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Does the invasive species *Reynoutria japonica* have an impact on soil and flora in urban wastelands?

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Abstract Invasive plants are recognised as a major threat to biodiversity. Although they are well-established in natural areas, the supposed negative impacts of invasive plants upon communities and ecosystems have so far been poorly investigated in urban areas, where invasions are a main issue for ecologists and for urban planners and managers. We propose to assess the effects of an invasive species along an invasion gradient in a typical urban habitat. We focused on the Japanese knotweed (*Reynoutria japonica* Houtt.), a widespread invasive species in Europe and North America. We considered eight urban wastelands invaded by this species in the heart of the Greater Paris Area, France. On each site, we ran four transects from the centre of the Japanese knotweed patch towards the uninvaded peripheral vegetation. We recorded the flora using the line intercept method, and several soil parameters (thickness of A horizon, abundance of earthworm casts, topsoil Munsell value, pH) every metre along each transect. The A horizon was thicker and the topsoil darker under *R. japonica* canopy. Thus, this invasive plant species seemed to influence soil organic matter pool. However, our results also steadily showed that *R. japonica* locally excluded and/or severely reduced the cover of many plant species through competition. Our study clarified the local effects of *R. japonica*: an influence on the soil organic matter, and a severe negative impact on wasteland plant communities. We suggest implications in both conservation and restoration ecology.

Keywords Competition, Organic matter, Japanese knotweed, Wasteland plant community, *Fallopia japonica*, *Polygonum cuspidatum*

Introduction

Human-mediated species introductions have dramatically increased in recent decades with the expansion of long-distance trade (Westphal et al. 2008) and are leading to global biotic homogenisation (McKinney and Lockwood 1999; Olden 2006). Biological invasions are now regarded as a significant component of global change (Vitousek et al. 1997), one of the major threats to biodiversity on Earth (Gurevitch and Padilla 2004), and a central issue in conservation biology. In plant communities, the success and impacts of invasive species (*sensu* Richardson et al. 2000) have been thoroughly studied in natural and semi-natural areas, where they are often harmful to native ecosystems. Not only can they alter floristic composition and diversity (Meiners et al. 2001), and the successional dynamics of vegetation (Yurkonis and Meiners 2004), but they can also disrupt soil properties (Ehrenfeld 2003; Vanderhoeven et al. 2005) and soil biota (Wolfe and Klironomos 2005; see Bohlen 2006 for a review). Moreover, several studies have suggested positive feedbacks: invasive species change soil biota and/or soil biogeochemistry in ways that benefit themselves at the expense of native species (Klironomos 2002; Levine et al. 2006). In addition, these positive feedbacks may contribute to ‘invasional meltdown’, i.e. a process by which invasive species aid one another, leading to an increased rate of establishment and/or an impact at the community level (Simberloff and Von Holle 1999; Simberloff 2006).

Urban areas are particularly prone to plant invasions (Trepl 1995; Pysek 1998): many exotic plants are deliberately introduced for ornamental purposes (Reichard and White 2001), and urban areas are focal points for trade and transport (roads, railways and waterways, harbours and airports), which convey many exotic plant species and maintain a high level of propagule pressure (Vilà and Pujadas 2001; McKinney 2004; von der Lippe and Kowarik 2007). Several consequences may arise from this increase in the probability of species introductions. Predicting the impact of invasions on urban plant communities is not straightforward. Since invasive plant species commonly establish in disturbed, vulnerable communities that are far from ecological equilibrium (Kowarik 1995; Niemelä 1999; Williams et al. 2005; Godefroid et al. 2007; Muratet et al. 2007), it can be assumed that they are a major threat to local urban plant communities. All the more since, according to the ‘invasional meltdown’ theory, invasive species tend to invade the same sites, and thus may have more impact on native communities than predicted by summing independent effects. In contrast, it can be assumed that existing strong urban pressures select against ‘weak’ species, and that the remaining species are successful competitors, which are less likely to be affected by invasive species, even the most competitive.

Despite the success of urban ecology (Miller and Hobbs 2002; Sukopp 2002; Adams 2005) and the emergence of reconciliation ecology (Rosenzweig 2003), it is striking to note that little has been published about plant invasions in cities. Thus there is a need to assess the impact of invasive plant species in urban areas.

In this study, we aimed at better understanding invasion processes and the threats caused by invasive species in cities. We examined the influence of an invasive plant species, Japanese knotweed (*Reynoutria japonica* Houtt.), on native plant communities and soil characteristics within invaded sites of the Greater Paris Area (France). This species is one of the “100 of the world’s worst invasive alien species” (Lowe et al. 2000) and is often considered a prominent disturbance by managers of urban green spaces and parks. We focused on urban wastelands, a habitat frequently colonised by *R. japonica* that plays a major role in urban biodiversity, since they are numerous and widespread, exhibiting the highest floristic richness of all urban plant communities (Muratet et al. 2007, 2008). In its introduced range, *R. japonica* has been shown to alter plant and invertebrate species diversity in natural riparian habitats (Gerber et al. 2008). We investigated whether similar floristic patterns occurred in urban areas. Furthermore, *R. japonica* influences soil nitrogen patterns in its native range (Hirose and Tateno 1984) and is known to produce large amounts of biomass, often more than do other plants of invaded communities (Dassonville et al. 2007). Therefore we hypothesised that *R. japonica* was likely to affect ecosystem process, resulting in changes in the soil organic matter pool.

Consequently, we addressed the following questions: (1) how does *R. japonica* influence the floristic richness and composition of plant communities in urban wastelands? (2) Does the presence of *R. japonica* modify the soil organic matter pool?

Materials and methods

Study species

Japanese knotweed (*R. japonica* Houtt., Polygonaceae) is a perennial geophyte with bamboo-like annual stems up to 3 m tall and a deep rhizome network, which forms dense patches (Beerling et al. 1994; Smith et al. 2007). Native to Japan and eastern Asia, it was introduced in Europe and North America in the early nineteenth century. *R. japonica* grows in riparian habitats, but this opportunistic species tolerates a broad range of soil and climate conditions, and is also widely distributed in artificial, disturbed areas such as wastelands or road and railway banks (Müller 2004).

Study area

The study was carried out in the heart of the Greater Paris Area ($48^{\circ}51^{\circ}\text{N}$; $2^{\circ}21^{\circ}\text{E}$; Fig. 1), which consists of about 70% urbanised areas (IAURIF 2003) and a human density of 8,501 versus 112 inhab./km² on average in France (INSEE 2006). The climate is oceanic with continental trends: the mean annual temperature is 11.7°C with 16°C thermal amplitude and an average annual rainfall of 641 mm. We chose to focus on urban wastelands, defined as abandoned lands where plant species grow with no human control (Muratet et al. 2007).

Site description

Eight sites, all invaded by *R. japonica*, were included in this study (Fig. 1). The site list was provided by the floristic database of the National Botanical Conservatory of the Parisian Region (CBNBP 2008). We mapped the sites and we calculated their area using a Geographic Information System (MapInfo 8.5, MapInfo Corporation 2006). Land Use Patterns (LUP) were provided by IAURIF (2003) and grouped into nine major LUP classes. Six successive LUP updates are available from 1982 to 2003, allowing estimating the age of sites. Consistently with the results of Muratet et al. (2007), six of the eight sites belonged either to “building sites and vacant urban” (BUILVAC) class, or to “open urban areas and rural” (OPENRUR) class. Only W3 and W4 were located respectively in “facilities” (FACI) and “transport” (TRAN) classes, corresponding to rather small unused spaces in the built matrix. Also consistently with Muratet et al. (2007), half of the sites (W2, W4, W6, W7) were older than 21 years (in the same LUP class since 1982), whereas two sites were less than 9 years old (W3, W5) and the two remaining (W1, W8) were of intermediate age. The characteristics of all sites are summarised in Table 1.

Sampling strategy

Paired-sites comparisons are controversial: when differences are observed between invaded and uninvaded sites, either they result from a differentiation triggered by the invasive species, or they merely reflect differences pre-existing the invasion event. Therefore other studies resort to within-site comparisons, where uninvaded control plots are located as close as possible to the ‘invasion front’ (e.g. Vanderhoeven et al. 2005; Dasonville 2008). Similarly,

we focused on sites already invaded by *R. japonica* and we compared patches of *R. japonica* (invaded area) with the surrounding uninvaded area.

Each site displayed at least one patch of *R. japonica* surrounded by a continuous herbaceous cover of different heights, sometimes mixed with shrubs. Patches were circular to oval-shaped, with a well-delineated ‘invasion front’. They ranged from 10 to 74 m², i.e. from 0.038 to 3.26% of the entire site (Table 1).

Floristic composition (Kerguelén 2003), floristic richness, vegetation cover and soil parameters were assessed along four transects arranged in a cross shape, running from the centre of the invaded area towards the peripheral uninvaded area, at right-angles to the ‘invasion front’ (see Wearne and Morgan 2004 or Maerz et al. 2005 for a similar design). Contrary to isolated paired plots, line transects allowed detecting gradual changes with the expanding of the invading population, so that the distance from the centre of the patch could be considered as a proxy of invasion time. Transects were centred on the invasion front, with an identical length within and outside the patch, thus the length of the whole transect ranged from 3 to 22 m. One of the sites (W5; Fig. 1) was destroyed before all four transects were inventoried, therefore only two transects were surveyed for W5.

Data collection

We conducted our study in May 2007. The vascular flora was sampled along each transect using the line intercept method (Canfield 1941): all vascular plant species other than *R. japonica* that intercepted the transect line were recorded every centimetre. We classified species as ‘native’ versus ‘exotic’, according to a list compiled by professional botanists of the National Botanical Conservatory of the Parisian Region (CBNBP 2008).

Transects were split up into 0.5 m sections (see Fig. 3). We calculated species richness and estimated the total cover (non-bare ground) of the herbaceous layer, *R. japonica* excepted, in each section.

Several soil parameters were recorded every metre along each transect, on a ground sample of 20 cm². We measured (1) the abundance of earthworm casts at the soil surface (observations grouped into three classes: 0 = no earthworm casts, 1 = few earthworm casts, i.e. covering less than 25% of the sample, 2 = abundant earthworm casts, i.e. covering more than 25% of the sample), (2) the thickness of the A horizon (cm), (3) the soil colour in the top five centimetres, according to the Munsell Soil Color Charts (Munsell Color Company 1975): the colour “value” ranges

from 1 to 5 and decreases with the amount of organic matter (Wills et al. 2007). We also recorded (4) soil pH-H₂O: the pH was measured using a Fisher Scientific pH-meter, 3 h after soil was oven-dried at 40°C for 36 h and mixed with deionised water (soil:water 1:5 v/v) for 5 min (AFNOR 1999).

Data analysis

We analysed the variation of four soil parameters formerly tested for independence: (1) thickness of the A horizon, (2) abundance of earthworm casts, (3) topsoil Munsell value (colour parameter) and (4) pH, through linear mixed-effect models with section as a fixed effect and site as a random effect (using nlme package, Pinheiro and Bates 2000). Because of insufficient replicates, sections '-11m' to '-6 m' were grouped into one single class ('<-5 m'), and similarly, sections '6 m' to '11 m' were grouped into class '>5m'.

Within each site, we assessed differences in floristic composition between invaded and uninvaded areas through distance-based redundancy analysis, an ordination method which compares distances among groups (dbRDA, Legendre and Anderson 1999). We calculated the floristic distances d_f between and within uninvaded areas and invaded areas via the Jaccard similarity s_f index as follows (using ADE4 package, Thioulouse et al. 1997):

$$d_f = \sqrt{1 - s_f}$$

where s_f is the fraction of species observed in both sites. We then performed dbRDA, to explore the relationship between floristic distances and the "invasion" variable. To graphically display the results, we used Nonmetric Multidimensional Scaling.

We analysed the variation of (1) species richness and (2) total percent cover as a function of the section's location (a proxy of 'invasion effect') using linear mixed-effect models with section as a fixed effect and site as a random effect. Because of insufficient replicates, sections '-11 m' to '-6.5 m' were grouped into one single class ('<-6 m'), and similarly, sections '6.5 m' to '11 m' were grouped into class '>6m'.

Due to the small number of replicates (2) and to lacking data for the A horizon and pH, site W5 was discarded from composition analyses, and from all analyses involving soil parameters, while it was kept for richness and cover analyses.

Statistical analysis were performed using R software (R 2.8.0, R Development Core Team 2008).

Results

Excluding *R. japonica*, a total of 83 species were observed along the 30 transects we inventoried, with an average of 23 ± 3 (SE) species per site (Table 1). Among these, 86.7% were native, 13.3% were exotic.

The species most frequently found were *Dactylis glomerata* (8/8 sites), *Galium aparine* (7/8 sites), *Elytrigia repens*, *Picris hieracioides*, and *Plantago lanceolata* (all in 6/8 sites).

Impact of *R. japonica* on wasteland soil

Globally, the thickness of the A horizon decreased significantly from the centre of *R. japonica* patches towards uninhabited periphery ($P < 0.0001$; Fig. 2a) while the topsoil Munsell value increased significantly ($P < 0.0001$; Fig. 2b). On average, the A horizon was (mean \pm SE) 2.77 ± 0.09 cm thick in invaded areas versus 1.72 ± 0.07 cm in uninhabited areas, and the topsoil Munsell value was 2.92 ± 0.07 under *R. japonica* versus 3.54 ± 0.06 in adjacent uninhabited vegetation.

On the contrary, the abundance of earthworm casts did not differ significantly along transects ($P = 0.64$), nor did the pH ($P = 0.17$).

Impact of *R. japonica* on wasteland flora

Species richness and total percent cover increased significantly from the centre of *R. japonica* patches towards adjacent uninhabited vegetation ($P < 0.0001$ for both models, Fig. 3a, b; *R. japonica* is excluded from analyses and figures).

Species composition differed significantly between uninhabited and invaded area for sites W6 and W7 ($P = 0.001$ for each of them in dbRDA tests, Fig. 4a, b). In contrast, no difference in species composition was detected between uninhabited areas and areas invaded by *R. japonica* for sites W1, W2, W3, W4 and W8 (respectively, $P =$

0.613, P = 0.690, P = 0.081, P = 0.067, P = 0.699; Fig. 4c).

Discussion

Impacts of *R. japonica* on soil

In the wastelands studied, soils were quite variable among sites, making it difficult to detect any effect of *R. japonica*. However, despite the variability of all edaphic parameters, we observed a thicker A horizon, as well as a darker topsoil under *R. japonica* as compared to the surrounding uninvaded area. Altogether, these results led us to argue for a strong influence of *R. japonica* on the soil organic matter pool.

These results could be explained by the massive production of annual aboveground and permanent belowground biomass (Dassonville et al. 2007). According to Maerz et al. (2005) and Dassonville (2008), *R. japonica* provides abundant but low-quality litter, and stems and leaves decay slowly, resulting in the accumulation of large, rough fragments, and in an increase in litter thickness (personal observation). Added to the slightly alkaline soils (pH ranging from 7.2 to 8.3), this organic matter supply could result in a darker topsoil and a thicker A horizon.

Ehrenfeld (2003) suggested that invasive plants could enhance productivity and nutrient availability in invaded areas via an abundant litter, thereby increasing their own success. Although our study did not allow showing evidence for such a process, it is possible that *R. japonica* contributes to its own growth and productivity by creating a positive plant-soil feedback.

Nevertheless, this apparent soil enrichment may especially benefit very common eutrophic species, such as the nitrophilous *G. aparine*, frequently observed in the invaded areas.

Impacts of *R. japonica* on plant communities in urban wastelands

As expected, there was an important decrease in herbaceous cover under *R. japonica*, largely due to the competitive exclusion of most grasses and forbs. Poaceae are a meaningful example: they covered almost 40% of uninvaded areas, but only 8% of invaded areas. The herbaceous community was also clearly poorer under *R. japonica*. We cannot completely exclude that *R. japonica* systematically established in quasi-bare grounds (differences would then pre-exist the

invasion). However, our sampling design allowed us detecting gradual changes along transects, therefore we assume that *R. japonica* was rather responsible for an impoverishment of plant communities, i.e. these differences in richness and cover followed, and not pre-existed, the invasion by *R. japonica*. This could be ascribed to the competitive ability of this invasive species. *R. japonica* could win both aboveground and belowground competition, thanks to a high growth rate, the production of large amounts of biomass (Beerling et al. 1994; Dassonville et al. 2007), the efficiency of leaves to intercept light (see the experiment conducted on the close *Reynoutria x bohemica* by Siemens and Blossey 2007), the early use of space and soil resources, and possible allelopathic interactions (Vrchotova and Sera 2008). Thus, *R. japonica* could become a long-term dominant species in invaded plant communities, forming dense, homogeneous, near monospecific patches.

Thus, after patterns of competitive exclusion were shown for European and North American riparian habitats (Gerber et al. 2008), we showed that similar patterns can be observed in urban areas also.

Despite the strong impacts on diversity and vegetation abundance, the effects of *R. japonica* on floristic composition are more questionable. Differences were significant in only two sites. On average the proportion of flora growing in invaded areas represented 59% of the species versus 90% in uninvaded areas. Some species were never or rarely recorded under *R. japonica*, like *P. lanceolata*, *Achillea millefolium* or *P. hieracioides*. On the contrary, others frequently coexisted with *R. japonica*, like *Rubus fruticosus*, *Urtica dioica* and *G. aparine*. Gerber et al. (2008) had already found the latter two species to persist under *R. japonica* canopy in riparian habitats.

However, across all sites, *R. japonica* patches represented at most less than 4% of the whole surface of the site. This slightly balances our results: *R. japonica* deeply impacts plant communities at local scale, but these effects are questionable at larger scale. In particular, in a very dynamic urban landscape, the high turnover of wasteland sites could prevent *R. japonica* to dramatically expand after establishing, and at the same time it could create new open spaces where plant species could find favourable conditions for growing.

Perspectives in restoration biology

Our study stressed the relevance of soil-plant relationships in the area of plant invasions. Soil characteristics can partly control the establishment of an invasive plant species, including in disturbed areas, as demonstrated by Kulmatiski et al. (2006) in abandoned agricultural fields. In return, invasive plant species, once established, can

modify soil abiotic and biotic components. As such, our observations supported the conceptual sketch suggested by Wolfe and Klironomos (2005): the arrival of an exotic species in an ecosystem influences the links between plant community composition, soil community composition, and ecosystem processes and properties. Studies of plant invasions increasingly explore the effects of invasive plants on soils (Vitousek 1990; Ehrenfeld 2003; Levine et al. 2003). Some authors emphasise that these effects matter for the restoration of local flora, as higher nitrate content, for example, can inhibit the growth of several native species, alter dominance relationships in the plant community, and hence curb the restoration process (Kourtev et al. 1999; Yu et al. 2005).

In practice, there are many attempts—rarely successful—to remove *R. japonica* from invaded communities, both in natural riparian communities and in less natural areas, like urban green parks or urban forest remnants. In order to prevent *R. japonica* from growing again, river managers and urban planners usually associate such removal projects with greening projects, by planting trees, shrubs, forbs and/or grasses, depending on the vegetation structure they intend to create or restore. Soils are rarely—if ever—taken account of in their scheme. Some authors reported that the effects of an invasive plant on soil properties are very likely to persist even after its removal (Kourtev et al. 1999; Dassonville et al. 2007). Similarly, we can expect that soils would remain modified after the eradication of *R. japonica*, and that this could influence the restoration process, by favouring some species traits, such as eutrophy. Kourtev et al. (1999) also suggested that persistent soil changes could favour other exotic species. Although our data did not show evidence of such facilitation, we recommend managers to be very watchful with the evolution of floristic composition during restoration process.

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Figure captions

Fig. 1 Map of the study area, showing the four administrative districts (French “départements”) of the heart of the Greater Paris Area. Each point corresponds to a survey wasteland site

Fig. 2 Thickness of A horizon (a) and topsoil Munsell value (b) along transects as a function of the section’s location (distance to invasion front inside or outside the patch)

Fig. 3 Species richness (a) and total percent cover (b) along transects as a function of the section’s location (distance to invasion front inside or outside the patch). *R. Japonica* is excluded from this figure

Fig. 4 Floristic composition in invaded areas (IA) and uninvaded areas (UA) illustrated by Nonmetric Multidimensional Scaling (NMDS). W6 (Noisy-le-Grand, a) and W7 (Paris, b) display significant differences between IA and UA. For W2 (Champigny-sur-Marne, c) floristic composition is similar in IA and UA

Table 1 Characteristics of the eight survey sites

Site	Borough	Site area (m ²)	Patch area (m ²)	Proportion of the site invaded by <i>R. japonica</i> (%)	Sampled species richness	Proportion of exotic species (%) on the transects
W1	Gennevilliers 1	11,600	66	0.56	31	16.1
W2	Champigny	1,300	42.7	3.28	13	15.4
W3	Châtenay-Malabry	5,200	10.2	0.19	15	0
W4	Colombes	5,400	74.2	1.37	33	9.1
W5	Gennevilliers 2	3,000	35.9	1.20	19	11.1
W6	Noisy-le-Grand	113,600	41.5	0.036	28	10.7
W7	Paris	35,900	17.6	0.049	20	5
W8	Villetaneuse	29,900	26.2	0.087	26	11.5
	Global				83	13.3
	Mean				23	9.9

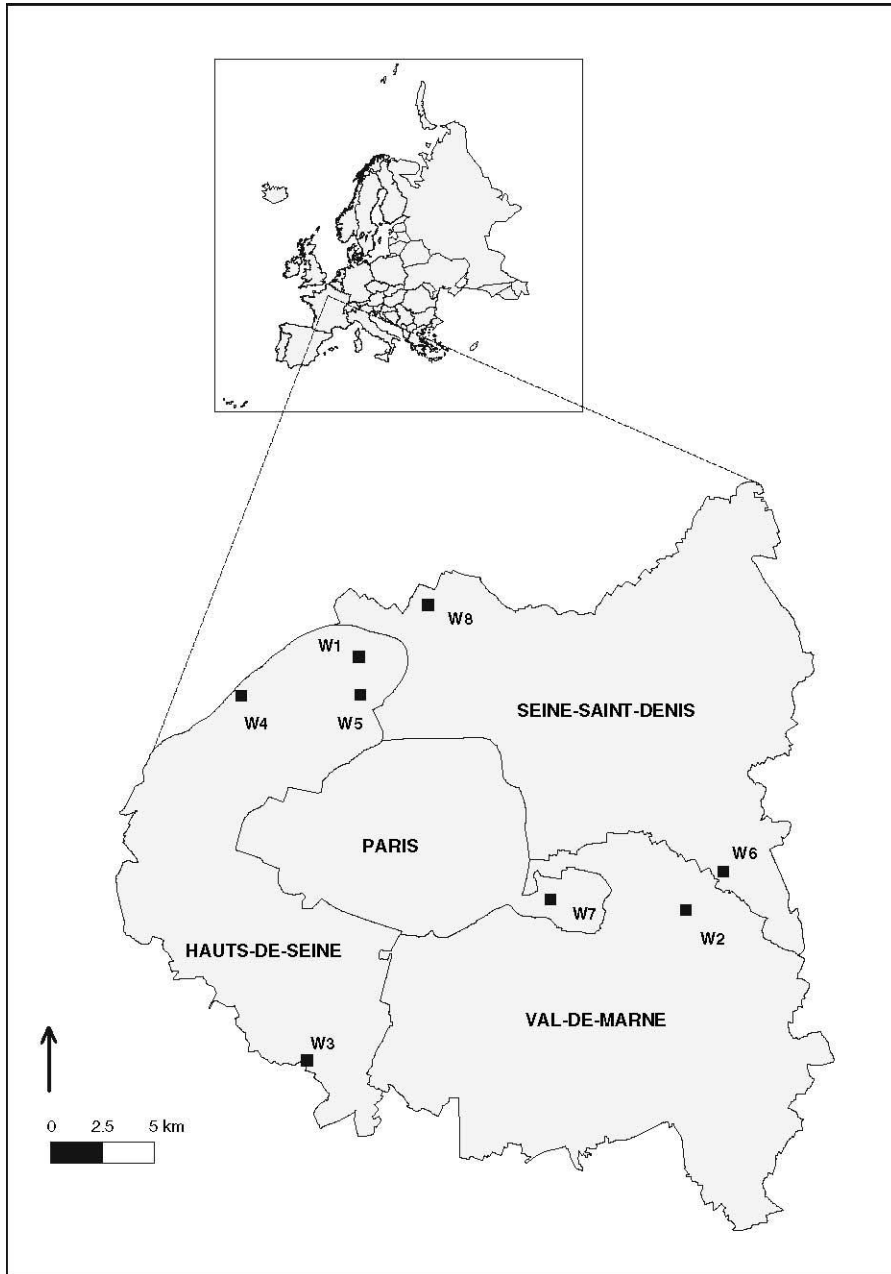


Fig. 1

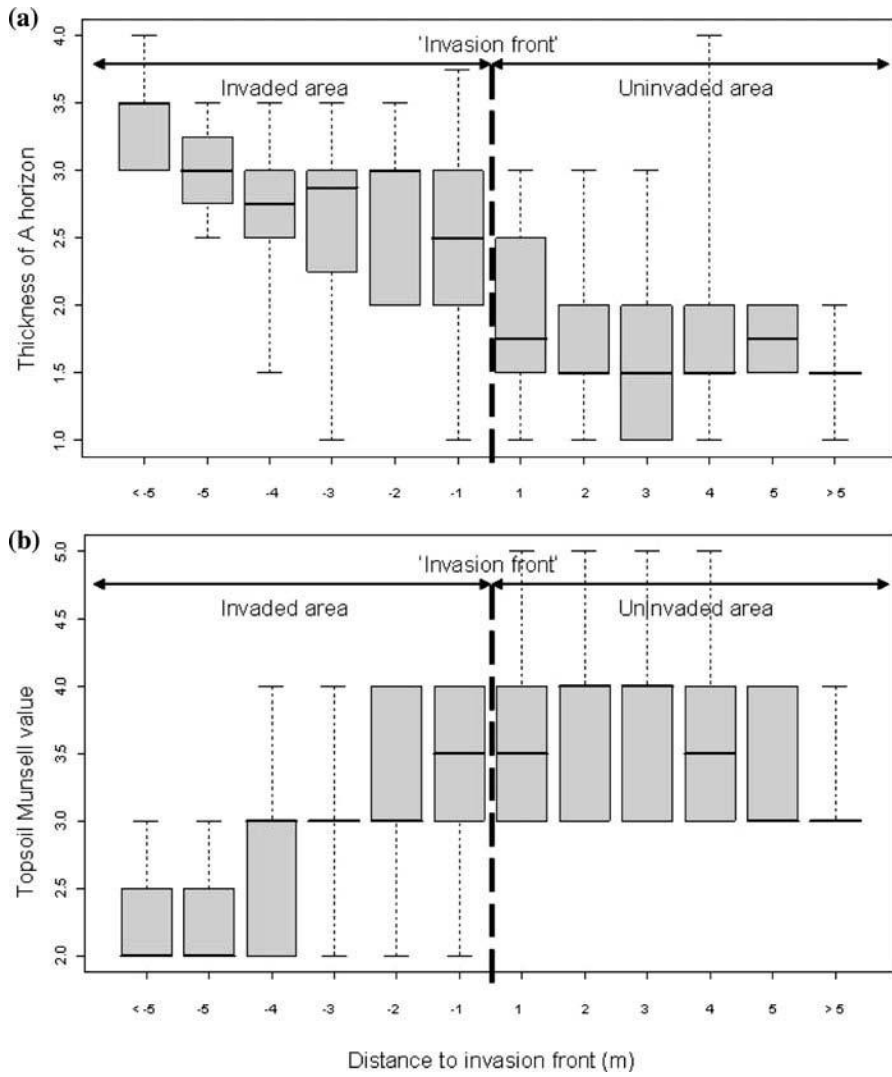


Fig. 2

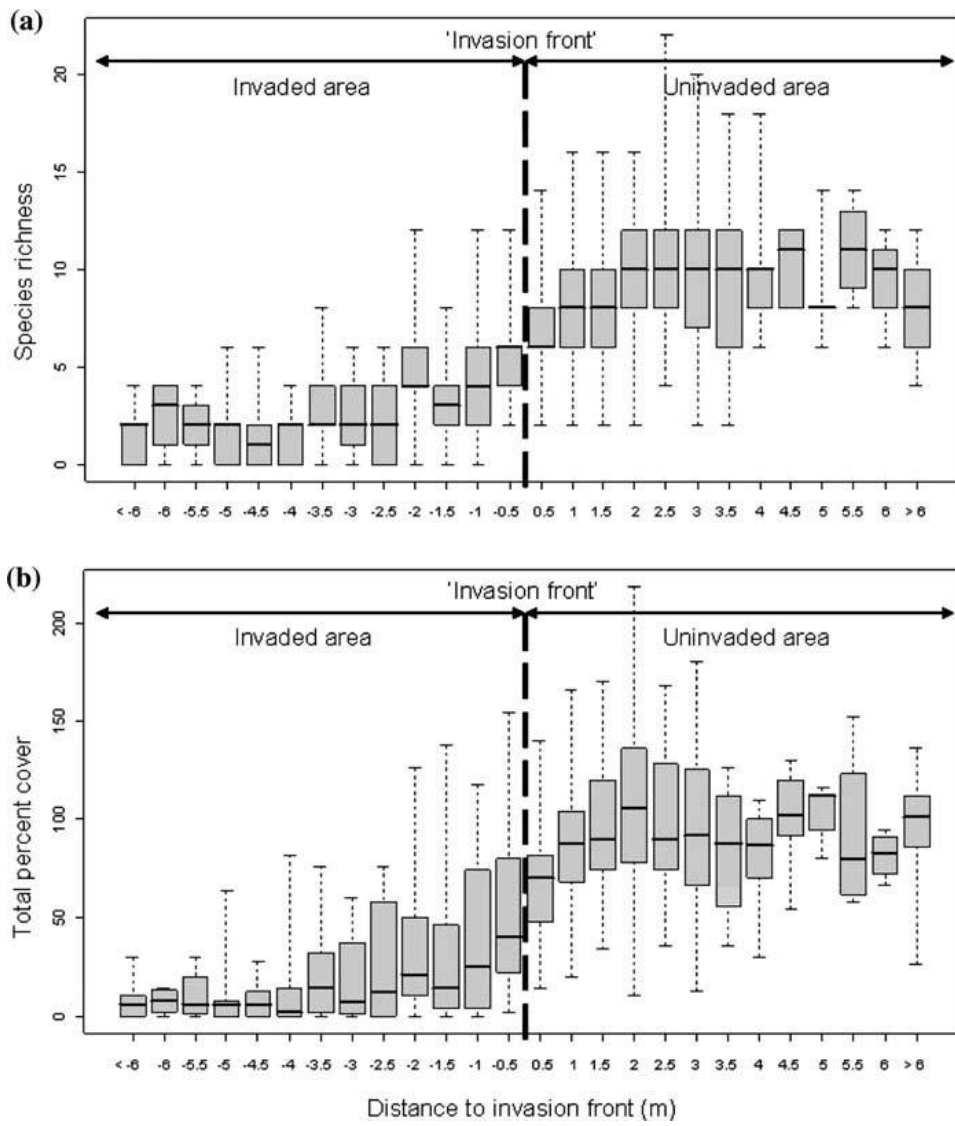


Fig. 3

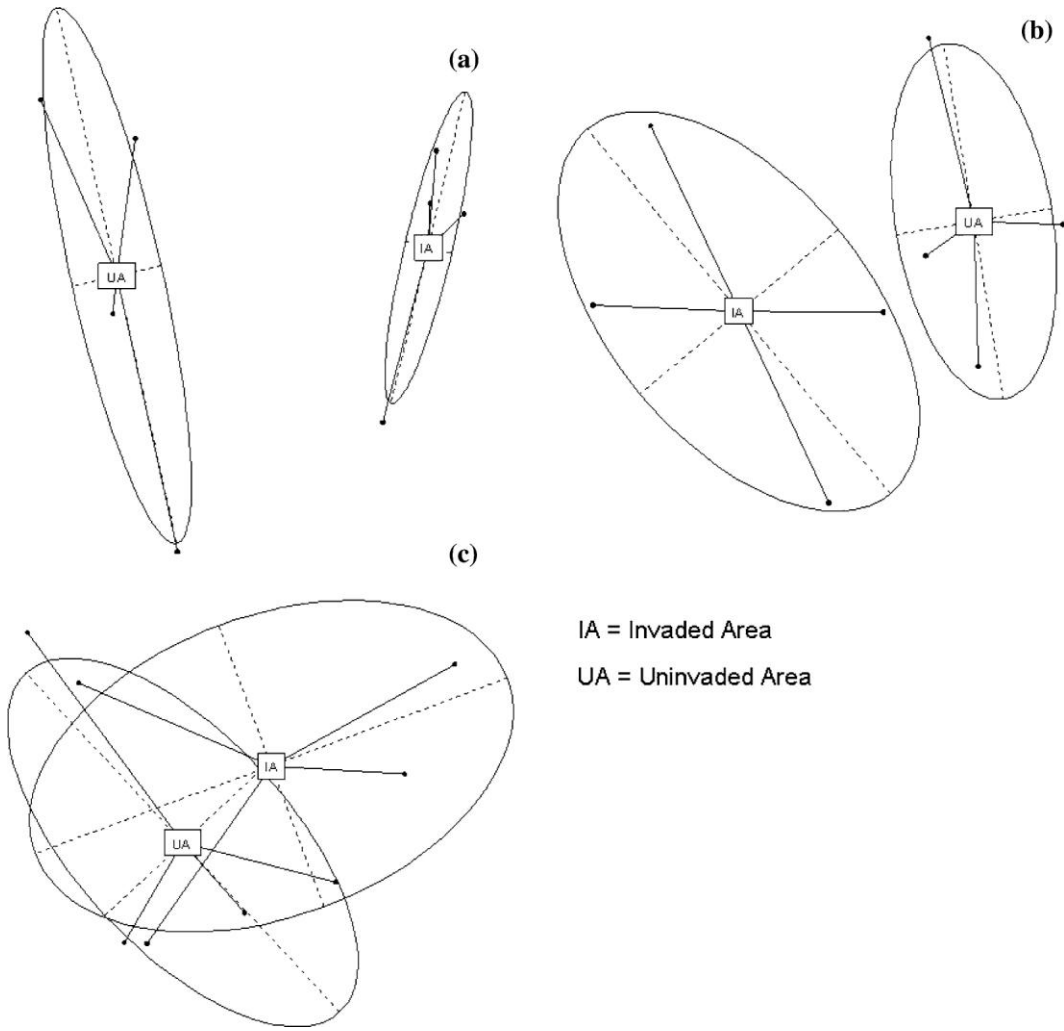


Fig. 4