

Can a large metropolis sustain complex herpetofauna communities? An analysis of the suitability of green space fragments in Rome

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Abstract

Urban areas are primary causes of species' range fragmentation and reduction. However, relatively few studies have attempted to describe the habitat variables influencing the diversity and conservation of amphibians and reptiles, particularly in Mediterranean Europe and in large metropolitan areas. We explored this broad conservation ecology problem by studying the richness and diversity patterns in relation to a suite of six independent habitat variables in Rome, one of the most ancient cities of the world. We considered all the green remnant areas ($n = 62$) of Rome, ranging 1 to >1000 ha in size, which are interspersed within a sea of urbanized matrix. A total of 10 amphibian and 15 reptile species were studied. Their presence/absence patterns were assessed and the effects of the various habitat variables on each species were predicted by a logistic regression model. A total of 1261 presence records (404 amphibians and 857 reptiles) were analysed. Fragment size and wood size within each fragment did correlate significantly with the species richness of both amphibians and reptiles, and there was a clear threshold effect after 50 ha of wooded surface. The presence of water bodies positively affected the species distribution. One amphibian and three reptiles inhabited exclusively fragments > 50 ha. The distance from the centre did not affect fragment species richness. The presence of most species of both amphibians and reptiles was positively influenced by the irregular versus circular shape of the wooded area. The legal protection of a given area did not influence the observed patterns but the total number of sheltered species. Overall, our study suggests that, in order to maintain the current diversity and population viability, it is necessary, in addition to water bodies' maintenance, to (1) preserve the wooded landscapes over 50 ha; (2) promote irregularly shaped increases in the wood surface; (3) maintain ecotonal boundaries.

Introduction

Although urbanized area accounts for only 1–6% of the total Earth surface (Meyer & Turner II, 1992), urban centres (towns and cities) may have a considerable impact on local and global ecosystems by fragmenting, isolating and altering natural habitats (Marzluff, 2001) and by modifying microclimate, energy flow and nutrient cycling (McDonnell & Pickett, 1990; Medley, Pickett & McDonnell, 1995; McDonnell *et al.*, 1997; Vitousek *et al.*, 1997; Grimm *et al.*, 2000). A direct consequence of the changes in ecological conditions associated with urbanization is the simplification and homogenization of community species richness (Blair, 2001). Moreover, the urbanization impact on natural areas may be extended well beyond the human settlements.

In Europe, over 70% of the population is concentrated within urban areas, with the level of urbanization steadily

increasing and with the risk that several natural areas surrounding cities may soon be exploited to give way to new buildings and residential areas (World Resources Institute, 1996; EEA, 1999). In this scenario, the conservation and restoration of remnant fragments within urban habitats is critical for a sustainable cohabitation between human beings and natural resources.

In general, agricultural and urban land covers have been shown to be negative or neutral for invertebrate species richness (i.e. beetles and spiders – Alaruiikka *et al.*, 2002; Deichsel, 2006; Sadler *et al.*, 2006; ants – Yamaguchi, 2004), and to negatively affect vertebrate species richness (for mammals and birds – Sauvajot *et al.*, 1998; Rottenborn, 1999; Parody, Cuthbert & Decker, 2001; Tikhonova, Tikhonov & Bogolomov, 2006; Wenguang *et al.*, 2008). Amphibians and reptiles have already been investigated as for their patterns of species occurrence and abundance in agricultural

areas (Beja & Alcazar, 2003; Fabricius, Burger & Hockey, 2003; Porej, Micacchion & Hetherington, 2004; Denoël & Ficotola, 2007; Piha *et al.*, 2007; Schutz & Driscoll, 2008), whereas such studies are almost neglected in urban habitats (Gagné & Fahrig, 2007). However, there is evidence that, at least in tropical regions, urban reptilian communities may still be rich (up to 17 sympatric species) despite being subject to strong anthropologic pressure and higher than normal mortality rates (Akani *et al.*, 2002, 2008). Overall, a negative influence of human disturbance (e.g. land cover use modification and habitat fragmentation) has been widely confirmed on herpetofauna species apart from a few lizards (Richter & Azous, 1995; Bonin *et al.*, 1997; Knutson *et al.*, 1999; Lehtinen, Galatowitsch & Tester, 1999; Aauri & de Lucio, 2001; Jellinek, Driscoll & Kirkpatrick, 2004; Rubbo & Kiesecker, 2005; Parris, 2006). To be more precise, an increasing proportion of urban territory surrounding remnant green areas may have negative effects on herpetofauna communities in terms of species abundance and composition (e.g. Mensing, Galatowitsch & Tester, 1998; Knutson *et al.*, 1999). However, recent works reported both neutral and negative effects of urban fragmentation on occurrence and abundance patterns in amphibians (i.e. Riley *et al.*, 2005) and reptiles (i.e. Kjos & Litvaitis, 2001; Jellinek *et al.*, 2004). With respect to agriculture, negative effects were reported on total amphibian abundance (Mensing *et al.*, 1998), and both positive and negative effects on the occurrence and abundance of individual species and guilds (Knutson *et al.*, 1999; Joly *et al.*, 2001; Trenham *et al.*, 2003; Gray, Smith & Brenes, 2004; Weyrauch & Grubb Jr, 2004). Moreover, patch boundaries and shape effect on herpetofaunal species movement is a topic that has almost been neglected in most previous studies (e.g. Wiens, Crawford & Gosz, 1985; Wiens, Schooley & Weeks, 1997; Collinge & Palmer, 2002). In urban landscapes, the boundaries between remnant natural patches and matrix, defining patch shape, are generally abrupt because the contrast between cement (roads and buildings) and green areas is not gradual. In such a landscape, animal movements are typically within-fragment rather than between-fragment constrained by the surrounding matrix (Collinge, 1996; Collinge & Palmer, 2002).

Large metropolitan areas may represent a peak of urbanization and therefore can be used as appropriate models to analyse the effect of habitat fragmentation in its extreme manifestation. Large cities and towns may include within their boundaries remnant green patches of variable size, including for example, natural reserves, urban parks and green areas for human recreation. These green areas may mitigate the effects of fragmentation and habitat loss for herpetofauna species. The Municipality of Rome (128 500 ha – Central Italy) is among the European capitals with the higher proportion of green areas (W. Veltroni, *in verbis*), with over 60% of its territory being devoted to green spaces (public gardens, woods included), with half of them consisting of protected areas (Blasi *et al.*, 2001; Celesti Grapow *et al.*, 2006). Most of these green spaces consist of remnant fragments from once larger natural areas that were

pre-existent to urbanization or villa's estate of late Roman nobility (Celesti Grapow, 1995; Bologna *et al.*, 2003).

Our aims in this study are to describe and interpret the occurrence patterns of amphibians and reptiles from remnant green spaces and the role of urban parks and reserves in mitigating fragmentation effects in Rome, a territory where urbanization (and consequently the human impact on natural resources) has been operating for over 2000 years. The presence/absence patterns of each species are analysed through logistic regression modelling techniques in order to link the species-specific relationships with the ecological characteristics of each green-space fragment and its degree of isolation. More specifically, based on previous studies performed in human-dominated habitats on herpetofauna and other taxa (e.g. Kjos & Litvaitis, 2001; Collinge & Palmer, 2002; Jellinek *et al.*, 2004; Rubbo & Kiesecker, 2005), we predict that: (1) in general, in an urban habitat, fragment vegetation structure (surface and shape of the wooded area) should be the driving parameter determining herpetofauna species richness independently of the isolation degree or the relative position of the target green area to the city centre; (2) the species richness of large predators (i.e. snakes) will be correlated to fragment size, vegetation structure complexity (presence of structured wood) but inversely to circularity of its wooded area shape and fragment distance from the city periphery (Kjos & Litvaitis, 2001); (3) secondary consumer species richness (potential prey of snakes, i.e. lacertids) will be independent from the above parameters and positively affected by prediction (2) (mesopredator release; Crooks & Soulé, 1999).

Materials and methods

Study area

Rome metropolis is inhabited by about 2.6 million people, with a mean population density of 1.860 inhabitants km⁻². The municipality of Rome extends over *c.* 129 000 ha, with 82 000 ha consisting of agricultural and non-urbanized (natural protected areas, river banks and untilled lands) areas. A ring highway [named 'Grande Raccordo Anulare' (GRA)] encircles the core of the city and includes 36 000 ha, less than half of which is urbanized. There are 16 protected green areas within Rome territory, for a total of about 14 000 ha. Nine of these areas are located inside the GRA. In addition, several green areas, mostly represented by historical estates or public gardens, are distributed on all sectors of the city. A few parks and gardens represent relicts of the late 'Campagna romana' (the traditional landscape composed by a mosaic of agricultural and woodland patches) and can be approximated to 'ecological islands', and some larger parks extend well outside the city, being connected in various ways to the surrounding 'Campagna romana'. Such a remarkable extension of relict green areas still supports the presence of a rich biodiversity, including more than 5150 species of insects and almost 200 of vertebrates inside the GRA, out of which some are endangered and listed in the appendices of European Conventions devoted to the protection of biodiversity

(Zapparoli, 1997, 2000; Bologna *et al.*, 1999, 2003; Bologna, Calvario & Sarrocco, 2001).

Protocol

In the study area, we selected four wedge-shaped sample plots (sectors) placed along four cardinal points (NW, NE, SE and SW), and tapered towards the centre of the city (Fig. 1). These four wedges were selected because each of them was representative of a different phytoclimate (Blasi, 1994). A wedge-shaped sub-sample was used because it allowed to analyse a full range of green areas situated from the inner centre to the external periphery of the city for that specific phytoclimate. This procedure allowed us to investigate the

whole range of urbanization degrees in the city through representative samples. Within each sector, we determined all the remnant green-space fragments and the surrounding urbanized areas by means of a GIS project, using CORINE (COoRdination of INformation on the Environment) Land Cover, a European Commission programme mainly devoted to assess the state of individual environments and the geographical distribution and state of natural areas (2006; four levels; working scale 1:25 000) (Ficetola *et al.*, 2004; Vos *et al.*, 2008). We considered as green-space fragments all the areas constituting polygons belonging to a sector and comprising predominantly the following CORINE Land Cover categories: (a) 1.4.1. green urban areas, areas with vegetation within the urban fabric, including parks, cemeteries

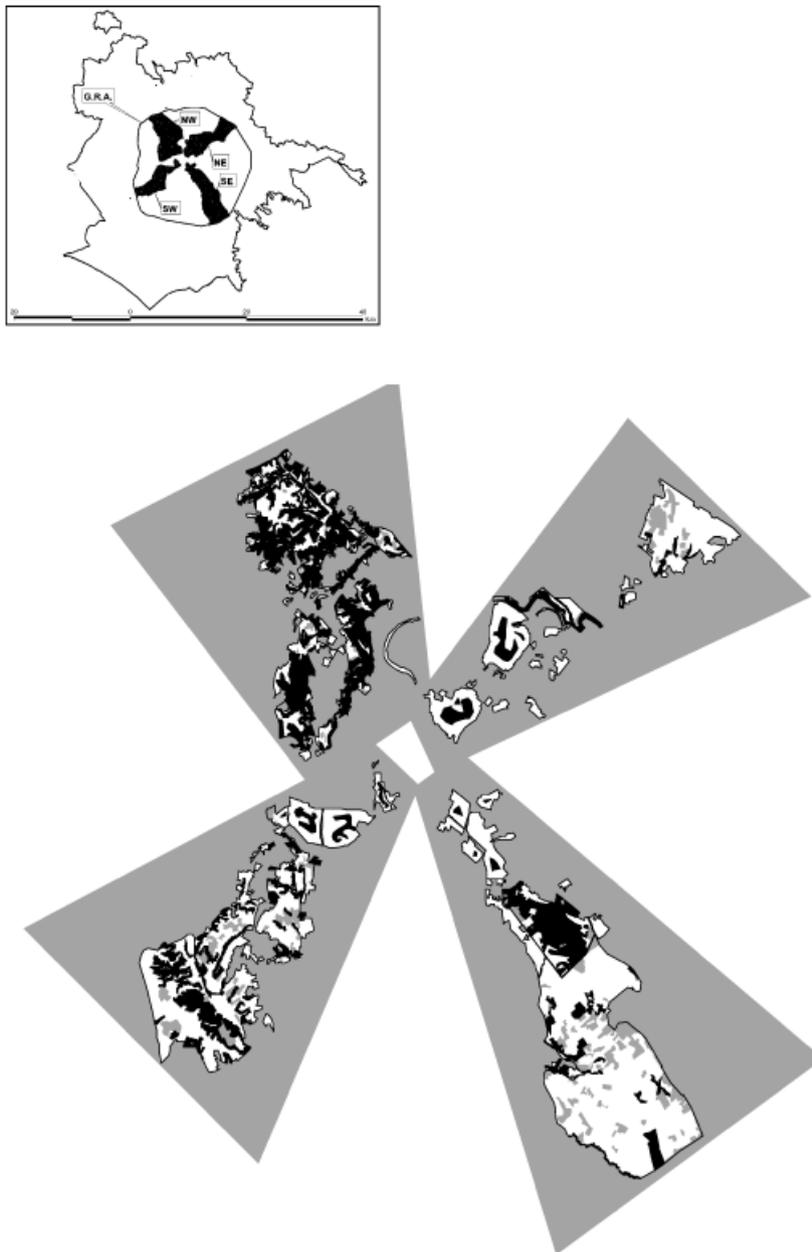


Figure 1 Small box: GIS-built map of the study area showing Municipality of Rome's borders and point counts' distribution within the four sectors (SW, SE, NE and NW; grey shadows) into the city area inside the Grande Raccordo Anulare ring highway. Rest of the figure: detail of remnant fragments (white patches) distribution within the four sectors; grey spaces indicate cemented urbanized areas (roads and buildings) surrounding remnant natural patches; white spaces indicate agricultural areas; black spaces indicate wooded areas.

with vegetation and mansions and their ground; (b) 2.4. heterogeneous agricultural areas; (c) 3. woods and semi-natural areas; (d) 4. wetlands. In the study area, CORINE Land Cover category 3 includes different wood typologies (i.e. oak, pine and chestnut forests, Mediterranean maquis). A total of 62 green-space polygons ranging from about 1 to > 1000 ha were individuated in the four study sectors.

We calculated the proportion of land cover categories covering each fragment surface (PLC). Given that fragments in the urban habitat were usually highly heterogeneous in terms of land cover composition (for instance, some areas had native and exotic woodlands and other areas had native-only woodlands, etc.), we quantitatively estimated PLC by condensing land cover information in woodland/urban (W/U) and woodland/agricultural (W/A) surface ratios. These ratios are the expression of the two main causes of fragmentation of remnant natural habitats (Harris & Silva-Lopez, 1992). Following the definition given by Semlitsch & Bodie (2003) for the terrestrial core habitat of a population, we created a 200 m buffer-zone surrounding each fragment that could well represent the potential edge effect due to the penetration of a mixed urban–agricultural matrix on a fragment interior area. As for the area covered by each green-space polygon, the same PLC was evaluated for the buffer area. The PLC within each fragment was not significantly related to the one in the surrounding matrix (W/U: $R = -0.097$, $P = 0.454$; W/A: $R = 0.179$, $P = 0.164$; Spearman's rank correlation). The PLC of fragments and buffer areas together provided a quantitative measure of the urbanization degree for the area comprising remnant natural habitats and for the area surrounding each fragment. Because the W/U and W/A ratios were significantly correlated within both the fragment and the matrix (fragment: $R = 0.306$, $P < 0.05$; matrix: $R = 0.507$, $P < 0.01$), we used only the W/U ratio as a representative measure of the whole fragmentation trend in the analysis.

The following structural and habitat variables were measured for each remnant fragment (Appendix S1):

- (1) fragment size (measured in hectare) (FRAG);
- (2) fragment shape, estimated by applying a shape index (Cs) ranging from 0 to 1. This index is calculated as follows:

$$\frac{(A/P)_x}{(A/P)_c}$$

where $(A/P)_x$ is the surface/perimeter ratio of the fragment x and $(A/P)_c$ is the surface/perimeter ratio of the circle of equal surface to fragment x . Hence, the more the fragment surface approximates that of a circle, the more the index tends to 1;

- (3) size of the wooded areas inside the fragment (WOOD);
- (4) shape of the wooded areas estimated using the Cs index as explained for the fragment (CsWOOD); functionally, this index is an indirect estimator of the ecotonal quantitative development. These ecotones are the interface connecting woodland habitat to open habitats (e.g. agricultural lands, pastures and artificial green habitats). These ecotonal areas are of particular importance for the biology of several

species for dispersion, feeding and thermoregulation activities (e.g. Luiselli & Capizzi, 1997; Rugiero & Luiselli, 2006);

(5) fragment land cover composition (PLC), calculated as the proportion of urban and agricultural land use extension in respect of woodland surface (W/A);

(6) water presence, we recorded in each fragment the presence of water bodies suitable for amphibians (but also for reptiles) in terms of hydroperiod (maintenance of water during most of the year), pollution (no pollution) and the presence of potential predators (absence or limited presence of fishes, crabs and other invertebrate predators);

(7) fragment isolation (CONN), measured as the average distance between the fragment and the surrounding fragments within a radius of 500 m [distance close to the maximum dispersal observed in the most vagile and large-sized species (colubrid snakes) in Central Italy. Ciofi & Chelazzi, 1991b, 1994].

Fragment size and presence of water (natural and artificial basins) were highly correlated with some other structural variables (Appendix S2). Because of this statistical correlation, we avoided introducing these variables in the logistic models, even though the importance of these parameters is acknowledged by other analyses (i.e. Mann–Whitney U -test for testing the effect of the presence/absence of water on the distribution of herpetofaunal species), in order to avoid statistical biases due to autocorrelation among multiple variables.

Field surveys were conducted from January 2004 to December 2006, and the various study sites were selected on the basis of an early dataset (1990–2003; Bologna, Capula & Carpaneto, 2000; Bologna *et al.*, 2003), including presence-only data for both amphibians and reptiles. In 2004–2006, we surveyed each green-area fragment from two times (fragments < 10 ha) to four times per year (fragments > 10 ha) during the reproduction period searching for animals throughout the whole fragment area, using the Visual Encounter Survey technique (Heyer *et al.*, 1994). We assumed that a given species was truly absent when it never appeared during the survey in the whole period (2004–2006). These studies revealed the occurrence within the study area of 10 species of amphibians (*Salamandrina perspicillata*, *Triturus carnifex*, *Lissotriton vulgaris*, *Bufo bufo*, *Bufo lineatus*, *Hyla intermedia*, *Rana dalmatina*, *Rana italica*, *Pelophylax lessonae* and *Pelophylax klepton esculentus*) and 15 of reptiles (*Testudo hermanni*, *Emys orbicularis*, *Hemidactylus turcicus*, *Tarentola mauritanica*, *Anguis fragilis*, *Chalcides chalcides*, *Lacerta bilineata*, *Podarcis muralis*, *Podarcis sicula*, *Natrix natrix*, *Natrix tessellata*, *Hierophis viridiflavus*, *Elaphe quatuorlineata*, *Zamenis longissimus* and *Vipera aspis*). Alien species (e.g. *Trachemys scripta*) were not analysed even if present. Green frog species (*Pe. lessonae* and *Pe. klepton esculentus*) were analysed together as a single unit because of their extreme morphological similarity, which prevented identification in the field.

For all species, we limited the analyses to presence/absence data in order to delineate the basic and general patterns of fragment occupancy by species. Given the high level of knowledge cumulated for each study area over the 16 years of study, we are led to think that absences really

represent true absences, and not 'false' absences due to inappropriate field research.

Statistical analyses

In order to determine the occurrence of an urbanization gradient affecting species richness from the centre to the peripheral areas of the city, the species richness (number of species) of each target fragment was correlated with fragment size, woodland size inside the fragment and fragment distance from the centre of the city (point of maximum degree of urbanization, urbanized land cover in the inner centre: 78%; average urbanized land cover in the periphery: 23%; GIS elaboration by the authors) by Spearman's rank correlation coefficient. The univariate GLM procedure was applied to top predators (snakes) and their prey (lacetids, although they are also secondary consumers) species abundance (dependent variables), using the size and the shape of the wooded areas inside each fragment (respectively, WOOD and CsWOOD) and the distance from the city centre (GRAD) as fixed factors, all of them adjusted for the size of the fragment (FRAG used as a covariate).

To assess which environmental variable influenced the presence/absence of the various species, we applied a logistic regression design (forward stepwise conditional design), with the model fit assessed by a χ^2 -test. The presence/absence of each species was entered as a dependent variable in the logistic regression model, and the various structural variables listed above were entered as independent variables.

Statistical analyses, all tests being two tailed and with α set at 5%, were performed by Statistica (StatSoft Inc., 2001 – Version 6).

Results

Species distribution and their relationships with fragments

A total of 1261 presence records (404 for amphibians and 857 for reptiles) were detected within the 62 residual natural areas in the study area (Table 1).

Water presence positively influenced the distribution of both amphibians ($U = 99$; $N_0 = 38$, $N_1 = 24$; $P < 0.00001$) and reptiles ($U = 249$; $N_0 = 38$, $N_1 = 24$; $P < 0.003$). Fragment size (FRAG) and wood size within each fragment (WOOD) were highly correlated with the number of species for both amphibians ($R_{\text{FRAG}} = 0.773$, $R_{\text{WOOD}} = 0.747$, $n = 62$; for both tests $P < 0.001$; Spearman's correlation) and reptiles ($R_{\text{FRAG}} = 0.853$, $R_{\text{WOOD}} = 0.763$; $P < 0.001$). The observed relationship for both amphibians and reptiles showed a clear visual threshold effect after 50 ha of wooded size (Fig. 2). Only one amphibian (*S. perspicillata*) and three reptiles (*Te. hermanni*, *El. quatuorlineata* and *V. aspis*) inhabited exclusively fragments > 50 ha. The distance from the centre did not affect fragment species richness (amphibians: $R = 0.004$, $P = 0.977$; reptiles: $R = -0.138$, $P = 0.284$).

The slopes of the general regression between fragment size and number of species, amphibians and reptiles pooled

Table 1 Number of presence records and number of colonized green areas per species for amphibians and reptiles in the study area

Species	Number of records	Number of occupied green areas (protected)	Presence only in green areas > 50 ha
Amphibians			
<i>Bufo bufo</i>	163	23 (14)	
<i>Bufo viridis</i>	40	11 (10)	
<i>Hyla intermedia</i>	23	7 (7)	
<i>Pelophylax</i> skl.	102	21 (13)	
<i>esculentus</i> ^a			
<i>Rana dalmatina</i>	6	5 (3)	
<i>Rana italica</i>	23	6 (5)	
<i>Salamandrina terdigitata</i>	5	1 (1)	X
<i>Triturus carnifex</i>	15	5 (4)	
<i>Triturus vulgaris</i>	27	7 (7)	
Total = 9	404		
Reptiles			
<i>Testudo hermanni</i>	5	5 (4)	X
<i>Emys orbicularis</i>	8	3 (3)	
<i>Hemidactylus turcicus</i>	19	22 (10)	
<i>Tarentola mauritanica</i>	54	31 (12)	
<i>Anguis fragilis</i>	14	7 (5)	
<i>Chalcides chalcides</i>	55	12 (11)	
<i>Lacerta bilineata</i>	79	13 (11)	
<i>Podarcis muralis</i>	221	35 (14)	
<i>Podarcis sicula</i>	248	30 (13)	
<i>Elaphe quatuorlineata</i>	7	3 (2)	X
<i>Zamenis longissimus</i>	29	14 (11)	
<i>Hierophis viridiflavus</i>	69	18 (12)	
<i>Natrix natrix</i>	30	10 (9)	
<i>Natrix tessellata</i>	1	1 (1)	
<i>Vipera aspis</i>	18	6 (5)	X
Total = 15	857		

^aGreen frog species are considered as a single unit as explained in the text.

together, for protected versus non-protected areas (respectively, 0.0105 and 0.0208), were not significantly different (ANCOVA: $F_{1,58} = 0.002$, $P = 0.966$), whereas Y -intercepts (respectively, 7.343 and 1.692) did differ significantly (ANCOVA: $F_{1,59} = 17.334$, $P < 0.0001$) (Fig. 3), thus showing that protection of a given area did not influence *per se* the relationship considered but the total number of sheltered species. The same type of relationships also occurred when considering amphibians (ANCOVA: slopes – $F_{1,58} = 0.304$, $P = 0.583$; Y -intercept – $F_{1,59} = 14.572$, $P < 0.001$) and reptiles separately (ANCOVA: slopes – $F_{1,58} = 0.107$, $P = 0.774$; Y -intercept – $F_{1,59} = 16.421$, $P < 0.001$).

Although WOOD and CsWOOD were significantly correlated to each other (Appendix S2), we introduced both variables in the GLM models because we wanted to test whether there is an interaction effect between the two above-mentioned variables. Top predator species (snakes) abundance distribution was not influenced by single fixed factors, whereas it was significantly conditioned by interactions among the fixed factors, representing constraints to

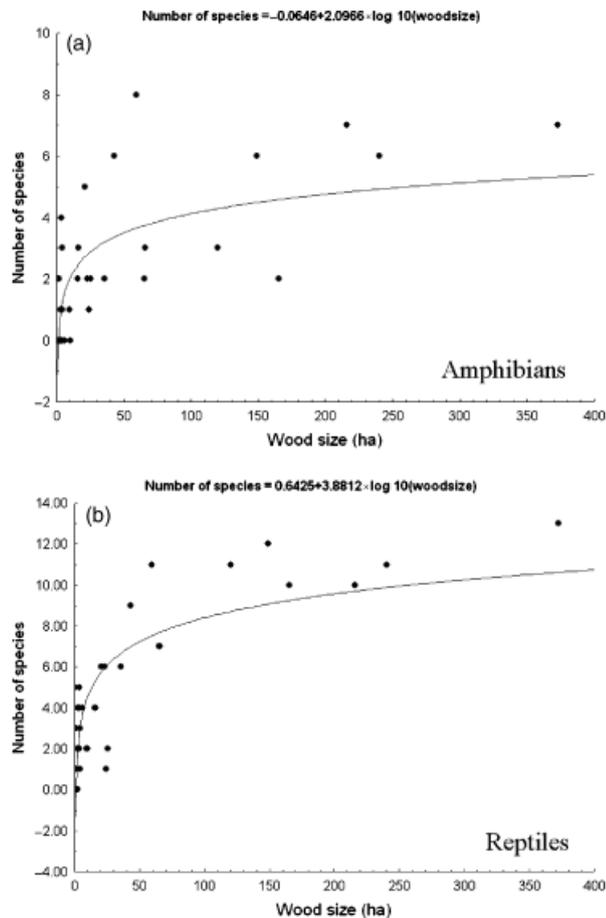


Figure 2 Threshold effect of wooded area surface on species richness for amphibians (a) and reptiles (b) in the study fragments. In both plots, a threshold effect around 50 ha is observed.

predator species distribution pattern (GLM procedure). In detail, snake species abundance was influenced by the interaction between GRAD and WOOD and among all the fixed factors ($WOOD \times GRAD \times CsWOOD$) (Table 2), that is, (1) the size of the wooded area influenced snake abundance only when the fragment is on average (mean among the four sectors) at least 7.25 ± 2.5 km from the city centre; (2) the shape of the wood is crucial (the less circular the better) when its surface reaches a size threshold and the fragment is located in peripheral areas of the city. Secondary consumer species (lacertids) abundance distribution among fragments was not constrained by any of the considered variables or by their interactions (GLM procedure), that is, the lizard species abundance distribution merely followed a positive relationship with the remnant fragment size (Table 2).

Species occurrence patterns

Regarding the logistic regression analysis, four species were excluded because of their low rate of occurrence in the residual natural fragments: *S. perspicillata* ($n = 1$) among amphibians, and *El. quatuorlineata* ($n = 3$), *Em. orbicularis*

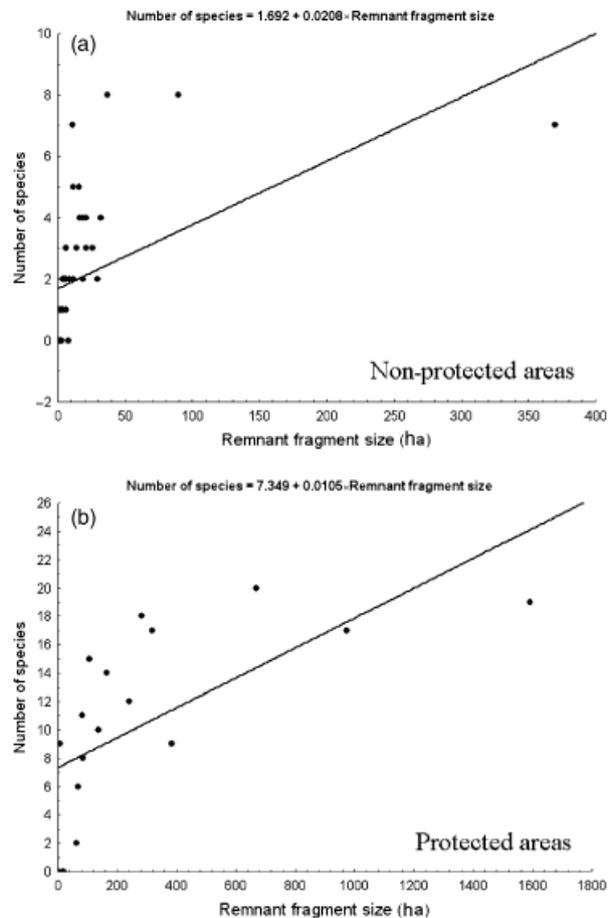


Figure 3 Analysis of covariance on non-protected (a) versus protected (b) areas revealed a significant difference in intercepts but not in slope coefficients, that is, protected areas sheltered more species than non-protected ones but following the same relationship species richness–area size.

($n = 3$) and *N. tessellata* ($n = 1$) among reptiles. Logistic regression analysis gave significant models for all the amphibian species studied and for most of the reptiles, except for the two wall lizards (*Po. muralis* and *Po. sicula*) and the two geckos (*He. turcicus* and *Ta. mauritanica*). The ecological equations for all species are listed in Table 3. Most species of both amphibians and reptiles were negatively linked to the shape index of the wooded area, whereas the distributions of two amphibian species were positively influenced exclusively by the distance from the centre of the town. There were no more than two independent variables influencing significantly the presence/absence of both amphibians and reptiles (Table 3). However, the presence/absence of most amphibians (seven out of eight) was affected by single variables, whereas the presence/absence of the majority of reptiles (five out of eight) was influenced by a combination of two different variables.

Closer inspection of the logistic regression equations (Table 3) revealed that a few variables did matter: CsW affected negatively the probability of being present in a

Table 2 Ecological equations and percentage of cases correctly classified by a logistic regression model (forward stepwise conditional design) for each species of amphibian and reptile in the study area

Species	Logit equation	Overall percentage correct
Amphibians		
<i>Bufo bufo</i>	$Y = 0.942 - 1.126 \times \text{CsW}$	74.2
<i>Bufo lineatus</i>	$Y = 0.602 - 1.069 \times \text{CsW}$	82.3
<i>Hyla intermedia</i>	$Y = 2.329 - 2.715 \times \text{CsW}$	88.7
<i>Rana dalmatina</i>	$Y = -5.510 + 0.903 \times \text{GRAD}$	93.5
<i>Rana italica</i>	$Y = 1.241 - 2.217 \times \text{CsW}$	91.9
<i>Rana sk. hispanica</i>	$Y = -1.769 - 1.195 \times \text{CsW}$	72.6
<i>Triturus carnifex</i>	$Y = -4.424 + 0.677 \times \text{GRAD}$	91.9
<i>Lissotriton vulgaris</i>	$Y = -0.087 - 1.110 \times \text{CsW}$	88.7
Reptiles		
<i>Testudo hermanni</i>	$Y = -8.511 + 1.629 \times \text{CONN}$	93.5
<i>Anguis fragilis</i>	$Y = -0.114 - 1.244 \times \text{CsW}$	90.3
<i>Chalcides chalcides</i>	$Y = 2.465 - 2.242 \times \text{CsW}$	83.9
<i>Lacerta bilineata</i>	$Y = -1.197 - 1.752 \times \text{CsW}$	85.5
<i>Hierophis viridiflavus</i>	$Y = -1.742 - 1.066 \times \text{CsW} + 0.937 \times \text{CONN}$	75.8
<i>Natrix natrix</i>	$Y = 0.553 - 1.198 \times \text{CsW}$	87.1
<i>Zamenis longissimus</i>	$Y = -1.772 - 1.609 \times \text{CsW}$	77.4
<i>Vipera aspis</i>	$Y = 0.232 - 1.467 \times \text{CsW}$	91.9

given fragment for most amphibians and reptiles (14 out of 16 significant models had this variable), that is, the more circular the fragment shape, the less likely the presence of most amphibian and reptile species. GRAD was positively correlated to the presence of two amphibian species (*R. dalmatina*, *Tr. carnifex*), that is, the larger the fragment distance from the city centre, the more likely their presence. With regard to reptiles, W/A influenced positively the presence of three species (*C. chalcides*, *La. bilineata* and *N. natrix*), and the proximity to other patches was positively correlated to the likelihood of presence for two other species (*Hi. viridiflavus* and *Te. hermanni*). If we only consider the equations that correctly classified at least 90% of the cases (sensu Arntzen & Alexandrino, 2004; Luiselli, 2006), it was found that only three species of amphibians (*R. dalmatina*, *R. italica* and *Tr. carnifex*) and three species of reptiles (*A. fragilis*, *Te. hermanni* and *V. aspis*) satisfied this criterion (Table 3), all these species being ecologically specialist.

Discussion

One of the main results of our study is that water presence strongly conditioned the presence/absence of both amphibians and reptiles. If this was entirely predictable for amphibians, given their dependence on local supply of water for reproduction and larval development to take place also at a large spatial scale (Qian et al., 2007; Soares & Brito, 2007), this was less obvious for reptiles. In our case, the dependence of reptiles on water may be linked to the correlation between water presence and wood size, and also with other main structural variables (for instance, fragment size; see Appendix S2). As it is evidenced that the presence of water is important

Table 3 Analysis of covariance (GLM) of the effects of three fixed factors (fragment variables) on predator and prey species richness in the remnant fragments in the study area

Source	Type III sum of squares	d.f.	Mean square	F	P
Predators (snakes)					
Corrected model	86.713	28	3.097	5.334	0
Intercept	5.776	1	5.776	9.949	0.003
FRAG ^a	8.8756	1	8.875	15.287	0.001
WOOD ^b	1.5076	2	0.754	1.298	0.287
CsWOOD ^b	3.0656	3	1.022	1.760	0.174
GRAD ^b	0.9176	3	0.306	0.527	0.667
WOOD × CsWOOD	2.775	4	0.694	1.195	0.331
WOOD × GRAD	8.867	6	1.478	2.545	0.039
CsWOOD × GRAD	8.941	7	1.277	2.200	0.060
WOOD × CsWOOD × GRAD	3.959	1	3.959	6.820	0.013
Error	19.158	33	0.581		
Total	140	62			
Corrected total	105.871	61			
Prey (lacertids)					
Corrected model	137.015	28	4.893	11.161	0
Intercept	1.348	1	1.348	3.074	0.089
FRAG ^a	6.557	1	6.557	14.956	0.001
WOOD ^b	0.265	2	0.133	0.303	0.741
CsWOOD ^b	1.213	3	0.404	0.923	0.441
GRAD ^b	0.741	3	0.247	0.563	0.643
WOOD × CsWOOD	2.810	4	0.703	1.602	0.197
WOOD × GRAD	3.279	6	0.547	1.247	0.308
CsWOOD × GRAD	5.303	7	0.758	1.728	0.137
WOOD × CsWOOD × GRAD	0.062	1	0.063	0.143	0.708
Error	14.468	33	0.438		
Total	288	62			
Corrected total	151.484	61			

^aCovariate.

^bFixed factor.

For abbreviation see 'Materials and methods'.

for Roman herpetofauna, conserving and managing the few water bodies (both natural and artificial) still available in Rome metropolis should be a priority for the years to come.

Our study also revealed, by means of some sets of analyses (linear and logistic regressions), that in Rome the herpetofauna richness and diversity are strongly influenced by the occurrence of large remnant woods (> 50 ha) and an irregular shape allowing for extended ecotonal habitats. This general pattern is observed in the whole study patch system for both amphibians and reptiles independently on: (1) the heterogeneity of the wooded area; (2) the isolation degree; (3) the position of the green areas relative to the city centre. We also evidenced that the shape and size of the wooded area and the periphericity of the fragment are particularly relevant independent of its size for high-level predators, but not for second-level predators. For instance, snakes, besides *Hi. viridiflavus*, are only found in the peripheral green areas with large and irregular-shaped woods, whereas lizards and geckos are not influenced by

fragment wooded area or position from the city centre. In addition, it is noteworthy that, among the four species that were found only in patches > 50 ha, three (*S. perspicillata*, *Te. hermanni* and *El. quatuorlineata*) are listed in Appendix S2 of 'Habitat' European directive, and are hence of particular conservation concern.

The lack of significance of the logistic regression equations for the two wall lizards (*Podarcis*) and the two geckos was likely due to their generalist habits and widespread occurrence.

The species richness of both amphibians and reptiles is consistent with one of the two main predictions of the island biogeography theory (IBT; MacArthur & Wilson, 1967), that is, the island surface (= size of the remnant fragments in our study case). Overall, despite the above-mentioned congruencies of the observed patterns with IBT, the fact that our green areas have a relict origin due to 2000 years of urbanization led us to think that species resilience and habitat carrying capacity properties are the main factors determining the current species distribution patterns. On the other hand, the second IBT prediction (isolation degree should influence the species colonization rate) did not match with our observations. We suggest that this should depend on (1) the almost perfect non-permeability of the urban matrix (apart from the anthropophilic wall lizards and geckos; Luiselli & Capizzi, 1999; Bologna *et al.*, 2003); (2) the reduced vagility of most species (with the exceptions of *Te. hermanni* and *Hi. viridiflavus* (Ciofi & Chelazzi, 1991a,b, 1994). Given that the species distribution in the remnant fragments is not influenced by (1) the isolation degree or (2) the proximity of the natural areas to the city periphery, and consequently to larger natural areas outside urban habitat, we suggested that the presence of species, particularly the largest (i.e. snakes), should be due to the extinction debt effect (Tilman *et al.*, 1994; Hanski & Ovaskainen, 2002). Indeed, because of habitat loss, most of the remnant fragments do not guarantee minimum area and habitat quality requirements for several species, and the urban matrix strongly prevents any dispersal outside the few suitable fragments. Thus, in these cases, threshold conditions for survival may no longer be met. For an effective conservation of Rome herpetofauna we suggest not only to enlarge the wooded size within each green area but also to start specific restoration programmes to introduce new genes into the severely depressed and isolated populations (Madsen *et al.*, 1999). In this regard, it is necessary to stress that caution should be exercised in re-stocking and re-introduction of plans (Ruokonen *et al.*, 2000). However, it should be strongly necessary (1) to actively promote, via wood management actions, increases of irregularly shaped wood surfaces; (2) to maintain ecotonal boundaries.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. List of the study fragments, including the main structural variables. Symbols: ID = fragment code (letter indicates wedge of fragment pertinence). TOT

AMP = number of amphibian species; TOT REP = number of reptile species; FRAG = fragment size (ha); Cs = fragment shape index; WOOD = wooded area size (ha); CsWOOD = wood shape index; CONN = connectivity index; GRAD = fragment distance from city centre; W = presence of water (absence: 0; presence: 1); STATUS = fragment conservation status (non protected: 0; protected: 1).

Appendix S2. Correlation statistics (Spearman Rank Correlation coefficient) among the main structural variables examined in this study (significant correlation in bold; $P < 0.05$). For the symbols see **Appendix S1**.

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