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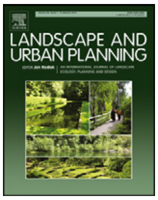

Investigating alien plant invasion in urban riparian forests in a hot and semi-arid region

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ABSTRACT

In this paper, we examined twelve riparian forests along urban–rural gradients in Austin, TX (USA), on the relationship among watershed urbanization and the invasion of alien woody species. We assessed the degree of biological invasion by measuring relative alien cover (RAC) of the riparian forests. We also measured environmental variables (15 in total) that characterize the study forests, including impervious surface percentage of corresponding watersheds, stream hydrology of adjacent streams, species diversity, canopy gap percentage, and soil nutrient contents of the riparian forests. Stream hydrology was quantified by the transfer function model. The results indicate that impervious surface percentage was related to stream hydrology: the more the impervious surface in a watershed, the faster streamflow recedes after the storm, and the longer dry period the riparian forest experienced ($R^2 = 0.722$). Impervious surface percentage was also related to RAC ($R^2 = 0.498$). Nonmetric multidimensional scaling (NMDS) grouped the 15 environmental variables into five dimensions. Multiple regression analysis of RAC on the five NMDS dimensions shows that RAC was related only to the dimension related to hydrological drought. Based on these results, we concluded that watershed urbanization facilitates the invasion of alien species in riparian forests by causing hydrologic drought, particularly in hot and semi-arid regions.

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1. Introduction

Riparian forests are prone to the invasion of alien plant species (Stohlgren et al., 1998; Tickner et al., 2001; Zedler and Kercher, 2004; Schnitzler et al., 2007). Floods disturb riparian forests mechanically as well as by creating long inundation periods (Hood and Naiman, 2000; Pettit and Froend, 2001). Once floods recede, the disturbed forests are readily invaded by alien species that can grow faster and are more tolerant to disturbance than native competitors (Nilsson et al., 1997; Morris et al., 2002; Glenn and Nagler, 2005; Stromberg et al., 2007). Stream channel further facilitates the dispersal of alien species by transporting their propagules from upstream habitats (Moggridge et al., 2009).

Urbanization aggravates this problem. Impervious surface generates more stormwater runoff and reduces groundwater recharge, which alter stream hydrology with severe flood damage during storm events and drought with longer intermittent periods (Leopold, 1968; Hollis, 1975; Rose and Peters, 2001; Groffman et al., 2002, 2003; Burns et al., 2005; Walsh et al., 2005). As a consequence, urban riparian forests become more vulnerable to the invasion of alien species that may be better able to cope with the fluctuating water levels (Medina, 1990; Moffatt et al., 2004; Maskell et al., 2006).

Previous studies reported that urban riparian forests were more invaded by alien species in urban than rural counterparts. King and Buckney (2000) found that elevated nutrient level washed off by stormwater runoff from urban watersheds contributed to biological invasion of stream and riparian ecosystems in northern Sydney, Australia. Maskell et al. (2006) discussed that increased flood damage by stream channelization decreased ecosystem resistance in Birmingham, UK. Some studies attributed the higher biological invasion to urban land use in surrounding areas. Moffatt et al. (2004) and Burton and Samuelson (2008) related urban development in surrounding areas to riparian invasion. Pennington et al. (2010) also showed that landscape variables, such as development density and the proximity to railway, are important predictors of invasion of riparian forests in Cincinnati, USA. Although these studies addressed different aspects of urbanization, they commonly identified hydrological disturbance as an underlying cause that results in biological invasion of urban riparian forests. However, only few studies (e.g., Greer and Stow, 2003; Burton et al., 2009)

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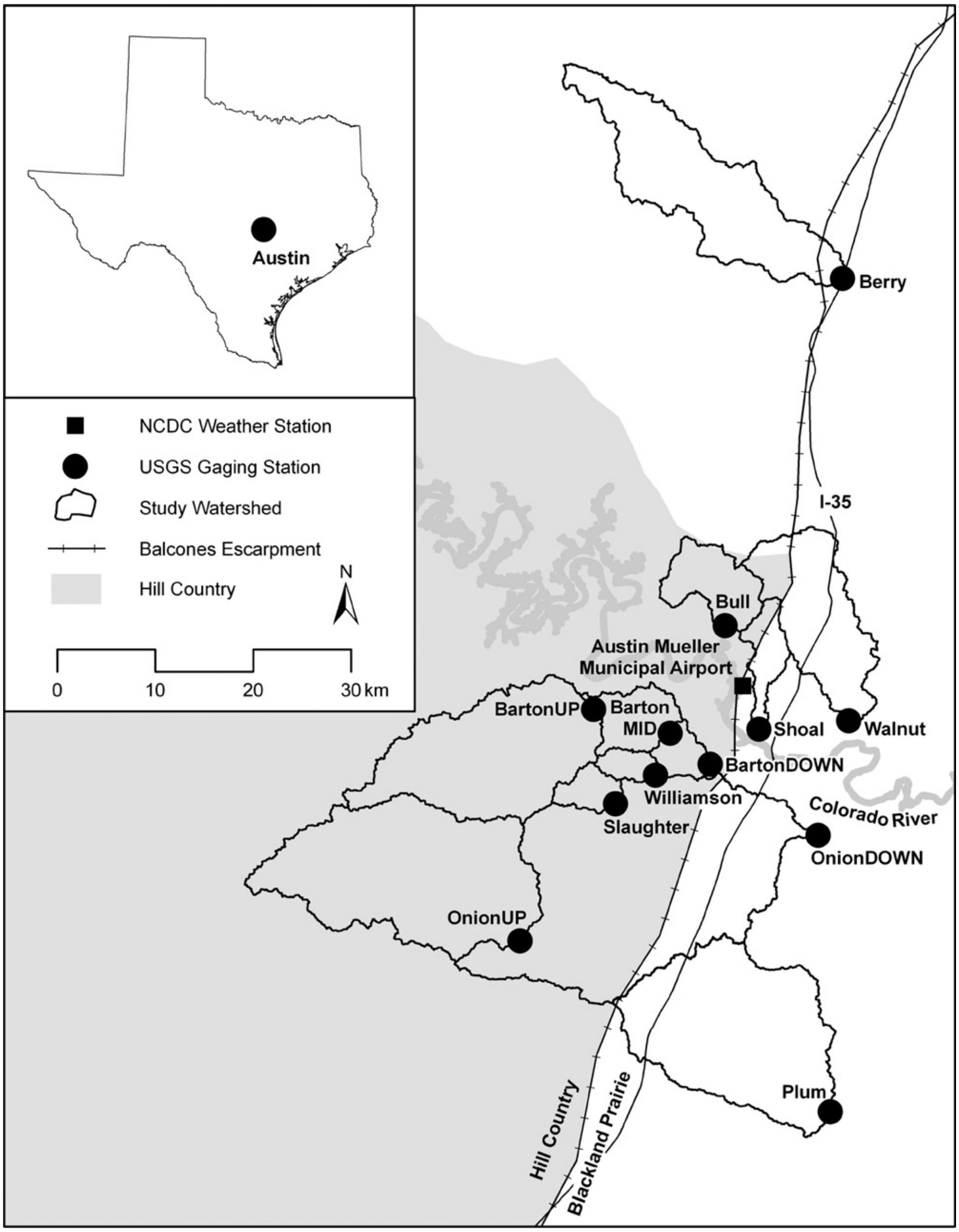

Fig. 1. Locations of twelve study sites.

directly examined the effect of hydrological disturbance on riparian invasion in urban areas.

The objective of this study is to determine whether hydrological alteration causes biological invasion in urban riparian forests. We studied twelve riparian forests in Austin, TX, USA, where rapid urbanization occurred in the last decades. We surveyed the abundance of alien species of the twelve forests and related them to 15 environmental variables, including stream hydrology, vegetation community structure, and nutrient levels of urban riparian forests. Based on these relationships, we discussed a causal pathway that links watershed urbanization and biological invasion in riparian forests.

2. Materials and methods

2.1. Study sites

The study sites are twelve riparian forests in Austin, TX, where two major ecoregions meet (Fig. 1). The eastern half of the city is the Blackland Prairie ecoregion, a flat to rolling plain with deeply incised streams. This ecoregion has deep and organic rich clay soil (Soil Conservation Service, 1974). Across the Balcones Escarpment to the west is the Hill Country, an eastern end of the Edwards Plateau ecoregion. This region has a rugged landscape with numerous hills and valleys formed by exposed limestone bedrocks (Marsh and Marsh, 1995; Woodruff and Wilding, 2008). The soil in this region is shallow and composed of gravel and clay loam weathered from the underlain Cretaceous limestone bedrock (Soil Conservation Service, 1974). Before urban development, the study area was covered by mixed forests dominated by live oak (*Quercus virginiana*), Ashe's juniper (*Juniperus ashei*), and honey mesquite (*Prosopis glandulosa*) (McMahan et al., 1984). The climate is semi-arid and subtropical with hot summer and mild winter. Mean annual precipitation is 854 mm mostly falling in spring and autumn. About 40% of annual precipitation falls during the three wettest months (May, June, and October) (US National Climatic Data Center (NCDC), 2010). Snow is rare in this region.

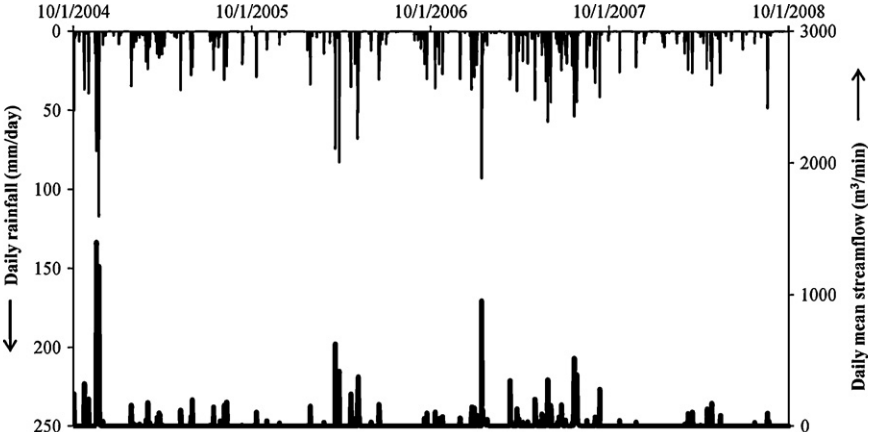

Fig. 2. Daily hyetograph and hydrograph for four water years (WY) of study period. Daily rainfall depth and daily mean streamflow were measured at Austin Mueller Municipal Airport and Shoal Creek gaging station (the most urbanized stream), respectively.

Austin is one of the fast growing cities in the United States. Population increased from 466,000 in 1990 to 758,000 in 2008 (US Census Bureau, 2009). Many natural areas were converted to urban land uses to serve the growing population. During the same period, impervious surface expanded from 21% to 31%. The Edwards Aquifer, a shallow karst aquifer underneath Austin, experienced a significant drop in the water level. Pumpage for municipal water supply and reduced groundwater recharge of stormwater were attributed to this hydrological drought. In the last decades, serious declines in baseflow were observed in many streams fed by the Edwards Aquifer (Sung and Li, 2010), which raises a concern about biological invasion in riparian forests. Biological invasion in riparian forests may threaten local and regional biodiversity because riparian forests serve as refugees for regionally rare wetland species in a hot and semi-arid region (Aguiar and Ferreira, 2005).

The twelve riparian forests were selected to couple the vegetation survey data with the daily mean streamflow data collected by the US Geological Survey (USGS). All streams are intermittent. Barton Creek and Onion Creek are monitored by multiple stations. The total number of streams is seven. Five streams are tributaries of the Colorado River, while Plum Creek and Berry Creek are tributaries of the Guadalupe River and the Brazos River, respectively.

2.2. Impervious surface estimation

We estimated impervious surface percentages of watersheds corresponding to the twelve study sites using a cloud free Landsat TM image acquired on February 7, 2008 (path/row: 27/37, level 1 terrain corrected). The watersheds were delineated from 1″ USGS National Elevation Dataset using Better Assessment Science Integrating Point & Non-point Sources (BASINS) (US Environmental Protection Agency, 2006). Previous studies (e.g., Wu and Yuan, 2007) recommended a growing season image for impervious surface mapping based on the finding that healthy foliage has a distinct spectral profile and can be easily separated from impervious surface. In this study, however, we selected the winter image because it can detect impervious surface under canopies of deciduous trees, and some evergreen trees and shrubs maintain spectral contrast between vegetation and impervious surface in winter.

We used support vector machine (SVM) to classify impervious surfaces from the Landsat TM image. SVM has been widely used for land cover mapping because of its strength in classifying pixels mixed with more than one surface material (Burges, 1998; Foody and Mathur, 2006). We classified the image into four land cover classes (impervious surface, forest, grassland, and waterbody). For each class, we selected 10 rectangular polygons (four to ten training pixels per polygon) representing pure pixels of that class. The polygons were evenly distributed throughout the study sites. For impervious surface, the training pixels covered diverse pavement and rooftop materials. Because these materials have varying spectral profiles that are not linearly separable from other land cover classes, we applied the non-linear kernel (the second order polynomial kernel) to improve classification accuracy. User-specific parameters of SVM were determined by trial-and-error so that SVM had the highest classification accuracy. We assessed the classification accuracies based on binary impervious surface maps, i.e., three non-impervious surface classes were combined. To assess classification accuracy, we collected 58 validation pixels. Initially, we selected 300 pixels randomly from the Landsat TM image and digitized impervious surfaces on-screen over the 2008 aerial photographs (0.5 m in spatial resolution), but only 29 pixels have impervious surface greater than 50% of the pixel's instantaneous field of view. Hence, we randomly selected 29 pixels from the remaining 271 non-impervious surface pixels. ENVI 4.5 was used for the image classifications (ITT VIS, 2008).

2.3. Quantifying stream hydrology

We quantified stream hydrology using TF model that describes the dynamic relationship between rainfall and streamflow in a watershed–stream system. We estimated parameters of the TF models with daily rainfall and daily mean streamflow for four water years (WY) between October 2004 and September 2008. The daily rainfall data were obtained from the weather station at Austin Mueller Municipal Airport (NCDC COOP ID: 410428) located approximately at the center of the study sites. Although there were other weather stations near some of the stream gaging stations, we did not use the data from those stations because they did not measure the rainfall from midnight to midnight (e.g., the Morgan weather station, the one near Plum Creek, read the rain gage at 8 am every day). The mean annual precipitation during the four WY study period was 863 mm, which is similar to the long term average (854 mm). Fig. 2 illustrates the daily hyetograph and hydrograph for Shoal Creek (the one near downtown Austin).

The TF model expresses daily mean streamflow as:

$$Y_t = \frac{\varpi(B)}{\delta(B)} X_t + N_t, \quad N_t = \frac{\theta(B)}{\phi(B)} e_t \tag{1}$$

where Y_t denotes the mean streamflow at day t in m^3/min, X_t is the rainfall depth at day t in mm, B is the backshift operator, $\varpi(B)$, $\delta(B)$, $\theta(B)$, and $\phi(B)$ are the backshift polynomials estimated by Box and Jenkins' (1976) procedure, N_t is unexplained noise at time t, and e_t is

Table 1
Description of variables analyzed in this study.

Variables	Description
Invasibility of riparian forest	
Relative alien cover (RAC, %)	(Sum of alien species cover/total species cover) × 100 (Magee et al., 2008)
Watershed characteristics	
Area of watershed	Delineated using national elevation data (NED) by BASINS (km^2)
Impervious surface (%)	Classified from 2008 Landsat TM image by support vector machine (SVM)
Stream hydrology	
$\hat{\varpi}_0$	Scale parameter from TF model (m^3/min/mm)
$\hat{\delta}_1$	Recession parameter from TF model [(m^3/min)/(m^3/min)]
Community structure	
Canopy gap (%)	(Lengths not covered by canopy of any woody species/length of transect) × 100
Species richness	Total number of woody plant species (Magurran, 1988)
Shannon diversity index	$-\sum_{i=1}^{S} p_i \ln p_i$ where S = number of species, p_i = the relative abundance of species i (Magurran, 1988)
Soil nutrient contents[a]	
pH	–
NO_3–N	(ppm)
P	(ppm)
K	(ppm)
Ca	(ppm)
Mg	(ppm)
S	(ppm)
Na	(ppm)

[a] Average concentrations in soil samples taken at two transects.

true white noise. For simplicity, we restricted the model by setting the orders of polynomial $\varpi(B)$ and $\delta(B)$ by 0 and 1, respectively. The simplified TF models can be written as:

$$Y_t = \frac{\varpi_0}{1 - \delta_1 B} X_t + N_t \tag{2}$$

where ϖ_0 is a parameter of the zero order term in the polynomial $\varpi(B)$, and δ_1 is a parameter of the first order term in the polynomial $\delta(B)$. Because daily streamflow rises during a storm event and ultimately returns to baseflow, rainfall–streamflow series is stationary and δ_1 must have a value between 0 and 1. Eq. (2) can be expressed as an infinite sum of weighted rainfalls on previous days:

$$Y_t = \varpi_0 \sum_{i=0}^{\infty} \delta_1 X_t + N_t \tag{3}$$

A set of parameters of $X_t, X_{t-1}, X_{t-2}, \ldots, X_{t-i} \ldots$, in Eq. (3), given as $\varpi_0, \varpi_0\delta_1, \varpi_0\delta_1^2, \ldots, \varpi_0\delta_1^i, \ldots$, are called the impulse response function and can be interpreted as the impacts of unit rainfall at day t on streamflow at day $t, t-1, t-2, \ldots, t-i, \ldots$ Because $0 < \delta_1 < 1$, the impacts decay exponentially at the rate of δ_1. Hence, δ_1 is a recession parameter indicating how fast streamflow decays after a storm. ϖ_0 is a scale parameter that represents the impact of unit rainfall on the streamflow at day 0. Higher ϖ_0 means greater streamflow during a storm event, and higher δ_1 means slower recession of streamflow after the storm (Sung and Li, 2010). The TF models were estimated by the maximum likelihood method using SAS 9.1 (SAS Institute Inc., 2004).

The TF model is conceptually equivalent to physically based hydrological models. For instance, Jakeman et al. (1990) showed that the impulse response function has the same hydrological meaning as the unit hydrograph. The TF model has an advantage over other hydrological analysis models because it can take into account antecedent rainfalls stored in reservoir, soil, and groundwater layer. The TF models have been successful to predict streamflow in various time intervals (e.g., Young and Beven, 1994; Mwakalila et al., 2001; Young, 2003).

2.4. Vegetation sampling

We surveyed woody vegetation communities near USGS gaging stations during October 2008 using line intercept method (Caratti, 2006). We placed two 30 m line transects parallel to stream bank – the first transect along the stream bank and the second one 5 m apart from the stream bank. Transects followed the curvature of the streamline. We measured lengths of canopies intersecting the transect lines for all woody vegetation taller than 0.6 m. To rule out the effect of gaging stations, such as physical damage during installation and regular maintenance, the transects began at 10 m downstream from the station. In the case that the transects

Table 2
Woody native and alien species (taller than 0.6 m) found at twelve study sites.

Scientific name	Common name	Number of site observed
Alien species		
Ligustrum lucidum	Glossy Privet	4
Ligustrum sinense	Chinese Privet	1
Melia azedarach	Chinaberrytree	8
Triadica sebifera	Chinese Tallow	3
Native species		
Acer negundo	Boxelder	4
Amorpha fruticosa	Desert False Indigo	1
Carya illinoinensis	Pecan	7
Celtis laevigata	Sugarberry	9
Celtis occidentalis	Common Hackberry	2
Celtis reticulata	Netleaf Hackberry	8
Cephalanthus occidentalis	Common Buttonbush	4
Cornus drummondii	Roughleaf Dogwood	5
Diospyros texana	Texas Persimmon	1
Eysenhardtia Texana	Texas Kidneywood	1
Fraxinus pennsylvanica	Green Ash	7
Fraxinus texensis	Texas Ash	4
Ilex decidua	Possumhaw	4
Ilex vomitoria	Yaupon	2
Juglans major	Arizona Walnut	1
Juglans nigra	Black Walnut	3
Juniperus ashei	Ashe's Juniper	1
Morus rubra	Red Mulberry	4
Parkinsonia aculeata	Jerusalem thorn	1
Platanus occidentalis	American Sycamore	5
Populus deltoides	Eastern Cottonwood	2
Quercus macrocarpa	Bur Oak	1
Quercus shumardii	Shumard's Oak	1
Rhus copallina	Winged Sumac	2
Robinia pseudoacacia	Black Locust	2
Salix nigra	Black Willow	3
Sideroxylon lanuginosum	Gum Bully	3
Taxodium distichum	Bald Cypress	1
Ulmus Americana	American Elm	8
Ulmus crassifolia	Cedar Elm	8

passed over areas highly disturbed by human activities (e.g., buildings, trails, street trees, and lawns), we selected an alternative location in the following order: (1) across the stream from the gaging station, (2) toward the upstream direction at the same side of the station, (3) toward the upstream direction across the stream from the station. Many of the study sites were narrow linear remnant forests encompassed by urban land uses, and the selected locations were the only available bank among the four alternatives. Thus, we were not able to survey vegetation communities in both sides of stream banks and/or along wetland–upland gradient. Nomenclature and native status of species followed the USDA NRCS PLANTS database (http://plants.usda.gov). Soil samples were taken at a depth of 30 cm in the middle of each transect and sent to the Texas A&M University Soil, Water and Forage Testing Laboratory (http://soiltesting.tamu.edu) for analyses of pH, nitrate–N (NO_3–N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sulfur (S), and sodium (Na) concentrations.

2.5. Data analysis

We summarized the vegetation survey data to characterize the vegetation communities of the riparian forests. The abundance of alien species was estimated by relative alien cover (RAC) (Magee et al., 2008). Canopy gap percentage, species richness, and Shannon diversity index were also calculated from the survey data to measure vegetation community structure (Magurran, 1988).

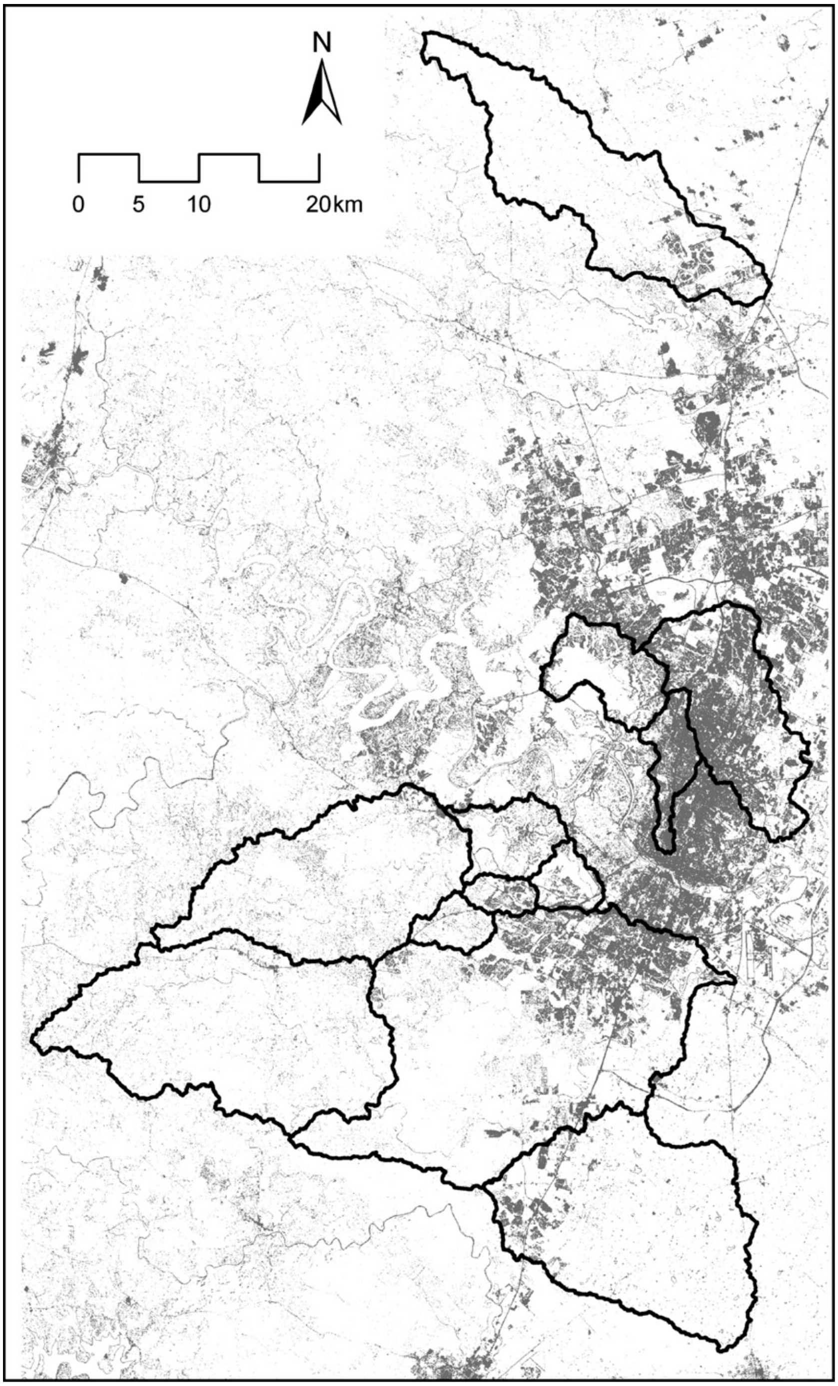

Fig. 3. Impervious surface (in halftone) detected from Landsat TM images (February, 2008) by support vector machine.

Table 3
Characteristics of watersheds for the twelve study sites.

USGS ID	Site name[a]	Ecoregion	Area of watershed (km^2)	% impervious surface
08158600	Walnut Creek	Blackland Prairie	138.4	57.4
08156800	Shoal Creek	Blackland Prairie	33.0	74.1
08159000	OnionDOWN Creek	Blackland Prairie[b]	838.0	13.1
08105100	Berry Creek	Blackland Prairie	213.9	6.3
08172400	Plum Creek	Blackland Prairie	287.0	7.1
08154700	Bull Creek	Hill Country	58.7	27.6
08158920	Williamson Creek	Hill Country	16.3	31.4
08158840	Slaughter Creek	Hill Country	22.7	10.4
08155200	BartonUP Creek	Hill Country	231.8	6.5
08155240	BartonMID Creek	Hill Country	278.0	8.3
08155300	BartonDOWN Creek	Hill Country	301.6	9.6
08158700	OnionUP Creek	Hill Country	320.2	3.9

[a] The suffixes, “UP”, “MID”, and “DOWN” represent the relative locations of USGS gaging stations at Barton Creek and Onion Creek.
[b] Watershed lying on two ecoregions.

Ultimately, we measured 15 environmental variables, including watershed characteristics, hydrological regimes, vegetation community structures, and soil nutrient contents (Table 1). Preliminary analyses showed that there were no significant differences between the first and second transects in RAC, two diversity indices, canopy gap percentage, and soil nutrient contents (bootstrap paired comparison at $\alpha = 0.05$, $n = 12$) (Canty and Ripley, 2009). Therefore, we pooled the data of the first and second transects for the rest of the analyses.

Many environmental variables confound each other, and they need to be decorrelated for unbiased estimation of their relationships to RAC. We extracted the smaller number of dimensions from a 15-dimensional input space using nonmetric multidimensional scaling (NMDS) (Everitt, 2005). The distances between the pairs of twelve sites in the input space were used to construct a dissimilarity matrix. To level out different measurement units for the environmental variables, they were rescaled from 0 to 1 prior to constructing the dissimilarity matrix. The number of dimensions to be extracted was determined based upon stress values – a measure of the overall error not captured by reduced dimensions – calculated by 1-norm Minkowski distance between original and estimated dissimilarities. The varimax method was used to rotate the dimensions (Everitt, 2005). Once NMDS extracted dimensions, we performed a regression analysis to predict RAC using the NMDS dimensions. Because the sample size is small ($n = 12$) and sensitive to the violation of the normality assumption, all statistical tests were based upon a bootstrapping procedure (95% confidence intervals were constructed from 999 resampled datasets). vegan (Oksanen et al., 2009) and boot (Canty and Ripley, 2009) packages in R were used to conducted NMDS and bootstrapping regression analysis, respectively.

3. Results

The vegetation survey of the twelve riparian forests documented thirty-four woody species (Table 2). Dominant native species included Pecan (*Carya illinoinensis*), Sugarberry (*Celtis laevigata*), Netleaf Hackberry (*Celtis reticulata*), Green Ash (*Fraxinus pennsylvanica*), American Elm (*Ulmus americana*), and Cedar Elm (*Ulmus crassifolia*). These species were found at more than half of the study forests. Of the thirty-four species, four were alien species. Those species were Chinaberry (*Melia azedarach*), Glossy Privet (*Ligustrum lucidum*), Chinese Privet (*Ligustrum sinense*), and Chinese Tallow (*Triadica sebifera*). Of the four alien species, *M. azedarach* was the most widespread in this region. Eight of the twelve sites were invaded by *M. azedarach*.

Fig. 3 illustrates the impervious surface map derived from the Landsat TM image by SVM classification. The classification accuracy of the map was 79.3% (46 of 58 validation pixels). Because we selected the equal number of validation pixels for impervious and non-impervious surfaces, classification accuracies were also 79.3% (23 of 29 validation pixels) for both classes. The Shoal Creek watershed located near downtown Austin had the highest impervious surface percentage at 74% (Table 3). The Walnut Creek was the second most urbanized watershed (57%). The watershed with the least impervious surface was the OnionUP Creek (4%), the southwestern-most watershed of the study sites. The Berry Creek watershed (6%) and the BartonUP Creek watershed (7%) also had low impervious surface percentages.

Impervious surface percentage was significantly related to $\hat{\delta}_1$ (Fig. 4a). The regression analysis showed the strong negative effect of impervious surface percentage on $\hat{\delta}_1$ ($R^2 = 0.722$), indicating that as a watershed is increasingly urbanized, streamflow more rapidly

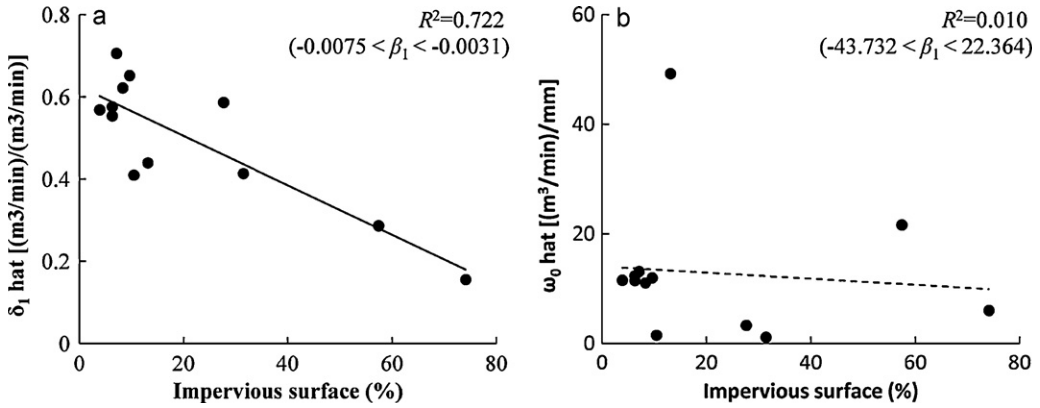

Fig. 4. Bivariate regressions of two parameter estimates of the transfer function (TF) model ($\hat{\delta}_1$ and $\hat{\varpi}_0$) on impervious surface percentage with 95% bootstrapping confidence interval for regression slope (β_1) in parenthesis. A solid regression line indicates a statistically significant relationship, while a dotted line represent an insignificant relationship at $\alpha = 0.05$.

Table 4
Pearson's correlations between four dimensions extracted by nonmetric multidimensional scaling (NMDS) and 15 environmental variables.

Variables	NMDS 1	NMDS2	NMDS3	NMDS4	NMDS5
Area of watershed	−0.429	−0.813*	0.174	0.275	0.038
% impervious surface	0.332	0.049	−0.832*	−0.415	0.095
$\hat{\varpi}_0$	−0.389	−0.857*	−0.152	0.113	−0.127
$\hat{\delta}_1$	−0.245	−0.015	0.925*	0.018	0.088
% canopy gap	0.207	0.219	−0.583*	0.323	0.050
Species richness	0.638*	−0.344	0.516	−0.394	0.089
Shannon diversity Index	0.597*	−0.514	0.338	−0.391	−0.144
pH	0.745*	0.424	0.236	0.384	0.100
NO_3–N	−0.385	−0.677*	−0.180	0.179	0.021
P	−0.678*	−0.088	−0.074	−0.254	0.328
K	−0.850*	0.202	−0.050	−0.231	0.254
Ca	0.829*	−0.456	−0.093	0.101	−0.112
Mg	−0.033	0.249	0.741*	0.042	0.081
S	0.041	0.176	0.214	−0.187	−0.673*
Na	−0.716*	0.242	0.242	−0.425	0.018

* Significant at $\alpha = 0.05$.

recedes after a storm. The relationship between impervious surface percentage and $\hat{\varpi}_0$ was not statistically significant (Fig. 4b).

We extracted five NMDS dimensions from the 15 environmental variables (Table 4). The five-dimensional model was selected based on stress values that began to stabilize from five-dimensional model. The first dimension (NMDS 1) indicates species diversity and soil nutrient levels of a riparian ecosystem: it was significantly related to species richness, Shannon diversity index, soil pH, P, K, Ca, and Na. The second dimension (NMDS 2) generally represents flow magnitude: it was significantly related to the area of watershed, $\hat{\varpi}_0$, and soil N. The third dimension (NMDS 3) can be interpreted as an indicator of hydrological drought. This dimension was significantly related to $\hat{\delta}_1$, impervious surface percentage, and canopy gap percentage ($r = -0.583$). Two other dimensions did not reflect any distinct environmental characteristics.

Of the five NMDS dimensions, only NMDS 3 – the one representing hydrological drought – explained the variation of RAC of the riparian forest. The result of multiple regression analysis indicates that after controlling the other environmental factors, only NMDS 3 was significantly related to RAC (Table 5). Bivariate regressions between RAC and 15 environmental variables further showed that RAC was significantly related to the three variables that were grouped into NMDS 3 ($\hat{\delta}_1$, impervious surface percentage, and canopy gap percentage) (Fig. 5).

4. Discussions

We found that RAC was related to impervious surface percentage, recession parameter $\hat{\delta}_1$, and canopy gap percentage. NMDS showed these three variables covary with each other. Canopy gaps play an important role in biological invasion because otherwise alien species would rarely have a chance to establish in forest floor (Knapp and Canham, 2000). From these results, we can conjecture a causal pathway from watershed urbanization to the invasion of alien species in riparian forests: (1) urbanization alters stream hydrology of a watershed by lengthening dry periods; (2) the altered hydrologic regime disturbs riparian forests and reduces ecosystem resistance; (3) alien species replace a native competitor within a same regenerating cohort; and (4) after several generations, the riparian forests become dominated by the alien invaders.

Table 5
95% bootstrapping confidence intervals (CIs) of regression slopes of relative alien cover (RAC) on five dimensions extracted from nonmetric multidimensional scaling (NMDS) ($R^2 = 0.764$).

NMDS dimensions	Parameter estimates	95% CIs	
		Lower bound (2.5%)	Upper bound (97.5%)
Intercept	0.134	–	–
NMDS1	0.065	−0.383	0.209
NMDS2	0.027	−0.132	0.446
NMDS3[a]	−0.221	−0.735	−0.004
NMDS4	−0.020	−0.197	1.631
NMDS5	0.315	−0.074	0.708

[a] The regression slope statistically significant different from 0 at $\alpha = 0.05$

As expected, hydrologic disturbance appears to be an underlying mechanism of the invasion of riparian forests in the study area, but the inferred mechanism differs from what conventional ecohydrological models assert. According to the conventional models, urbanization facilitates biological invasion in riparian forests by intensifying flood damage. However, in the study sites, RAC was not related to $\hat{\varpi}_0$, but to $\hat{\delta}_1$. No significant relationship between RAC and $\hat{\varpi}_0$ is apparently due to the very large effect of the area of the watershed on $\hat{\varpi}_0$ that overwhelms other effects. For a more accurate analysis of the effect of urbanization on stream hydrology, we investigated hydrological change in the study sites during the last two decades when Austin experienced rapid urbanization (Sung and Li, 2010). The result of the longitudinal comparison suggests that urbanization did not always increase streamflow during storm events. Peakflow increased only in the Blackland Prairie region (on average, $\hat{\varpi}_0$ increased 18%), but decreased in the Hill Country region (on average, $\hat{\varpi}_0$ decreased 23%). Unlike peakflow, baseflow decreased in both regions. These results indicate that urban streams in Austin became drier with fewer floods and longer intermittent periods as a result of watershed urbanization.

Drought in urban riparian forests has been reported in other regions. Groffman et al. (2003) found that urban riparian forests in Baltimore, USA, had higher soil denitrification potential, or drier soil condition, than rural ones. Burton et al. (2009) exhibited that watershed urbanization was related negatively to baseflow in an adjacent stream and positively to flood-intolerant species in riparian forests in Georgia, USA. Pennington et al. (2010) also reported that low-moisture-requiring species were abundant in urban than rural riparian forests in Cincinnati, USA. Many native riparian species adapt to a mesic environment, and are less likely to survive in a drier condition. Therefore, drought shifted riparian vegetation compositions to the ones invaded by drought-tolerant alien species.

The effect of drought can be much more serious in a hot and arid or semi-arid climate region where evapotranspiration quickly depletes moistures in soil. In this region, water is often a limiting factor for the plant growth. Groundwater is the sole source of water for phreatophytic riparian species, and decline in groundwater level makes a riparian forest less habitable for native riparian

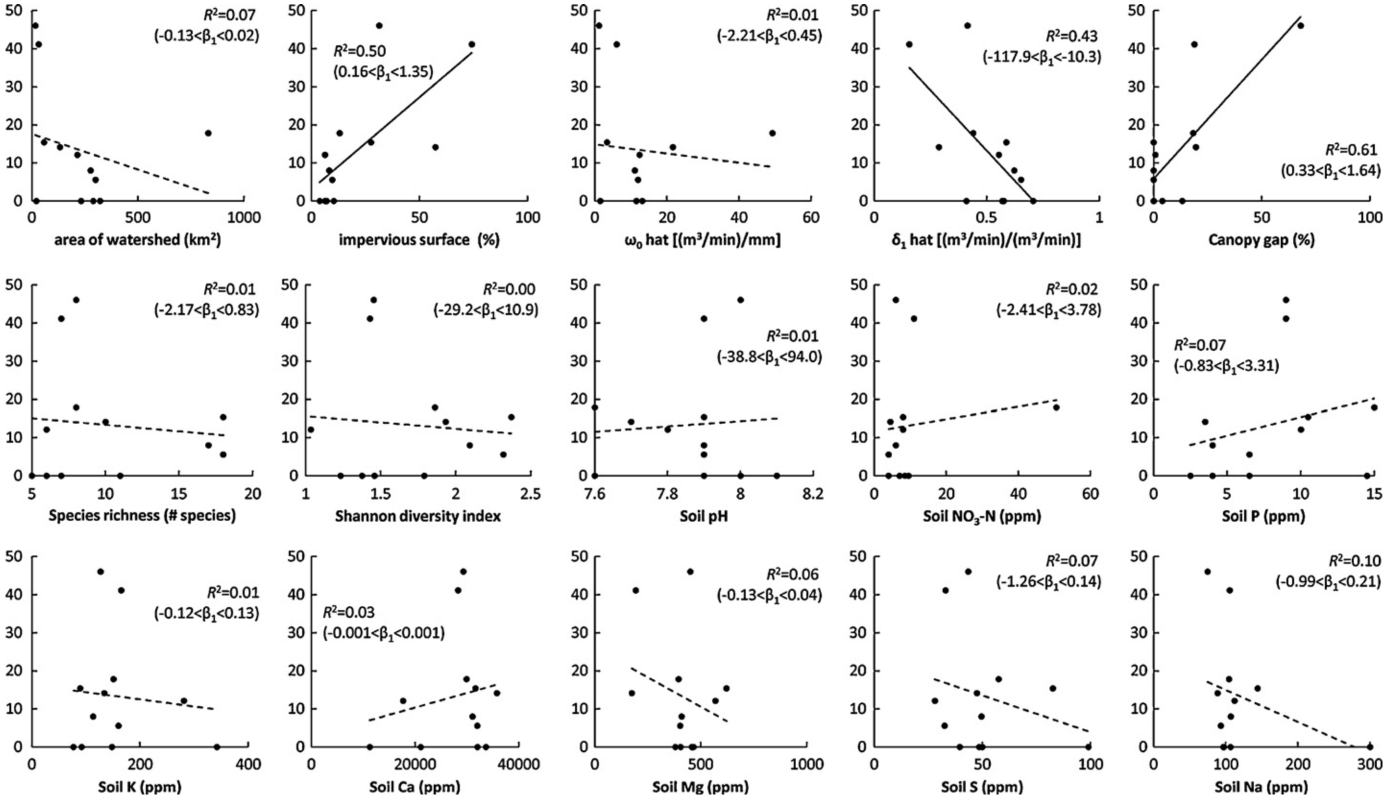

Fig. 5. Bivariate regressions of relative alien cover (RAC) on 15 environmental variables with 95% bootstrapping confidence interval for regression slope (β_1) in parenthesis. *Y*-axis represents RAC in percentage. Solid regression lines indicate statistically significant relationships, while dotted lines represent insignificant relationships at $\alpha = 0.05$.

species (Snyder and Williams, 2000; Horton and Clark, 2001). The adverse effect of drought is exemplified by the invasion of *Tamarix* spp. that rapidly spreads along riparian forests in US southwest. Previous studies revealed that the altered hydrologic regime, particularly by dams that regulate flood in riparian forests, created a drier environment that were favorable to *Tamarix* spp. over native riparian species (Glenn and Nagler, 2005; Stromberg et al., 2007).

The lack of relationship between RAC and the two species diversity indices suggests that Elton's theorem – a theorem stating that an ecosystem with higher species diversity is more resistant to biological invasion (Elton, 1958) – cannot be applied to urban riparian woody vegetation community (excluding vegetation shorter than 0.6 m). Under severe disturbance regimes, the effect of disturbance outweighs the effect of diversity, and allogenic process governs ecological succession (Planty-Tabacchi et al., 1995; Stohlgren et al., 1999; von Holle, 2005; Maskell et al., 2006; but see Levine, 2000). In other words, biological invasion was primarily determined by the disturbance regime, but not by the interspecific competition between native and alien species in riparian forests.

Finally, it is worth noting potential limitations of this study. First, we only surveyed woody vegetation taller than 0.6 m. An understory community responds more strongly to urbanization than canopy layer and so may be an early indicator of biological invasion in an urban riparian forest (Pennington et al., 2010). Although our survey scheme allowed us to exclude saplings whose establishment successes were not yet determined, species composition between saplings would have provided a more comprehensive understanding of biological invasion of urban riparian forest. Another limitation that we did not take into account is a source of alien propagules. All of the alien species found in the study sites were ornamental trees that were widely planted in surrounding urban areas. Hydrochorous dispersal is the major dispersal route in riparian forests, but alien species can also be dispersed anemochorously or vegetatively (Moggridge et al., 2009). More studies are needed to separate the effects of urbanization on the invasion of riparian forests via different dispersal paths.

5. Conclusion

In this study, we investigated twelve riparian forests in Austin, TX, and found that watershed urbanization increased the invasion of alien species in riparian forests. We infer that the change in hydrologic regime with an extended intermittent period made the riparian forests less suitable for native plants that adapt to a mesic environment and more favorable to disturbance-tolerant alien species.

Our results provide several important implications for urban planners and ecosystem managers. First, mechanical removal will not eliminate alien species from urban riparian forests. An empty space created by the removal would be quickly refilled by fast growing and disturbance-tolerant alien invaders as long as severe disturbances are continued. This leads to a corollary of the first implication: the remedy should come from a watershed-wide managerial scheme. Many municipalities have adopted stormwater management schemes, such as low impact development, to mitigate hydrologic alteration by development. However, the current management schemes focus mainly on flood control but often overlook hydrologic change in an urban watershed. For instance, a detention/retention basin, the most commonly used stormwater best management practice designed to reduce peak discharge by retaining water during storm events, does not necessarily maintain groundwater level unless they are carefully placed by considering local infiltration capacity. Many of these basins are even lined with impermeable materials for easy maintenance. The retained stormwater will quickly evaporate instead of infiltrate, especially in hot and semi-arid regions like Austin and aggravate drought through a consequent decrease in the baseflow. We recommend that urban planners and ecosystem managers take a watershed scale approach that manages local water cycle to protect riparian forests from the invasion of alien species.

References

Aguiar, F.C., Ferreira, M.T., 2005. Human-disturbed landscapes: effects on composition and integrity of riparian woody vegetation in the Tagus River basin, Portugal. Environ. Conserv. 32, 30–41.

Box, G.E.P., Jenkins, G.M., 1976. Time Series Analysis: Forecasting and Control. Holden-Day, Oakland, CA.

Burges, C.J.C., 1998. A tutorial on support vector machines for pattern recognition. Data Min. Knowl. Discov. 2, 121–167.

Burns, D., Vitvar, T., McDonnell, J., Hassett, J., Duncan, J., Kendall, C., 2005. Effects of suburban development on runoff generation in the Croton River basin, New York, USA. J. Hydrol. 311, 266–281.

Burton, M.L., Samuelson, L.J., 2008. Influence of urbanization on riparian forest diversity and structure in the Georgia Piedmont, US. Plant Ecol. 195, 99–115.

Burton, M.L., Samuelson, L.J., Mackenzie, M.D., 2009. Riparian woody plant traits across an urban–rural land use gradient and implications for watershed function with urbanization. Landscape Urban Plan. 90, 42–55.

Canty, A., Ripley, B., 2009. boot (Version 1.2-41) [Computer Software]. Available from: http://cran.r-project.org/web/packages/boot/boot.pdf.

Caratti, J.F., 2006. Line Intercept (LI). In: Lutes, D.C. (Ed.), FIREMON: Fire Effects Monitoring and Inventory System. Rocky Mountain Research Station, Natural Resources Research Center, Fort Collins, CO.

Elton, C.S., 1958. The Ecology of Invasions by Animals and Plants. Methuen & Company, London.

Everitt, B., 2005. An R and S-Plus Companion to Multivariate Analysis. Springer, London.

Foody, G.M., Mathur, A., 2006. The use of small training sets containing mixed pixels for accurate hard image classification: training on mixed spectral responses for classification by a SVM. Remote Sens. Environ. 103, 179–189.

Glenn, E.P., Nagler, P.L., 2005. Comparative ecophysiology of *Tamarix ramosissima* and native trees in western U.S. riparian zones. J. Arid Environ. 61, 419–446.

Greer, K., Stow, D., 2003. Vegetation type conversion in Los Peñasquitos Lagoon, California: an examination of the role of watershed urbanization. Environ. Manage. 31, 489–503.

Groffman, P.M., Bain, D.J., Band, L.E., Belt, K.T., Brush, G.S., Grove, J.M., Pouyat, R.V., Yesilonis, I.C., Zipperer, W.C., 2003. Down by the riverside: urban riparian ecology. Front. Ecol. Environ. 1, 315–321.

Groffman, P.M., Boulware, N.J., Zipperer, W.C., Pouyat, R.V., Band, L.E., Colosimo, M.F., 2002. Soil nitrogen cycle processes in urban riparian zones. Environ. Sci. Technol. 36, 4547–4552.

Hollis, G.E., 1975. The effect of urbanization on floods of different recurrence interval. Water Resour. Res. 11, 431–435.

Hood, W.G., Naiman, R.J., 2000. Vulnerability of riparian zones to invasion by exotic vascular plants. Plant Ecol. 148, 105–114.

Horton, J.L., Clark, J.L., 2001. Water table decline alters growth and survival of *Salix gooddingii* and *Tamarix chinensis* seedlings. Forest Ecol. Manage. 140, 239–247.

ITT VIS, 2008. ENVI (Version 4.5) [Computer Software]. Boulder, CO.

Jakeman, A.J., Littlewood, I.G., Whitehead, P.G., 1990. Computation of the instantaneous unit hydrograph and identifiable component flows with application to two small upland catchments. J. Hydrol. 117, 275–300.

Knapp, L.B., Canham, C.D., 2000. Invasion of an old-growth forest in New York by *Ailanthus altissima*: sapling growth and recruitment in canopy gaps. J. Torrey Bot. Soc. 127, 307–315.

King, S.A., Buckney, R.T., 2000. Urbanization and exotic plants in northern Sydney streams. Austral Ecol. 25, 455–461.

Leopold, L.B., 1968. Hydrology for Urban Land Planning-A Guidebook on the Hydrologic Effects of Urban Land Use. U.S. Department of Interior, Washington, DC.

Levine, J.M., 2000. Species diversity and biological invasions: relating local process to community pattern. Science 288, 852–854.

Magee, T.K., Ringold, P.L., Bollman, M.A., 2008. Alien species importance in native vegetation along wadeable streams, John Day River basin, Oregon, USA. Plant Ecol. 195, 287–307.

Magurran, A.E., 1988. Ecological Diversity and Its Measurement. Princeton University Press, Princeton, NJ.

Marsh, W.M., Marsh, N.L., 1995. Hydrogeomorphic considerations in development planning and stormwater management, Central Texas Hill Country, USA. Environ. Manage. 19, 693–702.

Maskell, L.C., Bullock, J.M., Smart, S.M., Thompson, K., Hulme, P.E., 2006. The distribution and habitat associations of non-native plant species in urban riparian habitats. J. Veg. Sci. 17, 499–508.

McMahan, C.A., Frye, R.G., Brown, K.L., 1984. The Vegetation Types of Texas Including Cropland, Wildlife Division. Texas Parks and Wildlife Department, Austin, TX.

Medina, A.L., 1990. Possible effects of residential development on streamflow, riparian plant communities, and fisheries on small mountain streams in central Arizona. Forest Ecol. Manage. 33/34, 351–361.

Moffatt, S.F., McLachlan, S.M., Kenkel, N.C., 2004. Impacts of land use on riparian forest along an urban–rural gradient in southern Manitoba. Plant Ecol. 174, 119–135.

Moggridge, H.L., Gurnell, A.M., Mountford, J.O., 2009. Propagule input, transport and deposition in riparian environments: the importance of connectivity for diversity. J. Veg. Sci. 20, 465–474.

Morris, L.L., Walck, J.L., Hidayati, S.N., 2002. Growth and reproduction of the invasive *Ligustrum sinense* and native *Forestiera ligustrina* (Oleaceae): implications for the invasion and persistence of a nonnative shrub. Int. J. Plant Sci. 163, 1001–1010.

Mwakalila, S., Campling, P., Feyen, J., Wyseure, G., Beven, K., 2001. Application of a data-based mechanistic modelling (DBM) approach for predicting runoff generation in semi-arid regions. Hydrol. Process. 15, 2281–2295.

Nilsson, C., Jansson, R., Zinko, U., 1997. Long-term responses of river-margin vegetation to water-level regulation. Science 276, 798–800.

Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Wagner, H., 2009. vegan (Version 1.17-2) [Computer Software]. Available from: http://cran.r-project.org/web/packages/vegan/vegan.pdf.

Pennington, D.N., Hansel, J.R., Gorchov, D.L., 2010. Urbanization and riparian forest woody communities: diversity, composition, and structure within a metropolitan landscape. Biol. Conserv. 143, 182–194.

Pettit, N.E., Froend, R.H., 2001. Variability in flood disturbance and the impact on riparian tree recruitment in two contrasting river systems. Wetl. Ecol. Manage. 9, 13–25.

Planty-Tabacchi, A.-M., Tabacchi, E., Naiman, R.J., Deferrari, C., Décamps, H., 1995. Invasibility of species-rich communities in riparian zones. Conserv. Biol. 10, 598–607.

Rose, S., Peters, N.E., 2001. Effects of urbanization on streamflow in the Atlanta area (Georgia, USA): a comparative hydrological approach. Hydrol. Process. 15, 1441–1457.

SAS Institute Inc., 2004. SAS (Version 9.1) [Computer Software]. Cary, NC.

Schnitzler, A., Hale, B.W., Alsum, E.M., 2007. Examining native and exotic species diversity in European riparian forests. Biol. Conserv. 138, 146–156.

Snyder, K.A., Williams, D.G., 2000. Water sources used by riparian trees varies among stream types on the San Pedro River, Arizona. Agric. Forest Meteorol. 105, 227–240.

Soil Conservation Service, 1974. Soil Survey of Travis County, Texas. U.S. Department of Agriculture, Washington, DC.

Stohlgren, T.J., Binkley, D., Chong, G.W., Kalkhan, M.A., Schell, L.D., Bull, K.A., Otsuki, Y., Newman, G., Bashkin, M., Son, Y., 1999. Exotic plant species invade hot spots of native plant diversity. Ecol. Monogr. 69, 25–46.

Stohlgren, T.J., Bull, K.A., Otsuki, Y., Villa, C.A., Lee, M., 1998. Riparian zones as havens for exotic plant species in the central grasslands. Plant Ecol. 138, 113–125.

Stromberg, J.C., Beauchamp, V.B., Dixon, M.D., Lite, S.J., Paradzick, C., 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid south-western United States. Freshwater Biol. 52, 651–679.

Sung, C.Y., Li, M.-H., 2010. The effect of urbanization on stream hydrology in hillslope watersheds in central Texas. Hydrol. Process. 24, 3706–3717.

Tickner, D.P., Angold, P.G., Gurnell, A.M., Mountford, J.O., 2001. Riparian plant invasions: hydrogeomorphological control and ecological impacts. Prog. Phys. Geog. 25, 22–52.

US Census Bureau, 2009. Population Finder: Austin City, Texas. http://factfinder.census.gov (retrieved 15.07.09).

US Environmental Protection Agency, 2006. Better Assessment Science Integrating point & Non-point Sources (BASINS) (Version 3.1) [Computer Software]. Washington, DC.

US National Climatic Data Center, 2010. Normal Monthly Precipitation. http://lwf.ncdc.noaa.gov/oa/climate/online/ccd/nrmlprcp.html (Retrieved 23.11.2010).

von Holle, B., 2005. Biotic resistance to invader establishment of a southern Appalachian plant community is determined by environmental conditions. J. Ecol. 93, 16–26.

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., Morgan II, R.P., 2005. The urban stream syndrome: current knowledge and the search for a cure. J. N. Am. Benthol. Soc. 24, 706–723.

Woodruff Jr., C.M., Wilding, L.P., 2008. Bedrock, soils, and hillslope hydrology in the Central Texas Hill Country, USA: implications on environmental management in a carbonate-rock terrain. Environ. Geol. 55, 605–618.

Wu, C., Yuan, F., 2007. Seasonal sensitivity analysis of impervious surface estimation with satellite imagery. Photogramm. Eng. Remote Sens. 73, 1393–1401.

Young, P., 2003. Top-down and data-based mechanistic modelling of rainfall-flow dynamics at the catchment scale. Hydrol. Process. 17, 2195–2217.

Young, P.C., Beven, K.J., 1994. Data-based mechanistic modelling and the rainfall-flow non-linearity. Environmetrics 5, 335–363.

Zedler, J.B., Kercher, S., 2004. Causes and consequences of invasive plants in wetlands: opportunities, opportunists, and outcomes. CRC Crit. Rev. Plant Sci. 23, 431–452.