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Fengqing Li, Qinghua Cai*, Wanxiang Jiang, Xiaodong Qu and Xinghuan Jia

State Key Laboratory of Freshwater Ecology and Biotechnology, Institute of Hydrobiology, Chinese Academy of Sciences, Wuhan, Hubei Province 430072, People's Republic of China
e-mail: qhcai@ihb.ac.cn

Research Paper

Flow-related disturbances in forested and agricultural rivers: influences on benthic macroinvertebrates

key words: flood, temporal variation, land-use type, benthic macroinvertebrate, community composition

Abstract

Flow disturbances and conversions of land-use types are two major factors that influence river ecosystems. However, few studies have considered their interactions and separated their individual effects on aquatic organisms. Using monthly monitoring data from two streams with different land-use types (*i.e.* forest and agriculture) in the subtropical Central China over three years, we accurately predicted the changes of macroinvertebrate communities under flood disturbances and land-use type conversions. The dominant taxa and main community metrics significantly declined following flash floods. Several mayflies and chironomid had rapid rates of recovery, which could reach high abundance in three months after floods. And most of the community metrics recovered more rapidly in the forested river than that in the agricultural river. Stepwise multiple regression (SMR) models were used to investigate the relationships between biotic metrics and hydrological and temporal variables. For example, SMR revealed that floods reduced the stability of benthic communities, and the length of low flow period was of considerable importance to the recovery of the fauna. Two-way ANOVA indicated that intra-annual fluctuation had more (*e.g.* the total abundance and wet biomass), equal (*e.g.* total richness, EPT richness, percent EPT abundance, and Margalef index), or less (*e.g.* tolerant value) influence on macroinvertebrate communities than land-use types. Consequently, the effects of floods on macroinvertebrates should be taken into account when macroinvertebrates are used as indicators for assessing river ecosystem.

1. Introduction

A distinguishing feature of river ecosystems is a high level of spatial and temporal heterogeneity. Their features are manifested as four dimensions, namely lateral, longitudinal, vertical and temporal dimensions (Ward, 1989), with disturbance being the primary heterogeneous driver. Disturbance has also been suggested to be the dominant factor affecting community structure in the river ecosystems (Lake, 2000; Olsen and Townsend, 2005).

Disturbance can differ in the spatial and temporal patterns in their intensity and duration. Their types are recognized (Lake, 2000; Hillman and Quinn, 2002): pulse disturbance (sharp

* Corresponding author

and strong influence in a short time), press disturbance (sharply increase or decrease and then reaching a new maintained constant level) and ramp disturbance (steady increase or decrease over time and without reaching an equilibrium endpoint). In most situations, a discrete disturbance regime is unlikely. Because most ecosystems are more likely to be dominated by a major type, and other mixed types. For example, an agricultural river may interact with ramp disturbances caused by pesticide use, and pulse disturbance may result from floods.

Flooding is a fundamental source of disturbances in many lotic ecosystems (Hillman and Quinn, 2002). Several experimental (Death, 1996) and observational studies (Death, 2002; Collier and Quinn, 2003) have suggested that increasing the strength of flood disturbances could temporarily reduce the abundance and diversity of periphyton and macroinvertebrate. Furthermore, flood disturbances could also influence production, consumption, and decomposition with consequences for food web and energy flow in river ecosystems (Chiu *et al.*, 2008; Rader *et al.*, 2008).

Widespread changes in land-use types have been shown to potentially alter habitat suitability for stream species (Hall *et al.*, 2001; Miserendino *et al.*, 2011). For example, deforested practices can increase the amount of surface runoff and sediment introduced to the stream systems (Walling and Fang, 2003). Frequently water quality, especially concentrations of nutrients, is affected both directly and indirectly by land-use conversions. This can, in turn, increase nutrient concentrations that cause overgrowth of algae and aquatic plants (Bixby *et al.*, 2009) which potentially affect the habitat suitability for endemic fauna (Richards and Host, 1994; Miserendino *et al.*, 2011). Some sensitive species, for instance, EPT (Ephemeroptera, Plecoptera, and Trichoptera), were found negatively correlated with open agricultural rivers (Hall *et al.*, 2001; Collier *et al.*, 2008). The loss of these sensitive species due to land-use exploitations can be critical to preserving macroinvertebrate diversity (Miserendino *et al.*, 2011).

As a link between terrestrial and aquatic ecosystems, streams may also influence many organisms in the riparian zone. The important linkages between aquatic and terrestrial ecosystems have been increasingly highlighted (Chan *et al.*, 2007; Li and Dudgeon, 2008). Not only the linkage of food webs between fish or macroinvertebrates and riparian communities (Chan *et al.*, 2007; Chiu *et al.*, 2008; Li and Dudgeon, 2008), but different land-use types have also been given as important factors that affect river habitat and aquatic communities (Richards and Host, 1994). Several studies have documented the change of benthic macroinvertebrate communities after a single severe flood (Collier and Quinn, 2003; Chiu *et al.*, 2008), but few studies have evaluated the interaction of land-use types and floods on long-term macroinvertebrate communities. Thus, in this study, we utilized the flood events to investigate how macroinvertebrates responded to the combined stressors of land-use types and floods.

Flows in the Chinese mainland river systems are naturally highly variable owning to the monsoon floods (Zhang *et al.*, 2005; Jiang *et al.*, 2006). Changes of land-use can reduce the predictability of floods in the subtropical regions, causing the deforested rivers to flood annually and severely (Jiang *et al.*, 2006). However, few studies report on the response of aquatic organisms to flood disturbance in the Asian monsoon region. Thus, we assessed the structure of benthic macroinvertebrates in the Xiangxi River watershed, and attempted to (1) evaluate the responses of macroinvertebrates to monsoon floods, and especially which type of land-use was expected to recover rapidly from the flooding, and (2) assess the differences in macroinvertebrate community with respect to recovery from flood disturbances in watersheds of two land-use types.

2. Materials and Methods

2.1. Study area and sampling sites

The present study was conducted in the Xiangxi River watershed, the largest tributary of Three Gorges Reservoir in Hubei province, Central China. The river originates from the mountains of Shen-

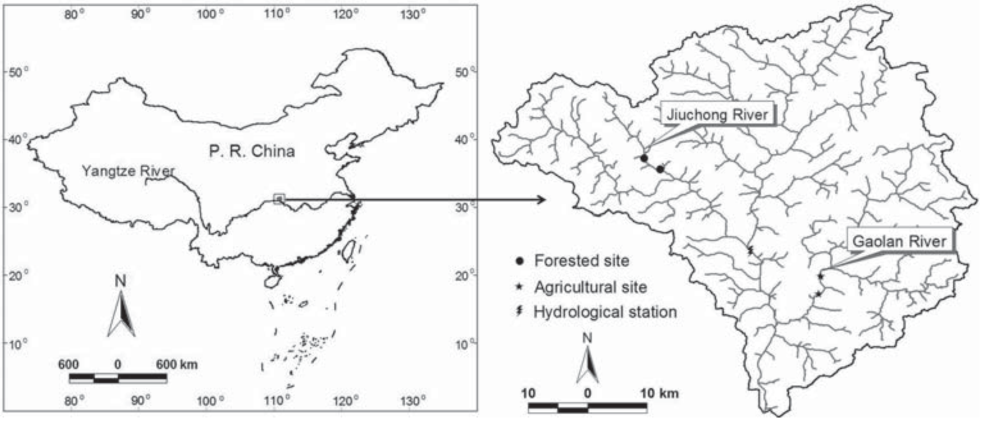

Figure 1. Locations of the sampling sites in the Xiangxi River in the People's Republic of China.

nongjia Nature Reserve, and flows for 94 km before joining the Yangtze River. The watershed area of the Xiangxi River is 3099 km^2, and the mean slope of 23.2° is from the headwaters to its confluence with the Yangtze River (Qu *et al.*, 2005; Wu *et al.*, 2009). The average annual precipitation in this watershed is 900–1200 mm. The Jiuchong River and the Gaolan River are two main tributaries of the Xiangxi River (Fig. 1; Li *et al.*, 2009a). According to the study of land-use types in the Xiangxi River watershed by Ye *et al.* (2009), the Jiuchong watershed is mainly covered with forests, while the Gaolan watershed is dominated by farmlands. Four sites were selected for this study based on the land-use characteristics. Two forested sites were established on the downstream of the Jiuchong River (3rd order, evergreen broadleaved forest accounts for 92% within a 100 m × 1000 m buffer range of the sampling sites), while two agricultural sites are located at the downstream of the Gaolan River (4th order, cropping accounts for 61% within a 100 m × 1000 m buffer range of the sampling sites). The elevations of this four sites range from 332 m to 623 m.

2.2. Flow-related data

Because there is only one hydrological station, these data cannot be used for the studied sub-catchments. Instead, watershed areas of the sampling sites, and those of the hydrological station site, were calculated using ArcGIS (version 9.2). Based on the watershed areas and the multi-year (1959–2007) average daily flows from the hydrological station, the water flow of each sub-watershed was calculated with the conversion method according to linear equation between the area and flow data.

Two variables were used to describe the relationships between flow-related variables and biotic metrics: mean flow of the sampling month (Q mean); the number of days since last flood (N days), where a flood criterion was arbitrarily defined as one where flows exceeded 150 $m^3 s^{-1}$ at the hydrological station. This high flow was around four times more than multi-year average flow (37.11 $m^3 s^{-1}$) (Clausen and Biggs, 2003). The strength of such flow can move medium-small sized substrates, and produce ecological consequences for the diversity of benthic algae and macroinvertebrates (Suren and Jowett, 2006).

2.3. Physicochemical variables and chlorophyll a

At each site, water temperature, river width, water depth, current velocity, total nitrogen (TN), total phosphorus (TP) and chlorophyll *a* were measured monthly from July 2004 to June 2007. No samples were collected in April 2007 in the forested river because of a landslide. The seasons in north subtropical China are: summer (June to August), and winter (December to February).

Depth and velocity were measured at around 10 equally spaced transects (about 1 m) in each site, according to the width of the river. Water was sampled in sulphuric acid-washed (pH < 2) plastic bottles, which were immersed at least 30 cm below the water surface (if the depth allowed), filled and capped to exclude air, preserved by adding sulphuric acid (analytical grade) to adjust pH to < 2, and then stored at 4 °C for physicochemical analyses (Huang *et al.*, 2000). Water temperature, pH and conductivity were measured *in situ* with a HACH instrument. The TN was determined using the alkaline potassium persulphate digestion-ultraviolet spectrophotometric method (at 220 and 275 nm; GB11894–89), and the TP was measured using the ammonium molybdate spectrophotometric method (at 700 nm; GB11893–89) in the laboratory.

Benthic algae, from three randomly selected rocks (diameter, around 25 cm) from each site, were scrubbed from substrates (23.75 cm^2 with a round cover) thoroughly with a brush and then rinsed several times with distilled water. We composited samples from each set of rocks into a single sample, and then stored samples in the dark in an ice container until we processed them in the laboratory. Chlorophyll *a* was extracted in 10 ml of 90% acetone for 24 h at 0 ºC, and pigment concentrations were measured spectrophotometrically at 630, 645, 665 and 750 nm (APHA, 1989).

2.4. Benthic macroinvertebrates

Benthic macroinvertebrates were sampled monthly over the study period. A 0.42 mm mesh Surber sampler (30 cm × 30 cm) was used to take three replicates at different habitat types (*i.e.* riffle, run and pool) within a 100 m reach. All stones within the sampler frame were scrubbed with a soft brush to remove attached organisms. In areas of unconsolidated substrates, the river bed was sampled to a depth of about 10 cm. The samples were pooled and preserved in 10% formalin. Biological samples were identified to genus or species following Kawai (1985) and Morse *et al.* (1994). All individual invertebrates were weighed using a Sartorius balance (precision 0.1 mg) for obtaining wet weights.

2.5. Data analysis

Calculations of total abundances (numbers of organisms per 1 m^2), wet biomass (weights per 1 m^2), richness (numbers of species in the sampler), and diversity (Margalef biodiversity index, which was calculated as: $\frac{S-1}{\log_2 N}$, where S is the number of species in the sample, and N is the total abundance in the sample) were straightforward. One-way ANOVA was conducted to compare the differences of physicochemical variables between flow states and land-use types.

Stepwise multiple regression (SMR) analysis was used to predict the effects of hydrological variables (*i.e.* Q mean, river width, water depth and current velocity), and temporal variable (*i.e.* the number of days since last flood) on periphyton biomass (chlorophyll *a*). We used macroinvertebrate metrics (*i.e.* dominant taxa, total abundance, wet biomass, total richness, EPT richness, percent EPT abundance, and Margalef index) (Collier and Quinn, 2003; Suren and Jowett, 2006). Stepwise methods were used in both SMR models (using $P = 0.10$ for entry into the model), and the model with the highest predictive power was finally chosen.

Two-way ANOVA were performed using month and land-use types as factors for total abundance, wet biomass, total richness, EPT richness, percent EPT abundance, Margalef index, and tolerant value. Here we selected the tolerant value because it related to species resistance and resilience (Parkyn and Collier, 2004). The interaction term was used to assess whether consistent temporal responses occurred between land-use types.

3. Results

3.1. Habitat characteristics

Daily flow at the Xiangxi River hydrological station varied greatly over the study period, ranging from 1.02 to 435 $m^3 s^{-1}$. Hydrographs were characterized by higher summer flows (average daily flow was 122.36 $m^3 s^{-1}$), and lower winter flows (11.08 $m^3 s^{-1}$) (Fig. 2). According to the flood criterion, we recognized 11 flood events in total. Floods in excess of 300 $m^3 s^{-1}$ occurred five times, in July 5th 2006 (435 m $m^3 s^{-1}$), July 17th 2004 (360 $m^3 s^{-1}$), August 29th 2005 (343 $m^3 s^{-1}$), August 21st 2005 (317 $m^3 s^{-1}$), and June 4th 2004 (303 $m^3 s^{-1}$).

The values of river width, water depth, and current velocity indicated that the two studied rivers were small to medium in size (Table 1). Physicochemical measures showed that there were significant differences of most water quality variables between the forested and agricultural river, whereas pH, TN and TP were exceptions (Table 2). Generally, most of concentrations of environmental variables were higher in the agricultural river.

The concentration of chlorophyll *a* indicated that algal biomass was low in forested rivers and ranged from 0.05 mg m^{-2} (July 2004) to 16.53 mg m^{-2} (November 2005). In contrast, the algal biomass was much higher in the agricultural river and ranged from 0.08 mg m^{-2} (July 2004) to 144.97 mg m^{-2} (December 2004) (Fig. 3). There was a significant difference in the

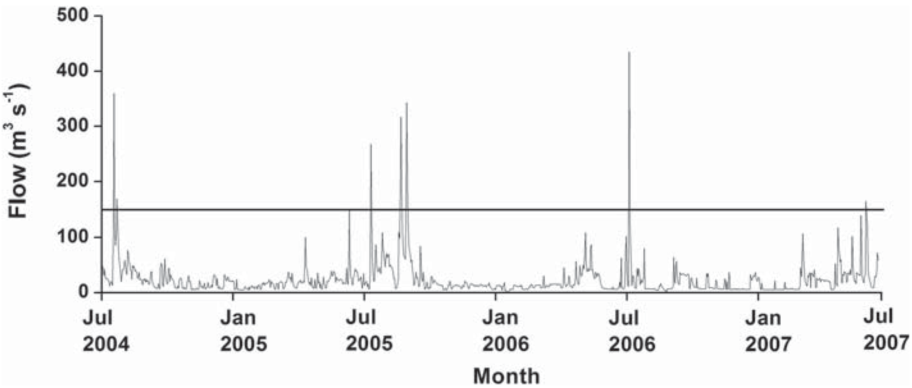

Figure 2. Hydrograph of daily flows, at the Xiangxi River hydrological station, from July 2004 to June 2007. The horizontal line refers to flood criteria.

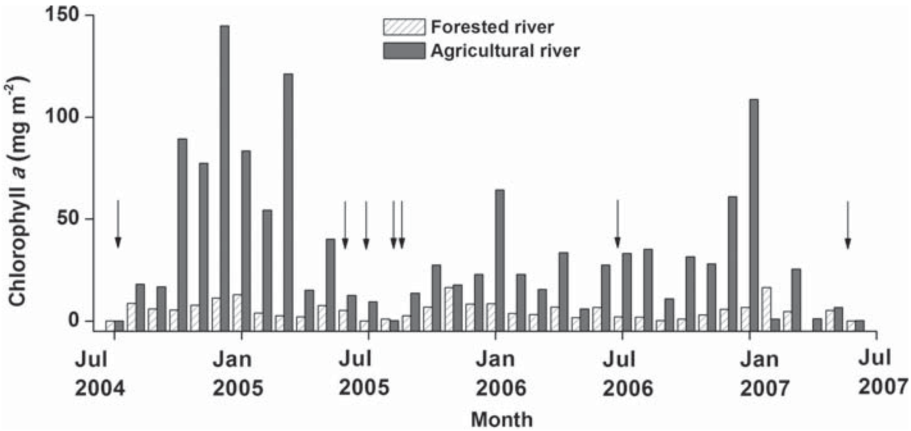

Figure 3. Histogram of benthic algal chlorophyll *a*, measured in the forested and agricultural rivers, from July 2004 to June 2007. The arrow refers to the date of flood. Some dates overlap.

Table 1. Main environmental variables in both rivers during the summer flood period, the winter low flow period and all seasons in the forested and agricultural rivers from July 2004 to June 2007.

Variable	Forested river (Mean ± SD)			Agricultural river (Mean ± SD)		
	Summer	Winter	All seasons	Summer	Winter	All seasons
River width (m)	7.83 ± 3.66	6.38 ± 1.18	6.95 ± 2.29	19.76 ± 6.47	16.71 ± 4.89	17.44 ± 5.71
Water depth (m)	0.35 ± 0.09	0.30 ± 0.14	0.31 ± 0.10	0.47 ± 0.14	0.40 ± 0.11	0.44 ± 0.14
Current velocity (m s^{-1})	0.80 ± 0.37	0.44 ± 0.18	0.56 ± 0.30	0.81 ± 0.25	0.56 ± 0.32	0.72 ± 0.30
Water temperature (°C)	17.97 ± 3.46	8.26 ± 2.57	13.02 ± 5.00	22.09 ± 2.48	10.55 ± 1.96	16.21 ± 5.21
pH	8.42 ± 0.48	8.53 ± 0.62	8.44 ± 0.71	8.41 ± 0.28	8.34 ± 0.43	8.33 ± 0.62
Conductivity (μS cm^{-1})	237.63 ± 50.33	251.72 ± 38.66	257.93 ± 44.38	284.56 ± 72.74	306.94 ± 40.61	304.22 ± 62.16
TN (mg L^{-1})	1.19 ± 0.93	0.67 ± 0.28	0.89 ± 0.57	1.74 ± 1.04	0.99 ± 0.36	1.18 ± 0.71
TP (mg L^{-1})	0.02 ± 0.01	0.02 ± 0.01	0.03 ± 0.03	0.05 ± 0.04	0.03 ± 0.02	0.04 ± 0.03
Chlorophyll *a* (mg m^{-2})	2.89 ± 3.25	9.03 ± 4.04	5.38 ± 4.27	15.24 ± 14.14	67.67 ± 42.01	35.55 ± 33.91

Table 2. Statistics of significance tests of main environmental variables in the forested and agricultural rivers from July 2004 to June 2007.

Variable	Forested river *vs* agricultural river		Summer flow *vs* winter flow: Forested river		Summer flow *vs* winter flow: Agricultural river	
	F	*P*	*F*	*P*	*F*	*P*
River width	102.14	< **0.01**	1.3	0.27	1.28	0.27
Water depth	20.45	< **0.01**	0.97	0.34	1.25	0.28
Current velocity	5.23	**0.03**	6.99	**0.02**	4.82	**0.04**
Water temperature	6.81	**0.01**	45.68	< **0.01**	111.1	< **0.01**
pH	0.42	0.52	0.14	0.71	0.11	0.74
Conductivity	12.49	< **0.01**	0.44	0.51	0.59	0.45
TN	3.11	0.08	1.92	0.18	5.36	**0.03**
TP	3.22	0.08	0.67	0.43	1.38	0.26
Chlorophyll *a*	24.52	< **0.01**	12.62	< **0.01**	12.59	< **0.01**

algal biomass between these two rivers ($P < 0.05$, Table 2). Furthermore, we also found a strong negative relationship between chlorophyll *a* and stream flow (log-transformed, forested river: $r = 0.57$, $n = 35$, $P < 0.05$; agricultural river: $r = 0.56$, $n = 36$, $P < 0.05$).

Several environmental variables differed significantly between floods and low flow periods in the same river. Examples can be seen in the current velocity, water temperature, and chlorophyll *a* in the forested river, and current velocity, water temperature, chlorophyll *a*, and TN in the agricultural river (Table 2). Generally, the concentration of chlorophyll *a* in winter was higher than that in summer ($P < 0.05$). Furthermore, the seasonal variation of TN was significantly different in the agricultural river ($P < 0.05$, Table 2). However, the seasonal difference of TN was not significant in the forested river ($P > 0.05$, Table 2).

3.2. Benthic macroinvertebrates

A total of 125 and 94 macroinvertebrate taxa were collected from the forested and agricultural river, respectively. Macroinvertebrates were composed predominantly of Ephemeroptera, Trichoptera, Plecoptera, Coleoptera, and Chironomidae. The fauna in the forested river were dominated numerically by the mayfly *Baetis* sp. (which made up to 21.84% of the total abundance). This was followed by the elmid *Stenelmis* sp. (9.01%), then the mayflies *Epeorus* sp. (7.52%), *Heptagenia* sp. (6.02%), the caddisfly *Glossosoma* sp. (5.98%) and the stonefly *Nemoura* sp. (5.21%). The patterns in the agricultural river were dominated by the caddisfly *Ceratopsyche* sp. (26.09%), the mayflies *Baetis* sp. (18.10%) and *Serratella* sp. (14.28%), the Dipteran *Antocha* sp. (6.34%) and the chironomid *Orthocladius* sp. (6.32%). The dominant taxa were higher in winter and early spring during the relative stable flow period in both rivers (Fig. 4).

The density of both dominant taxa dropped markedly, after a flash flood in both rivers, and remained at lower values for three to eight months (Fig. 4). However, densities of *Baetis* sp. soon recovered indicating they may be pioneer taxa (Fig. 4). Rapidly colonizing taxa also included *Orthocladius* sp., *Nemoura* sp. and *Epeorus* sp. Interestingly, most dominant taxa in the agricultural river could withstand the flood disturbances. For example, the density of clinger habitat taxa *Ceratopsyche* sp., *Serratella* sp., and *Antocha* sp. decreased less than other dominant taxa after the flash flood (Fig. 4).

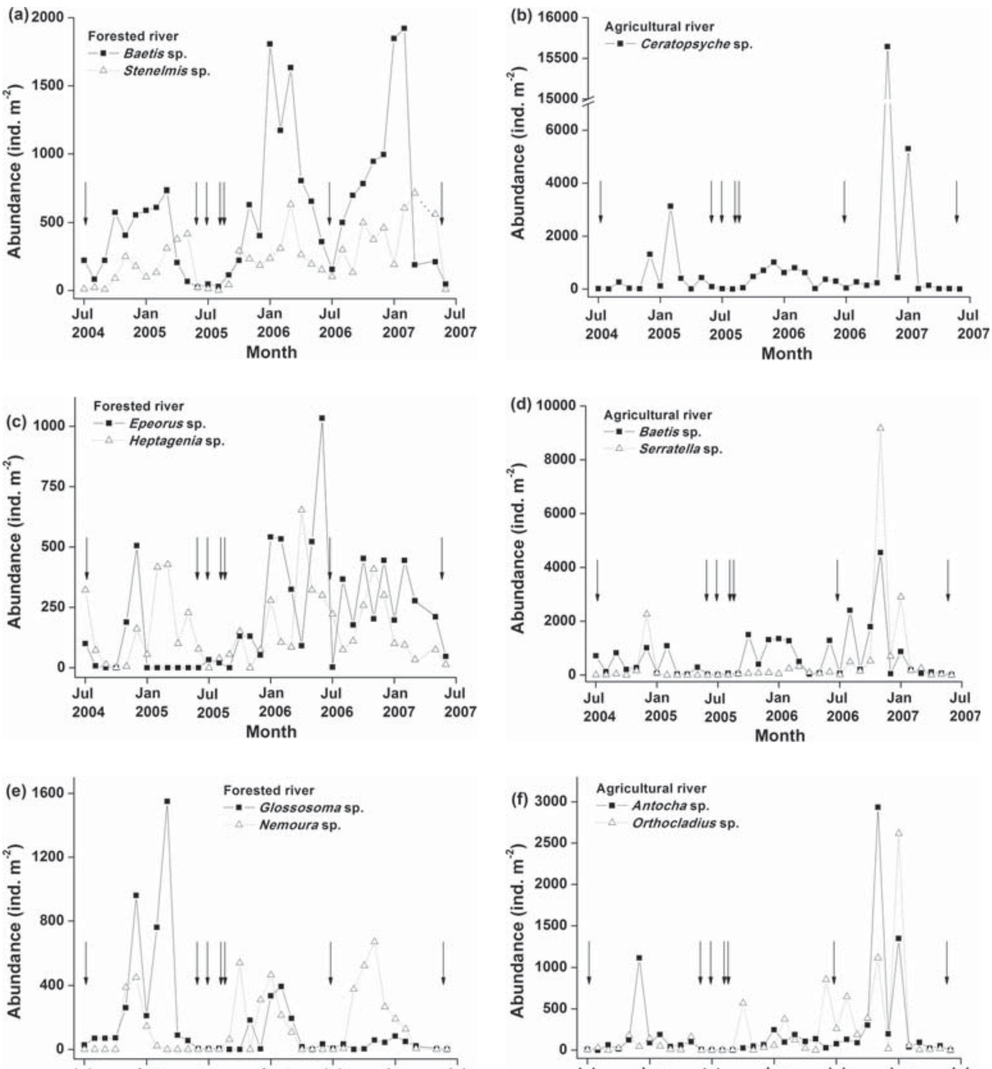

Figure 4. Total abundance of dominant taxa in the forested and agricultural rivers from July 2004 to June 2007. The arrow refers to the date of flood. Some dates overlap.

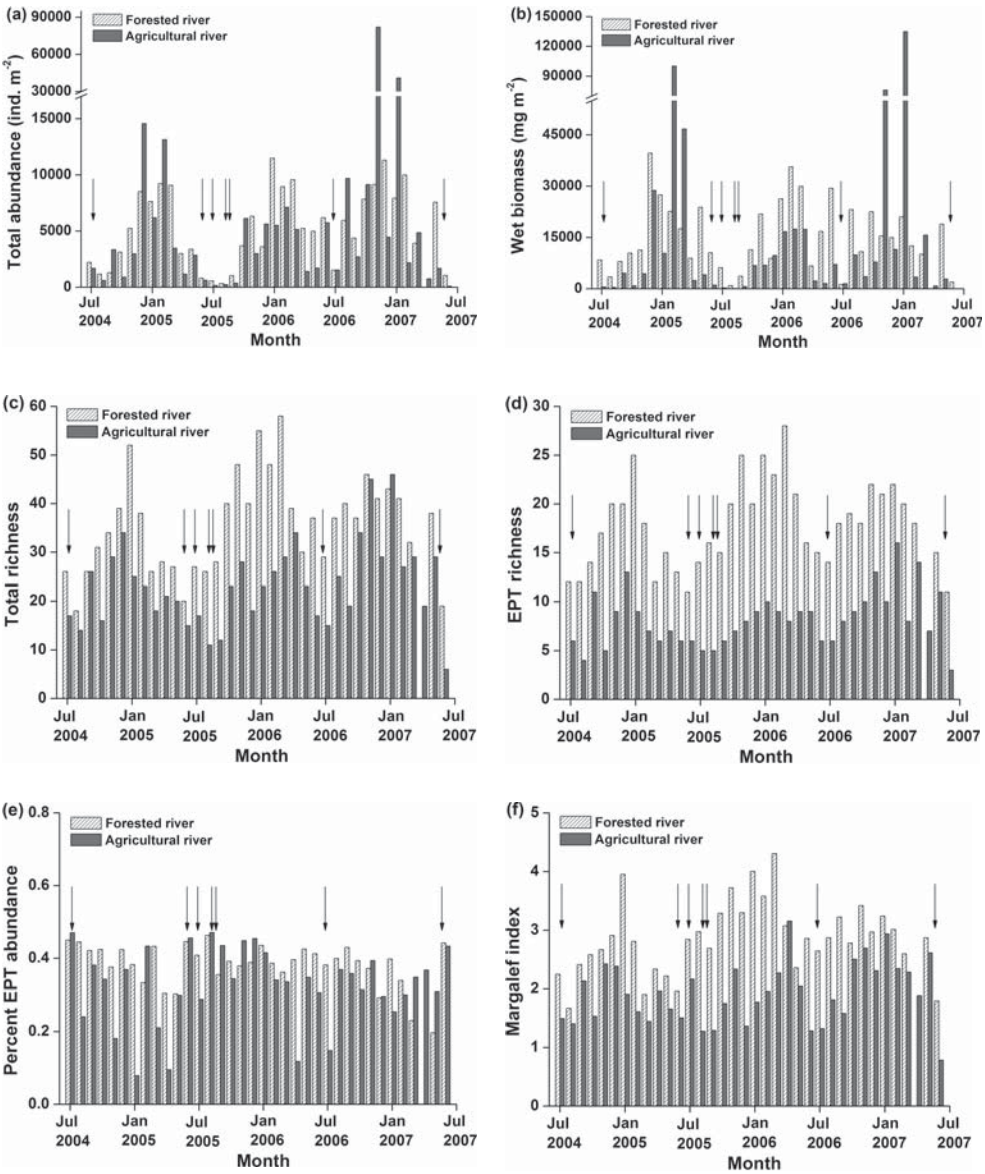

Figure 5. Histogram of the benthic macroinvertebrate metrics in the forested and agricultural rivers (from July 2004 to June 2007). The arrow refers to the date of flood. Some dates overlap.

The total abundance fluctuated seasonally and significantly in both rivers ($P < 0.05$), with lowest values being recorded in summer, and highest values peaking in winter and early spring (Fig. 5). The above pattern was also evident for wet biomass, total richness, EPT richness and Margalef index. However, that pattern was not detected for percent EPT abundance (Fig. 5). There was a reduced proportion of community metrics after the monsoon floods. For example, see May 2005, June 2006, and May 2007 in Table 3. Total abundance and wet biomass exhibited higher reduced proportions (approximately 80%) than total richness, EPT richness, and Margalef index (approximately 20%–40%, Table 3). In addition, relatively higher reduced proportions can be found in the agricultural river than that in the forested river (Table 3).

Community metrics fluctuated significantly with time after the flood in both rivers ($P < 0.05$). The rate of recovery (reaching the maximum values) of community metrics was four to nine months following floods (Table 4). In general, most of the community metrics recovered more rapidly in the forested river than that in the agricultural river (Table 4). Interestingly, the wet biomass recovered more rapidly than the total abundance in the forested river (Table 4), in contrast, a different pattern can be found in the agricultural river.

Table 3. Reduced proportions of community metrics after the monsoon floods (May 2005, June 2006, and May 2007) in the forested and agricultural rivers.

Time	Total abundance	Wet biomass	Total richness	EPT richness	Percent EPT abundance	Margalef index
Forested river						
May–June 2005	75.74%	55.78%	25.93%	15.38%	–47.04%	11.50%
June–July 2006	75.31%	95.89%	21.62%	6.67%	7.43%	7.38%
May–June 2007	86.06%	89.65%	50.00%	26.67%	–124.43%	37.58%
Mean	79.04%	80.44%	32.52%	16.24%	–54.68%	18.82%
Agricultural river						
May–June 2005	77.95%	73.57%	25.00%	0.00%	–52.16%	9.03%
June–July 2006	73.06%	78.72%	11.76%	0.00%	51.99%	–3.13%
May–June 2007	95.03%	97.98%	79.31%	72.73%	–39.70%	70.02%
Mean	82.01%	83.42%	38.69%	24.24%	–13.29%	25.31%

Table 4. Summaries of the time cost to reach the maximum values after the monsoon floods for each community metrics in the forested and agricultural rivers from July 2004 to June 2007. All differences are relative to the July.

Time	Total abundance	Wet biomass	Total richness	EPT richness	Percent EPT abundance	Margalef index
Forested river						
July2004–June2005	7.0	5.0	6.0	6.0	0.0	6.0
July2005–June2006	6.0	7.0	8.0	8.0	1.0	8.0
July2006–June2007	5.0	1.0	4.0	4.0	11.0	4.0
Mean	6.0	4.3	6.0	6.0	4.0	6.0
Agricultural river						
July2004–June2005	5.0	7.0	5.0	5.0	0.0	4.0
July2005–June2006	7.0	7.0	9.0	6.0	1.0	9.0
July2006–June2007	4.0	6.0	6.0	6.0	11.0	6.0
Mean	5.3	6.7	6.7	5.7	4.0	6.3

3.3. *Stepwise multiple regression (SMR) models*

SMR models were utilized to investigate the influences of environmental variables on aquatic organisms. Although the predictive power of the SMR analyses varied greatly, several significant models were still produced. A large number of the predictive powers were relatively good, and the highest value of R^2 could reach 0.76 (the density of mayfly *Baetis* sp. in the forested river, Table 5). However, the predictive powers of some models were poor (Table 5).

Hydrological variables were negatively related to most biotic metrics (Table 5). The number of days since last flood (*N* days) were positively related to abundance. This suggests that abundance declined after flash floods, and recovered with the increasing of low flow days. Of the hydrological variables, current velocity was selected by 17 regression models. Both species were sensitive to the increase of current velocity in the forested river, while in the agricultural river, *Baetis* sp., *Serratella* sp., *Antocha* sp., and *Orthocladius* sp. were also sensitive to current velocity. Of the temporal variables, *N* days was selected by 18 models. Generally, taxa were defined as slow recovery if they spent a long time to reach the maximum abundance, thus, *Epeorus* sp., *Heptagenia* sp. and *Nemoura* sp. were recognized as

Table 5. Relationships between benthic macroinvertebrate metrics and temporal and hydrological variables with stepwise multiple regression analyses in the forested and agricultural rivers from July 2004 to June 2007.

Metric	*Q* mean	*N* days	River width	Water depth	Current velocity	*F*	R^2	*P*
Forested river								
Baetis sp.		0.48	1.00		–3.04	32.73	0.76	< 0.01
Stenelmis sp.		0.35			–4.02	26.37	0.62	< 0.01
Epeorus sp.			–3.31			4.40	0.12	0.04
Heptagenia sp.			–1.67	–8.33		6.45	0.29	< 0.01
Glossosoma sp.		0.91	2.62			10.02	0.39	< 0.01
Nemoura sp.	–5.39					19.55	0.37	< 0.01
Chlorophyll *a*	–1.19	0.20			–1.77	10.33	0.50	< 0.01
Total abundance		0.34			–1.89	34.27	0.68	< 0.01
Wet biomass		0.18	0.19	–0.08	–0.06	32.49	0.50	< 0.01
Total richness		0.09			–0.53	18.87	0.54	< 0.01
EPT richness		0.08			–0.35	16.39	0.51	< 0.01
Percent EPT abundance					0.09	5.01	0.13	0.03
Margalef index		0.05				11.01	0.25	< 0.01
Agricultural river								
Ceratopsyche sp.		0.80		–4.35		14.13	0.46	< 0.01
Baetis sp.	–1.58				–0.69	17.69	0.35	< 0.01
Serratella sp.		0.79	3.07		–4.65	12.38	0.57	< 0.01
Antocha sp.		0.63			–4.09	22.45	0.58	< 0.01
Orthocladius sp.	–2.14			7.20	–7.11	16.04	0.60	< 0.01
Chlorophyll *a*		0.47	–1.43		–1.51	23.59	0.69	< 0.01
Total abundance	–0.79	0.31			–2.93	22.23	0.68	< 0.01
Wet biomass		0.82		–7.01		50.27	0.75	< 0.01
Total richness		0.13			–0.85	21.02	0.56	< 0.01
EPT richness		0.07			–0.85	15.22	0.48	< 0.01
Percent EPT abundance			0.10		0.15	4.74	0.22	0.02
Margalef index		0.04			–0.34	7.40	0.31	< 0.01

rapid recovery in the forested river, while *Baetis* sp. and *Orthocladius* sp. were considered rapid recovery in the agricultural river (Table 5). Although a positive relationship between *N* days and *Baetis* sp. in the forested river was found, we thought its recovery rate was still fast (Fig. 4).

We further investigated the effects of flow-related variables on the total abundance and the wet biomass of macroinvertebrates. Regressions of the total abundance, the wet biomass and the stream flow yielded negative relationships in both rivers (Fig. 6). In contrast, linear regression analyses revealed significantly positive relationships between these two community metrics and *N* days (Fig. 6). In addition, the y-intercepts of wet biomass and *N* days in Fig. 6 can be used to assess the relative differences in recovery. For example the y-intercept, of the fourth panel in Figure 6, is around 3.5 for the forested river and it is around 2 for the agricultural river. This suggests that there is more invertebrate biomass remaining in the forested river immediately following the floods. Similar patterns can be also found in the relationships between the community metrics and the stream flow (Fig. 6).

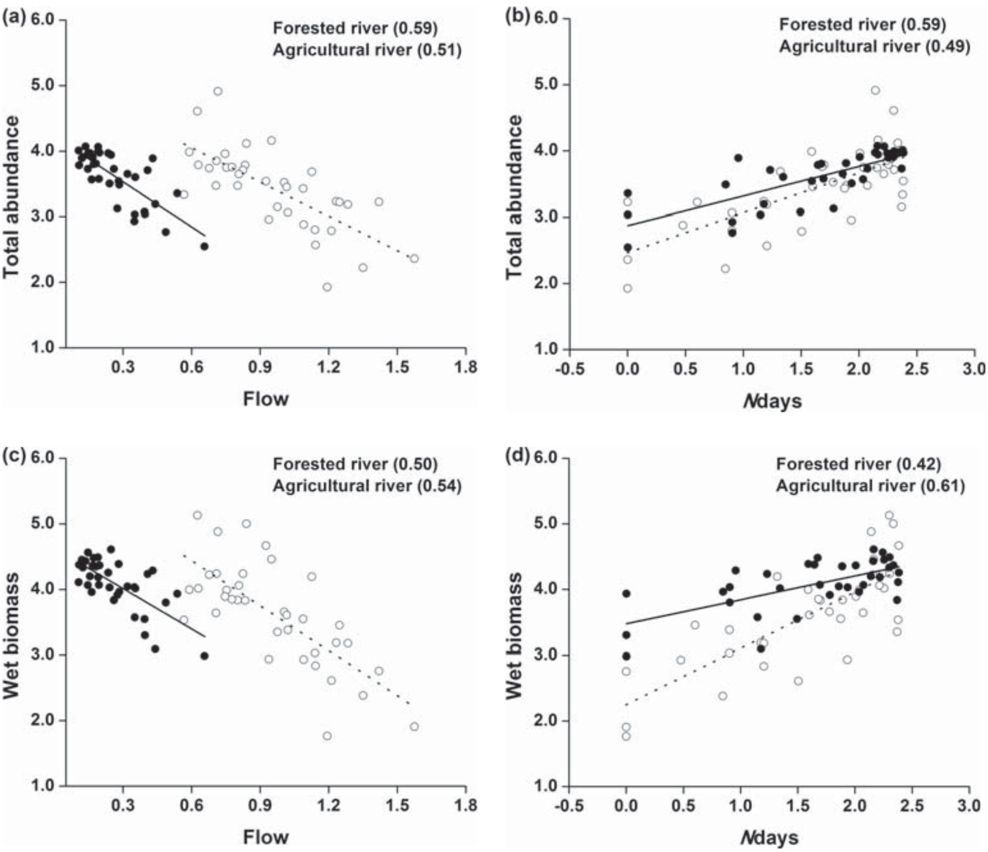

Figure 6. Linear regressions between total abundance and wet biomass of benthic macroinvertebrates, and some flow-related variables, in the forested and agricultural rivers from July 2004 to June 2007. All data were log-transformed. The closed circle refers to the forested river, and the opened circle refers to the agricultural river. The value in the bracket refers to R^2.

Table 6. Summary of the results of two-way ANOVA for the macroinvertebrate metrics in the forested and agricultural rivers from July 2004 to June 2007.

Source	*df*	*F*	*P*	Partial eta squared (η^2)
Total abundance				
Month	11	4.07	< 0.01	0.49
Land-use	1	2.97	0.02	0.05
Month × Land-use	11	0.31	0.98	0.07
Wet biomass				
Month	11	4.64	< 0.01	0.52
Land-use	1	13.83	< 0.01	0.23
Month × Land-use	11	1.02	0.44	0.19
Total richness				
Month	11	4.65	< 0.01	0.52
Land-use	1	43.02	< 0.01	0.48
Month × Land-use	11	0.30	0.98	0.07
EPT richness				
Month	11	5.88	< 0.01	0.58
Land-use	1	100.16	< 0.01	0.68
Month × Land-use	11	0.50	0.89	0.10
Percent EPT abundance				
Month	11	2.95	0.02	0.23
Land-use	1	7.69	< 0.01	0.14
Month × Land-use	11	0.73	0.70	0.15
Margalef index				
Month	11	2.96	< 0.01	0.41
Land-use	1	57.54	< 0.01	0.55
Month × Land-use	11	0.51	0.89	0.11
Tolerant value				
Month	11	1.40	0.21	0.25
Land-use	1	109.70	< 0.01	0.70
Month × Land-use	11	0.71	0.72	0.15

3.4. Two-way ANOVA for the benthic macroinvertebrate communities

There were significant effects of months and land-use types on most biotic metrics (Table 6). We could rank the order of variables which contributed to the total inertia as follows: month > land-use > (month × land-use) for the total abundance and the wet biomass according to the partial eta squared (η^2) (Table 6). The total richness, the EPT richness, the percent EPT abundance, and the Margalef index showed relatively equal effects by the months and land-use types (Table 6). However, the tolerant value indicated less effects by the month than by the land-use types (Table 6).

4. Discussion

Effects of monsoon floods on the macroinvertebrate communities in these two subtropical land-use rivers have been studied over a three-year period. Results suggest that the dominant macroinvertebrate taxa and community metrics dramatically declined following the flash flood in both rivers. Such a change has also been noted in other comparable studies (LYTLE, 2000; McCabe and GOTELLI, 2000; MAIER, 2001; SUREN and JOWETT, 2006; CHIU *et al.*, 2008). Most of the community metrics appeared to recover within a shorter time, after the initial flash flood, in the forested river than that in the agricultural river. This is consistent with the findings by COLLIER and QUINN (2003) and PARKYN and COLLIER (2004) in the New Zealand rivers. Furthermore, we also found that the influence of intra-annual fluctuation on the community composition was more important, especially using the total abundance and wet biomass as indicated factors.

4.1. Effects of floods on macroinvertebrates

The dominant taxa and community metrics declined as flood disturbances increased in both rivers (Table 3, Fig. 4, 5). Such impacts were likely caused by physical removal, abrasion of the transportation substrates and mortality (COLLIER and QUINN, 2003; OLSEN and TOWNSEND, 2005). Other studies highlighted the importance of reducing food resources, for example stone surface biofilms, following disturbance (DEATH, 1996; BOND and DOWNES, 2000; DEATH, 2002). WAIDE *et al.* (1999) and DEATH (2002) also agreed with the latter idea. They supposed macroinvertebrates colonized a kind of habitat and they discovered low concentrations of algal biomass and noted that the organisms would move to other places to search food.

Despite the large effects of floods on macroinvertebrate communities, their combined effects were relative short. This reflects the low resistance and high resilience of many taxa (PARKYN and COLLIER, 2004). To persist in lotic ecosystem, three main strategies are used: (1) morphological or physiological adaptations, for example hooks, claws or body shapes, that enable individuals to withstand flood flows (LANCASTER and BELYEA, 1997; COLLIER and QUINN, 2003), in our case, as a net-spinning expert, caddisfly *Ceratopsyche* sp. can withstand small floods; (2) moving to nearby smaller tributaries or other rivers (COLLIER and QUINN, 2003; PARKYN and COLLIER, 2004), and (3) using patterns in their life cycle to escape the effects. A common characteristic of aquatic insects inhabiting temporary habitats is a rapid larval development (WISSINGER *et al.*, 2003). This strategy allows them to complete their development before a flood or drought occurs (WISSINGER *et al.*, 2003). For example, a previous study (LI *et al.*, 2009b) showed that the presence of late instar larvae in spring indicated that *Nemoura sichuanensis* cannot adapt to withstand a longer summer flood period in the Xiangxi River. *Nemoura sichuanensis* may use dormancy during their egg stage to safeguard against catastrophic situations of flood or otherwise (LI *et al.*, 2009b).

After beginning recoveries, biotic metrics in both rivers were variable. This reflected the differences in the trajectories of succession and population expansion for various macroinvertebrate taxa (Fig. 4, 5). A range of taxa-specific recovery patterns was observed, which supported the works of COLLIER and QUINN (2003). Chironomids (in our case was *Orthocladius* sp.) and some mayflies were recolonized as exhibiting a rapid recovery following disturbances. The recovery of total abundance took about 5–6 months (Table 4, Fig. 5) and, although this was much slower than the 8–30 days observed following a flood in the Acheron River, Australia (DOEG *et al.*, 1989), it was similar to the Mangaotama River in New Zealand (COLLIER and QUINN, 2003). Interestingly, the total abundance recovered more rapidly than wet biomass in the agricultural river, two interpretations could be: (1) a reduction in the number and mass of large body sized larvae after floods; (2) oviposition or first instar larvae would be abundant but have lower biomass when both of pulse and ramp disturbances exited. In addition, it appeared that the biomass recovered faster in the forest streams than in agriculture streams (Table 4, Fig. 5). This trend suggests that recolonization and reproduction mechanisms could be more central to the functioning and resilience of the invertebrate community in these systems.

4.2. Effects of land-use type on macroinvertebrate responses to flooding

In terms of LAKE's (2000) disturbance response types, macroinvertebrate communities exhibited significant pulse responses in both rivers. Community compositions were also affected by management practices, such as the use of fertilizers and pesticides, in an agricultural river (LI *et al.*, 2008). The integrated pulse and ramp responses were relatively evident in this mountainous agricultural river. For example, most of the community metrics showed slower rates of recovery in the agricultural river than that in the forested river (Table 4). However, significant ramp disturbances were reported in the lowland agricultural (ANDERSON *et al.*, 2003) or pasture rivers (COLLIER and QUINN, 2003; PARKYN and COLLIER, 2004). Moreover, floods can disturb the sediments and reduce the concentrations of fertilizers and pesticides in the mountainous river channels (ZHANG *et al.*, 2005; JIANG *et al.*, 2006). Thus, the effects of ramp responses by macroinvertebrate communities decreasd in this agricultural river.

Aquatic organisms can be seriously damaged following sudden increases in stream flow. In contrast, clinger habitat taxa, such as some caddisflies, can build cases that one tightly attached to the surface of large stones (*e.g. Ceratopsyche* sp. and *Stenopsyche* sp.). This helps them to withstand flash floods (COLLIER and QUINN, 2003; PARKYN and COLLIER, 2004). In our study, the caddisfly *Ceratopsyche* sp., the mayfly *Serratella* sp., and the Dipteran *Antocha* sp. exhibited greater resistance to floods in the agricultural river (Fig. 4). This suggests that they were not only tolerant to flood disturbance but were also the dominant taxa in the agricultural river as clingers. In contrast, the dominant taxa in the undisturbed forested river were swimmers, sprawlers and clingers. However, the community metrics in the agricultural river showed a different picture, i.e., reduced proportions were found in the agricultural river (Table 4). This suggests a lower resistance of community metrics to floods in agricultural rivers than in forested rivers. The longer recovery rate of community metrics in agricultural rivers may reflect the cropping patterns, a deterioration of the bank habitat, and increased flows during the flood events. Thus, decreasing the suitability of river channel (LI *et al.*, 2008).

Besides floods, other types of disturbance (*e.g.* drought, pesticide, mineral and coal) can also influence river ecosystems (HARPER and PECKARSKY, 2005; SUREN and JOWETT, 2006; POND *et al.*, 2008; SMUCKER and VIS, 2009), and interactions between disturbance types. The land-use context of particular sites could also strongly influence the disturbance impacts and rates of post disturbance recovery (COLLIER and QUINN, 2003).

4.3. Two-way ANOVA for the benthic macroinvertebrate communities

In comparison with the land-use types, temporal variables appeared more important for macroinvertebrate communities, especially using the total abundance and the wet biomass (Table 6). This, however, does not support the second study objective. Although there were strong effects of land-use type and temporal variables on several macroinvertebrate metrics in our study, we detected few interactions between these metrics, suggesting consistent temporal responses occurred between land-use types. However, tolerant values showed a different picture, and the partial eta squared value indicated that the land-use type was more important than the temporal variable (Table 6). This suggests that macroinvertebrate resilience under different land-use types is quite different, which confirmed the results that macroinvertebrates in the forested river had faster rates of recovery than those in the agricultural river.

Compared to the effects of land-use types on stream ecosystems, the influences of hydrology, water quality, stream temperature, river morphology, and food resources, were more directly important for aquatic organisms (Collier and QUINN, 2003). Thus, in the East Asian monsoonal region, floods and thermal regimes are major determinant factors for the variations of aquatic communities (Cai et al., 2001).

We used a three-year study in the Xiangxi River watershed to provide a view of the influence of flood disturbance on macroinvertebrates. Our data suggest that it is essential to take the impacts of intra-annual flow variations in the mountainous rivers into account, when using macroinvertebrates to assess the river ecosystems. This is of great importance in the Chinese mainland where monsoons may cause flash floods that might have catastrophic influences on aquatic organisms.

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