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Long-term impacts of Argentine ant invasion of urban parks in Hiroshima, Japan

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Abstract

Ant communities are well suited for monitoring changes in ecosystems. Although numerous studies have examined the responses of ant communities to environmental disturbance, relatively few long-term studies on ant communities have been undertaken in urban environments. We examined species richness in nine urban parks in Hiroshima, Japan, and compared the survey results with data collected at the same sites by using the same methods in 1999. In both surveys, total of 25 species was recorded: 23 species in 1999 and 20 species in 2012. Non-metric multidimensional scaling analysis revealed that the ant communities consisted of two distinct groups, which could in turn be characterized by three patterns of ant community changes in between the two groups. The first of these community change patterns was characterized by a shift within group 1, but the number of species remained constant (approx. 10 species). The second pattern was characterized by a shift within group 2, but the number of species remained low (approx. 4 species). The third pattern was characterized by a shift from group 1 to group 2 as the abundance of *Linepithema humile* (Mayr) increased over time. Unlike the first and second patterns, the number of ant species in communities of the third type decreased significantly. These findings suggest that *L. humile* has a marked effect on the species diversity of indigenous ant communities in urban environments.

Key words: ant community, *Linepithema humile*, long-term, urban park

INTRODUCTION

Ant communities are well suited for monitoring ecosystems because the distribution patterns of individual species are relatively stable (Alonso and Agosti 2000). In addition, the stability, sedentary nature, and sensitivity to environmental perturbation also make ant communities suitable for assessing the general state of an environment (Alonso and Agosti 2000, Yamaguchi 2005, Ribas et al. 2012). Consequently, numerous studies have examined ant responses to a variety of environmental disturbances (Human and Gordon 1996, Thompson and Mclachlan 2007, Menke et al. 2011).

Most of the studies conducted to date on ants in urban environments have concentrated on the negative effect of the urbanization on ant biodiversity (Holway and Suarez 2006, Sanford et al. 2009) or the relationship between invasive and native species (Miyake et al. 2002, Touyama et al. 2003, Sunamura et al. 2007). However, given the difficulties associated with conducting long-term observations in urban environments, the amount of ecological data collected in urban environments is typically limited (Buczkowski and Richmond 2012). Consequently, most of the studies on ants in urban environments have been

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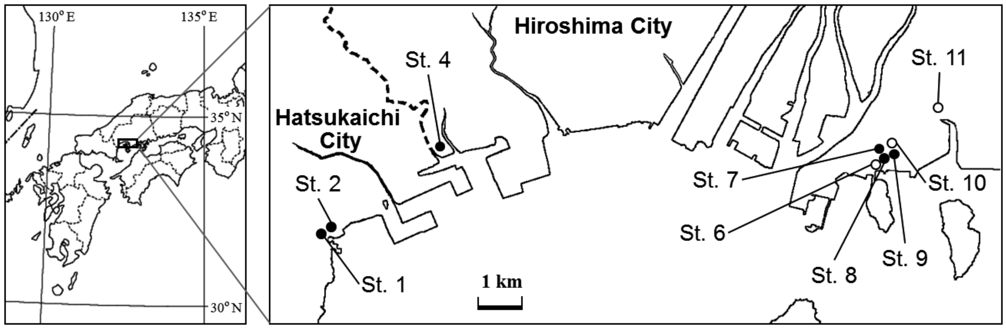

Fig. 1. Location of the nine urban parks in Hiroshima and Hatsukaichi surveyed in this study. Black dots (●) denote the sites have been invaded by *L. humile* and white dots (○) denote the sites that have not been uninfested. St. 1, Ougien Dai-ichi Kouen; St. 2, Ougien Dai-ni Kouen; St. 4, Minori Kouen; St. 6, Ujina Kaigan Dai-ichi Kouen; St. 7, Ujina Dai-ni Kouen; St. 8, Ujina Dai-san Kouen; St. 9, Ujina Dai-yon Kouen; St. 10, Ujina Higashi Dai-ni Kouen; St. 11, Tan-na Nishi Kouen.

performed at a single point in time along a gradient of urbanization (e.g., urban vs. rural areas, urban periphery vs. inner city, and urban green spaces vs. residential areas), or surveys have been conducted in several sites that have similar environmental condition (Yamaguchi 2004, 2005, Stringer et al. 2009). While these approaches have allowed us to simplify the complicated effects of urbanization (Antonova 2005) and increased our understanding of the impact of invasive species on native communities (Touyama et al. 2003, Stringer et al. 2009), these approaches also have shown several associated problems. To date, numerous studies have been undertaken at different sites for better understanding of the effect of urbanization on ant diversity in urban. However, it is difficult to assess the effect of urbanization on ant diversity in urban environments in isolation because different sites have subtle differences as microhabitat characteristics (Buczkowski and Richmond 2012). According to Porter and Savignano (1990), invasion of the fire ant *Solenopsis invicta* in the southeastern USA had a negative impact on the populations of other insects, particularly native ants, in the region. However, for 12 years after the fire ants had invaded an area, the diversity of local populations of native ants and other insects had returned to pre-invasion levels (Morrison 2002). Consequently, although one-off experiments conducted to assess the effects of urbanization or invasive species on native populations may be technically sound, the overall value of data that these studies generate is somewhat limited as they reflect conditions at a single point in time.

In order to better understand the ecology of urban environments, we examined changes in the community structure of ants in urban parks over a decade. We also clarified the impacts of invasion by the Argentine ant *Linepithema humile* (Mayr) on species diversity of indigenous ants by using long-term data. In conjunction with the field data collected in this study, we reanalyzed the data of Touyama et al. (2003) to assess the effects of urbanization and *L. humile* abundance on indigenous ant communities. The findings of the study are considered useful for the conservation and management of urban biodiversity.

MATERIALS AND METHODS

Study area

We resampled the nine urban parks among 12 urban parks (Touyama et al. 2003) in Hatsukaichi and neighboring Hiroshima (Fig. 1) to compare changes in ant assemblages over time because the three urban parks have been changed and trimmed for the construction of high-way road and apartment.

All of the urban parks surveyed were small (less than 0.5 ha) with a centrally situated playground consisting of bare soil, playing equipment, and occasionally wisteria trellises. Organic matter in the topmost soil layer (e.g., leaf litter and humus) was limited to areas under shrubs, as fallen leaves and weeds were usually removed during routine park maintenance. The environmental conditions in the surveyed parks were thus considered to be very stable over time.

Ants were collected using the same time-unit sampling method as that employed in 1999. The method consisted in collecting as many species as possible within a specified time period (10 min, repeated 10 times at each park in this study) by conducting visual and manual searches for ants on the ground surface, under stones and bark, and around the bases of trees (Touyama 2000, Ogata 2001).

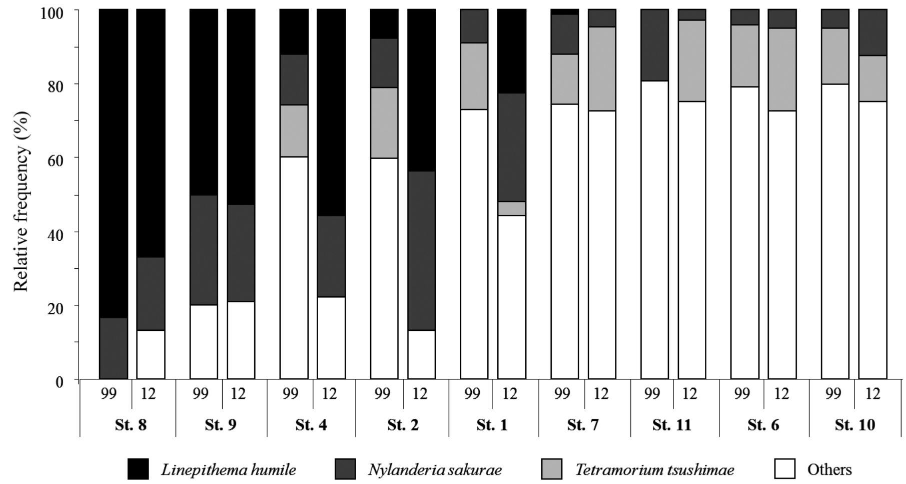

Fig. 2. Changes in relative abundance of three common ant species in each park over time.

Ant specimens were then sorted and identified to species by using taxonomic keys provided by the Japanese Ant Database Group (2003).

Data analysis

Using the frequency data for each species, we performed a non-metric multidimensional scaling (NMDS) analysis to examine the similarities between all parks over time. This ordination technique represents samples as points in a two-dimensional space, such that the relative distances of all points are in the same rank order as the relative similarities of the samples (Van der Gucht et al. 2005). We used the Bray-Curtis index as a similarity measure for different parks. The Bray-Curtis index is considered to be well suited for multivariate statistics because it is less affected by the numbers of rare species in the samples (Krebs 1989). The NMDS goodness of fit was estimated using a stress function which ranged from 0 to 1, with values approaching zero indicating a good fit. An analysis of similarity (two-way crossed ANOSIM) with 999 permutations was used to test for significant differences between groups. In addition, the similarity percentage tests (SIMPER) were conducted to identify which species contributed the most to the average measure of dissimilarity between groups (Clarke 1993). All analyses were performed using Primer ver. 5 (Clarke and Gorley 2001).

Finally, to assess temporal species turnover, we computed Jaccard's index (JI) from the presence-absence data in the ant data matrix; JI = $a/(a + b + c)$, where a is the number of species observed in both years, b is the number of species observed only in 1999, and c is the number of species only observed in 2012 (Radomski and Goeman 1995, Stuart et al. 2012).

RESULTS

Changes in the number of ant species over time

Table 1 shows a list of the species identified in each park together with their abundance. A total of 25 species are shown; 23 species were recorded in the 1999 survey, and 20 species were recorded in the present survey. The differences between the two datasets are due to the inclusion of two previously unobserved species, *Temnothorax koreanus* (Teranishi) and *Hypoponera nubatama* Terayama & Hashimoto, and the apparent disappearance of five previously observed species, *Nylanderia flavipes* (F. Smith), *Pachycondyla chinensis* (Emery), *Pheidole noda* F. Smith, *Crematogaster osakensis* Forel, and *Lasius hayashi* Yamauchi & Hayashida.

The number of ant species exhibited several different inclinations among the examined parks. First, in parks where *L. humile* was dominant (e.g., St. 8 and St. 9 in Table 1), the number of the indigenous species remained relatively low in both 1999 and 2012. Second, in parks where the incidence of *L. humile* increased over time (e.g., St. 1, St. 2, and St. 4 in Table 1), the incidence of the indigenous ant species decreased significantly over time (Mann-Whitney U test, $P < 0.05$). Third, in all of the parks except St. 10, the number of indigenous species decreased, but these differences were not statistically significant (Mann-Whitney U test, $P > 0.05$).

Table 1. Relative frequency (0 to 1, $n = 10$) of ant species in the urban parks surveyed by time-unit sampling[*] in 1999 and 2012

Sites / Species	St. 8		St. 9		St. 4		St. 2		St. 7		St. 11		St. 10		St. 6		St. 1	
	1999	2012	1999	2012	1999	2012	1999	2012	1999	2012	1999	2012	1999	2012	1999	2012	1999	2012
Linepithema humile	1	1	1	1	0.7	1	0.4	1	0.1									0.6
Nylanderia sakurae	0.2	0.3	0.6	0.5	0.8	0.4	0.7	1	0.8	0.2	0.9	0.1	0.3	0.6	0.2	0.2	0.5	0.8
Camponotus vitiosus			0.2		0.2			0.2		0.1	0.3	0.2		0.3	0.2	0.3	0.1	0.1
Nylanderia flavipes			0.2				0.1		0.1		0.1							
Tetramorium tsushimae					0.8		1		1	1		0.8	0.9	0.6	0.8	0.9	1	0.1
Pristomyrmex punctatus					0.4		0.6		0.6	0.2	0.6	0.3	0.7	0.4		0.5	0.9	
Formica japonica					1		0.9		1	1		0.3	1	1	1		1	
Temnothorax spinosior					0.4	0.2	0.8						0.1				0.5	
Temnothorax congruus					0.1	0.1			0.4	0.1							0.1	
Ochetellus glaber					0.1		0.2		0.4	0.1	0.1		0.4		0.5		0.2	0.4
Crematogaster matsumurai					0.2		0.1		0.4	0.6	0.3	0.2		0.5		0.1	0.1	0.5
Pachycondyla chinensis					0.8						0.1				1		0.5	
Cardiocondyla nuda					0.2		0.2								0.2	0.4	0.2	
Camponotus japonicus							0.2						0.1	0.1				
Lasius japonicus									0.4	0.3	0.4		0.5	0.2	0.2	0.7	0.2	
Pheidole noda									0.9		0.3		0.9					
Pheidole indica											1	1		0.2	0.3			
Solenopsis japonica		0.1						0.1	0.1	0.3		0.1	0.1	0.3			0.2	0.1
Nylanderia amia						0.1			0.2	0.1	0.4	0.6	0.9	0.2		0.6		
Messor aciculatus					0.1										0.4	0.3		
Crematogaster osakensis									0.1		0.1							
Lasius hayashi									0.2		0.1							
Monomorium chinense				0.4					0.5	0.3				0.4				
Temnothorax koreanus		0.1								0.1								
Hypoponera nubatama																		0.1
Number of species	2	4	4	3	13	5	11	4	16	13	13	9	11	12	10	9	13	8

[*]Time-unit sampling: 10 replications of 10-min sampling in each park. The site numbers correspond to Fig. 1.

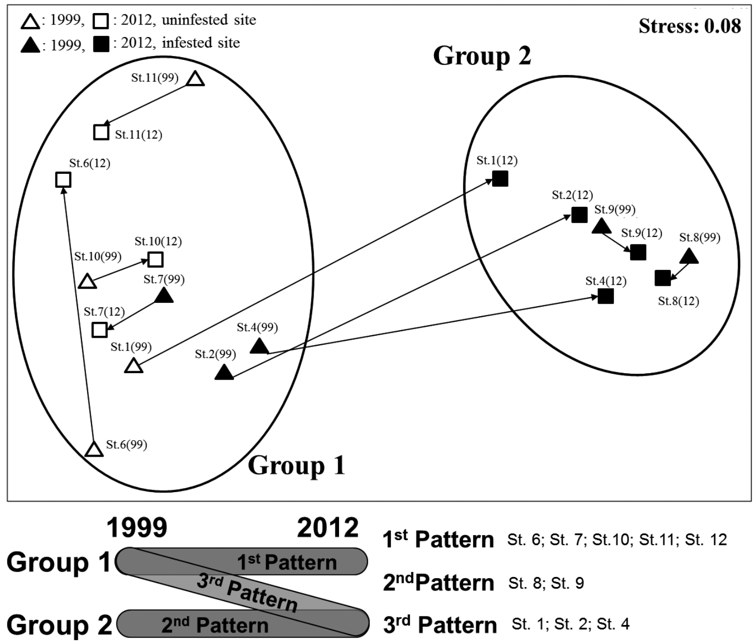

Fig. 3. The non-metric multidimensional scaling (NMDS) ordination to show community composition and tracking patterns of ant community composition over time.

Changes in occurrence patterns of each ant species over time

Fig. 2 shows the relative frequency of three ant species that were common to all of the parks surveyed in both 1999 and 2012. Interestingly, *L. humile* maintained either high (e.g., St. 8 and St. 9) or increasing (e.g., St. 1, St. 2, and St. 4) frequencies of occurrence over time. For example, the incidence of *Tetramorium tsushimae* Emery was high in those parks where *L. humile* was absent and was markedly lower in those parks where *L. humile* had increased over time.

Spearman's rank correlation coefficient analysis revealed a significantly negative correlation between the incidence of *L. humile* and other ant species, such as *T. tsushimae* (r_s = −0.67, P = 0.002), *Pristomyrmex punctatus* F. Smith (r_s = −0.68, P = 0.002), *Formica japonica* Motschoulsky (r_s = −0.61, P = 0.007), *Crematogaster matsumurai* Forel (r_s = −0.54, P = 0.020), *Lasius japonicus* Santschi (r_s = −0.76, P = 0.000), *Pheidole indica* Mayr (r_s = −0.55, P = 0.019), and *Nylanderia amia* (Forel) (r_s = −0.63, P = 0.006). However, no correlation was observed between the incidence of *P. noda*, *C. osakensis*, and *L. hayashi* and invasion by *L. humile*. The incidence of *Nylanderia sakurae* (Ito) remained constant over time irrespective of invasion *by L. humile* (Table 1).

Changes in ant community composition over time

The NMDS analysis separated ant communities into two groups (Fig. 3); stress was low (0.08), indicating a good degree of fit. A two-way crossed ANOSIM showed that the species composition of the ant communities at the sites did not change over time (R = 0.157, P = 0.119), except when *L. humile* abundance increased, as this caused a significant change of the ant communities (R = 0.478, P = 0.004).

Additionally, SIMPER analysis revealed that five species were the main contributors to the differences between two groups (overall dissimilarity = 77.38%, see Fig. 3). The representative species of group 1 were *F. japonica* and *T. tsushimae*, whilst *L. humile* was the most representative of group 2 (Table 2).

In addition, we identified three general patterns of change in ant community structure between groups 1 and 2 in response to changes in *L. humile* incidence over time (Fig. 3). Although the first and second patterns both shifted within their respective groups, the third pattern shifted from group 1 to group 2, in response to an increase in *L. humile* abundance over the 13 years between surveys. The index values of the estimated mean of species turnover in the first and second patterns were 0.52 and 0.55, respectively, while that associated with the third pattern was higher, at 0.72.

Table 2. Results of the similarity percentage (SIMPER) analysis for five ant species representing more than 50% of the cumulative contribution to average dissimilarity between two groups

Species	Group 1 mean abundance	Group 2 mean abundance	% contribution to dissimilarity	Cumulative %
Linepithema humile	0.16	0.86	14.38	14.38
Formica japonica	0.75	0.00	13.20	27.58
Tetramorium tsushimae	0.73	0.13	12.27	39.85
Pristomyrmex pungens	0.43	0.07	7.20	46.05
Nylanderia sakurae	0.54	0.46	6.20	53.25

DISCUSSION

In this study, we clarified the effect of urbanization and invasion by *L. humile* on the following three ant community characteristics in urban parks over a ten-year period: 1) changes in the number of ant species over time, 2) changes in occurrence patterns of each ant species over time, and 3) changes in ant community composition over time.

First, the number of ant species decreased significantly after an invasion or increase in *L. humile* abundance. Further, the number of ant species in parks where were either not invaded by *L. humile* or had post-high intensities of *L. humile* did not change significantly, suggesting that the presence of *L. humile* had a negative impact on the number of indigenous ant species (Miyake et al. 2002, Sunamura et al. 2007).

Second, both urbanization and high numbers of *L. humile* had a marked effect on indigenous ant species over time. In case of *P. noda*, *C. osakensis*, and *L. hayashi*, they were not observed over time, implying that their absence in invaded parks was due to urbanization and not due to invasion by *L. humile*. However, high frequencies of *L. humile* had a negative impact on the incidence of several indigenous ant species in urban parks, including *T. tsushimae* and *P. punctatus*. Thus, the obtained results implied that ant species surveyed in this study, except for *N. sakurae*, have been affected by continuous urbanization or *L. humile* invasion.

Third, NMDS analysis showed that the ant communities of the surveyed parks could be divided into two groups, and that the compositions of the ant communities in each of these groups were significantly different. Three patterns of change in ant community structure were identified in both groups; only the third pattern exhibited a shift from group 1 to group 2. This shift was caused by an increase in *L. humile* abundance over time. Furthermore, the species turnover rate in groups exhibiting the third pattern was higher than the species turnover rate in groups exhibiting the other two patterns. Therefore, the ant community composition may be simplified under the influence of new invasions or increases in *L. humile* abundance.

In conclusion, because most urban parks are maintained regularly for the benefit of visitors and are located in developed areas, these habitats are considered to be very difficult for ant species to invade. However, in the event that colonization by an invasive species does occur, the impact on any existing indigenous ant assemblages is expected to be negative. The findings of this study demonstrated that invasion by *L. humile* decreased the species richness, and that species turnover accelerated the abundance of *L. humile* increased over time, simplifying the indigenous ant community structure in urban parks (Oliveras et al. 2005). Consequently, a continuous monitoring and ecological study of *L. humile* is considered necessary in order to preserve ant biodiversity in urban environments.

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