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Arthropod abundance and diversity in street trees of south Texas, USA

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ABSTRACT

In urban areas, street trees provide a variety of ecological services, including biodiversity conservation. In this study we examined arthropod diversity on native and non-native street trees sampled during the fall of 2010 and spring of 2011 in McAllen, Texas, one of the most rapidly growing urban areas in the country. Eighty-eight street trees were sampled by removing arthropods from the lower canopy foliage using a hand held vacuum. Arthropods were collected into nylon bags, identified to order, and counted by morphospecies. Overall, street trees supported a significant and diverse population of arthropods: a total of 1,971 arthropods were collected, from which 12 different orders and 102 different morphospecies were identified. We found arthropod abundance was higher on street trees native to the Lower Rio Grande Valley compared to non-native trees, especially for beetles, wasps, bees, ants, and spiders. This difference was particularly striking in spring when trees were flushed with new growth. The significant deficiency of arthropods on non-native trees is indicative of their relatively low value for maintaining entomological fauna. Local land managers who aim to include biodiversity conservation in their efforts thus should enhance the urban forest through the conservation of existing native remnant trees and promoting the use of native tree species in landscaping.

Additional Index Words: Lower Rio Grande Valley, McAllen, urban forest, biodiversity, ecological services, native species

Urbanization, or the rapid proliferation of built environments to match a growing population, is often associated with the loss or disruption of natural ecosystems (Brown & Freitas, 2002; McKinney, 2002; Santos et al., 2009; Walker et al., 2009). Few areas in the United States have experienced a more precipitous population growth than the Lower Rio Grande Valley (LRGV), a burgeoning area along the US-Mexico border in south Texas (Huang et al., 2011). For example, Hidalgo County-- the largest of the four counties (Cameron, Hidalgo, Starr, and Willacy) that comprise the LRGV--grew by 48.5% between 1990 and 2000, and is recognized as one of the fastest growing areas in the nation (DISC, 2002). With this trend of precipitous growth, cities in the LRGV are experiencing the most rapid urbanization in the country, reflected by dramatically changing land cover and land use patterns (Huang et al., 2011). Only a small fraction of natural vegetation remains in the LRGV (Jahrsdoerfer & Leslie, 1988), and thus urbanization undoubtedly will continue to have a tremendous impact on native biodiversity and ecosystems (Paull et al., 2003).

In heavily urbanized environments, trees of the urban forest are important habitat corridors for local fauna (Pirnat, 2000; Rudd et al., 2002; Alvey, 2006). For example, Fernández-Juricic (2000) found in Madrid, Spain, that tree-lined streets play an important role in providing habitat connectivity for birds. In Sao Paolo, Brazil, remnant urban forests have become vital in maintaining diverse populations of insects, especially butterflies (Brown & Freitas, 2002). Arthropods are commonly used as an indicator of the health of ecological food webs in managed forest systems (Langor & Spence, 2006; Maleque et al., 2006).

In this study, we surveyed street trees in the city of McAllen, Texas, the most rapidly growing city in the LRGV, in an effort to better understand the potential of street trees--both native and non-native--to support significant populations of arthropods in the quickly urbanizing area. We conducted timed surveys using a leaf vacuum to document the abundance of arthropods found on 88 street trees. Furthermore, we compared arthropod assemblages found on urban trees native to the LRGV to that collected from non-native trees to reveal patterns and processes that may be important to consider in the maintenance and extension of the urban forest in the LRGV.

MATERIALS AND METHODS

This survey was conducted along public walkways that parallel Bicentennial and 2nd streets, two of the main North-South thoroughfares in McAllen which were sparsely populated with native and non-native trees in 2006 or 2007. The street trees were planted by city employees, spaced two to ten meters apart surrounded by lawn. These areas are maintained with regular mowing and irrigation with limited or no understory plants, and never treated with insectide (M. Kroeze, personal communication). Trees along these streets were selected non-randomly with the main criterion that (1) the tree had a full canopy, and (2) the lower part of the canopy was no higher than 2.5 meters above ground level, so that the leaves could be reached with a leaf vacuum. Nylon stockings were fitted between the joints of the plastic tube of a leaf vacuum (Ryobi™ RV09053; Cambridge, Ontario, Canada) to collect arthropods as they were aspirated by the device. For each tree sampled, the lower foliage of each tree was vacuumed for one minute using a slow, left to

Table 1. Sampled subset of common street trees in McAllen, TX

Scientific Name	Family	Common Name	Status	# trees sampled (Fall, Spring)
Acacia minuata	Fabaceae	huisache	Native	6 (2,4)
Callistemon viminalis	Mytaceae	bottle brush	Non-native	3 (0,3)
Casurina equistifolia	Casurinaceae	casuarina	Non-native	4 (0,4)
Celtis laevigata	Ulmaceae	sugar hackberry	Native	5 (1,4)
Chilopsis linearis	Bignoniaceae	desert willow	Non-native	1 (1,0)
Cordia boissieri	Boranginaceae	Mexican olive	Native	6 (2,4)
Diospyros texana	Ebenaceae	Texas persimmon	Native	5 (3,2)
Ehretia anacua	Boranginaceae	anacua	Native	4 (2,2)
Ficus benjamina	Moraceae	ficus	Non-native	1 (1,0)
Koelreuteria paniculata	Sapindaceae	golden raintree	Non-native	3 (3,0)
Lagerstroemia indica	Lythraceae	crape myrtle	Non-native	10 (5,5)
Magnolia grandiflora	Magnoliaceae	magnolia	Non-native	2 (0,2)
Parkinsonia aculeata	Fabaceae	retama	Native	4 (2,2)
Phoenix dactylifera	Arecaceae	date palm	Non-native	4 (0,4)
Prosopis glanduosa	Fabaceae	mesquite	Native	6 (2,4)
Quercus macrocarpa	Fagaceae	bur oak	Non-native	3 (2,1)
Quercus virginiana	Fagaceae	live oak	Native	2 (1,1)
Sabal mexicana	Arecaceae	sabal palm	Native	2 (2,0)
Salix nigra	Salicaceae	black willow	Native	6 (3,3)
Sophora secundiflora	Fabaceae	mountain laurel	Native	5 (1,4)
Syagrus romanzoffiana	Arecaceae	queen palm	Non-native	1 (0,1)
Ulmus crassifolia	Ulmaceae	cedar elm	Native	2 (1,1)
Vitex agnus-cactus	Lamiacea	vitex	Non-native	1 (1,0)
Washingtonia robusta	Arecaceae	Washingtonia palm	Non-native	2 (0,2)

right sweeping motion. Arthropods collected at each sampling event were chilled to immobilize them so that they could be easily identified and counted in indoor settings. Relatively few juveniles were collected (<2% of all total arthropods collected), and thus were omitted from the total counts. Each collection was sorted separately first by order using the entomological expertise of J. A. G. and secondly by referencing keys and descriptions provided by Borror and DeLong (1964) . Within each order, morphospecies were determined based on phenotypical differences, such as size, color, and general appearance. Each apparent morphospecies was numbered and then crossed-referenced among collections to insure that morphospecies were not double-counted. Vouchers for each morphospecies are stored at the USDA-ARS Subtropical Agriculture Research Center (Weslaco, TX).

Numbers of spiders were counted but were not further identified, and they were entered as a single entry in terms of species abundance. To account for intra-annual differences, one collection was made in the fall (November 2010) and in the spring (March 2011). Originally, the same trees were meant to be sampled across seasons, but the LRGV experienced several below freezing days in both December 2010 and February 2011, and thus many trees surveyed in the fall had severe dieback when revisited in the spring. In light of this, we treated each sampling event independently, thus we have a total of 88 sampled trees as part of this study (53 native/35 non-native trees). Depiction of species accumulation curves (Fig.1) confirm that sampling effort was sufficient in each case.

To estimate the potential of urban trees to support significant populations of arthropods, we use both Shannon diversity indices (H') and arthropod richness (number of unique morphospecies) as basic proxies for the more complex concept of ecological diversity (Magurran, 2004). To test the hypotheses that entomological diversity is highest among native trees, we used two way analyses of variance to compare average morphospecies richness and average H' values. Data were analyzed using SYSTAT 13 where tree status (Native and Non-native) and season (Fall and Spring) were considered fixed factors (Table 1). Tree status (native or non-native) was based on the classification proposed by Everitt et al. (2002). Where necessary, data were statistically transformed to meet requirements of normality and homoscedasticity, and considerations were made for unbalanced design (Weerahandi, 1995). Holm-Sidak pairwise tests were conducted where there were significant interactions between fixed factors. Within-season comparisons of average number of arthropods collected by order were made using separate independent t-tests (Fig. 2).

RESULTS

The species accumulation curves depicted in Figure 1 demonstrate that timed-survey using a leaf vacuum is an adequate technique for capturing arthropods in street trees. Cumulatively, few new species were recorded after sampling 30 trees. This is particularly true when sampling non-native trees, where few new species were captured after sampling 15 trees.

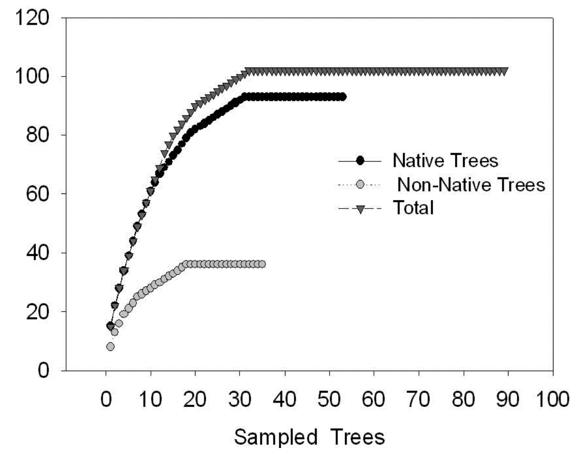

Fig. 1. Species accumulation curves for arthropods collected from street trees (McAllen, TX) using one-minute timed leaf-vacuum surveys.

A total of 1,971 arthropods were collected and identified to order, from which 102 different morphospecies were identified (Table 2). Coleoptera, Hymenoptera, and Hemiptera (27, 26, and 22 morphospecies) represented almost 75% of the total morphospecies identified. Overall arthropod biodiversity across all urban trees was relatively high (H'Total =2.74). The insect orders of Coleptera, Hemiptera, Diptera, and Lepidoptera had the highest measures of species diversity and were well-represented in our survey (Table 2). In many cases, such as with Mantodea, Neuroptera, and Orthoptera, we only captured one individual. Conversely, in the case of aggregating organisms such as thrips (Thysanoptera) and mites, we trapped several individuals at once. Although we did not further differentiate spiders (all included as a single morphospecies, Table 2), we collected an average of 1.31 spiders per sampling event.

Ninety-two percent of the different species collected (94 morphospecies) where collected from native trees, whereas only 36 of the different morphospecies collected (35%) where found on non-native trees. Although disproportionately sampled (53 and 35 native and non-native trees respectively) species accumula-

Table 2. Total number of species and individuals collected from street trees (n=88), including Shannon diversity indices calculated by order.

Order	*Species Abundance*	*Total individuals collected*	*Ave. indiv/ sample (n=88)*	*H'*
Coleoptera	27	207	2.35	2.53
Hemiptera	22	169	1.92	2.18
Diptera	9	122	1.39	1.56
Lepidoptera	10	70	0.80	1.46
Hymenoptera	26	439	4.99	0.79
Mantodea	1	1	0.01	---
Neuroptera	1	1	0.01	---
Orthoptera	1	1	0.01	---
Thysanoptera	1	434	4.93	---
Trichoptera	1	1	0.01	---
Trombidiformes (Mites)	2	411	4.67	0.02
Aranea (Spiders)	1*	115	1.31	---
TOTALS	**102**	**1971**	**22.40**	**2.74**

tion curves as depicted in Fig 1 suggest that native trees harbor greater arthropod species diversity than non-native trees. Results from the two-way analysis of variance of both species abundance (log-transformed) and diversity (H') showed a significant interaction between a tree's status and the season in which it was sampled (respectively $F_{1,85}=6.22$, $P=0.012$, and $F_{1,85}=4.618$, $P=0.035$). Results from pairwise multiple comparisons (Table 3) suggest that both species abundance and diversity is highest on native trees in the spring, when trees are often flushed with new leaves. Each fixed factor in the log-transformed species abundance model and the species diversity model was also significant: overall a greater average abundance of arthropods was found per tree sampling in the spring ($F_{1,87}=30.75$, $P<0.001$), and across seasons, native trees were found to have a greater abundance of arthropods than non-native trees (an average of 6.1 morphospecies sampled in native trees versus an average of 2.9 unique morphospecies in non-native trees, $F_{1,87}=9.75$, $P<0.001$). In the ANOVA model of Shannon diversity indices (H'), season was not found to be significant, although there was a significant difference in diversity measures between native (1.23 ± 0.07 SE) and non-native trees (0.87 ± 0.09) ($F_{1,87}=8.71$, $P=0.004$). Differences in average species abundance were further analyzed using separate two-sample t-tests. Across seasons, we found a higher abundance of certain orders of arthropods in native street trees than in non-native trees (Fig 2). For example, in the fall, we found significantly higher populations of Hymenoptera ($t=11.764$, $df=33$, $P = <0.001$) and Coleoptera ($t=2.132$, $df=33$, $P = 0.041$) on native trees than non-native trees. In the spring sampling, Hymenoptera ($t= 2.86$, $df=51$, $p=0.006$), Thysanoptera ($t=3.42$, $df=51$, $p=0.001$), and Diptera ($t=2.69$, $df=51$, $p=0.010$) were particularly more abundant on native trees. The spider abundance was greater on native trees in both fall ($t = 2.246$, $df=33$, $P = 0.032$) and spring ($t = 2.125$, $df=51$, $P = 0.038$).

Table 3. Results from pairwise comparisons of the average values of the number of unique morphospecies and overall species diversity (H') in native and non-native street trees in McAllen, TX USA. Trees were sampled in the fall (November 2010) and spring (March 2011) using a vacuum sampler. Different letters indicate significant differences in average values (p<0.05).

	Morphospecies abundance	Shannon Diversity index (H')
NATIVE		
Fall	4.36 ± 0.54 a	0.96 ± 0.11 A
Spring	7.84 ± 0.46 b	1.47± 0.10 B
NON-NATIVE		
Fall	2.92 ± 0.71 a	0.90 ± 0.16 A
Spring	2.91± 0.55 a	0.85 ± 0.12 A

DISCUSSION

Biodiversity is frequently the key element used to inform or prioritize conservation actions, which are

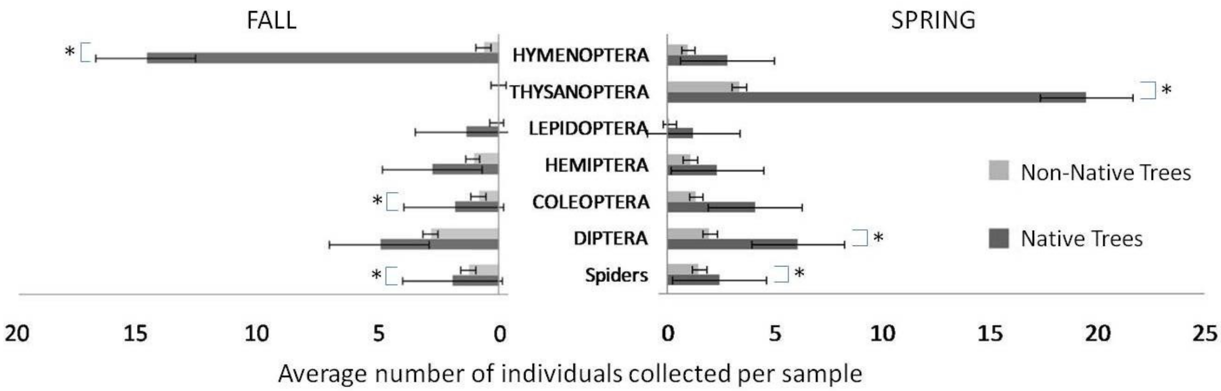

Fig. 2. Comparisons of average number of most abundant arthropods on native and non-native trees, by order. Asterisk denotes significant difference ($p<0.05$)

typically centered on the preservation of large intact natural habitats (Myers et al., 2000). However, in urban areas such as the LRGV where few undisturbed intact areas remain, individual street trees are paramount for sustaining biodiversity (McKinney, 2002; Alvey, 2006). Arthropods can be a good indicator of biodiversity in urban ecosystems (Langor & Spence, 2006). Since arthropods have diverse behavior and life histories, more consideration should be given to consistent year-round sampling using a diversified sampling regime (i.e. pitfall traps, malaise traps, etc.) to perhaps reveal other patterns of arthropod diversity. For example, although use of vacuum traps is an excellent technique for collecting arthropods, it is a poor method for collecting and estimating population of caterpillars, an important food source for breeding birds (Burghart et al, 2009). Still, as this research demonstrates, seasonal, timed sampling using a leaf vacuum can be an adequate way of getting conservative estimates of arthropod diversity and abundance.

Biodiversity conservation in these burgeoning urban areas is particularly pertinent to the LRGV as it maintains a robust ecotourism industry centered on the observation of avifauna and entomofauna, especially butterflies (Mathis & Matisoff, 2004). Our finding that street trees maintain a rich diversity of insect herbivores and arthropod predators confirms the ecological importance of street trees, especially in urban areas where other conservation areas or green spaces are absent. As this work demonstrates, urban trees can harbor significant populations of insects, which in turn serve as a critical source of protein to terrestrial and insectivorous birds (Burghardt et al., 2009). In addition, as urban trees are watered and maintained, they continue to provide seeds, fruits, and nectar to plant-feeding birds, which is especially important in times of drought.

We found that native tree species harbor a disproportionate abundance of arthropods, adding to the growing evidence that urban areas that maintain native vegetation can preserve more biodiversity (Chace & Walsh, 2006; Tallamy & Shropshire, 2009 ; Burghardt et al., 2010; Perre et al., 2011). Native trees planted as ornamentals confer not only key ecological services through the maintenance of biodiversity but prove to be more resilient as landscaping plants as they are more adapted to the intra- and inter-annual variations in climate that are common to the LRGV. Many of the non-native tropical tree species used as ornamentals in McAllen, some of which were included in the fall sampling of this survey, perished in the extended sub-freezing temperatures that occurred in the LRGV in December 2010 and Februrary 2011 (M. Kroeze, pers communication).

Thus, in this context urban area decision makers such as city planners and home owners, can readily incorporate ecological considerations along with other socio-economic implications of street trees, such as energy conservation through shade, homeowner satisfaction, stormwater management, and carbon sequestration . Often, these ecological and socio-economic implications can overlap. For example, other studies of McAllen's urban forest have found that percent forest cover in neighborhoods is significantly correlated with average home value (Racelis and Kroeze, unpublished data). Land managers in rapidly urbanizing areas should consider planting more trees, particularly trees native to the region, if biodiversity conservation of arthropods is to be integrated into local development plans.

In McAllen, more than half (55%) of the urban trees are considered native to the LRGV, many of which were there before the neighboring construction (Kroeze & Racelis, 2010). However, as urbanization and development expand in the area, many of these ecologically important remnant trees are being cut down, and if replaced are usually replaced by non-native ornamentals. As such, urban tree conservation and tree species selection within urban areas can have considerable effects on biodiversity, especially in the LRGV. As more research emerges on how these ecological benefits can be translated economically (Costanza et al., 1997; Bolunda & Hunhammar, 1999; Pickett et al., 2008; Kroeze & Racelis, 2010), urban area managers can more readily recognize the considerable conservation value of street trees and incorporate these considerations in future development plans.

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