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Exploring the dynamic correlation of landscape composition and habitat fragmentation with surface water quality in the Shenzhen river and deep bay cross-border watershed, China

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ABSTRACT

The dynamics of land cover and landscape structure are important variables that influence the environmental and ecological quality of a watershed. In this study, a complex hypothesis was proposed that assumed water quality was closely correlated with landscape composition and habitat fragmentation and that this relationship varied at different spatial scales, namely, in riparian buffers with different widths. This hypothesis was tested using a case study in the Shenzhen River and Deep Bay watershed, which is the border region between the Hong Kong Special Administrative Region and Shenzhen in China. To effectively explore the correlation of habitat fragmentation with surface water quality, a compound indicator was proposed that can be used to disentangle the effect of habitat fragmentation from that of habitat loss alone. The results of the redundant analysis suggested that the surface water quality in our study area was strongly correlated with landscape composition and habitat fragmentation, and the correlations varied with riparian buffer widths. Compared to habitat fragmentation, landscape composition seemed to be the dominant contributor to the variation in water quality. The cross-border comparison between Hong Kong and Shenzhen suggested the riparian buffers with the strongest linkages were different for the two sides of the watershed, likely due to their special combinations of landscape characteristics and other socio-economic contexts. Whether habitat loss and fragmentation had negative effects on water quality depended on the habitat types and water quality variables that were examined. These findings can be helpful in offering useful information for future watershed management and landscape planning.

1. Introduction

Water quality is considered to be affected by a wide range of factors, including local topography, soil permeability, stream density, temperature, and land-use activities (Sliva and Williams, 2001; Didham et al., 2012; Aronson et al., 2014; Tanaka et al., 2016). Landscape characteristics have critical influences on hydrological processes, energy flows and nutrient cycles (Grimm et al., 2000; Lee et al., 2009). The importance and effectiveness of landscape approaches in studies on water quality dynamics have been increasingly recognized (Uuemaa et al., 2007; Amiri and Nakane 2009; Liu et al., 2012; Zhou et al., 2012; Shen et al., 2015; Clément et al., 2017). The variations in landscape characteristics are believed to most likely affect hydrological conditions and ecological processes by altering the types and amounts of pollutants entering aquatic systems (Griffith, 2002; Xiao and Ji, 2007; Shen et al., 2015). Habitat loss and fragmentation are essential processes that occur during landscape change in terms of both composition and configuration (Fahrig, 2003; Smith et al., 2009). Intense modification of the landscape, increased consumption and pollution of ecosystems have directly and indirectly resulted in habitat loss, fragmentation and degradation (Maron and Fitzsimons, 2007; Giam et al., 2010).

Landscape composition has been widely reported to have a strong correlation with water quality change. The increase in forest land (a type of natural habitat area) could contribute to the reduction of water pollution (Lee et al., 2009; Liu et al., 2012), whereas the increase in urban area (*i.e.*, the decline in habitat area) is suggested to be positively correlated with the degradation of water quality in terms of all pollutants (Uuemaa et al., 2007; Lee et al., 2009). Habitat fragmentation is defined as the process of breaking up a large, intact area of a habitat type into smaller, intact units that are separated by a matrix of human-converted land cover (Forman and Godron, 1986; Haddad et al., 2015). Some landscape metrics related to habitat fragmentation, such as patch

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density, patch edge, and contagion, have been evaluated in terms of their correlation with water quality. It was reported that the nutrient and organic parameters (e.g., TN, NO_3-N, TP, and COD_{Mn}) were positively correlated with the patch density of cropland, orchard and grassland (Ding et al., 2016). A negative correlation between contagion and water pollution was detected (Bu et al., 2014), which indicates that degraded water quality usually occurs in highly fragmented landscapes (Uuemaa et al., 2005). Although these metrics used to measure certain aspects of habitat fragmentation are related to water quality, to date, the direct correlation between habitat fragmentation and river water quality has been poorly addressed.

Indeed, the process of habitat fragmentation is not independent of habitat loss (loss of habitat amount). Habitat loss could cause an increase in the distances (isolation) between habitats within a landscape (Goodsell and Connell, 2002), and habitat fragmentation will inevitably produce a loss of habitat (Andrén, 1994). Although habitat loss and fragmentation have been considered so tightly linked that disentangling their effects is meaningless (Didham et al., 2012), they are two distinct processes that are not always independent of one another (Fahrig, 2003; Fahrig, 2017). The impacts of habitat loss can sometimes be mitigated by connectivity with the surrounding habitat (Fahrig, 2003). Therefore, it is crucial to distinguish the effects between habitat loss alone and true fragmentation in terms of their different effects on species and ecosystems (Fahrig, 1998; Schmiegelow and Mönkkönen, 2002; Mortelliti et al., 2011). Many landscape metrics adopted for measuring habitat fragmentation in a study might have high correlations with each other (Hargis et al., 1998). However, few studies have been able to differentiate the effect of habitat fragmentation from habitat loss. Therefore, an indicator that can disentangle the process of fragmentation from that of habitat loss alone should be developed. This could contribute to the determination of whether habitat fragmentation is actually correlated with water quality, respectively. Furthermore, a study investigating the relationship between landscape configuration and water quality in a large number of watersheds (590) suggested that, for smaller watersheds ($< 250\ km^2$), landscape configuration explained most of the variation in water quality; however, for larger watersheds ($> 250\ km^2$), the proportions of different land-cover types become more important (Clément et al., 2017). An indicator measuring pure habitat fragmentation could also contribute to examine whether landscape composition or habitat fragmentation has a stronger correlation with water quality.

The relationship between water quality and landscape characteristics can be evaluated at different spatial scales, including the watershed, sub-watershed, buffer zone and other smaller proportions of the watershed (Griffith, 2002; Smith et al., 2009; Zhou et al., 2012; Shen et al., 2015). Riparian buffer landscapes, composed of patches or corridors of vegetation, wetlands, agricultural crops and urban settlements, are interfaces between terrestrial and aquatic environments (Apan et al., 2002; Shen et al., 2015). Variation in the riparian landscape plays a significant role in influencing entire ecological and hydrological conditions, including surface water quality, soil erosion and wildlife and fish habitat (Apan et al., 2002; Xiao and Ji, 2007). Some studies maintain that the explanatory ability of the landscape characteristics of riparian buffer zones would increase with the width of the buffer, e.g., from 100 m to 200 m, 400 m, 800 m or 1500 m (Zhao et al., 2015), whereas other studies recognize that the landscape characteristics of 100-m-wide buffer zones have the greatest influence on water quality (Shen et al., 2015). A consensus has not yet been achieved due to the specificity of each watershed and the variables of both human disturbance and the collected datasets (Shen et al., 2014; Shen et al., 2015). Although an increasing number of studies have been conducted in recent years, many questions remain unanswered, e.g., quantifying and disentangling the impact of habitat fragmentation on water quality at multiple spatial scales.

Cross-border or trans-boundary studies are very popular in Europe and North America, and this issue has attracted a large range of researchers from different disciplines, such as geography, political science, environmental science, and sociology (Perkmann, 2003; Nelles and Durand, 2014). In China, one of the most notable cases of a cross-boundary study is the one between Hong Kong and mainland China (Chan, 1998; Lee, 2002). Many scholars have shed light on cross-boundary policy integration, environmental problems, and transportation development (Shen, 2004; Gu and Yim, 2016; Lam et al., 2017). However, the understanding of the effects of habitat fragmentation on water quality in the border region is still very poor. This study attempts to investigate the correlations of landscape composition and habitat fragmentation with water quality degradation in the Shenzhen River and Deep Bay watershed, which is located in the border region between the Hong Kong Special Administrative Region and Shenzhen, China. The entire watershed has an area of approximately $726.72\ km^2$. To examine this issue, this study tested the following working hypotheses: 1) landscape composition and habitat fragmentation were closely related to surface water quality, and the effects of landscape composition had a stronger correlation with variation in water quality than did habitat fragmentation (independent of habitat loss); 2) the correlation of landscape composition and fragmentation with water quality varied with the widths of riparian buffers, and the riparian buffers with the strongest linkages were different for the Shenzhen and Hong Kong sides of the watershed; and 3) the most significant effects of habitat loss and fragmentation on water quality were negative.

2. Materials and methods

2.1. The Shenzhen river and deep bay cross-border watershed

The city of Shenzhen and Hong Kong SAR, located in the southeast of the Pearl River Delta, China, are neighbors along the Shenzhen River and the Deep Bay area. The Shenzhen River and Deep Bay cross-border watershed (abbreviated as the Shenzhen River cross-border watershed) is a border region, as shown in Fig. 1. The whole area extends from 22°23′10″ to 22°40′47″ N and 113°51′39″ to 114°12′54″ E. The Shenzhen River cross-border watershed neighbors Dapeng Bay (Mirs Bay) to the east and the Pearl River Estuary to the west, connecting the Shenzhen Special Economic Zone (SEZ) to the north and the SAR to the south. The northern area of the Shenzhen River cross-border watershed covers most of the Shenzhen SEZ and a small portion of the town of Buji in the Longgang district, whereas the south lies in the northern part of the New Territories of Hong Kong, primarily including the Yuen Long and North Districts. Approximately $412.95\ km^2$ of the watershed is in the city of Shenzhen, which is equivalent to almost 60% of the total area. The Shenzhen River is not only the major river but also the natural border within this watershed. It originates near Niuweiling in the Wutong Mountains and flows approximately 37 km westward into its estuary in Deep Bay. Its major tributaries include the Liantang, Shawan, Buji, Futian, and Huanggang rivers on the Shenzhen side, and the River Indus, River Beas, and River Ganges on the Hong Kong side. The Yuen Long River, Kam Tin River, Tin Shui Wai River and the Dasha River drain directly into Deep Bay. Two ecologically important wetlands exist in the estuary of the Shenzhen River. The Mai Po and Inner Deep Bay Ramsar Site on the Hong Kong side of the watershed is a wetland of international importance that provides habitat for East Asian migratory birds.

Since the late 1970s, the Shenzhen River cross-border watershed has experienced serious habitat loss and fragmentation (Ng et al., 2011; Xie and Ng, 2013). Nearly 50% of the wastewater on the Shenzhen side of this watershed never reaches the sewage system, and substantial non-point wastewater inputs from agriculture eventually enter Deep Bay, which has significantly impacted the water quality there. The data from the Environmental Protection Department of Hong Kong (2003) indicate that the water quality of the major rivers and nullahs in the Inner Deep Bay watershed are generally poor. In 2003, the Inner Deep Bay area had the poorest water quality, characterized by high ammonia and

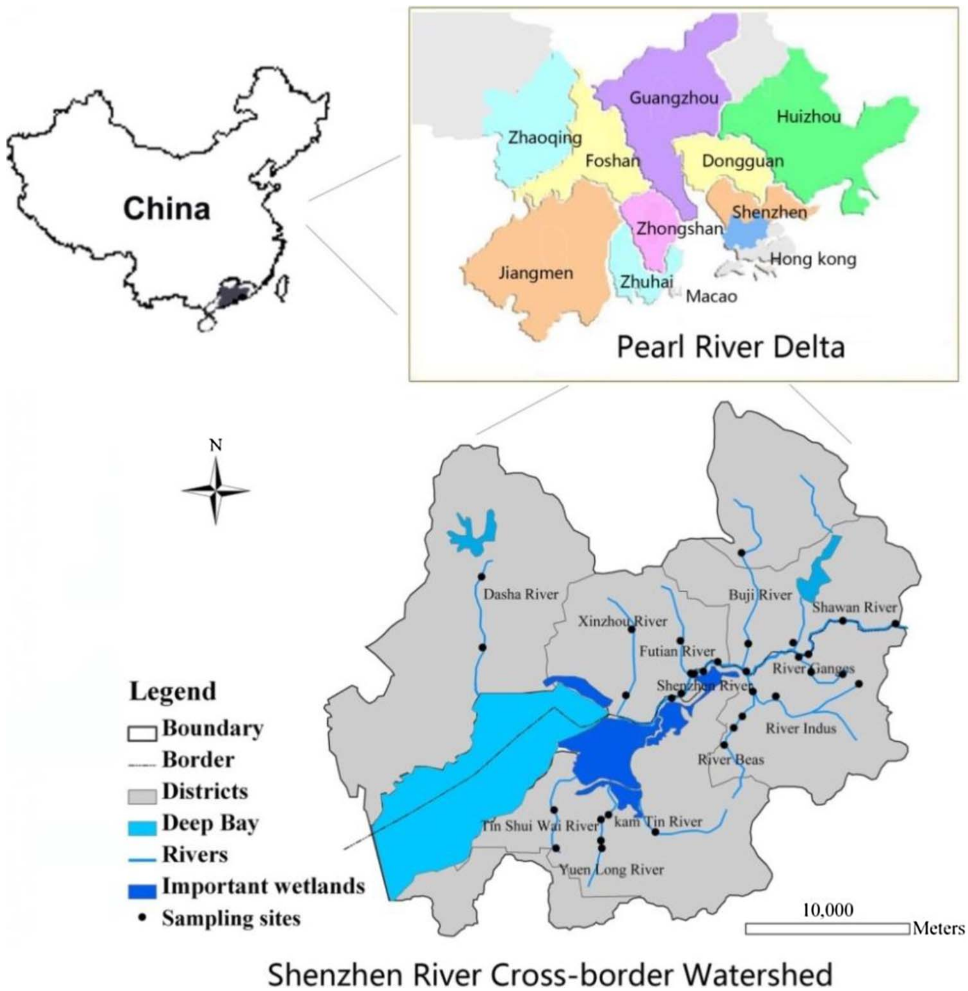

Fig. 1. Location of the study area.

low dissolved oxygen, among open marine waters.

2.2. Methods and data

2.2.1. Indicators measuring landscape composition and habitat fragmentation

Landscape composition can be easily quantified by calculating the percentage of each habitat type, which is preferable to quantifying the total area of each landscape type due to the varying spatial scales. In contrast, habitat fragmentation is much more complicated to measure, and it is especially difficult to disentangle habitat fragmentation from habitat loss. According to Fahrig (2003), four effects of the fragmentation process have been identified: (a) decrease in the amount of habitat, (b) increase in the number of habitat patches, (c) reduction in the sizes of habitat patches, and (d) increase in the isolation of patches (*i.e.*, the tendency for patches to be relatively isolated in space or distant from other patches of the same, or ecologically similar, class) (McGarigal, 2002). As noted above, habitat fragmentation will inevitably cause the loss of habitat. Fig. 2 illustrates five different processes of habitat loss, some of which include the process of habitat loss only (*i.e.*, A, C and D) and some that refer to habitat loss with fragmentation (*i.e.*, B and D).

As shown in Fig. 2, changes (increase, decrease or no change) in the number of patches (NP), mean patch size (MPS) and mean isolation (MI) vary among the five processes illustrated. Accordingly, the use of only one indicator would be insufficient to detect habitat fragmentation. However, to disentangle the effects of fragmentation, an indicator must be identified that can distinguish independent habitat loss from habitat fragmentation. Through comparison among different single and combined metrics, only the compound indicator NP/MPS (abbreviated as N/S) can satisfy the above requirements. This indicator increases with the number of patches and decreases with the mean patch size. Furthermore, with an increase in N/S, the degree of fragmentation will rise, as shown in Fig. 2(B and D), whereas during the process of habitat loss only, N/S will decrease. In this respect, the indicator N/S used in this study has no significant positive correlation with habitat loss. Briefly, in this study, three landscape metrics, including NP, MPS, and the percentage of landscape (PLAND), were selected and analyzed. The software package FRAGSTATS 3.3 (McGarigal et al., 2002) was used to compute the selected metrics. When analyzing the effects of habitat loss and fragmentation, the percentage of habitat area (*e.g.*, forest habitat) and the compound indicator N/S were used, and before calculating the metrics of N/S, both the indicators of NP and MPS were computed.

2.2.2. Data acquisition for land cover and water quality

The quantification of landscape composition and habitat fragmentation is based on land-cover maps derived from five sets of Landsat TM images from the years 1988, 1993, 1998, 2003, and 2008. The resolution of all images used was 30 m × 30 m; maintaining a consistent grain size is important because the landscape metrics are generally sensitive to the spatial resolution. The detailed image processing techniques are as follows. First, topographic maps with a 1: 50,000 scale were employed for geometric correction under the D_Krasovsky_1940 coordinate system with the Albers projection. The mixed unsupervised and supervised classification method was used to yield five land-cover categories for each image, namely, cultivated land, forest land, yuan

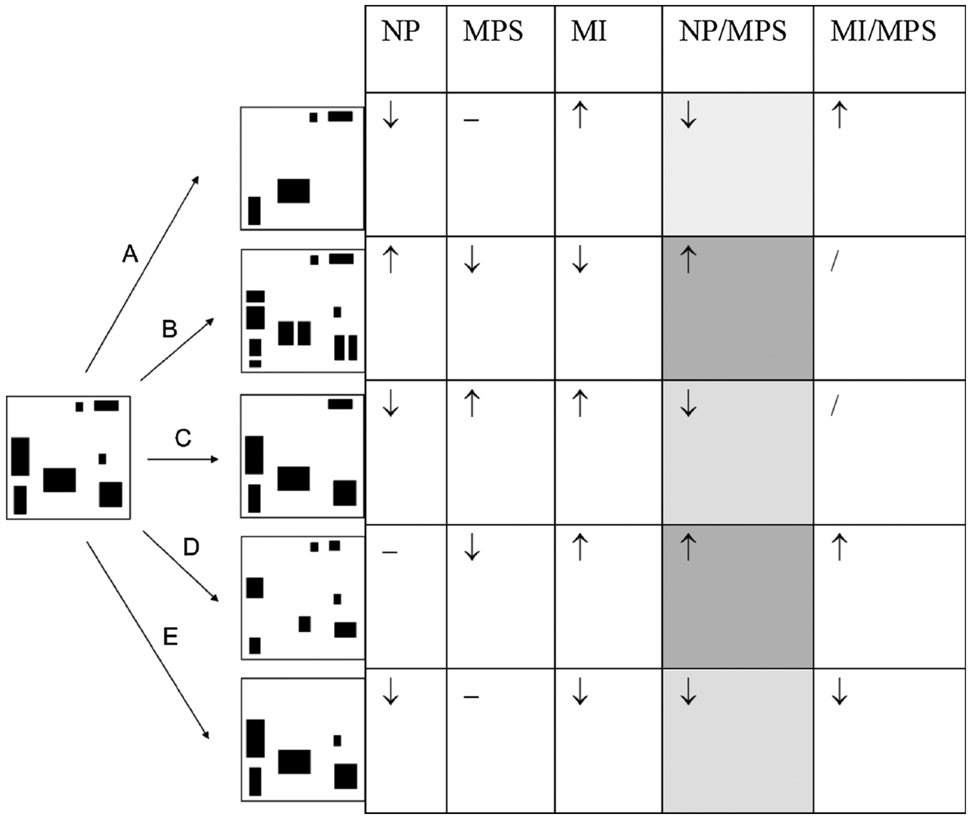

Fig. 2. Illustration of the different processes of habitat loss alone and with habitat fragmentation and the variation in landscape metrics (adapted from Fahrig (2003)). NP: number of patches; MPS: mean patch size; MI: mean isolation.

land, urban area and water area. To minimize classification error, the images were further corrected using a GIS model with the assistance of a digital elevation model (Yu and Ng, 2007). The overall accuracy approached 95%.

Because of the limited availability of water quality data, only four rivers (the Dasha, Buji, Xinzhou and Shenzhen rivers) on the Shenzhen side of the watershed and six rivers (the River Ganges, River Indus, River Beas, Yuen Long River, Kam Tin River, and Tin Shui Wai River) on the Hong Kong side were included in this study. The spatial locations of the water sampling sites for these rivers are shown in Fig. 1. The correlation of landscape composition and habitat fragmentation with water quality variation was analyzed using riparian buffers with varying widths: 100 m, 500 m, 1000 m, 1500 m, 2000 m and 2500 m. Water quality data collected over a span of 20 years (from 1988 to 2008) were acquired from the Environment Protection Department in Hong Kong and the Environment Protection Bureau in Shenzhen. Suspended solids (SS), biological oxygen demand (BOD), dissolved oxygen (DO), ammonia-nitrogen (AN) and total phosphorus (TP) were the selected water quality variables. As these data were sampled at different intervals, the annual average concentrations were used for this analysis. The concentrations for each water quality variable were averaged for five time periods, namely, 1988–1990, 1992–1994, 1997–1999, 2002–2004 and 2006–2008, which correspond to the years of the Landsat images (1988, 1993, 1998, 2003, and 2008).

2.2.3. Statistical analyses

The Kolmogorov-Smirnov (K-S) test was used to detect the normality of distribution of the variables for water quality, landscape composition and habitat fragmentation. To meet the requirements for statistical analyses, the raw data of all variables were log-transformed. A one-way analysis of variance (ANOVA) was adopted to test for significant differences among the five periods of water quality data and between the Hong Kong and Shenzhen sides of the watershed. Using CANOCO 4.5 (Braak and Smilauer, 2002), a redundancy analysis (RDA) was then employed to investigate how landscape composition and habitat fragmentation explained the variability in water quality. Before the RDA, the water quality data, including SS, BOD, DO, AN and TP, were imported into the software to test if the DCA gradient shaft length was less than 3 (Shen et al., 2015). A manual variable selection process was chosen to identify the significant variables at multiple scales based on the results of the Monte Carlo permutation method ($n = 499$). In the RDA, five water quality variables, as well as five landscape metrics (*i.e.,* three metrics measuring landscape composition and two metrics measuring habitat fragmentation) were considered. The metrics of landscape composition included the percentages of forest land, cultivated land and urban area. For habitat fragmentation, the compound indicator N/S was used.

3. Results

3.1. Temporal and spatial changes in water quality

3.1.1. Temporal change in water quality

To examine the temporal change in water quality, the mean concentration of each water quality variable and its standard error were calculated for five periods, namely, 1988–1990, 1992–1994, 1997–1999, 2002–2004 and 2006–2008. According to the ANOVA tests, as shown in Table 1, statistically significant differences were observed for the mean concentrations of SS, DO and BOD, but not for AN and TP, during the five periods. The mean concentrations and standard errors of SS, DO and BOD indicated that water quality generally improved between 1988 and 2008, with the highest water quality measured between 2002 and 2004, as shown in Fig. 3. The concentration of SS was highest between 1992 and 1994, and then it gradually decreased until 2008, reaching its lowest value in 2004. The concentration of BOD gradually decreased to its lowest level in 2004

Table 1
Results of the ANOVA tests for variation in water quality variables among the five time periods.

	F	p-value
SS	2.912	0.035*
DO	2.616	0.049*
BOD	3.211	0.022*
AN	1.873	0.134
TP	0.249	0.909

* $p < 0.05$.

Table 2
Results of the ANOVA tests for spatial differences in water quality variables.

Variables	F	p-value
SS	4.822	0.035*
DO	10.753	0.002**
BOD	4.062	0.05*
AN	3.644	0.062
TP	0.705	0.407

* $p < 0.05$.
** $p < 0.01$.

and had a small increase during the period 2006–2008. In contrast, the DO concentration gradually increased and achieved its maximum in 2002–2004. Additionally, the mean concentrations of AN and TP did not significantly vary among the five periods, likely because they barely fluctuated during the latter four periods. However, a decreasing trend can be observed compared to the first time period.

3.1.2. Comparison of the water quality in Shenzhen and Hong Kong

A similar analysis was conducted to compare the variation between the Shenzhen and Hong Kong sides of the watershed. The ANOVA test (see Table 2) indicated that statistically significant differences existed in the mean concentrations of SS, DO and BOD but not in the values for AN and TP. Generally, the mean concentrations and standard errors of SS and TP indicated poorer water quality on the Shenzhen side than on the Hong Kong side, whereas the opposite was true for the BOD and AN concentrations, as shown in Fig. 4. The DO concentration was also higher on the Hong Kong side.

3.2. Variations in landscape composition and habitat fragmentation at different riparian buffer widths in the entire watershed

Land-cover maps for the years 1988, 1993, 1998, 2003, and 2008 are shown in Fig. 5. In 1988, the land cover of the entire watershed was dominated by forest land and water area. In contrast, by 1998, forest land and urban area dominated the entire landscape. By 2008, due to the substantial increase in urban area, cultivated land declined by approximately 75.5%. The water area shrank to about two-thirds of its original area during this time frame. It can also be observed that yuan land had very little variation, with a 0.8% decrease per year. There were no great changes in terms of its amount and spatial configuration in the watershed during the study period. Thus, the composition and fragmentation of yuan land has not been analyzed here.

The percentages and fragmentation of forest land and cultivated land at riparian buffers with different widths are shown in Fig. 6. The percentage of forest land (P_F) slightly decreased between 1988 and 1993, but it experienced a sharp increase from 1993 to 1998 within the 1000-m riparian buffer widths. Since 1998, P_F has not experienced major variation. The most serious fragmentation of forest land (measured by F_N/S) was found in the 100-m riparian buffer width, where a

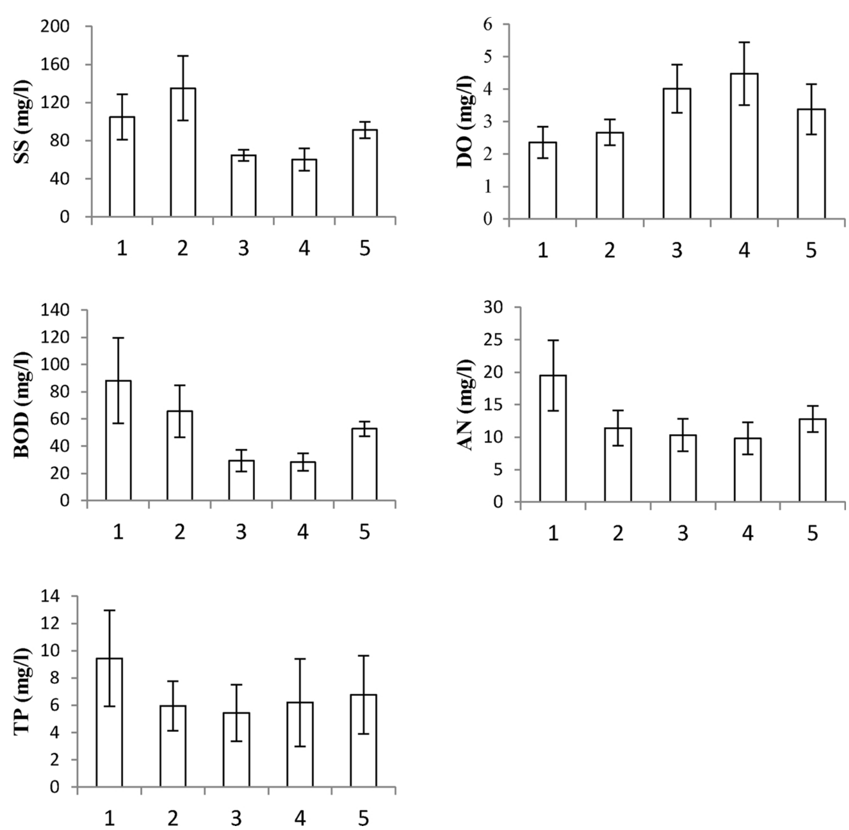

Fig. 3. The mean concentration and standard error of each water quality variable in five different time periods: (1)1988–1990; (2)1992–1994; (3) 1997–1999; (4) 2002–2004; and (5) 2006–2008.

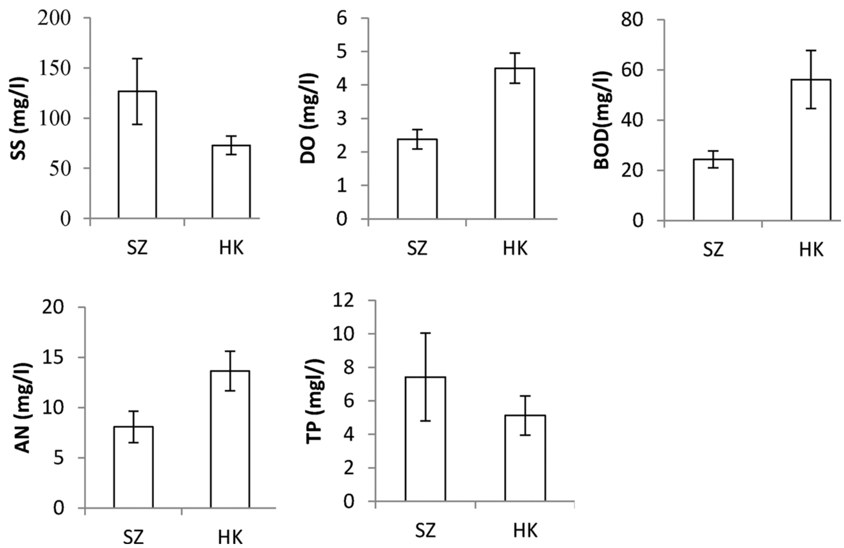

Fig. 4. The mean concentration and standard error of each water quality variable on the Shenzhen (SZ) and Hong Kong (HK) sides of the watershed during five periods: (1)1988–1990; (2) 1992–1994; (3) 1997–1999; (4) 2002–2004; and (5) 2006–2008.

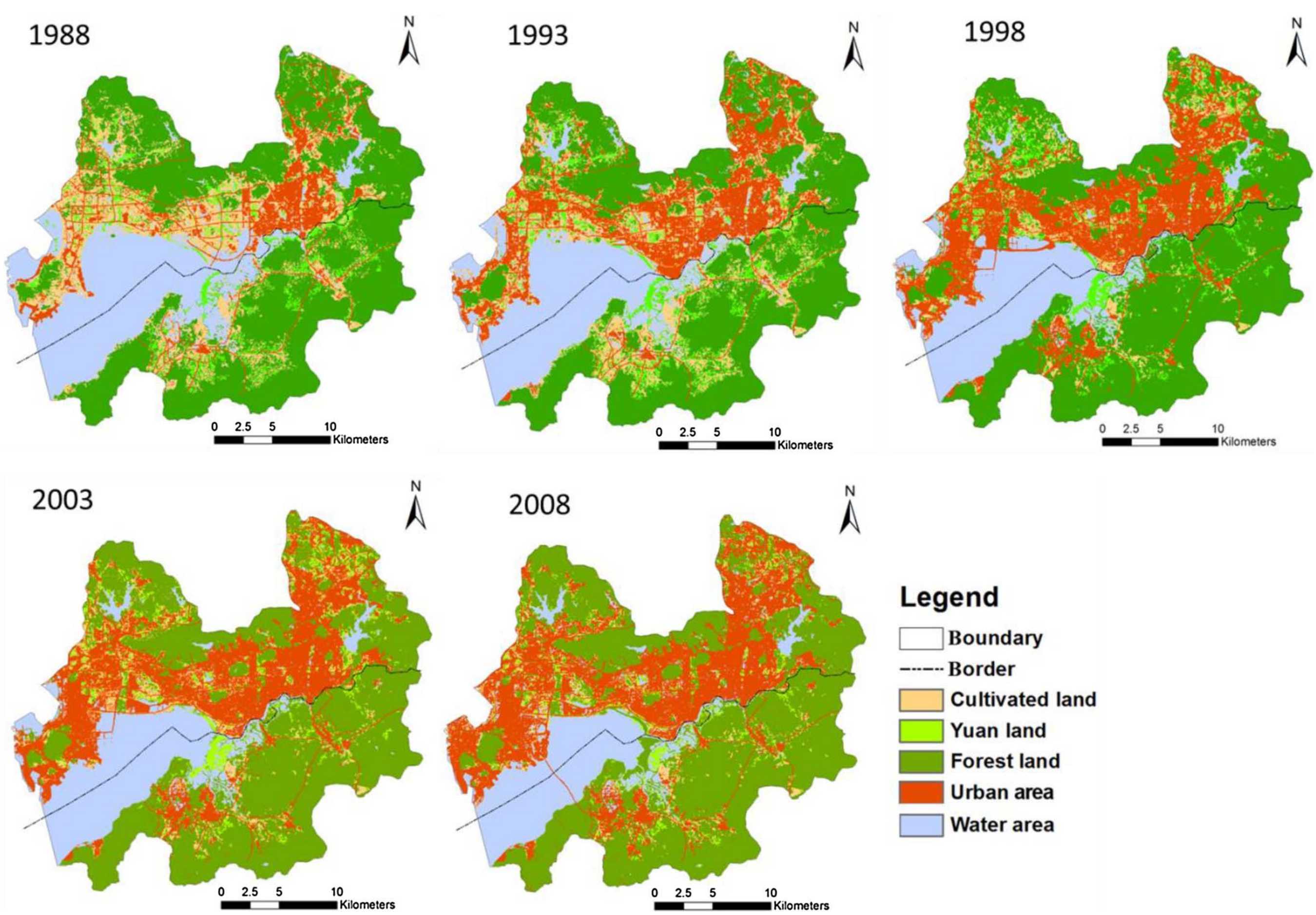

Fig. 5. Maps of land-cover change, from 1988 to 2008.

large number of forested patches had been separately into small parcels. Between the 200-m and 2500-m riparian buffer widths, the forest land had not further fragmented during the period of 1998–2008. Thus, it might be concluded that forest loss and fragmentation only occurred at the same time between 1988 and 1998. In contrast, the percentage of cultivated land (P_C) substantially dropped between 1988 and 1998 and then remained mostly constant until 2008. The situation was different after 1998, and the fragmentation of cultivated land (measured by C_N/S) significantly intensified. The fragmentation and removal of cultivated patches occurred at the same time in all riparian buffers between 1998 and 2008. Furthermore, the percentage of urban area (P_U) significantly increased during the period of 1988–1998, and then it gently rose until 2008, except in the 100-m buffer widths. In all, the landscape at the examined buffer zones experienced considerable

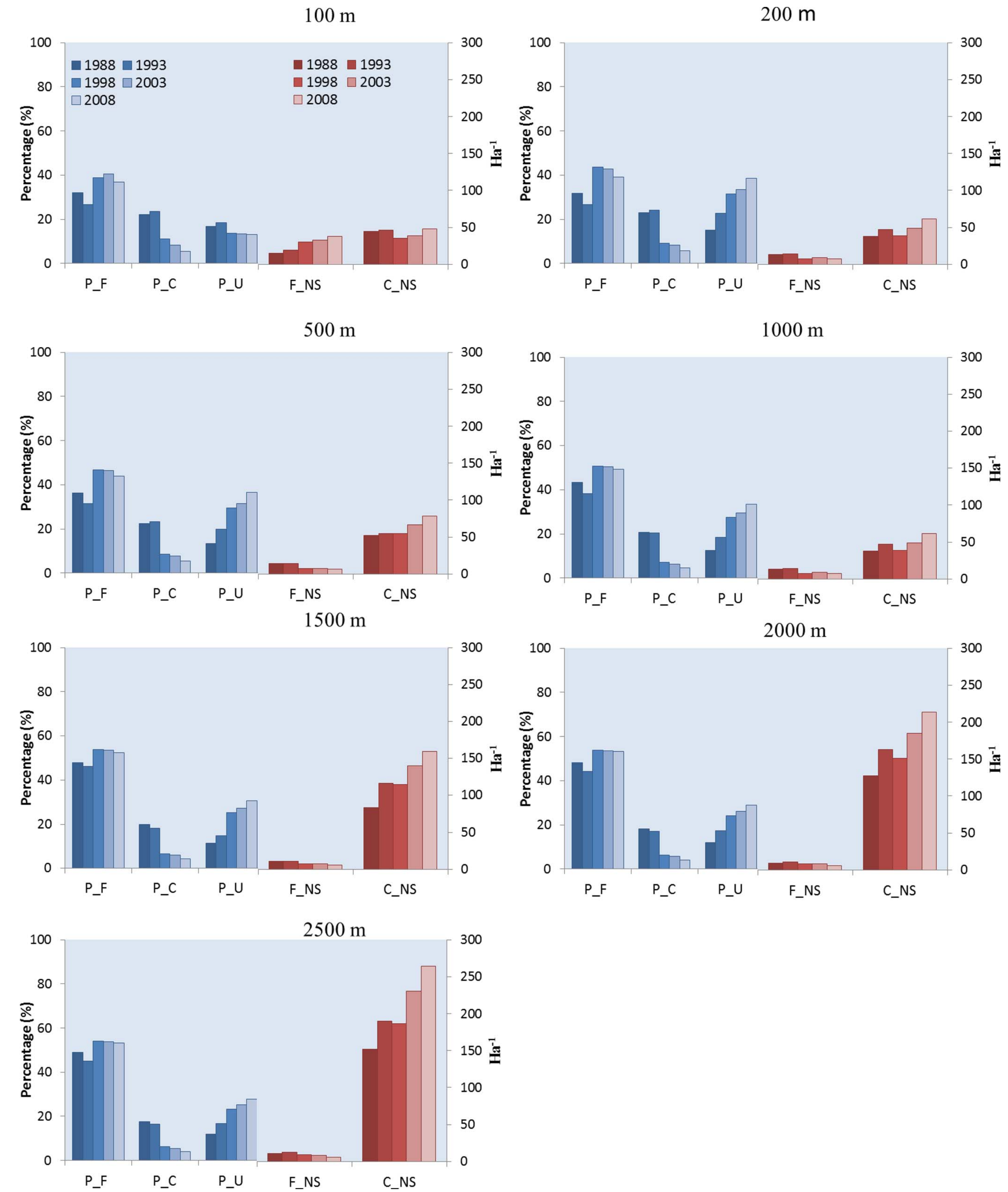

Fig. 6. Variations in metrics for measuring landscape composition and habitat fragmentation at different riparian buffer widths.

habitat loss between 1988 and 1998, whereas habitat fragmentation appeared to be the predominant process between 1998 and 2008 in these buffer zones.

3.3. The correlation of landscape composition and habitat fragmentation with water quality at different riparian buffer widths in the entire watershed

The results of the RDA of different riparian buffer widths throughout the entire watershed are shown in Tables 3, 4 and Fig. 7.

Table 3
Redundancy analysis using the water quality variables and landscape/habitat metrics for multiple buffer widths.

Riparian buffer widths	Axis 1	Axis 2	Axis 3	Axis 4	Total explained variance
100 m					
Eigen values	0.114	0.084	0.031	0.014	24.4%
Cumulative percentage correlation of habitat-water quality data	46.7	81.0	93.9	99.8	
200 m					
Eigen values	0.099	0.095	0.063	0.022	27.9%
Cumulative percentage correlation of habitat-water quality data	35.7	69.6	92.2	99.9	
500 m					
Eigen values	0.142	0.076	0.075	0.022	31.7%
Cumulative percentage correlation of habitat-water quality data	44.8	68.9	92.4	99.4	
1000 m					
Eigen values	0.118	0.072	0.05	0.028	26.9%
Cumulative percentage correlation of habitat-water quality data	43.7	70.6	89.2	99.7	
1500 m					
Eigen values	0.116	0.053	0.032	0.018	22.1%
Cumulative percentage correlation of habitat-water quality data	52.3	76.1	90.8	99.1	
2000 m					
Eigen values	0.115	0.077	0.048	0.009	24.8%
Cumulative percentage correlation of habitat-water quality data	46.1	77.0	96.5	100.0	
2500 m					
Eigen values	0.114	0.077	0.054	0.006	25.2%
Cumulative percentage correlation of habitat-water quality data	45.4	76.1	97.4	99.8	

Table 4
Significant metrics and their explanatory abilities for water quality variables across multiple buffer widths.

Riparian buffer widths	Significant variables	Explained variance (%)	Proportion of total explained variance (%)	Riparian buffer widths	Significant variables	Explained variance (%)	Proportion of total explained variance (%)
100 m	P_U	0.080	32.8	1000 m	P_U	0.104	38.7
	F_N/S	0.055	22.5		P_F	0.054	20.1
	C_N/S	0.047	19.3		C_N/S	0.052	19.3
200 m	P_U	0.087	32.1	1500 m	P_U	0.107	48.4
	C_N/S	0.061	21.4		P_F	0.047	21.3
	P_F	0.058	21.4	2000 m	P_U	0.107	43.1
	P_C	0.044	14.3		C_N/S	0.065	26.2
500 m	P_U	0.100	31.2		P_C	0.053	21.4
	C_N/S	0.066	21.9	2500 m	P_U	0.108	42.8
	P_F	0.059	18.8		C_N/S	0.061	24.2
	F_N/S	0.048	15.6		P_C	0.057	22.6

The correlations of water quality with landscape composition and habitat fragmentation varied with increasing buffer width, which ranged from 100 m to 2500 m. As shown in Table 3, the metrics of landscape composition and habitat fragmentation at multiple scales explain approximately 30% of the water quality variation in the watershed in the full model. The first two ordinate axes accounted for approximately 70% of the total correlation reflected by all axes. As the buffer width increased from 100 m to 500 m, the explanatory power increased from 24.4% to 31.7%. However, when the buffer width increased over 500 m, the explanatory power decreased slightly. Thus, the strongest linkage between water quality and landscape composition and/or habitat fragmentation occurred at the 500-m riparian buffer width.

Table 4 suggests that the significant landscape metrics varied in their ability to explain the variation in the water quality variables at each riparian buffer width. More specifically, P_U and C_N/S were significantly correlated with water quality variables at almost all riparian buffer widths. In contrast, F_N/S was only significantly related to water quality variables at the 100-m and 500-m buffer widths. However, these variables have different explanatory abilities for different buffer widths. For example, for P_U, the proportion of the total explained variance ranged from 31.2% to 48.4%. P_F could only significantly explain the variation in water quality between the 200-m and 1500-m buffer widths, and the proportion of the total variance explained by P_F was approximately 20%.

For the full model, the explanatory abilities of landscape composition and habitat fragmentation exhibited different characteristics. As suggested by our method, the process of habitat fragmentation (as measured by F_N/S and C_N/S) was differentiated from the process of habitat loss only. For all buffer widths in the entire watershed, the proportion of the total explanatory ability of habitat fragmentation (*i.e.*, the sum of the values of the explained variance for F_N/S and C_N/S) ranged from 20.8% to 45.5%, with an average value of 31.4%. Landscape composition (*i.e.*, the sum of the values of the explained variance for P_C, P_F and P_U) explained almost 70% of the variation in the water quality variables. Determining whether habitat loss and fragmentation caused water quality degradation was complicated. As shown in Fig. 7, the concentration of TP was positively correlated with P_U in all buffer zones, which suggested that the urban sprawl, directly resulting in the loss of habitat amount, might be a major contributor to the concentration of TP. Within the 1500-m buffer width, P_F was negatively related to the concentrations of TP and BOD, and F_N/S was positively associated with the concentrations of TP and BOD. These results indicate that both a larger area and better connectivity of forest habitat would decrease the concentrations of TP and BOD. However, P_C and/or C_ N/S were positively associated with the concentrations of AN (except at the 1000-m buffer width), which indicates that a larger area and better connectivity of cultivated land would increase the concentration of AN. Finally, it should be noted that P_U was negatively correlated with the concentrations of SS and DO.

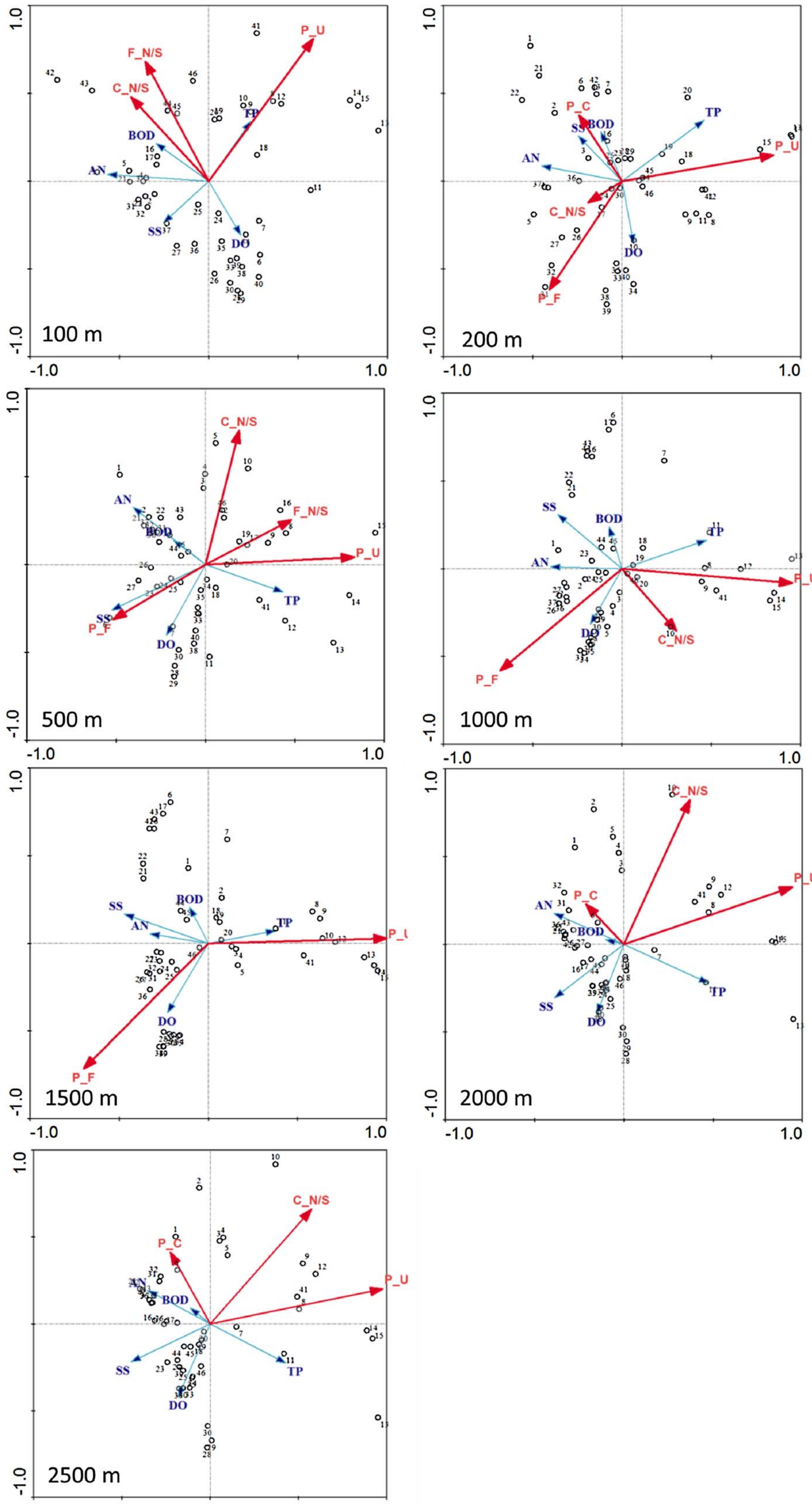

Fig. 7. Redundancy analysis biplots showing the correlations between landscape/habitat metrics and water quality variables across multiple buffer widths for the entire watershed.

Table 5
Redundancy analysis of the water quality variables and landscape/habitat metrics for multiple buffer widths on the Hong Kong side of the watershed.

Riparian buffer widths	Axis 1	Axis 2	Axis 3	Axis 4	Total explained variance
100 m					
Eigen values	0.347	0.072	0.021	0.007	44.8%
Cumulative percentage correlation of habitat-water quality data	77.4	93.5	98.1	99.8	
200 m					
Eigen values	0.311	0.062	0.045	0.009	42.8%
Cumulative percentage correlation of habitat-water quality data	72.6	87.2	97.8	100.0	
500 m					
Eigen values	0.416	0.143	0.057	0.011	62.7%
Cumulative percentage correlation of habitat-water quality data	66.4	89.2	98.2	100.0	
1000 m					
Eigen values	0.362	0.081	0.058	0.013	51.4%
Cumulative percentage correlation of habitat-water quality data	70.4	86.2	97.5	100.0	
1500 m					
Eigen values	0.333	0.068	0.040	0.012	45.3%
Cumulative percentage correlation of habitat-water quality data	73.4	88.3	97.2	100.0	
2000 m					
Eigen values	0.255	0.078	0.013	0.007	35.3%
Cumulative percentage correlation of habitat-water quality data	72.2	94.4	98.0	99.9	
2500 m					
Eigen values	0.254	0.080	0.014	0.005	35.3%
Cumulative percentage correlation of habitat-water quality data	72.0	94.6	98.5	99.9	

3.4. Comparison between the Hong Kong and Shenzhen sides of the watershed

3.4.1. Correlations on the Hong Kong side of the watershed

Tables 5 and 6 and Fig. 8 show the results of the RDA of riparian buffer widths on the Hong Kong side of the watershed. Overall, water quality was significantly correlated with landscape composition and habitat fragmentation for all buffer widths. As shown in Table 5, in the full model, the metrics of landscape composition and habitat fragmentation on the Hong Kong side explained more than 40% of the spatial variation in the water quality within the 1500-m buffer width, and these metrics had the greatest explanatory ability, 62.7%, for the 500-m buffer width. The first two ordinate axes accounted for more than 86% of the total correlation reflected by all the axes. When the buffer width increased from 200 m to 500 m, the explanatory power increased from 42.8% to 62.7%. However, the explanatory power decreased to 35.3% for both the 2000-m and 2500-m buffer widths. Thus, the strongest relationship between water quality and landscape composition and/or habitat fragmentation was found in the 500-m riparian buffer.

As shown in Table 6, the landscape composition and habitat fragmentation metrics that significantly explained the variation in water quality varied with buffer width. More specifically, P_U was the only variable that was significantly correlated with all water quality parameters. However, different explanatory abilities were observed for different buffer widths. For example, for P_U, the proportion of the total explained variance ranged from 12.6% to 62.6%. F_N/S was significantly related to water quality at all buffer widths except 200 m and 2500 m. P _F was significantly related to the variation in water quality for riparian buffer widths between 200 m and 2000 m, and its proportion of total explained variance ranged from 9.3% to 51.2%.

For the full model, the explanatory abilities of landscape composition and habitat fragmentation showed different characteristics. On the Hong Kong side of the watershed, landscape composition seemed to be the dominant contributor to the variation in water quality for most buffer widths, and the proportion of total explanatory ability of landscape composition ranged from 68.3% to 83.5%, with an average value of 74.3%. Indeed, it remained unclear whether habitat loss and fragmentation caused water quality degradation. As shown in Fig. 8, the concentration of TP was positively correlated with P_C and/or negatively associated with P_F. Thus, the decrease in forest land (as a type of natural habitat) led to the degradation of water quality, whereas the decrease in cultivated land contributed to the improvement of water quality. P_F was negatively correlated with the concentrations of AN and BOD, and F_N/S was positively associated with the concentrations of AN and BOD. These results indicate that both a greater area and

Table 6
Significant metrics and their explanatory abilities for water quality variation within multiple buffer widths on the Hong Kong side of the watershed.

Riparian buffer widths	Significant variables	Explained variance (%)	Proportion of total explained variance (%)	Riparian buffer widths	Significant variables	Explained variance (%)	Proportion of total explained variance (%)
100 m	F_N/S	0.089	19.9	1500 m	P_C	0.158	34.9
	P_U	0.114	25.4		F_N/S	0.070	15.5
	C_N/S	0.124	27.7		P_U	0.057	12.6
200 m	P_C	0.072	16.8		P_F	0.153	33.8
	P_U	0.062	14.5	2000 m	P_U	0.191	54.1
	P_F	0.219	51.2		P_F	0.056	15.9
500 m	P_C	0.106	16.9		F_N/S	0.072	20.4
	F_N/S	0.134	21.4	2500 m	P_U	0.221	62.6
	P_F	0.090	14.4		C_N/S	0.041	11.6
	P_U	0.233	37.2				
	C_N/S	0.065	10.4				
1000 m	P_C	0.134	26.1				
	F_N/S	0.111	21.6				
	P_F	0.048	9.3				
	P_U	0.194	37.7				

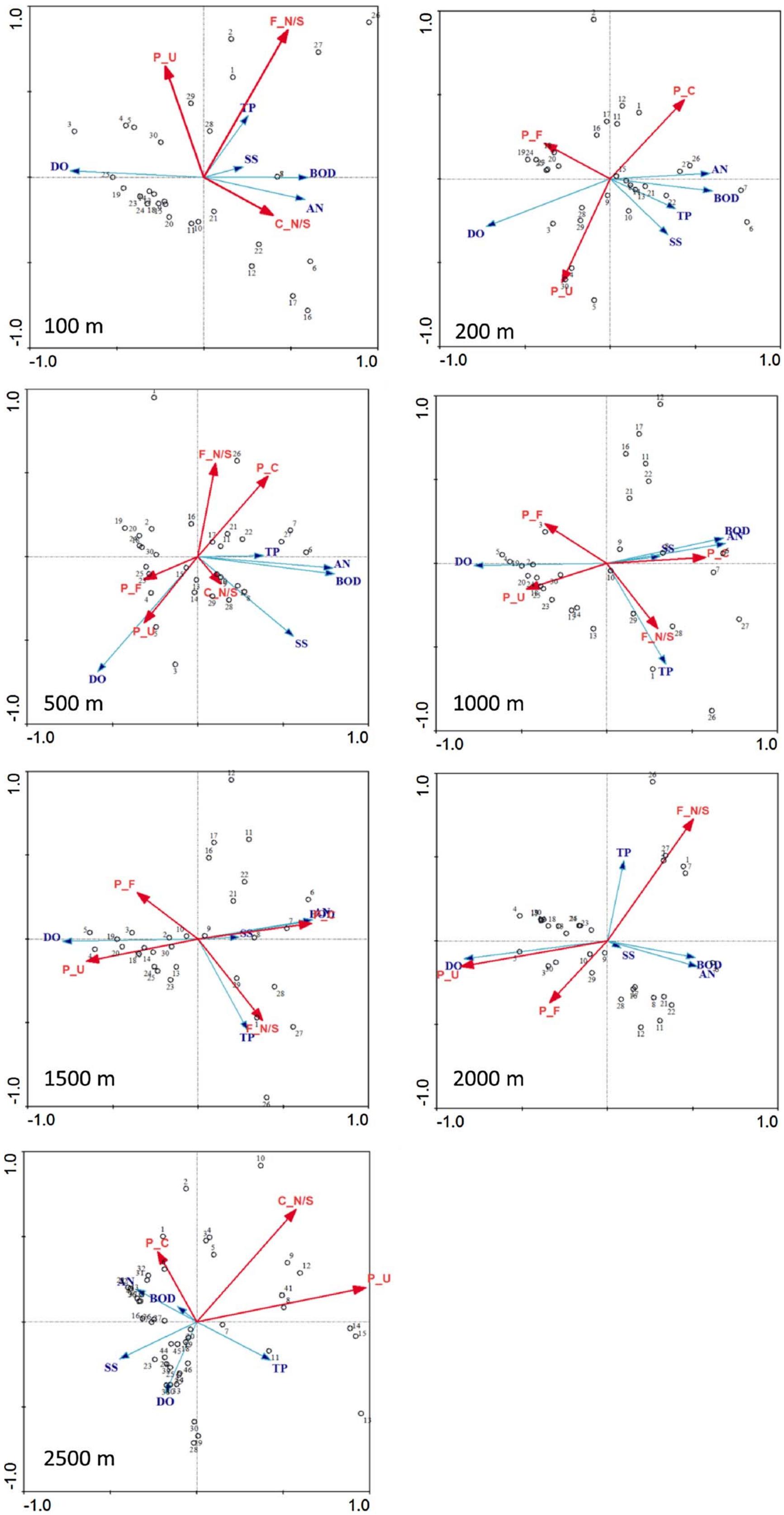

Fig. 8. Redundancy analysis biplots showing the correlations between landscape/habitat metrics and water quality variables within multiple buffer widths on the Hong Kong side.

Table 7
Redundancy analysis using the water quality variables and landscape/habitat metrics for multiple buffer widths on the Shenzhen side of the watershed.

Riparian buffer widths	Axis 1	Axis 2	Axis 3	Axis 4	Total explained variance
100 m					61.9%
Eigen values	0.341	0.241	0.028	0.009	
Cumulative percentage correlation of habitat-water quality data	55.2	94.1	98.6	100.0	
200 m					66.4%
Eigen values	0.380	0.235	0.037	0.010	
Cumulative percentage correlation of habitat-water quality data	57.3	92.7	98.3	99.8	
500 m					56.1%
Eigen values	0.332	0.182	0.035	0.012	
Cumulative percentage correlation of habitat-water quality data	59.2	91.7	97.9	100.0	
1000 m					53.9%
Eigen values	0.316	0.187	0.032	0.004	
Cumulative percentage correlation of habitat-water quality data	58.6	93.3	99.2	100.0	
1500 m					61.2%
Eigen values	0.350	0.180	0.070	0.012	
Cumulative percentage correlation of habitat-water quality data	57.2	86.7	98.1	100.0	
2000 m					54.3%
Eigen values	0.317	0.208	0.015	0.004	
Cumulative percentage correlation of habitat-water quality data	58.3	96.5	99.3	100.0	
2500 m					55.8%
Eigen values	0.335	0.209	0.011	0.003	
Cumulative percentage correlation of habitat-water quality data	60.0	97.4	99.5	100.0	

better connectivity of forest habitat would decrease the concentrations of AN and BOD. However, P_C and C_N/S were both positively associated with the concentration of AN, which indicated that both a greater area and better connectivity of cultivated land would increase the concentration of AN. Finally, the percentage of urban area was negatively correlated with the concentrations of TP, BOD and AN and positively correlated with the concentration of DO, which suggests that urban expansion, directly resulted in habitat loss, has not resulted in the degradation of water quality on the Hong Kong side of the watershed.

3.4.2. Correlations on the Shenzhen side of the watershed

The results of the RDA on the Shenzhen side of the watershed, shown in Tables 7, 8 and Fig. 9, suggest that the concentrations of BOD, AN and TP were significantly correlated with landscape composition and habitat fragmentation. As shown in Table 7, the landscape composition and habitat fragmentation explained more than 50% of the spatial variation in water quality in the full model, and the metrics for the 200-m buffer width had the largest explanatory ability, 66.4%, on the Shenzhen side of the watershed. The first two ordinate axes accounted for more than 90% of the total correlation reflected by all axes except the 1500-m buffer width. Thus, the explanatory ability on the Shenzhen side of the watershed was somewhat higher than that on the Hong Kong side of the watershed.

As shown in Table 8, although water quality at all riparian buffer widths was strongly related to landscape composition and/or habitat fragmentation, no metrics significantly explained all water quality parameters for all buffer widths. P_U was significantly related to the variation in water quality except at the 200-m buffer width, and P_C was strongly correlated with water quality for the 1500 m and below buffer width. Additionally, different explanatory abilities were observed among the buffer widths. More specifically, for P_U, the proportion of the total explained variance ranged from 20.0% to 48.2%. P_F was not closely associated with water quality at any buffer width, and F_N/S only significantly explained the variation in water quality for the buffer widths of 100 m and 200 m, with proportions of total explained variance of 36.3% and11.9%, respectively.

For the full model, the explanatory abilities of landscape composition and habitat fragmentation were distinct. For all buffer widths on the Shenzhen side, the proportion of the total explanatory ability for habitat fragmentation ranged from 15.8% to 57.5%, with an average value of 36.7%. Landscape composition remained the dominant contributor at most buffer widths, with the exception of 200 m. Similarly, the correlation of landscape composition and habitat fragmentation with water quality, as shown in Fig. 9, varied among the variables. P_C was negatively associated with the concentrations of TP, BOD and AN and positively correlated with the concentrations of SS and DO. Therefore, the decrease in cultivated land did not improve water quality, which contrasted with the trends observed on the Hong Kong side of the watershed. It could be concluded that the increase in the amount of urban land converted from cultivated land may have had a more serious influence on water quality, as suggested by the increases in BOD and TP and the decrease in DO. C_N/S was positively correlated with the concentration of AN, which indicated that the fragmentation of cultivated land further increased AN. Moreover, in contrast with the Hong Kong side of the watershed, P_F was not a significant predictor of water quality variance on the Shenzhen side of the watershed.

Table 8
Significant metrics and their explanatory abilities for water quality variation within multiple buffer widths on the Shenzhen side of the watershed.

Riparian buffer widths	Significant variables	Explained variance (%)	Proportion of total explained variance (%)	Riparian buffer widths	Significant variables	Explained variance (%)	Proportion of total explained variance (%)
100 m	P_C	0.215	34.7	1000 m	P_U	0.254	47.1
	P_U	0.124	20.0		P_C	0.160	29.7
	F_N/S	0.225	36.3	1500 m	P_U	0.252	41.2
200 m	P_C	0.231	34.8		C_N/S	0.152	24.8
	C_N/S	0.303	45.6		P_C	0.128	20.9
	F_N/S	0.079	11.9	2000 m	P_U	0.259	47.7
500 m	P_U	0.248	44.2		C_N/S	0.159	29.3
	C_N/S	0.179	31.9	2500 m	P_U	0.269	48.2
	P_C	0.096	17.1		C_N/S	0.149	26.7

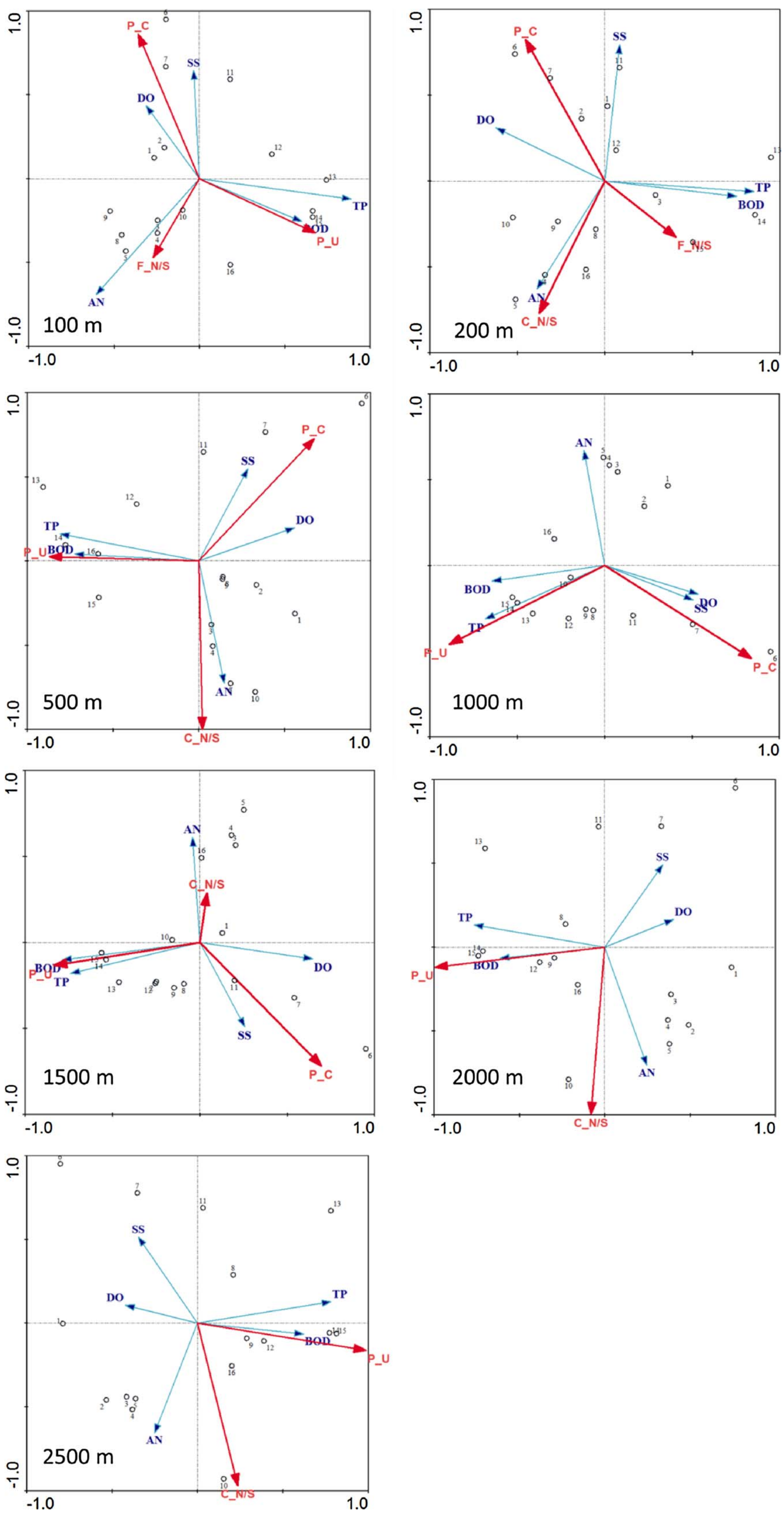

Fig. 9. Redundancy analysis biplots showing correlations between landscape/habitat metrics and water quality variables within multiple buffer widths on the Shenzhen side.

4. Discussion

4.1. Disentangling the effect of habitat fragmentation from habitat loss alone

To examine the correlation between habitat fragmentation with water quality degradation, the indicators of habitat fragmentation should be carefully chosen or developed. Many indicators exist that reflect some aspects of habitat fragmentation and have been widely used to investigate the effects of landscape configuration on water quality (Amiri and Nakane 2009; Guo et al., 2010; Liu et al., 2012; Zhou et al., 2012; Huang et al., 2013; Shen et al., 2015). However, these indicators did not satisfy the requirements of this study. First, an indicator by itself should ideally reflect the variation in habitat fragmentation. Indicators, such as the number of patches, will increase with the degree of habitat fragmentation and with the number of habitat patches due to the different increments of habitat area. Second, an indicator should be able to disentangle the process of habitat fragmentation independent on habitat loss. In most cases, although habitat fragmentation results in habitat loss, it is distinct from habitat loss alone. However, some indicators, such as the mean patch size, will decrease during both the processes of habitat fragmentation and habitat loss without fragmentation. Third, for application purposes, an indicator should be easily understood by decision makers and easily applied in practical management. In this study, based on the concept and process of fragmentation, the compound indicator N/S was adopted to measure habitat fragmentation. As N/S increases, the degree of habitat fragmentation rises, and this indicator is not positively correlated with habitat loss, which could avoid the general problem of collinearity (Smith et al., 2009). However, the indicators measuring edge characteristics, such as edge density and total edge which have been reported to be closely related to water quality (Clément et al., 2017), was not included in this study and should be further studied. Therefore, the key values of the metrics proposed in this study should be considered as a preliminary exploration that can provide guidance for future research.

4.2. Influences of landscape composition and habitat fragmentation on water quality

In this study, the variation in water quality was tightly correlated with landscape composition and habitat fragmentation in all riparian buffer widths. At multiple buffer widths, the metrics of landscape composition and habitat fragmentation explained more than 40%–50% of the variation in water quality on both the Hong Kong and Shenzhen sides of the watershed. Other studies reached the same conclusion, *i.e.*, changes in landscape structure resulted in changes in the flow of energy and nutrients of surface runoff (Shen et al., 2014). However, our explanatory ability was somewhat less than that in other studies (Shen et al., 2015; Zhao et al., 2015), partially because not all rivers in our study area were included in this study due to the lack of necessary data. Additionally, in our study area, the average explanatory abilities of habitat fragmentation were 31.4%, 25.7% and 36.7% for the entire watershed, the Hong Kong side of the watershed, and the Shenzhen side of the watershed, respectively. Landscape composition was generally more strongly correlated with the variation in water quality than was habitat fragmentation. Therefore, we accepted the first hypothesis. This finding is in accordance with Clément et al. (2017), which suggested that the proportions of different land-cover types played a dominant role in influencing water quality for relatively large watersheds ($> 250\,km^2$), as landscape proportions were the most significant structural characteristics that influenced landscape pattern. In our study area, as shown in Figs. 3, 5 and 6, considerable variations in water quality variables occurred during the period from 1988 to 1998, when landscape composition seems to be the most significant process of landscape change. As a result, landscape composition had a stronger correlation with variation in water quality than did habitat fragmentation.

Whether habitat loss and fragmentation had a negative effect on water quality depended on the habitat types and water quality variables that were examined. More specifically, the decrease in forest land (as a type of natural habitat) led to the degradation of water quality, whereas the decrease in cultivated land contribute the improvement of water quality on the Hong Kong side of the watershed; in contrast, the inverse relationship was observed on the Shenzhen side of the watershed. It has been suggested that riparian forest might act as a "sink" or a "filter" to reduce pollution in streams produced by human activities in the surrounding area (Lowrance et al., 1997; Kotowska et al., 2016). With an increase in the proportion of forest land, as well as the proximity of forest land to streams, water pollutants (e.g., nitrate levels) decline in coastal estuaries (Basnyat et al. 1999). However, the amount of agriculture land exerts either positive or negative impacts on different water quality variables (Rothenberger et al. 2009; Zhou et al. 2012). A positive relationship between percent agriculture and biological integrity scores was observed in catchments due to the negative correlation between the extent of agriculture and urban land (Snyder et al. 2003). On the Hong Kong side of our study area, the loss of cultivated land may have been caused by the transformation of land to urban area that has well-developed sewage treatment systems; this would result in the improvement of water quality. Therefore, we partially rejected the third hypothesis, and suggested that habitat loss and fragmentation do not always result in the degradation of water quality. Indeed, habitat fragmentation was also found to be positive or negative correlated with biodiversity (Fahrig, 2017).

4.3. Effect of buffer width and its implications

Variation in the riparian landscape plays a significant role in altering ecological and hydrological conditions, such as surface water quality, soil erosion and wildlife and fish habitat (Apan et al., 2002; Xiao and Ji, 2007). In this study, the relationship between water quality and landscape composition/habitat fragmentation significantly varied with riparian buffer width on the Hong Kong side of the watershed, and the strongest relationship occurred within the 500-m riparian buffer. In contrast, on the Shenzhen side of the watershed, the relationship between water quality and landscape composition/habitat fragmentation did not vary substantially with riparian buffer width, and the strongest linkage occurred in the 200-m riparian buffer. Furthermore, the question of which scale the strongest relationship between landscape configuration and water quality occurs has been widely discussed (Zhao et al., 2015; Tanaka et al., 2016). In this study, the relationship and the strongest linkage at the specific riparian buffers were different for the Hong Kong and Shenzhen sides of the watershed, as each side has a unique combination of landscape characteristics and other comprehensive influence factors that affect the water quality (Huang et al., 2013). It is also suggested that the utility of landscape indicators related to water quality is dependent on the percent of the watershed occupied by source-cover types (Clément et al., 2017). Therefore, we supported the second hypothesis. In all, to manage water quality impacts, a multiscale perspective that considers different related factors (e.g., landscape composition or habitat fragmentation) and scales (e.g., buffer widths or watersheds) should be adopted (Ding et al., 2016). Additionally, in this study, the annual average concentrations of the water quality variables were examined, but seasonal influences should be considered because they could also change the relationship between landscape characteristics and surface water quality (Rothenberger et al., 2009; Huang et al., 2013). Seasonal influences are of great importance in our study area, where there is a clear distinction between the dry and wet seasons. Therefore, this variable should be included in future studies.

4.4. Effects of urban sprawl on water quality

Numerous studies have suggested urban expansion has negative effects on water quality (Rothenberger et al., 2009; Guo et al., 2010; Zhou et al., 2012). However, in the case of the Shenzhen River cross-border watershed—especially on the Hong Kong side—the percentage of urban area was negatively related to water quality degradation. Although the urban area has expanded in this watershed over the past three decades, the water quality appears to have slightly improved based on the above analyses. This scenario might be because non-point source pollution from agricultural activity was previously the primary source of water pollution; however, as urban development on the Hong Kong side of the watershed increased, most agricultural activities ceased, and land use has been transformed to urban residential, commercial and recreational uses, all of which have well-developed sewage treatment systems. In contrast, on the Shenzhen side of the watershed, rapid urbanization has also transformed much of the previously cultivated land to urban use (Xie and Ng, 2013), but water quality has deteriorated. Moreover, as shown in Fig. 9, the fragmentation of cultivated land has strengthened the impact on water quality. This different scenario seen on the Shenzhen side of the watershed could be mainly due to the different timing of when urbanization occurred, compared to urbanization in Hong Kong. By 1988, large-scale urban expansion on the Hong Kong side of the watershed was complete, and the sewage treatment system and the related management policies for water resources were well developed in the urban area. However, at this time, Shenzhen had only just entered into its phase of rapid urban sprawl and intensive urban development. Urbanization in Shenzhen was closely associated with industrialization and dramatic human population growth, which would increase the pressure on water quality (Varis and Vakkilainen, 2001; Schmiegelow and Mönkkönen, 2002). More importantly, the pro-growth strategy, poor sewage treatment systems, and insufficient and ineffective environmental protection policies might enhance the negative impacts on water quality (Lee, 2002).

5. Conclusions

In this study, we examined the correlation of landscape composition and habitat fragmentation with variation in surface water quality in the Shenzhen River and Deep Bay cross-border watershed. A compound indicator (NP/MPS) was proposed to disentangle the fragmentation effect from that of habitat loss alone. Five variables were selected to represent water quality, namely, the concentrations of SS, BOD, DO, AN and TP. ANOVA tests were used to examine whether water quality changed over time and with landscape spatial variation, and redundancy analysis was used to examine whether landscape composition and habitat fragmentation in different riparian buffer widths were correlated with—or had effects on—the variation in water quality. The three hypotheses proposed in the introduction were tested. The reasons dictating whether the hypotheses were accepted or rejected have been well explained. In all, the adopted method was effective in providing a sound foundation for future watershed management and landscape planning.

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