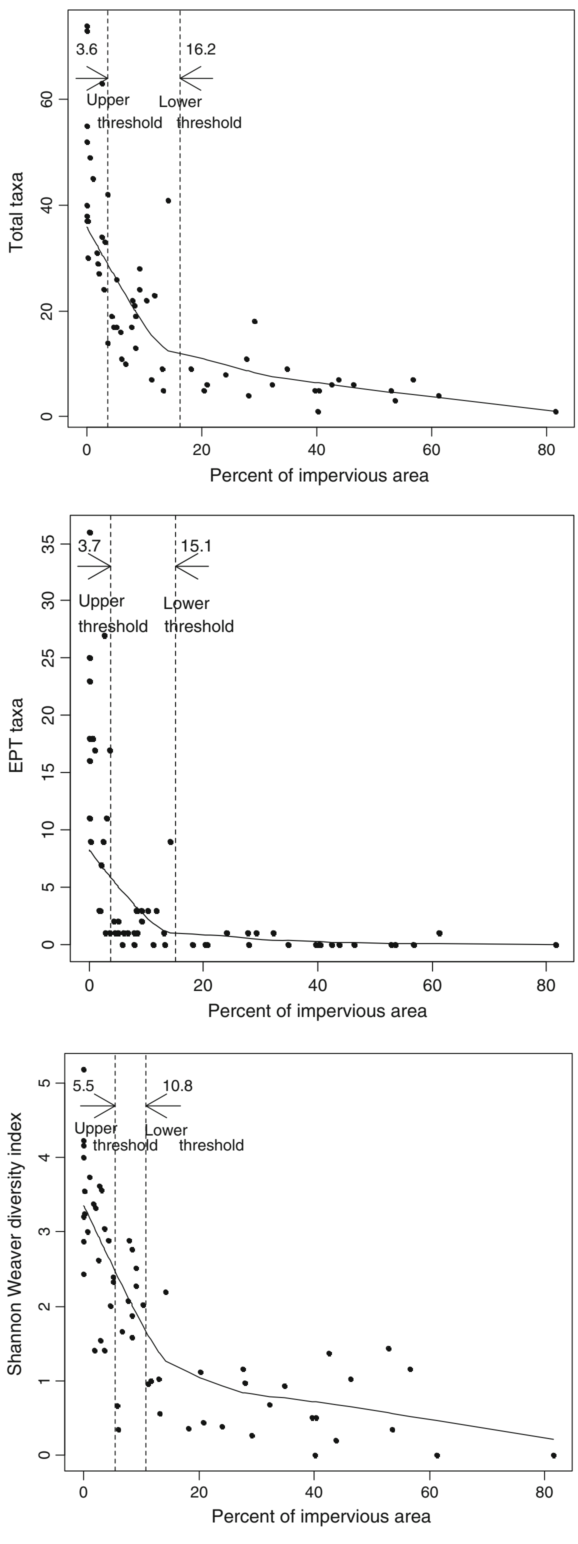

Fig. 3 Response patterns of total taxa richness, EPT taxa richness, and Shannon–Wiener diversity index to PIA

disturbed streams. All above-mentioned metrics were significantly correlated with percentages of farmland, urban land, and PIA. The strong associations among watershed urban land use and macroinvertebrate assemblage metrics are in consistent with the urban stream syndromes described by studies from North America and Europe (Paul & Meyer, 2001; Walsh et al., 2005; Miserendina et al., 2008; Cuffney et al., 2010) and East River in southern China (Zhang et al., 2010a). Our results, in conjunction with other studies, imply that the impacts of watershed urban development on stream biological communities have very similar symptoms worldwide. This conclusion has important implication because the problem of urbanization effects on stream systems in China or elsewhere could be addressed more effectively by borrowing experiences from other countries where successful management techniques and policies have been developed and implemented.

Our study also provided two different patterns compared with other studies. One of the findings is that the total density of all macroinvertebrate taxa is not significantly different between reference and disturbed streams, and is not significantly correlated with either farmland or urban land. Although total density is not a commonly used metric in bioassessment because most macroinvertebrate samplings are semi-quantitative. Roy et al. (2003) has reported a linear relationship between urban land use and total macroinvertebrate density for streams in Georgia, USA. Such a disassociation of macroinvertebrate total density with farmland and urban land uses in our study may be resulted from a strong natural variation among the sampling sites regardless of the disturbance gradients. The second finding is that the relative abundance of Chironomidae had a negative relationship with farmland and urban land uses; their relative abundance is significantly lower in disturbed (19%) than in reference streams (50%); and their average density is higher in reference (917 individuals/m^2) than in disturbed streams (448 individuals/m^2). This finding is different from what have been reported that the abundance of Chironomidae is expected lower in reference than in non-point source disturbed streams (e.g., Gresens et al., 2007). Although we are uncertain what has caused this unexpected result, the low channel gradient nature of the study streams and the undistinguished pollution tolerance among Chironomidae taxa in our analysis could be potential reasons, and deserves further investigation. Our study, in conjunction with the others, indicates that urban land uses are more damaging, on a per-unit-area basis, than agricultural land use.

Percentage of total impervious area in a stream watershed has been considered a key measure of urban land use and is widely used to describe the responsive relationships of aquatic assemblages or individual taxon to the impacts of urbanization (Wang et al., 2000; Allan, 2004; King et al., 2011). In our study, the response patterns of total taxa richness, EPT taxa richness, and Shannon–Wiener diversity index to urban land use were sharply linear-to-gradual that was described by Walsh et al. (2005), but not linear described by Cuffney et al. (2010) or wedge-shaped described by Paul et al. (2009). The patterns of such relationships appear to be related to what types of lands that urban land was converted from (Cuffney et al., 2010). For our study area, almost all the urban lands were converted from farmland. Stream benthic assemblages had likely been previously impacted by farmland before their watersheds were urbanized, and the effect of urbanization on benthic macroinvertebrate assemblages was an addition to the impacts of agricultural land use. Zhang et al. (2010a) also found that family richness had relatively linear relationship with PIA in East River. For watersheds that have low urban and are predominated by farmland, the stream health condition is predominantly influenced by agricultural activities and other non-urban factors. However, when urban land use exceeds its lower threshold, stream condition declines drastically as imperviousness values increase until urban imperviousness reaches its upper threshold; beyond the upper threshold of imperviousness, stream condition is consistently poor.

The response patterns and thresholds of macroinvertebrate metrics to level of urbanization have been recognized as a useful tool for setting benchmark for watershed development planning and for prioritizing high valued stream systems for protection and rehabilitation (Wang & Lyons, 2003; Paul et al., 2009). However, the reported lower and upper impervious thresholds vary considerably depending on the biological metrics used and the geographic region studied. Reported adverse impacts thresholds (equitant to the lower threshold of this study) of percent of watershed total impervious area vary from 5 to 18% (Wang et al., 2000; Paul & Meyer, 2001; Morse et al., 2003; Ourso & Frenzel, 2003; Taylor et al., 2004), and some macroinvertebrate taxa are impacted at as low as less than 2% (King et al., 2011). Our study found that the lower thresholds for Shannon–Wiener diversity index (5.5%), for total taxa richness (3.6%), and for EPT taxa richness (3.7%) are much lower than most of the aforementioned studies and higher than the values reported by King et al. (2011) from Maryland streams in USA. One of the potential causes for the varied urban imperviousness thresholds could be the manner in which imperviousness is measured and calculated. Hence, natural resource managers and researchers should use the thresholds developed specifically for the region they manage or study and use the same method that has been used to measure and calculate imperviousness for developing the threshold when they use urban imperviousness as a tool to set benchmarks for watershed development planning and for prioritizing high valued stream systems for protection and rehabilitation.

As urban areas rapidly expend to meet the needs of increasing in per capital housing demand, population growth, and work-force migration from rural areas to cities in China, it becomes critically important to develop and use biological assessment tools to inform management agencies and to educate public for establishing stream criteria. Our study demonstrates that the major stream health symptoms of watershed urbanization for tributaries of Qiantang River are similar to East River in southern China (Zhang et al., 2010a) and those reported in the other countries. Our study also provides some insights to the rarely studied river system in that watershed urbanization has significantly modified stream channel morphology and increased values of conductivity, TN, ammonia, phosphate, and COD for the middle section of the Qiantang River. Our study also provides evidence that stream benthic macroinvertebrate metrics are effective indicators of impacts of watershed urban development. Such impacts occur at about 4% watershed total impervious area (average from three indicators), which is lower than most threshold values reported from other countries. The urban land-use threshold we identified from relationships among percent watershed total imperviousness and total taxa richness, EPT taxa richness, and Shannon–Wiener diversity index could potentially be used for setting benchmarks for watershed development planning and for prioritizing high valued stream systems for protection and rehabilitation.

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