Critical steps to ensure the successful reintroduction of the Eurasian red squirrel

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Abstract

Critical steps to ensure the successful reintroduction of the Eurasian red squirrel.— Wildlife reintroduction strategies aim to establish viable long–term populations, promote conservation awareness and provide economic benefits for local communities. In Portugal, the Eurasian red squirrel (*Sciurus vulgaris*) became extinct in the 16th century and was reintroduced in urban parks in the 1990s, mainly for aesthetic and leisure purposes. We evaluated the success of this reintroduction in two urban parks and here described the critical steps. We assessed habitat use, population density and abundance, and management steps carried out during reintroduction projects. Reintroductions have been successful to some extent given squirrels are present 20 years after release. However, populations in both parks are declining due to the lack of active management and poor quality habitat. Successful reintroduction of Eurasian red squirrel in areas without competition of alien tree squirrels involves three critical main stages. The pre–project stage includes studies on habitat quality, genetic proximity between donors and closest wild population, and health of donor stocks. In the release stage, the number of individuals released will depend on resource variability, and the hard release technique is an effective and economically viable method. Post–release activities should evaluate adaptation, mitigate mortality, monitor the need for supplementary feeding, provide veterinary support, and promote public awareness and education.

Key words: Conservation, Management, Release, Rodentia, *Sciurus vulgaris*, Urban park

Resumen

Pasos fundamentales para garantizar la eficacia de la reintroducción de la ardilla roja.— El objetivo de las estrategias de reintroducción de fauna silvestre es establecer poblaciones viables a largo plazo, fomentar la concienciación con respecto a la conservación y aportar beneficios económicos para las comunidades locales. La ardilla roja (*Sciurus vulgaris*), que estaba extinta en Portugal desde el s. XVI, fue reintroducida en varios parques urbanos en la década de los años 90, principalmente con fines estéticos y recreativos. Evaluamos la eficacia de esta reintroducción en dos parques urbanos y describimos los pasos fundamentales de la misma. Se evaluaron la utilización del hábitat, la densidad y abundancia de la población y las medidas de gestión adoptadas durante los proyectos de reintroducción. Las reintroducciones fueron relativamente eficaces dado a que las ardillas seguían presentes 20 años después de la liberación. No obstante, las poblaciones en ambos parques están disminuyendo debido a la falta de una gestión activa y a la mala calidad del hábitat. La reintroducción eficaz de la ardilla roja en zonas donde no hay ardillas arborícolas exóticas conlleva tres etapas fundamentales. La etapa previa al proyecto comprende estudios sobre la calidad del hábitat; la proximidad genética entre los donantes y la población silvestre más cercana, y la salud de las poblaciones donantes. En la etapa de liberación, el número de individuos liberados dependerá de la variabilidad de los recursos disponibles; asimismo, se ha observado que la técnica de liberación dura es un método eficaz y viable desde el punto de vista económico. Las actividades posteriores a la liberación deberían analizar la adaptación, mitigar la mortalidad, hacer un seguimiento de la necesidad de aportar alimentación complementaria, prestar apoyo veterinario y fomentar la sensibilización pública y la educación.

Palabras clave: Conservación, Gestión, Liberación, Rodentia, *Sciurus vulgaris*, Parque urbano

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Introduction

Animal translocation is an ancient process used by humans to relocate species from one place to another (Griffith et al., 1989; Hodder & Bullock, 1997; Armstrong & Seddon, 2007; Seddon et al., 2007; Ewen et al., 2012). Griffith et al. (1989) defined animal translocation as the intentional release to establish, re–establish or increase the population of a given species. Reintroduction is currently one of the most popular translocation strategies used in the management of species (Armstrong & Seddon, 2007; Seddon et al., 2007; Ewen et al., 2012). Wildlife reintroductions are conducted to establish viable populations, enhance long–term survival of a given species, settle long–term economic benefits for local communities, and to promote conservation awareness (IUCN, 1998). Reintroductions should be carefully planned by a multidisciplinary team, and follow a three–step protocol, focusing on the pre–project activities, release stages and post–released activities (IUCN, 1998). Such projects require complex planning, implementing and monitoring species and habitats according to their biology, socio–economic impact on local communities, and legal requirements (Caughley & Gunn, 1996; IUCN, 1998; Armstrong & Seddon, 2007; Seddon et al., 2007; Ewen et al., 2012; Harrington et al., 2013). Parameters of success change in each project but should follow the principles of long–term survival of species while providing benefits for the local community and fostering conservation awareness (IUCN, 1998, 2012). The potential positive impact of reintroductions depends on temporal, spatial, and taxonomic factors (Ewen et al., 2012), and if reintroductions are not properly carried out they can damage both donor and receptor populations as well as ecosystems (Hodder & Bullock, 1997; Armstrong & Seddon, 2007; Seddon et al., 2007; Ewen et al., 2012). Therefore, publication and dissemination of successful and unsuccessful cases contribute to improve current reintroduction protocols (Armstrong & Seddon, 2007; Seddon et al., 2007; Ewen et al., 2012; IUCN, 2012).

Mammals, together with birds, are the most frequently chosen groups for releases with conservation purposes (Griffith et al., 1989; Seddon et al., 2005, 2007). Although most of reintroductions focus on ungulates and carnivores (Seddon et al., 2005), rodents such as the edible dormouse *Glis glis* in Poland (Jurczyszyn, 2006) and the European ground squirrel *Spermophilus citellus* in Central Europe (Matějů et al., 2010) have also been released in the last 20 years. The most commonly reported reintroductions among rodents are those concerning the Eurasian red squirrel *Sciurus vulgaris* reintroductions, with a considerable number of programmes being implemented in Europe over the last 30 years (Swinnen, 1988; Fornasari et al., 1997; Wauters et al., 1997a, 1997b; Poole & Lawton, 2009).

Although the Eurasian red squirrel is a widespread Palearctic species (Lurz et al., 2005; Shar et al., 2008; Bosch & Lurz, 2012), some of its populations, particularly in the United Kingdom and Italy, are threatened or extinct due to habitat loss, hunting, disease and competition with alien tree squirrels *Sciurus carolinensis*, *Callosciurus erythraeus* and *C. finlaysonii* (Gurnell, 1987; Wood et al., 2007; Bosch & Lurz, 2012; Bertolino & Lurz, 2013). In Portugal, the Eurasian red squirrel became extinct in the 16th century due to significant habitat loss, and it only reappeared in extreme northern areas around the 1980s (Mathias & Gurnell, 1998; Ferreira et al., 2001). One decade later, isolated reintroductions occurred in some urban parks, but no monitoring has been conducted since then to understand population dynamics and their status or to evaluate management and success. In order to determine whether Eurasian red squirrel reintroductions carried out in Jardim Botânico da Universidade de Coimbra and in Parque Biológico de Gaia were successful or not, we estimated population viability through density, abundance, and habitat use in released sites. We also evaluated stepwise reintroduction in both urban parks, based on the IUCN guidelines (IUCN, 1998, 2012) to highlight critical steps and suggest actions ensure the long–term persistence and viability of these Eurasian red squirrel populations.

Material and methods

Study area

The Parque Biológico de Gaia (PBG), which was created in 1983, initially covered 2 ha but has been extended to include 35 ha (Oliveira, 2013). It is situated in Vila Nova de Gaia, northern Portugal (41º 05' N and 8º 33' W; fig. 1A) and it is managed by a municipal company (Oliveira, 2013). This urban park is composed of open areas, a wildlife rehabilitation center and monospecific forests of black alder *Alnus glutinosa*, oak *Quercus robur* or cork oak *Q. suber* (fig. 1A). It also has enclosures distributed throughout open areas in the park containing wildlife that could not be rehabilitated or that provide examples of native and exotic fauna. The aim is to promote environmental education, as the case of the Eurasian red squirrel.

The Jardim Botânico da Universidade de Coimbra (JBUC) was created in 1772 as part of the Museu de História Natural da Universidade de Coimbra (UC, 2013). It covers 13 ha in Coimbra city, central Portugal (40º 12' N and 8º 25' W; fig. 1B). This park comprises mainly gardens and forests of alien flora (fig. 1B).

Eurasian red squirrel survey

From 7th October to 11th November 2013, 15 walking transects of 100 m were established in each study site. All transects were surveyed in the morning and afternoon for seven days to avoid biases from squirrel behavior (Gurnell et al., 2001). Transects were performed one after other to avoid double counting of individuals moving from one transect to another. Transects were selected to include all habitat types and to cover most of the area at each of the two urban parks, but with at least 20 m distance from each

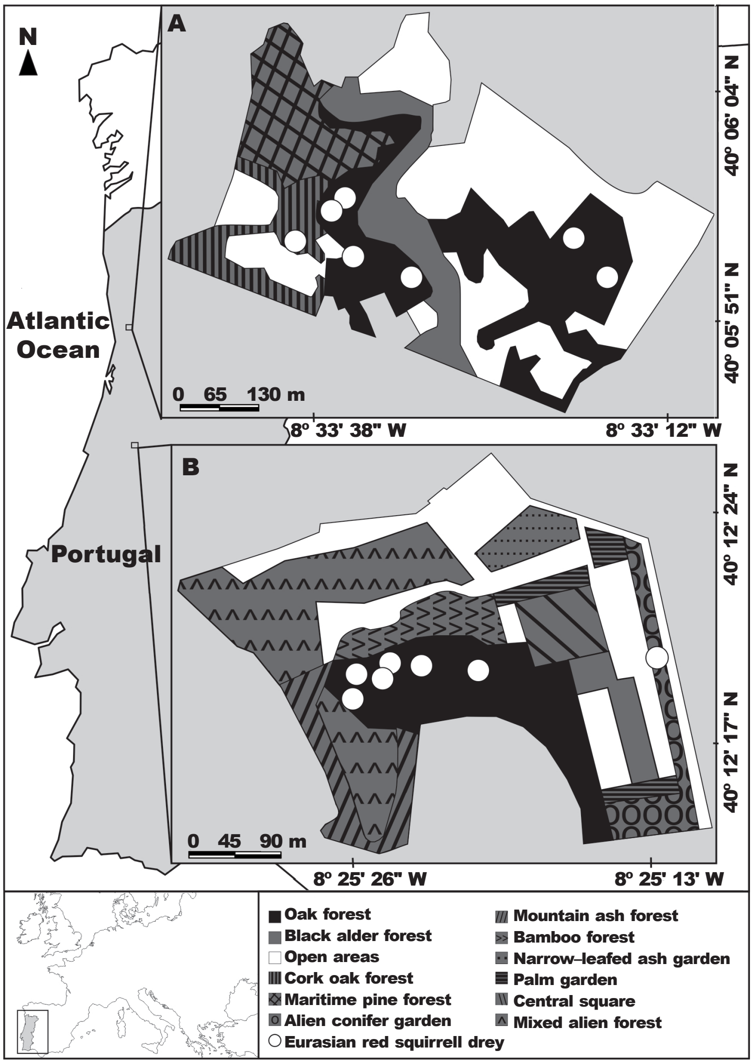

Fig. 1. Study area in northern and central Portugal with inset showing vegetation type and distribution of dreys (white circles) in the Parque Biológico de Gaia at Vila Nova de Gaia (A), and in the Jardim Botânico da Universidade de Coimbra at Coimbra (B).

Fig. 1. Zona de estudio situada en el norte y el centro de Portugal con recuadros que muestran el tipo de vegetación y la distribución de los nidos de ardilla (círculos blancos) en el Parque Biológico de Gaian en Vila Nova de Gaia (A) y en el Jardín Botánico de la Universidad de Coimbra, en Coimbra (B).

other also to avoid double counting (fig. 1; Gurnell et al., 2001). We counted squirrels using the distance sampling method with direct observation using binoculars 8–16 x 40 (Gurnell et al., 2001), given that both parks had great visibility with small and clear forests. Squirrel surveys were conducted between 8:00 and 16:00 in autumn when higher numbers of squirrels can be found (Tonkin, 1983; Wauters et al., 1992; Bosch & Lurz, 2012). The distance from the observer to the squirrel was measured using a telemeter, and compass bearings were taken to determine the angle between the animal and the transect line (Buckland et al., 1993; Gurnell et al., 2001). We measured the distance of squirrels once and did not consider individuals again after moving to a different position.

Population density and abundance were estimated using Distance Sampling 6.0 software (Thomas et al., 2010). Estimates were stratified and based on Conventional Distance Sampling. Half–normal, hazard and negative exponential rate models for the detection function were fixed against the records using a cosine function (Thomas et al., 2010). Models assumed certainty of detection and measurements (Thomas et al., 2010). The selection of the best model and adjustment term were based on the lowest Akaike information criterion (AIC).

Table 1. Best–fitting models according to Akaike information criterion (AIC) and degree of freedom (df) values to estimate the population density of Eurasian red squirrels at the Parque Biológico de Gaia (PBG) and at the Jardim Botânico da Universidade de Coimbra (JBUC), Portugal, in autumn 2013.

Tabla 1. Los mejores modelos según el criterio de información de Akaike (AIC) y los valores del grado de libertad (df) para estimar la densidad de la población de ardilla roja en el Parque Biológico de Gaia (PBG) y en el Jardín Botánico de la Universidad de Coimbra (JBUC), en Portugal, en otoño de 2013.

	PBG			JBUC		
Model	Negative exponential	Half–normal	Hazard	Negative exponential	Half–normal	Hazard
AIC	378.09	378.28	376.65	17.51	17.56	19.56
df	60	59	59	3	3	2

Habitat use

Eurasian red squirrels prefer mature native forests that can provide them with an abundant supply of food (Bosch & Lurz, 2012). We assessed vegetation type, location of dreys (*i.e.* squirrel nests) and food availability to understand habitat use in both urban parks. The survey was conducted in the PBG in October 2013 and in the JBUC in November 2013. Vegetation type (fig. 1) was mapped with a geographic information system in ArcView GIS 9.2 software (ESRI, 2008). The geographical limits of forests and gardens having the same composition and dominance were confirmed in the field. The number of trees to determine dominant species was verified in 10 x 10 m quadrats randomly within the study sites. Due to different area sizes, 80 quadrats were located in PBG and 60 in JBUC.

Dreys were mapped to determine preferences in relation to vegetation type (fig. 1). Drey counts were obtained by direct observation in a 3 km transect at each site. Transects to count dreys were larger than transects to count individuals because dreys were fixed and double counting was unlikely. We determined the position of dreys, tree species chosen and drey height (Cagnin et al., 2000; Kopij, 2009). Old or abandoned dreys were excluded from counts (Wauters & Dhondt, 1988; Cagnin et al., 2000; Kopij, 2009). The significance of the distribution of dreys in relation to height was measured using a one–way ANOVA in Bioestat 5.0 software (Ayres et al., 2007). Tukey's *post hoc* test (*F*) was applied to determine the significance of any differences (Zar, 1999).

Food availability focused on three aspects: number of feeders, relative abundance and richness of edible mushrooms, and energetic content of natural seeds (cones, acorns, hackberries, and nuts). Feeders with supplementary food were counted directly. Relative abundance of Basidiomycota was estimated by counting fungal bodies or remains with characteristic squirrel bites in stipe and cap on the ground in the same 10 x 10 m quadrats where the vegetation type was measured. Only mushrooms eaten by squirrels during the surveys or reported in the literature were considered as a component of Eurasian red squirrel diet (Fogel & Trappe, 1978; Bertolino et al., 2004). Fungi identification and nomenclature follows Crous et al. (2004).

In each 10 x 10 m quadrat, the number of trees with fruits of each species was counted. Only tree species already reported in the literature (Lurz et al., 2005; Bosch & Lurz, 2012) or those seen being consumed during fieldwork were considered as a component of the Eurasian red squirrel diet. A quadrat of 5 x 5 m was placed below every tree bearing fruit inside the 10 x 10 m quadrat to count fallen cones, acorns, hackberries or nuts. The remains of fruits consumed by Eurasian red squirrels were also recorded and identified by characteristic squirrel bites. Only natural sources of seeds were evaluated given that the composition of seeds offered in feeders varied widely. Seed counts provided an estimate of seed availability (calculated as 10^3 seeds/ha, Bosch & Lurz, 2012). We used data on seed production and calorific content obtained from the literature (Grodziński & Sawicka–Kapusta, 1970; Demir et al., 2002; Wauters et al., 2002; Bosch & Lurz, 2012; Stock et al., 2013) to measure the mean energy value (10^3 kJ/ha^{-1}) and standard deviation (± SD) related to seed counts per habitat.

Reintroduction management

Reintroductions were considered successful as viable populations were established, long–term benefit for local communities were achieved, and improvements in conservation awareness were made, in accordance with IUCN guidelines (IUCN, 1998, 2012). Qualitatively data on management were assessed by unstructured interviews with park managers and employees, and by consulting official documents. We investigated release histories according to motivation, year of release, reintroduction technique (*e.g.,* soft release in which animals are first acclimatized with new habitat in enclosures before release, or hard release in which individuals are directly released into the new environment; see Ewen et al., 2012), supplementary feeding, veterinary support, choice of donor population, number of individuals released, reinforcements, population expansion or decrease, presence/absence of dispersal into surrounding areas (distances greater than 500 m and less than 35 km from release sites), and the presence/absence of squirrels killed on roads.

Results

Eurasian red squirrel population

During the surveys, we observed 61 individuals in the PBG and four in the JBUC. The best relative fit model and adjustment term for the population in the PBG was a hazard–rate cosine based on the lowest AIC score (table 1). In contrast, the best fit for the JBUC population was a negative exponential cosine model based on the lowest AIC score (table1). Estimated abundance and density were higher in PBG (N = 47 squirrels, D = 1.33 squirrel/ha) than in JBUC (N = 2 squirrels, D = 0.17 squirrel/ha). The detection probability in the PBG was 44% whereas in the JBUC it was 51.5%. The encounter rate was 56% and 48.5% for the PBG and JBUC, respectively.

Habitat use

Seven squirrel dreys were found placed in the oak and cork oak forest at PBG (fig. 1B). Squirrels placed a significant portion of dreys in the height of 13 m in forests dominated by *Q. robur* and *Castanea sativa* (F = 8.35, df = 11, P < 0.01). Seven dreys were found around 14 m high in the oak forest in the JBUC (F = 15.74, df = 14, P < 0.01). One drey was found on *Pseudotsuga menziesii* in the alien conifer garden (fig. 1B).

During autumn 2013, only three tree species were fruiting in the PBG: *Q. robur*, *C. sativa* and *P. pinaster* (fig. 1A, table 2). The black alder forest had higher seed productivity (143.8 ± 163.6 x 10^3 seeds/ha) and energetic content (9,524 x 10^3 kJ/ha^{-1}) due to the high concentration of fruiting *C. sativa* (table 2). We counted 471 fungal bodies from 28 species in the PBG. Only 16% of mushrooms were edible to the Eurasian red squirrel (table 3). *Russula* spp. showed significant relative abundance of edible fungi in the PBG, with *R. cyanoxantha* and *R. decipiens* together accounting for 59.4% (table 3). Ongoing supplementary feeding in the PBG consisted of five feeders daily supplied with birdseed to attract birds, but these were also used by squirrels. Squirrels were observed eating mainly sunflower seeds.

Only the oak and mixed forests had fruiting trees in the JBUC during surveys (table 2), namely *Pinus pinea*, *Quercus robur*, and *Celtis australis*. Fruits of this last tree were seen being eaten by squirrels during fieldwork. The oak forest had higher seed productivity (974.0 ± 534.9 x 10^3 seeds/ha) and energetic content (20,779 x 10^3 kJ/ha^{-1}) due to the high productivity of seeds per cone of *P. pinea* (table 2). In the JBUC, we counted 33 fungal bodies of seven species and 70% of them were edible to the Eurasian red squirrel (table 3). As in PBG, the genus *Russula* was also an important food source in JBUC, with a relative abundance of 30.4% for *R. foetens* in the diet, which together with *Amanita gemmata* represented 91.2% of available edible mushrooms in the JBUC (table 3). Supplementary feeding was not recorded in this urban park during the study.

Table 2. Estimation of seed production in each habitat at the Parque Biológico de Gaia (PBG) and at the Jardim Botânico da Universidade de Coimbra (JBUC), Portugal, in autumn 2013: S. Seed (10^3 seeds/ha); Sec. Seed energetic content (10^3 kJ/ha^{-1}). Habitats: Of. Oak forest; Baf. Black alder forest; Oa. Open area; Cof. Cork oak forest; Mpf. Maritime pine forest; Ecg. Exotic conifer garden; Maf. Mountain ash forest; Bf. Bamboo forest; Nag. Narrow–leafed ash garden; Pg. Palm garden; Cs. Central square; Mef. Mixed exotic forest.

Tabla 2. Estimación de la producción de semillas en cada hábitat en el Parque Biológico de Gaia (PBG) y en el Jardín Botánico de la Universidad de Coimbra (JBUC), Portugal, en otoño de 2013: S. Semillas (10^3 seeds/ha); Sec. Contenido energético de las semillas (10^3 kJ/ha^{-1}). Hábitats: Of. Robledal; Baf. Alisal; Oa. Zona despejada; Cof. Alcornocal; Mpf. Pinar de pino negral; Ecg. Plantación de coníferas exóticas; Maf. Bosque de eucalipto regnans; Bf. Bosque de bambú; Nag. Plantación de fresno de hoja pequeña; Pg. Palmeral; Cs. Cuadrado central; Mef. Bosque mixto exótico.

Urban parks	Habitat	S	Sec
		Measured parameters	
PBG			
	Of	3.6 ± 2.2	96.4
	Baf	143.8 ± 163.6	9,524
	Oa	–	–
	Cof	–	–
	Mpf	74.6 ± 35.2	1,859
JBUC			
	Of	974.0 ± 534.9	20,779
	Oa	–	–
	Ecg	–	–
	Maf	–	–
	Bf	–	–
	Nag	–	–
	Pg	–	–
	Cg	–	–
	Mef	56.3 ± 19.5	3,821

Table 3. Relative abundance of mushrooms (Basidiomycota) recorded in the diet of Eurasian red squirrel (*Sciurus vulgaris*) and found at the Parque Biológico de Gaia (PBG) and at the Jardim Botânico da Universidade de Coimbra (JBUC), Portugal, in autumn 2013: * Consumption seen during fieldwork.

*Tabla 3. Abundancia relativa de hongos (Basidiomycota) observada en la alimentación de la ardilla roja (*Sciurus vulgaris*) y encontrada en el Parque Biológico de Gaia (PBG) y en el Jardín Botánico de la Universidad de Coimbra (JBUC), en Portugal, en otoño de 2013: * Consumo visto durante el trabajo de campo.*

Urban park	Taxon	Fungal bodies	Relative abundance (%)	Source
PBG				
	Boletus aestivalis	1	1.3	Fogel & Trappe (1978)*
	Cantharellus cibarius	12	16.2	Fogel & Trappe (1978)
	Pholiota alnicola	9	12.1	Fogel & Trappe (1978)
	Amanita rubescens	1	1.3	Fogel & Trappe (1978)
	Russula cyanoxantha	30	40.5	*
	Russula decipiens	14	18.9	*
	Xerocomus chrysenteron	7	9.4	Fogel & Trappe (1978)*
	Total	**74**	**100**	
JBUC				
	Agaricus campestris	2	8.7	Fogel & Trappe (1978)
	Amanita gemmata	14	60.8	Fogel & Trappe (1978)
	Russula foetens	7	30.4	*
	Total	**23**	**100**	

Reintroduction management

Table 4 summarizes reintroductions attendance to IUCN stepwise. Both reintroductions in PBG and JBUC aimed at enhancing parks aesthetics and enable people to become familiar with this species (table 4). Park managers used the Eurasian red squirrel historical population observations of Antunes (1985) as proving of the species historical range in Portugal. Both urban parks acquired squirrels from commercial creators with veterinary control and support which lowered possibilities of diseases or parasites.

The PBG released 12 squirrels In 1997, and a further 40 couples between 1998 and 2001 using a hard release approach. The animals were from Azé (France). The squirrels in the PBG have continuous veterinary support, because a wildlife rehabilitation center is located therein (table 4), and continuous feeding is provided through bird feeders. Two main failures were detected in the reintroduction project in PBG: the absence of genetic comparison between donors and the closest wild population, and a lack of long–term, technical monitoring.

Twelve squirrels from Madrid (Spain) were hard released at JBUC in 1994. Four squirrel feeders in the forest were active only during the first year (table 4). As in the PBG, the reintroduction project at this park did not consider genetic comparison between donors and the closest wild population. This project ended one year after releases and no post–project management measures and/or long–term monitoring were conducted (table 4).

Discussion

To date, reintroductions of Eurasian red squirrels in Portugal have been successful to some extent given that squirrels are still present in the urban parks almost 20 years later. However, the populations of squirrels are decreasing in both urban parks. Studies found densities from 0.03 to 1.80 squirrels/ha in mixed woodlands (Wauters & Dhondt, 1988; Cagnin et al., 2000; Magris & Gurnell, 2002; Vilar, 1997), figures that are similar to our estimate (0.17 squirrels/ha in JBUC and 1.33 squirrels/ha in PBG). Considering urban parks of limited area and resources, the density in the PBG is similar to densities found in Belgium (Wauters et al., 1997a) and Spain (Vilar, 1997, while the density in the JBUC is lower. The difference in density of the reintroduced populations is mainly related to post–release management since the PBG had squirrel population reinforcements but the JBUC did not. Habitat quality also regulates species

Table 4. Conditions of Eurasian red squirrel (*Sciurus vulgaris*) reintroductions in the Parque Biológico de Gaia (PBG) and in the Jardim Botânico da Universidade de Coimbra (JBUC), Portugal.

*Tabla 4. Condiciones de las reintroducciones de ardilla roja (*Sciurus vulgaris*) en el Parque Biológico de Gaia (PBG) y en el Jardín Botánico de la Universidad de Coimbra (JBUC), en Portugal.*

Phase / Aspect	PBG	JBUC
Pre–project		
Main motivation	Aesthetic, leisure and environmental education	Aesthetic, leisure and environmental education
Origin of donor population	Azé (France)	Madrid (Spain)
Captive or wild squirrels	Captive	Captive
Subspecies of donor population	*Sciurus vulgaris fuscoater*	*Sciurus vulgaris infuscatus*
Study of historical range of extinct populations	Antunes (1985)	Antunes (1985)
Study of genetic individual variability of donor population	No	No
Governmental permits	Not required	Not required
Veterinary certification of health and absence of parasites	Yes	Yes
Other certifications	Origin and transportation	Origin and transportation
Release		
Year of release	November 1997	June 1994
Number of squirrels released	12 (6♀♀ and 6♂♂)	12 (6♀♀ and 6♂♂)
Method (soft or hard)	Hard release	Hard release
Supplementary feeding	Five feeders for birds, but used by squirrels	Four squirrel feeders
Kind of supplementary feeding	Birdseed with sunflower seed	Walnuts, hazelnuts, and others
Post–project		
Population monitoring	No	No
Veterinary support	Yes	No
Continuity of supplementary feeding	Yes, to date	Stopped after one year
Manager's general feeling about squirrels abundance	Decrease and need for reinforcement	Population explosion in next three years
Population reinforcement	Yes (three)	No
Subspecies used to reinforcement	*Sciurus vulgaris fuscoater*	–
Number of individuals (origin, and year of population reinforcement)	10 couples (Epe, Netherlands, X 1998) 15 couples (Epe, Netherlands, VII 2001) 15 couples (Azé, France, VIII 2001	– – –
Manager's general feeling about squirrel abundance after reinforcement	Population explosion in next five years	–
Squirrels seen in nearby areas (> 500 m and < 35 km from release site)	Yes	Yes
Name of five localities where squirrels were seen	Sermonde, Vila Chã, Serra da Agrela, Serra da Freita, and Marco de Canaveses	Mata do Buçaco, Serra da Lousã, Alfarelos, Serra do Sicó, and Soure
First year of squirrels seen in these nearby localities	2010	2001
Squirrels killed on nearby roads	Yes	Yes

abundance and density and is of great importance for the success or failure of reintroductions (Ewen et al., 2012). Squirrel dreys in both urban parks were predominantly placed in native oak forests and near food sources, reinforcing the need for high quality habitat and food diversity for the maintenance of these populations. The studied parks had few fruiting trees compared with other studies (Bosch & Lurz, 2012). Forests in the JBUC had a higher energetic content than those in the PBG but the diversity of native food items was poorer. In contrast, forests in the PBG had less energetic content, but they presented richer and more abundant additional food items, such as edible mushrooms. Additionally, the PBG had continuous supplementary food, mainly through bird feeders also used by squirrels, whereas the JBUC only had feeders in the year following the reintroduction.

In terms of species identity for conservation purposes, genetic proximity was only adequately considered in the JBUC where the subspecies *Sciurus vulgaris infuscatus* was reintroduced, while in the PBG the subspecies *S. v. fuscoater* was released. Although both subspecies occur in the Iberian Peninsula, only *S. v. infuscatus* occurs naturally in Portugal (Mathias & Gurnell, 1998; Lurz et al., 2005; Bosch & Lurz, 2012). Further studies on Eurasian red squirrel distribution, taxonomy and genetic diversity in the Iberian Peninsula should consider the influence of *S. v. fuscoater* presence in Portugal, as has been done to other subspecies in the United Kingdom (see Hale & Lurz, 2003; Hale et al., 2004).

Post–project monitoring was not explicitly considered in either park. It is there not fully understood whether dispersal to vicinity (table 3) was natural or due to stress of limited resources. Deficiency in post–release actions, such as monitoring health and abundance, is responsible for the long–term decrease in Eurasian red squirrel populations in both urban parks but active adaptive management could improve the current situation (Ewen et al., 2012; Runge, 2013). Future actions should consider improving habitat quality by means of specific feeders for Eurasian red squirrels, and replacement of alien trees for native oak forest. Monitoring population health, adaptation and demographic variation will endorse the long–term success of the reintroductions. In addition, managers should ensure active human community involvement so that effective education would not only foster knowledge of species but also concern for its needs (IUCN, 2012).

Critical steps for successful reintroduction of Eurasian red squirrels in areas without competition of alien tree squirrels should follow three stages, consisting of pre–project activities, release stages and post–release activities (IUCN, 1998, 2012). Pre–project activities should include studies on (1) habitat quality, (2) genetic proximity between donors and the closest wild population, and (3) the health of donor stocks.

In the release stage, (1) the number of individuals released should consider 35 to 85 individuals to achieve a long–term viable population in an area of high resource variability, and 55 to 175 individuals in areas of low resource variability (Wood et al., 2007) and (2) hard release technique proved to be a good and cheaper method to Eurasian red squirrel reintroductions (Swinnen, 1988; Fornasari et al., 1997). Finally, post–release activities should (1) evaluate population adaptation, (2) mitigate mortality, (3) monitor the need for supplementary feeding, (4) provide veterinary support, and (5) promote continuous public awareness and education (IUCN, 1998, 2012).

Reintroductions for aesthetic and leisure purposes are not usually concerned about strictly following conservation protocols unless required by law. However, these reintroductions for aesthetic and leisure purposes have significant effects on wildlife management and conservation (Hodder & Bullock, 1997). Therefore, we strongly suggest that reintroductions with aims other than conservation should also have standardized international guidelines, regulations and monitoring.

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References

Antunes, M. T., 1985. *Sciurus vulgaris* no Cabeço da Arruda, Muge: presença e extinção em Portugal. *Arqueologia*, 12: 1–16.

Armstrong, D. P. & Seddon, P. J., 2007. Directions in reintroduction biology. *Trends in Ecology and Evolution*, 23: 20–25.

Ayres, M., Ayres, M. Jr., Ayres, D. L. & Santos, A. A. S., 2007. *BioEstat 5.0: aplicações estatísticas nas áreas das ciências biológicas e médicas*. Sociedade Civil Mamirauá, Belém.

Bertolino, S. & Lurz, P. W., 2013. *Callosciurus* squirrels: worldwide introductions, ecological impacts and recommendations to prevent the establishment of new invasive populations. *Mammal Review*, 43: 22–33.

Bertolino, S., Vizzini, A., Wauters, L. A. & Tosi, G., 2004. Consumption of hypogeous and epigeous fungi by the red squirrel (*Sciurus vulgaris*) in subalpine conifer forests. *Forest ecology and management*, 202: 227–233.

Bosch, S. & Lurz, P. W. W., 2012. *The Eurasian red squirrel*. Westarp Wissenschaften, Hohenwarsleben.

Buckland, S. T., Anderson, D. A., Burnham, K. P. &

Laake, J. L., 1993. *Distance sampling: estimating abundance of biological populations*. Chapman & Hall, London.

Cagnin, M., Aloise, G., Fiore, F., Oriolo, V. & Wauters, L. A., 2000. Habitat use and population density of the red squirrel, *Sciurus vulgaris meridionalis*, in the Sila Grande mountain range (Calabria, South Italy). *Italian Journal of Zoology*, 67: 81–87.

Caughley, G. & Gunn, A., 1996. *Conservation biology in theory and practice.* Blackwell Science, Cambridge and Massachusetts.

Crous, P. W., Gams, W., Stalpers, J. A., Robert, V. & Stegehuis, G., 2004. MycoBank: an online initiative to launch mycology into the 21st century. *Studies in Mycology*, 50: 19–22.

Demir, F., Doğan, H., Özcan, M. & Haciseferoğullari, H., 2002. Nutritional and physical properties of hackberry (*Celtis australis* L.). *Journal of Food Engineering*, 54: 241–247.

ESRI, 2008. *ArcView: Release 9.2.* Environmental Systems Research Institute, Redlands.

Ewen, J. G., Armstrong, D. P., Parker, K. A. & Seddon, P. J., 2012. *Reintroduction biology: integrating science and management.* Wiley–Blackwell, New Jersey.

Ferreira, A. F., Guerreiro, M., Álvares, F. & Petrucci–Fonseca, F., 2001. Distribución y aspectos ecológicos de *Sciurus vulgaris* en Portugal. *Galemys*, 13: 155–170.

Fogel, R. & Trappe, J. M., 1978. Fungus consumption (mycophagy) by small animals. *Northwest Science*, 52: 1–31.

Fornasari, L., Casale, P. & Wauters, L., 1997. Red squirrel conservation: the assessment of a reintroduction experiment. *Italian Journal of Zoology*, 64: 163–167.

Griffith, B., Scott, J. M., Carpenter, J. W. & Reed, C., 1989. Translocation as a species conservation tool: status and strategy. *Science*, 245: 477–480.

Grodziński, W. & Sawicka–Kapusta, K., 1970. Energy values of tree–seeds eaten by small mammals. *Oikos*, 21: 52–58.

Gurnell, J., 1987. *The natural history of squirrels*. Facts on File Publications, Oxford.

Gurnell, J., Lurz, P. W. W. & Pepper, H., 2001. Practical techniques for surveying and monitoring squirrels. *Forestry Commission Practice Note*, 11: 1–12.

Hale, M. L. & Lurz, P. W. W., 2003. Morphological changes in a British mammal as a result of introductions and changes in landscape management: the red squirrel (*Sciurus vulgaris*). *Journal of Zoology*, 260: 159–167.

Hale, M. L., Lurz, P. W. W. & Wolff, K., 2004. Patterns of genetic diversity in the red squirrel (*Sciurus vulgaris* L.): Footprints of biogeographic history and artificial introductions. *Conservation Genetics*, 5: 167–179.

Harrington, L. A., Moehrenschlager, A., Gelling, M., Atkinson, R. P., Hughes, J. & Macdonald, D. W., 2013. Conflicting and complementary ethics of animal welfare considerations in reintroductions. *Conservation Biology*, 27: 486–500.

Hodder, K. H. & Bullock, J. M., 1997. Translocations of native species in the UK: implications for biodiversity. *Journal of Applied Ecology*, 34: 547–565.

IUCN, 1998. *Guidelines for reintroductions.* IUCN/SSC Re–introduction Specialist Group, Gland and Cambridge.

– 2012. *Guidelines for reintroductions and other conservation translocations*. IUCN/SSC Species Survival Commission, Gland and Cambridge.

Jurczyszyn, M., 2006. The use of space by translocated edible dormice, *Glis glis* (L.), at the site of their original capture and the site of their release: radio–tracking method applied in a reintroduction experiment. *Polish Journal of Ecology*, 54: 345–350.

Kopij, G., 2009. Habitat and drey sites of the red squirrel *Sciurus vulgaris* Linnaeus 1758 in suburban parks of Wroclaw, SW Poland. *Acta Zoologica Cracoviensia Series A: Vertabrata*, 52: 107–114.

Lurz, P. W. W., Gurnell, J. & Magris, L., 2005. *Sciurus vulgaris*. *Mammal Species*, 769: 1–10.

Magris, L. & Gurnell, J., 2002. Population ecology of the red squirrel (*Sciurus vulgaris*) in a fragmented woodland ecosystem on the Island of Jersey, Channel Islands. *Journal of Zoology*, 256: 99–112.

Matějů, J., Říčanová, Š., Ambros, M., Kala, B., Hapl, E. & Matějů, K., 2010. Reintroductions of the European ground squirrel (*Spermophilus citellus*) in Central Europe (Rodentia: Sciuridae). *Lynx*, 41: 175–191.

Mathias, M. L. & Gurnell, J., 1998. Status and conservation of the red squirrel (*Sciurus vulgaris*) in Portugal. *Hystrix*, 10: 13–19.

Oliveira, N. G., 2013. *Parque Biológico de Gaia: 1983/2013*. Parque Biológico de Gaia, Vila Nova de Gaia.

Poole, A. & Lawton, C., 2009. The translocation and post release settlement of red squirrels *Sciurus vulgaris* to a previously uninhabited woodland. *Biodiversity Conservation*, 18: 3205–3218.

Runge, M. C., 2013. Active adaptive management for reintroduction of an animal population. *Journal of Wildlife Management*, 77: 1135–1144.

Seddon, P. J., Armstrong, D. P. & Maloney, R. F., 2007. Developing the science of reintroduction biology. *Conservation Biology*, 21: 303–312.

Seddon, P. J., Soorae, P. S. & Launay, F., 2005. Taxonomic bias in reintroduction projects. *Animal Conservation,* 8: 51–58.

Shar, S., Lkhagvasuren, D., Bertolino, S., Henttonen, H., Kryštufek, B. & Meinig, H., 2008. *Sciurus vulgaris*. IUCN Red List of Threatened Species, v.2013.1. http://www.iucnredlist.org. (Accessed on 6 November 2013).

Stock, W. D., Finn, H., Parker, J. & Dods, K., 2013. Pine as fast food: foraging ecology of an endangered cockatoo in a forestry landscape. *Plos One*, 8: e61145.

Swinnen, C., 1988. Reintroduction of the red squirrel (*Sciurus vulgaris* L.) in an isolated park habitat. *Parasitica*, 44: 89–91.

Thomas, L., Buckland, S. T., Rexstad, E. A., Laake, J. L., Strindberg, S., Hedley, S. L., Bishop, J. R. B., Marques, T. A. & Burnham, K. P., 2010. Distance software: design and analysis of distance sampling

surveys for estimating population size. *Journal of Applied Ecology*, 47: 5–14.
Tonkin, J. M., 1983. Activity patterns of the red squirrel (*Sciurus vulgaris*). *Mammal Review*, 13: 99–111.
Vilar, J. P., 1997. Ecoetologia i biologia de l'esquirol (*Sciurus vulgaris*, Linnaeus, 1758) en dos hàbitats de predictibilitat alimentària contínua que difereixen en l'abundància d'aliment. Ph. D. Thesis, University of Barcelona.
Wauters, L. A., Casale, P. & Fornasari, L., 1997b. Post-release behaviour, home range establishment and settlement success of reintroduced red squirrels. *Italian Journal of Zoology*, 64: 169–175.
Wauters, L. A. & Dhondt, A. A., 1988. The use of red squirrel (*Sciurus vulgaris*) dreys to estimate population density. *Journal of Zoology*, 214: 179–187.
Wauters, L. A., Somers, L. & Dhondt, A. A., 1997a. Settlement behaviour and population dynamics of reintroduced red squirrels *Sciurus vulgaris* in a park in Antwerp, Belgium. *Biology Conservation*, 82: 101–107.
Wauters, L. A., Swinnen, C. & Dhondt, A. A., 1992. Activity budget and foraging behaviour of red squirrels (*Sciurus vulgaris*) in coniferous and deciduous habitats. *Journal of Zoology*, 227: 71–86.
Wauters, L. A., Tosi, G. & Gurnell, J., 2002. Interspecific competition in tree squirrels: do introduced grey squirrels (*Sciurus carolinensis*) deplete tree seeds hoarded by red squirrels (*S. vulgaris*)? *Behavioral Ecology and Sociobiology*, 51: 360–367.
Wood, D. J., Koprowski, J. L. & Lurz, P. W. W., 2007. Tree squirrel introduction: a theoretical approach with population viability analysis. *Journal of Mammalogy*, 88: 1271–1279.
Zar, J. H., 1999. *Biostatistical analysis*, *4th ed.* Prentice–Hall Inc., New Jersey.