

# Invasion dynamics of an introduced squirrel in Argentina

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1 **Abstract**

2           Biological invasions are one of the major threats to both ecosystem and  
3 economic functioning. Their management typically involves culling of the pest or  
4 removal of its habitat. The Asiatic Red-Bellied Beautiful Squirrel *Callosciurus*  
5 *erythraeus* is the first known introduction of a squirrel into South America. It  
6 established from five releases in 1973, using exotic trees to spread through  
7 Argentinean Pampas. It now causes substantial economic damage in agricultural and  
8 urban areas across >680km<sup>2</sup>, and its continued spread threatens indigenous species.  
9 We developed a spatially explicit model of the invasion for the likely range of life-  
10 history parameters, matched against empirical data on patch occupancy in 2004. The  
11 two best-fitting models suggest the current population to be on the cusp of an  
12 explosive expansion. These models were used to predict future trends under  
13 alternative scenarios of strategic culling or habitat removal aimed at slowing the  
14 spread. The predictions for 18 years into the future were that (i) the present lack of  
15 systematic management will lead to a 5-fold increase in area of occupancy, (ii)  
16 removal of habitat down to half carrying capacity will thin the population but  
17 accelerate its spread, (iii) 10 yrs of culling above the maximum sustainable yield  
18 (MSY) will precipitate declines in abundance and patch occupancy towards  
19 extinction, but with immediate recovery upon cessation of the cull. We recommend  
20 continuous culling above the MSY in priority patches, aimed at slowing arrival to  
21 valuable conservation areas. This study demonstrates the need for prompt action to  
22 terminate invasions before they establish. The squirrel invasion is now irreversible  
23 after 30 years of slow spread across fragmented habitat. Although culling requires  
24 public awareness campaigns and sustained governmental commitment, it is now the  
25 best feasible strategy for managing this invasion.

# 1 **Introduction**

2           Biological invasions are now considered one of the major threats to biodiversity  
3 and ecosystem functioning, together with habitat degradation (Mack et al. 2002).  
4 Although exotic species are key components of the economies of several countries  
5 (e.g., most crops), they can provoke huge economic losses when they escape human  
6 control (e.g., many agricultural pests; Wittenberg and Cock 2001). The problem of  
7 biological invasions has long been recognised (Elton 1958) but concern about exotics  
8 has increased over the last decade. Recent studies have provided new evidence of the  
9 complexity of invasion dynamics and the interdisciplinary approach needed to inform  
10 management plans for preventing or mitigating negative impacts (Mack et al. 2002,  
11 Grosholz 2005).

12           Ecological models can provide valuable insight into the process of biological  
13 invasions, both for predicting the direction of colonisation and for exploring the  
14 consequences of alternative management options. Spatially-explicit population  
15 models (SEPMs) present a powerful tool in this context because of their capacity to  
16 incorporate the complexity of real-world landscapes in their predictions of spatial  
17 distribution, abundance, and viability of populations (Dunning et al. 2006). SEPMs  
18 have been applied mainly to predictions of population viability for endangered or  
19 harvested species (e.g., see examples in Akçakaya 2000 and Akçakaya et al. 2004).  
20 The same approach has enormous potential for modelling the new environment of an  
21 invader in terms of corridors and barriers to dispersal and suitable habitat for  
22 reproduction, to predict its future distribution, quantify environmental impacts,  
23 compare alternative management strategies, and guide new research by identifying  
24 key variables that most affect the model predictions (e.g., Rushton et al. 1997, Lurz et  
25 al. 2001, Tattoni et al. 2006).

1           We developed a spatially explicit model to predict the spread of the Red-Bellied  
2 Beautiful Squirrel *Callosciurus erythraeus*. This tree squirrel is native to southeast  
3 Asia and was introduced to Argentina as a pet in 1970 (Aprile and Chicco 1999). By  
4 1973, 2 to 5 animals had established from a single release point in the Pampas region.  
5 After 31 years of slow spread, the wild population occupied a region of 680 km<sup>2</sup>  
6 around the release point by 2004 (Guichón et al. 2005). This is the first known  
7 introduction of a squirrel into South America. Although no systematic studies have  
8 yet been conducted on the ecology of the species in Argentina, anecdotal data (Aprile  
9 and Chicco 1999) and recent observations (Guichón, unpublished data) indicate that  
10 its feeding and nesting habits are associated with exotic trees in commercial and  
11 ornamental plantations in rural and urban areas of the pampas. This represents another  
12 example of how the success of new exotics can be facilitated by human-modified  
13 environments and positive interactions among introduced species (Mack et al. 2000,  
14 Grosholz 2005).

15           In the Argentine Pampas, economic damage has been reported by some  
16 inhabitants and local producers to fruit plantations, forestations, and electric and  
17 irrigation systems. The presence of this attractively patterned species is welcomed by  
18 other inhabitants, however, who regard it as a local tourist attraction given that no  
19 native squirrels inhabit this region, and some of whom catch individuals to sell or  
20 keep as pets (Guichón et al. 2005). Observations of predation on the nests of native  
21 birds have been reported in the Pampas (Pereira et al. 2003). Studies conducted in  
22 Japan, where it was first introduced in 1935, have reported high population growth  
23 and adaptability to new environments, damage of plantations, garden trees, and  
24 cables, and predation of native birds (Tamura et al. 1988, 1989, Azuma 1998,  
25 Miyamoto et al. 2004). The species was also introduced into France in the 1970s,

1 where it now causes damage to trees and crops (Jouanin 1992). In its native habitat, it  
2 has been subjected to control actions to limit economic damage to the forestry  
3 industry caused by its debarking of coniferous trees which then become infected with  
4 a fungus (Lin and Yo 1981, Yo et al. 1992a).

5       The current distribution of this squirrel in Argentina lies within a highly  
6 modified rural area of the Pampas region to the west of the city of Buenos Aires  
7 (Guichón et al. 2005). The northerly spread is within 30 km of the Paraná River Delta  
8 which sustains important commercial forests and unique and bio-diverse riparian  
9 forests composed of both temperate and subtropical flora and fauna (Malvárez et al.  
10 1999). The Otamendi Natural Reserve within this delta protects several endemic birds  
11 potentially susceptible to nest predation by the Red-Bellied Squirrel. No programme  
12 of proactive management has yet been developed for this population, and currently the  
13 only control is sporadic lethal trapping by local producers in response to damage.  
14 Proactive control actions require long-term commitment and funding from  
15 government, NGOs, wildlife managers and scientists, and the local community (Mack  
16 et al. 2000, Genovesi 2005), which may take several years. Some localized control  
17 actions could be taken more immediately, however, to slow the invasion towards  
18 critical areas, and to establish methods and test the commitment of stakeholders  
19 (Bertolino and Genovesi 2003). Here we simulate alternative control scenarios as a  
20 first step to evaluating the feasibility and success of both localized and region-wide  
21 actions.

22       The objectives of this study are (1) to develop a spatially explicit population  
23 model (SEPM) to simulate the invasion dynamics of the Red-Bellied Squirrel in the  
24 Pampas region and (2) to evaluate population viability and spread under various  
25 scenarios in order to recommend a management plan to slow the invasion. In

1 particular we modelled the alternatives of harvesting and habitat removal. Local culls  
2 to harvest the population above its maximum sustainable yield can potentially  
3 precipitate extinction, but require sustained effort. Removing the squirrel's habitat can  
4 be done in a single operation but may have a stabilising effect on the population,  
5 associated with its reduced efficiency of exploiting resources (Kent et al. 2003).

## 6 **Materials and Methods**

### 7 **Study area**

8 The study area comprised 8,100 km<sup>2</sup> (90×90 km) of northeastern Buenos Aires  
9 Province, Argentina, centred at the liberation site of the squirrels close to the town of  
10 Jáuregui in the district of Luján (34°36'S, 59°11'W). Ten Red-Bellied Squirrels were  
11 bought from a pet shop in Holland and introduced to Argentina in 1970. They were  
12 kept caged for 3 years until 2 to 5 individuals escaped or were liberated, originating a  
13 wild population. The species now principally occupies highly fragmented woodland  
14 patches in rural and urban areas of the Pampas (Aprile and Chicco 1999). Native  
15 grasslands of the Pampas region have been extensively modified by agriculture,  
16 livestock, and urbanisation. In our study area, fruit (mainly *Actinidia* sp., *Citrus* spp.,  
17 *Juglans* spp., *Prunus* sp.) and timber plantations (mainly *Eucalyptus* spp., *Populus*  
18 spp.) add to an arboreal vegetation comprising exotic species used as wind curtains,  
19 for shade, or as ornaments in gardens and along roads, railways, and rivers (mainly  
20 *Ailanthus* sp., *Araucaria* spp., *Casuarina* spp., *Cupressus* spp., *Gleditsia triacanthos*,  
21 *Ligustrum* spp., *Melia azedarach*, *Morus* spp., *Pinus* spp., *Quercus* spp., *Tilia* spp.).  
22 All of these introduced trees are used by the squirrels (Aprile and Chicco 1999,  
23 Guichón et al. 2005, Guichón, unpublished data). The temperate moist climate has

1 mean temperatures of 23.8°C in January and 9.1°C in July, and mean annual  
2 precipitation of 944 mm.

### 3 ***The model***

4 We developed a spatially-explicit, stage-structured stochastic model to simulate  
5 the invasion dynamics of the Red-Bellied Squirrel in the Pampas region. Given that  
6 no previous studies have been conducted on the demography of the Red-Bellied  
7 Squirrel in Argentina, we took estimates of life history parameters principally from  
8 published data on the squirrel in its indigenous habitat in Taiwan and in its colonised  
9 range in Japan. We used the population viability analysis program RAMAS/GIS 4.00  
10 (Akçakaya 2002), which is designed to link a metapopulation model with landscape  
11 data from a geographical information system (GIS).

### 12 Habitat data

13 Published data on density and home range area were used to estimate habitat  
14 suitability for the different habitat types. The lowest density of Red-Bellied Squirrels,  
15 at 2 to 3 individuals ha<sup>-1</sup>, was reported in a Taiwanese plantation of Japanese Fir  
16 (*Cryptomeria japonica*) (Lin and Yo 1981), whilst 5 to 7 adults ha<sup>-1</sup> were found in  
17 temperate mixed forest in Japan and in native habitat (Tamura et al. 1989). Home  
18 range size and overlap was always larger in males than females (Tamura et al. 1989,  
19 Yo et al. 1992b). Tamura et al. (1987, 1989) reported mean home range sizes of  
20 females of 0.5 to 0.7 ha, which greatly overlapped with male home ranges but not  
21 with those of other females, and mean home range sizes for males of 2.2 to 3.8 ha,  
22 which greatly overlapped with both male and female home ranges. On average, 8  
23 males overlapped each female home range (Tamura et al. 1989), leading to densities

1 as high as 7 to 9 sexually mature squirrels ha<sup>-1</sup>. These data were used to estimate  
2 carrying capacity of the main habitat types identified in our study region.

3 Squirrels in the Pampas region reproduce in both urban and woodland areas  
4 (Aprile and Chicco 1999, Guichón, unpublished data), as was reported for Japan  
5 (Miyamoto et al. 2004). We defined 4 types of suitable habitat (woodland, residential,  
6 suburban, and urban) in a matrix of non-suitable habitat. Woodland habitat included  
7 all forested areas, woodland patches, and wooded corridors; in the absence of more  
8 specific information, all woodlands were considered of equal quality. Residential,  
9 suburban, and urban areas differed in their increasing degree of urbanization and  
10 decreasing degree of arboreal cover. Non-suitable habitat was defined as open areas  
11 with no trees, mainly comprising arable and pasture fields and water bodies.

12 Woodland was considered to have the highest habitat suitability, followed by  
13 residential, suburban, and urban, with non-suitable habitat receiving a zero value  
14 (Table 1). This habitat classification was used to construct a 90×90-km habitat map  
15 with a resolution of 40 m to encompass the current distribution of the squirrel, using  
16 GIS Idrisi 32 (Clark Labs) and ArcGIS 8.3 (Esri). With no land-cover maps available  
17 for this area, we built this habitat map by visual interpretation of a satellite image  
18 Landsat 7 TM orbit 225–084, October 2002 (licensed by PRODITEL, Universidad  
19 Nacional de Luján), and then calculated habitat values as the number of individuals  
20 that a 40×40-m cell of each habitat type could support.

21 The minimum habitat suitability value below which the habitat is not suitable  
22 for reproduction and the distance that individuals travel for daily activity provide the  
23 link between the habitat map and the metapopulation model. Miyamoto et al. (2004)  
24 considered that if the distance between woods was less than 163 m, which is the  
25 largest linear dimension of a female home range (Tamura et al. 1987), females could



1 move among them to establish home ranges and reproduce. Lin and Yo (1981)  
2 reported maximum movements of 326 m and 484 m for mature and immature  
3 females, respectively; however, these distances may be an overestimate for daily  
4 movements within non-contiguous wooded areas. The ‘perceptual range’ of a species  
5 describes the maximum distance from which an animal can perceive the presence of  
6 remote patches of habitat (Lima and Zollner 1996) and it becomes relevant for  
7 modelling gap crossing. We assumed daily movements of 160 m, after combining  
8 published information on the use of space by the Red-Bellied Squirrel with data from  
9 other sciurids (Zollner 2000: perceptual range of 120 m for *Tamias striatus* and 300 m  
10 for *Sciurus carolinensis*; Gurnell et al. 2002: movement between patches up to 200 m  
11 apart by *S. vulgaris*; Verbeylen et al. 2003: trivial range of *S. vulgaris* encompasses  
12 patches separated by up to 50 m; Selonen and Hanski 2004: fields 100-150 m wide  
13 seldom traversed by *Pteromys volans*). The estimate of daily movements takes into  
14 account the potential for squirrels to use cables, isolated trees and bushes, and high  
15 vegetation on field borders when crossing open fields and rivers (personal  
16 observation). Hedgerows, bushes, and isolated trees increase the connectivity of  
17 fragmented forest for *S. vulgaris* and *S. carolinensis* (Fitzgibbon 1993, van Apeldoorn  
18 et al. 1994, Wauters et al. 1994, Selonen and Hanski 2004). Therefore, in our habitat  
19 map, nearby cells within 160 m with habitat suitability at least equalling the threshold  
20 (i.e., woodland and urban habitat) were grouped together as a ‘patch’. The group of  
21 individuals occupying a discrete patch is hereafter referred to as a ‘population’,  
22 connected through dispersal to neighbouring populations. Nearest edge-to-edge  
23 distances yielded inter-patch distances. We used highways and main rivers to divide  
24 patches close enough to be assigned to the same patch but that would not be easily  
25 crossed by squirrels. The carrying capacity of each patch was calculated as the total

1 habitat suitability within the patch (i.e., the sum of habitat values of woodland,  
2 residential, suburban, and urban cells). The habitat is extremely fragmented because  
3 of small forested patches used for shade or wind curtains in this agricultural  
4 landscape. Patches with a carrying capacity ( $K$ )  $<$  75 individuals, were absorbed into  
5 the dispersal matrix as ‘stepping-stone’ fragments that facilitate dispersal between  
6 permanent local populations. These smaller patches accounted for only 8% of suitable  
7 area and their incorporation as independent patches would have vastly inflated the  
8 patch matrix without contributing substantially to the metapopulation dynamics. In  
9 effect, their inclusion in the matrix of unsuitable habitat gave us a conservative  
10 estimate of the expansion rate across a total of 497 suitable patches.

#### 11 Demographic data

12 We modelled the dynamics of the population within each patch with a  
13 composite age-structured Leslie matrix, with 3 age classes for each sex: juveniles in  
14 their first year of life, yearlings in their second year, and adults in their third year and  
15 above. We assumed that: (1) all births in the population take place in a relatively short  
16 breeding period in spring, (2) the population is censused immediately before the  
17 annual birth pulse, (3) no deaths occur between litters within the reproductive season,  
18 and (4) the Leslie matrix applies across all populations.

19 We used mean yearly survival rates,  $p_x$ , reported by Tamura (2003) for the Red-  
20 Bellied Squirrel in Japan, except for the survival rate of adults older than 3 years for  
21 which we used a value of 0.3 (Table 1), influenced by Gurnell (1987) who reported  
22 that patterns of survival appear similar among Holarctic tree squirrels: once they have  
23 achieved their second year of life, year-to-year survival is 50-70 %. The mean number  
24 of viable offspring per individual of each age (fecundity,  $F_x$ ) was calculated as the  
25 age-specific offspring production (maternity,  $m_x$ ) multiplied by the survival of

1 juveniles ( $p_j$ ) (Akçakaya 2002). As in most tree squirrels (Gurnell 1987), the mating  
2 system of the Red-Bellied Squirrel is promiscuous. Females care for the young, and  
3 each female may mate with 4 to 11 different males while a male can mate with at least  
4 6 females within a year (Tamura et al. 1988, 1989). In our model we assumed that  
5 each male can mate with up to 6 females each year. We also assumed that all females  
6 breed each year and that the sex ratio is 1:1 (Yo et al. 1992a). Females establish their  
7 own home ranges when they become sexually mature at about 1 year old, and they  
8 may have 1 to 3 litters per year (Tamura et al. 1988, 1989). Multiplying the mean  
9 juvenile survival of 0.6 (Tamura 2003) by a litter size of 2 (from a range of 1-4 given  
10 by Tamura 1999) yields a value of 1.2 weaned juveniles per litter, which is within the  
11 range estimated for the Red-Bellied Squirrel, of 1.1 to 1.4 (Tamura et al. 1988,  
12 Tamura 1999). We obtained estimates of fecundity ( $F_x$ ) per year from the survival of  
13 juveniles ( $p_j = 0.6$ ) times the litter size (2) times the number of litters per year  
14 (juveniles: 1, yearlings and adults: 2) times the sex ratio (0.5) (Table 1). The resulting  
15 Leslie matrix predicted a finite rate of increase,  $\lambda = 1.53 \text{ year}^{-1}$ ; because of uncertainty  
16 in these values for the Red-Bellied Squirrel in Argentina, their influence on model  
17 outputs was explored by varying these estimates by  $\pm 10\%$  (Table 1).

18 Antagonistic behaviours and hierarchies among both male and female Red-  
19 Bellied Squirrels reportedly influence their habitat use, foraging, and reproductive  
20 activities (Tamura et al. 1988, Yo et al. 1992a, 1992b). We therefore assumed density-  
21 dependent population growth on all vital rates (mortality as well as fecundity) due to  
22 contest competition.

### 23 Dispersal

24 Dispersal refers to the movement of squirrels between habitat patches. We  
25 assumed that dispersal occurs annually and that dispersal rate (i.e., the proportion

1 dispersing from a source population to a target population) was determined by: (1) the  
2 distance between the source and the target populations, and (2) the age and sex of the  
3 potential disperser. We modelled dispersal rate as a negative exponential function of  
4 distance (Akçakaya 2002), where distances were measured between patch edges. No  
5 study of dispersal distances is available for this species, and only one record of a  
6 dispersal event has been reported, of over 1 km (Lin and Yo 1981). The mean longest  
7 linear dimension of a home range in Japanese populations has been estimated to be  
8 435 m for males and 130 m for females (Tamura et al. 1987), while Taiwanese  
9 populations have a mean of 250 m (Lin and Yo 1981). A distance of 1 km is well  
10 within the range of dispersal of *Tamiasciurus hudsonicus*, *S. carolinensis*, and *S. niger*  
11 (Goheen et al. 2003). For *S. vulgaris*, dispersal distances range from 0.6 to 1.3 km  
12 (Lurz et al. 1997), up to 3.2 to 3.5 km (Magris and Gurnell 2002) and a maximum of 4  
13 km (Verbeylen et al. 2003). *Pteromys volans* has a mean and maximal dispersal  
14 distance of 2.5 and 9 km, respectively (Selonen and Hanski 2004). Taking into  
15 account differences in body size and mode of dispersal, we assumed mean and  
16 maximum dispersal distances of 1 km and 5 km respectively, and varied both  
17 estimates by 50% of their value to account for uncertainty (Table 1). As some  
18 individuals are likely to move beyond this maximum dispersal distance, model  
19 predictions were intentionally conservative. We assumed that juvenile males had the  
20 highest dispersal rate and that adult females had the lowest rate (Table 1) based on  
21 Tamura et al. (1989) that reported male- and juvenile-biased dispersal, although they  
22 observed transient squirrels of both sexes and all ages. We evaluated the effect of  
23 density-dependent dispersal in population spread by varying dispersal rate in direct  
24 proportion to the size of the source population (Akçakaya 2002), using an arbitrary

1 constant of proportionality (Table 1), which results in a greater proportion of  
2 individuals leaving a population at high than at low densities.

### 3 Stochasticity

4 Environmental stochasticity was incorporated at each time step of 1 year by  
5 random sampling from lognormal distributions around the set of vital rates used to  
6 project the dynamics of each population, with means taken from the Leslie matrix  
7 described above and standard deviations of 5% (Akçakaya 2000). Demographic  
8 stochasticity was incorporated by sampling the numbers of survivors and dispersers  
9 from a binomial distribution, and the numbers of offspring from a Poisson distribution  
10 (Akçakaya 2002).

### 11 Validation of the model

12 We varied the following 3 input parameters to account for uncertainty  
13 (Akçakaya and Sjögren-Gulve 2000): survival and fecundity rates, mean dispersal  
14 distance, and maximum dispersal distance (Table 1). Three levels (low, medium, and  
15 high estimates) of each parameter were tested using complete Latin square sampling  
16 in the standard form, which delivers a spatially balanced design (Gomes et al. 2004)  
17 suitable for identifying any interacting pairs amongst the three input variables. This  
18 design yields a total of 9 parameter combinations. We explored numerous other  
19 combinations in order to verify that the inputs to the balanced Latin square were  
20 representative of the full set of combinations. Each model was run for 1,000  
21 replications, always specifying an initial abundance of 5 squirrels (3 adult females and  
22 2 adult males) in the patch where they were originally liberated in 1973.

23 We first modelled the spatial distribution of the squirrels over 31 years,  
24 representing its spread from 1973 to 2004, in order to compare the predicted patch

1 occupancy to that observed in field reconnaissance in 2004 (Guichón et al. 2005).  
2 This comparison provided a valid test of the model inputs, because the dataset of  
3 observations was independent of the one used to generate the model. Although the  
4 observed data are limited to presence/absence at a single point in time, they are the  
5 only available information for the species in Argentina, and provide a useful reference  
6 against which to discard significant under- or overestimations of regional occupancy.

7 All of the patches lying within the observed distribution of squirrels in 2004  
8 were classified as ‘occupied’, which yielded 114 km<sup>2</sup> of occupied suitable habitat.  
9 The total area predicted to be invaded after 31 years was calculated by adding up the  
10 areas of the patches that had an average of at least 5 individuals at the end of the  
11 simulation period. The ‘simple matching coefficient’ (SMC, Krebs 1999) was used to  
12 measure the similarity between the observed area of distribution and the area  
13 predicted to be occupied by squirrels under the different parameter combinations  
14 (Rushton et al. 1997). To this end, the proportion of pixels with matching predicted  
15 and observed occupancy was calculated as:  $(a+b)/c$ , where  $a$  is the number of pixels  
16 predicted and observed to be occupied (positive matches),  $b$  is the number of pixels  
17 predicted and observed to be unoccupied (negative matches), and  $c$  is the total number  
18 of pixels of suitable habitat and thus belonging to a patch (matches + mismatches).

### 19 Prediction of expansion and alternative management scenarios

20 The 2 models that most closely matched the observed spread of squirrels to date  
21 were selected to predict the dynamics of invasion over the following 20 years (i.e.,  
22 over a total simulation period of 51 years up to 2024), under different management  
23 scenarios taking effect within the next 10 years. Longer simulation periods were also  
24 analysed to address particular questions, such as the year of invasion of Otamendi.

1           The first scenario described a continuation of the present condition of no  
2 systematic management. We then simulated 2 control strategies to slow down the  
3 expansion of squirrels: (1) increase mortality by removing individuals; (2) decrease  
4 available resources of food and refuge by a reduction in patch carrying capacities (see  
5 the Discussion for an appraisal of ways to implement these strategies). We simulated  
6 the removal of squirrels for 10 consecutive years from 2008 in the form of yearly culls  
7 from each population of a small amount more than the maximum sustainable yield  
8 ( $MSY \times 1.1$ , where MSY is attained at half carrying capacity under logistic growth).  
9 We simulated a reduction in available patch resources from 2008 onwards by halving  
10 the carrying capacity of each patch. These management actions were modelled for  
11 three areas of implementation: (1) all patches; (2) a priority area of 59 patches  
12 selected on the basis of the direction of the wave of advance and of the conservation  
13 value of the area beyond the front of invasion at Otamendi Natural Reserve (see  
14 Results); and (3) 2 central patches selected because of their location and large size  
15 (the release patch and a neighbouring patch that summed up 54% of the invaded area  
16 by 2004), where local producers would be more willing to control squirrels. Invaded  
17 area is expressed as the sum of all habitats occupied by at least 5 squirrels on average.  
18 Spread rate was estimated following Andow et al. (1993) from the square-root of the  
19 mean  $R_i^2$  where  $R_i$  are the shortest and longest radial increases in the minimum  
20 convex polygon encompassing the invaded area (Jenness 2004).

## 21 **Results**

22           The habitat suitability map shows the highly fragmented habitat occupied by the  
23 Red-Bellied Squirrel in the Pampas region, with contiguous tracts only in the north-  
24 eastern urbanised area (Fig. 1). The  $90 \times 90$ -km<sup>2</sup> study area encompasses 1,117 km<sup>2</sup> of  
25 suitable habitat (14% of total area), which partitioned into 353 km<sup>2</sup> of woodland and

1 764 km<sup>2</sup> of residential, suburban, and urban areas. This suitable habitat contained 497  
2 patches with  $K > 75$ , ranging in size from 0.09 to 95 km<sup>2</sup> and covering 1,027 km<sup>2</sup>  
3 (13% of total area). All of these patches had some positive dispersal rate under the set  
4 of tested models, indicating some degree of connectivity for all.

#### 5 ***Validation of the model***

6 Medium and high estimates of population growth ( $\lambda$ ) overestimated the area of  
7 squirrel occupation after the first 31 years, in comparison to the observed distribution  
8 of 114 km<sup>2</sup> (Table 2). The congruence of predicted and observed areas was relatively  
9 high for the model with low  $\lambda$ , regardless of dispersal distance (SMC = 0.96). The  
10 inclusion of density-dependent dispersal increased similarity between observed and  
11 predicted areas of occupancy in all models (Table 2), yielding lowest differences  
12 between observed and predicted occupied areas for models 2 and 3 with low  $\lambda$ , and  
13 model 4 with medium  $\lambda$ . Since the predictions of models 3 and 4 fell just either side of  
14 the observed area, and both had high SMCs, we used these 2 models for predicting the  
15 expansion of squirrels into the future. Model 3 yielded slightly conservative projects  
16 from low  $\lambda$  combined with a high mean, and low maximum, dispersal distance; model  
17 4 slightly overestimated the likely rate of spread from a higher  $\lambda$  combined with low  
18 mean and maximum dispersal distances.

#### 19 ***Prediction of expansion and alternative management scenarios***

20 In the absence of control measures, the models predicted expansion to  
21 occupancy of 707 to 808 km<sup>2</sup> of suitable habitat in the 20 years following 2004 (Table  
22 3) within a total convex polygon of 3,102 to 3,308 km<sup>2</sup>. The area invaded by 2024  
23 comprises 63-72% of habitat susceptible to colonisation within the 90x90 km map,  
24 including most of the available urban settlements and the forests of the Delta Region



1 with natural and commercial afforestations (Fig. 2). In particular, the models  
2 predicted invasion of the Otamendi Natural Reserve in the next 16 to 24 years (Table  
3 3). Given that the predicted expansion is mainly north-easterly and Otamendi is a first  
4 step into the Delta Region in this direction, we selected a priority area (see Fig. 2)  
5 where urgent management actions could be conducted to slow or prevent colonisation  
6 in this direction, until a long-term strategy can be undertaken with broad consensus.  
7 Culling squirrels in this area was at least as effective at delaying their arrival in  
8 Otamendi as culling in all patches, and more effective than culling the central area  
9 which did not have any delaying effect (Table 3). Habitat removal to reduce  $K$  only  
10 shortened the time to reach Otamendi, particularly when applied in the priority area  
11 (Table 3).

12 The models predicted a relatively slow expansion in occupancy during the first  
13 34-37 years after liberation in 1973 and then a steep increase in the invaded area (Fig.  
14 3a). This indicates that the spread of squirrels may be at the cusp of an explosive  
15 expansion. The rate of spread shows a tendency to increase (Fig. 3b), though large  
16 variations are observed (mean  $\pm$  SD, model 3:  $1.28 \pm 1.14$  km per year and model 4:  
17  $1.44 \pm 1.23$  km per year).

18 The increase in abundance was rather flat for the first 20 years of exponential  
19 growth, and then increased dramatically (Fig. 4). High numbers predicted by model 4  
20 showed signs of an intermediate plateau 30 to 40 years into the invasion, in response  
21 to density-dependent regulation within colonised patches (Fig. 3a and 4b). Culling and  
22 reducing  $K$  resulted in broadly similar patterns of predicted abundance, with the  
23 former tending to outperform the latter (Fig. 4). Numbers of squirrels and occupancy  
24 area could be significantly diminished by culling each population with a constant  
25 harvest for 10 years, though populations would start to recover immediately upon

1 interruption of the harvest (Fig. 4, note the particularly sharp reversal in fortune of the  
2 'cull all' scenario when culling stops after 10 years). It would take more than 30 years  
3 (model 3) or 60 years (model 4) of continuous harvest in all patches to eradicate  
4 squirrels. No significant advantages result from reducing the number of squirrels in  
5 central patches though concentrating efforts in a priority area would restrict  
6 abundance and area of occupancy while delaying the predicted invasion of Otamendi  
7 by 10 to 15 years (Table 3). Decreasing the carrying capacity of all or only priority  
8 patches has an unwanted effect of increasing the spread rate, such that Otamendi is  
9 reached before the no-action scenario (Table 3).

## 10 **Discussion**

11 Range expansion and squirrel numbers have grown slowly for the first 2 to 3  
12 decades after liberation, but the models predict a steep increase in abundance and  
13 regional distribution over coming years. The pattern of rapid growth following slow  
14 establishment is typical of the exponential expansion of many introduced species  
15 (Crooks and Soulé 1999). This is the first attempt to predict future expansion of the  
16 Red Bellied Squirrel in Argentina and we acknowledge some limitations in the  
17 construction and validation of the model, mainly due to sparse life history and  
18 population data for our study area. As with other studies using sparse data (e.g., Lurz  
19 et al. 2001, Tattoni et al. 2006), our input parameters for models are estimated from  
20 diverse sources of empirical information on behavioural and population ecology from  
21 the squirrel's native and exotic ranges. The use of a Latin Square of simulations at  
22 least ensures realistic orders of magnitude for the invasion dynamics of this species.

23 Density-dependent dispersal reduced predicted range expansion when  
24 population rate of increase was medium and high but increased predicted spread for  
25 low values, and even showed a rescue effect from extinction when lower rates were

1 used (e.g.  $\lambda = 1.23$ , unpublished outputs). These results are consistent with the  
2 population regulating effects of density dependence (e.g., Akçakaya et al. 2007). The  
3 first priority for improving model predictions should be to obtain local and seasonal  
4 estimates of reproduction and survival rates in different habitat types (work in  
5 progress).

6         One of the principal aims of this modelling approach is to make  
7 recommendations based on objective evaluation of the success of alternative control  
8 actions. Management measures could be oriented towards mitigating damage,  
9 restricting colonization of new areas, regulating abundance, or complete eradication.  
10 The probability of achieving the desired result can be evaluated by simulating  
11 population responses to different control measures. Our results show that reducing the  
12 carrying capacity of the habitat would lower squirrel numbers but an unwanted  
13 accelerating spread would lead to a faster colonization of a key conservation area  
14 (Otamendi). Non-lethal techniques have a potentially greater efficiency than lethal  
15 control, since they avoid the density-dependent consequences that can follow culling,  
16 and are likely to be seen as more acceptable in welfare and ethical terms (Barr et al.  
17 2002, Baker et al. 2007). Decreasing the carrying capacity of suitable patches could  
18 be accomplished by reducing accessibility (e.g., place sheet metal and cones to  
19 impede or reduce use of trees and cables), persistence (e.g., nest destruction), and  
20 suitability of resources (e.g., pruning trees to thin canopy and reduce suitability for  
21 nesting; replace trees used for food and nesting by alternative, less suitable tree  
22 species). Regardless of the viability of implementing this action, however, it must be  
23 rejected as a viable option because of the predicted acceleration of the invasion front.

24         In contrast, model predictions show that culling squirrels could be an effective  
25 measure to restrict invasion and reduce abundance, provided it targets priority areas.

1 Culling all patches would have the largest impact on numbers and range distribution,  
2 but implementing such a vast campaign has doubtful feasibility. Culling populations  
3 only in central patches could create a vacuum that would rapidly be filled by local  
4 growth and dispersal from neighbouring patches. However, concentrating effort and  
5 resources in the priority area would delay colonization of Otamendi. This Natural  
6 Reserve would be the first step into the Paraná River Delta which hosts a unique and  
7 biodiverse region of riparian forests represented by both temperate and subtropical  
8 species and also important commercial forests (Malvárez et al. 1999). Removing  
9 squirrels would require public awareness campaigns and long-term commitment of  
10 governmental authorities. Broad public support should be achieved through a  
11 communication strategy informing about the problem and addressing ethical issues,  
12 particularly given the charismatic appearance of the squirrel (Barr et al. 2002,  
13 Bertolino and Genovesi 2003). Additionally, an adequate legal basis is needed to  
14 initiate control actions, given that local authorities refuse to implement any measure  
15 until the species is declared a pest. A rapid response to new invasions is the best case  
16 scenario to prevent vast invasions, when technical, financial, and logistic needs are  
17 more easily achieved (Mack et al. 2000, Genovesi 2005). When control actions are  
18 delayed, eradication of introduced mammals is usually not viable for financial,  
19 technical, or social reasons. Actions to diminish negative effects caused by already  
20 established exotic species are expensive and not always successful, as in the case of  
21 attempts in the United Kingdom to control *S. carolinensis* in order to protect *S.*  
22 *vulgaris* and reduce bark stripping (Sheail 1999, Huxley 2003, Lawton & Rochford  
23 2007). Nevertheless, localized intensive control of introduced mammals in selected  
24 areas may help to reduce some negative effects and slower their spread (Bertolino and  
25 Genovesi 2003).

1           Because we seldom understand ecological systems completely, applications of  
2 models need to be flexible and management must be able to adapt to new information  
3 (Lyman et al. 2002). Our assessment of the most realistic and effective long-term  
4 action is to target culling in priority areas, and to mitigate damage in commercial  
5 plantations through local control. It is important that all such operations are closely  
6 monitored, and that the predictive models are updated with information on control  
7 actions and on vital rates. With more empirical information it will be possible to carry  
8 out sensitivity analyses to determine the key variables that drive expansion, which can  
9 then guide the collection of future data.

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Table 1. Parameter estimates used as model inputs (with range of values in parenthesis where applicable).

Parameter	Classification	Value (range)
Habitat quality ( $K$ )	Woodland	8 ind.ha <sup>-1</sup>
	Residential	6 ind.ha <sup>-1</sup>
	Suburban	4 ind.ha <sup>-1</sup>
	Urban	2 ind.ha <sup>-1</sup>
	Non-suitable	0 ind.ha <sup>-1</sup>
Survival rates ( $p_x$ )	Juveniles	0.8 (0.72-0.88)
	Yearlings	0.6 (0.54-0.66)
	Adults	0.3 (0.27-0.30)
Fecundity rates ( $F_x$ )	Juveniles	0.6 (0.54-0.66)
	Yearlings	1.2 (1.08-1.32)
	Adults	1.2 (1.08-1.32)
Finite rate of increase ( $\lambda$ )		1.53 year <sup>-1</sup> (1.38-1.68)
Mean dispersal distances		1 km (0.5-1.5)
Maximum dispersal distances		5 km (2.5-7.5)
Male:female relative dispersal	Juveniles	1.0:0.4
	Yearlings	0.7:0.3
	Adults	0.4:0.2
Density-dependent dispersal constant		0 (0.001)

Table 2. Model inputs to the Latin Square design: finite rate of increase ( $\lambda$ ), mean dispersal distance (MeanDD), and maximum dispersal distance (MaxDD), predicted area of occupancy after 31 years of invasion (difference from observed occupancy of 114 km<sup>2</sup> in 2004 in parentheses), and simple matching coefficient (SMC). The last 2 columns show predictions for models incorporating density dependent dispersal.

	$\lambda$	MeanDD	MaxDD	Area (km <sup>2</sup> )	SMC	Density dependent dispersal	
						Area (km <sup>2</sup> )	SMC
1	1.38	0.5	7.5	73 (-41)	0.96	88 (-26)	0.97
2		1.0	5.0	72 (-42)	0.96	111 (-3)	0.96
3		1.5	2.5	73 (-41)	0.96	93 (-21)	0.98
4	1.53	0.5	2.5	374 (+260)	0.70	124 (+10)	0.94
5		1.0	7.5	672 (+558)	0.46	283 (+169)	0.84
6		1.5	5.0	447 (+333)	0.68	404 (+290)	0.72
7	1.68	0.5	5.0	804 (+690)	0.33	361 (+247)	0.72
8		1.0	2.5	825 (+711)	0.31	412 (+298)	0.70
9		1.5	7.5	934 (+820)	0.20	493 (+379)	0.64

Table 3. Predicted expansion and abundance ( $\pm$  SD) of squirrels by the year 2024, and time to reach the Otamendi Natural Reserve under alternative management scenarios.

Scenario	Area	Area by 2024 (km <sup>2</sup> )		Abundance by 2024		Year reaches Otamendi	
		Model 3	Model 4	Model 3	Model 4	Model 3	Model 4
No control	-	707	808	55,251 (25,136)	122,557 (44,171)	2031	2023
Cull squirrels	all	140	381	17,525 (9854)	35,086 (9985)	2044	2038
	priority	288	408	36,277 (13,861)	40,380 (10,591)	2041	2038
	central	707	810	45,173 (24,755)	121,259 (42,752)	2031	2023
Reduce <i>K</i>	all	826	893	30,570 (14,173)	74,543 (24,526)	2024	2016
	priority	818	811	39,704 (16,800)	99,972 (29,096)	2021	2016
	central	713	793	45,712 (21,811)	110,861 (39,299)	2029	2023

Figure 1. (a) Location of study site in Argentina, (b) square showing the study area with the release point of the squirrels in the centre (×), the area covered by the city of Buenos Aires and its large suburbs, the Lower Delta Region of the Paraná River, Otamendi Natural Reserve, and the central and priority areas used for modelling alternative management scenarios, and (c) map showing suitable habitat for squirrels (wooded habitat in black and urban habitats in grey) and the matrix of unsuitable habitat (non-forested areas in white), indicating the range expansion observed by 2004 (polygon).

Figure 2. Area of occupancy predicted under current scenario of no control by (a) model 3 and (b) model 4 in 2004 (black), 2014 (dark grey), and 2024 (light grey).

Figure 3. Model predictions under current scenario of no control: (a) range expansion since liberation of squirrels in 1973 (area estimated using the minimum convex polygon method) predicted by models 3 (dashed line) and 4 (solid line), and (b) spread rate (dots indicate new invaded area every 3 years after liberation with trend line showing predicted increase of colonisation speed) predicted by models 3 (open circles, dashed line) and 4 (black squares, solid line). Arrow indicates year 2007.

Figure 4. Predicted abundance of population by (a) model 3, and (b) and model 4 since liberation of squirrels in 1973, under alternative scenarios described in Methods (no control, black solid line; culling, dashed lines; reduce k, grey solid line). Standard deviations are not shown for graph clarity though values never reached zero, see Table 2 for SD of no control scenario. Note different scales of y-axes in (a) and (b). Arrows indicate year 2007.

Figure 1 – Guichón & Doncaster

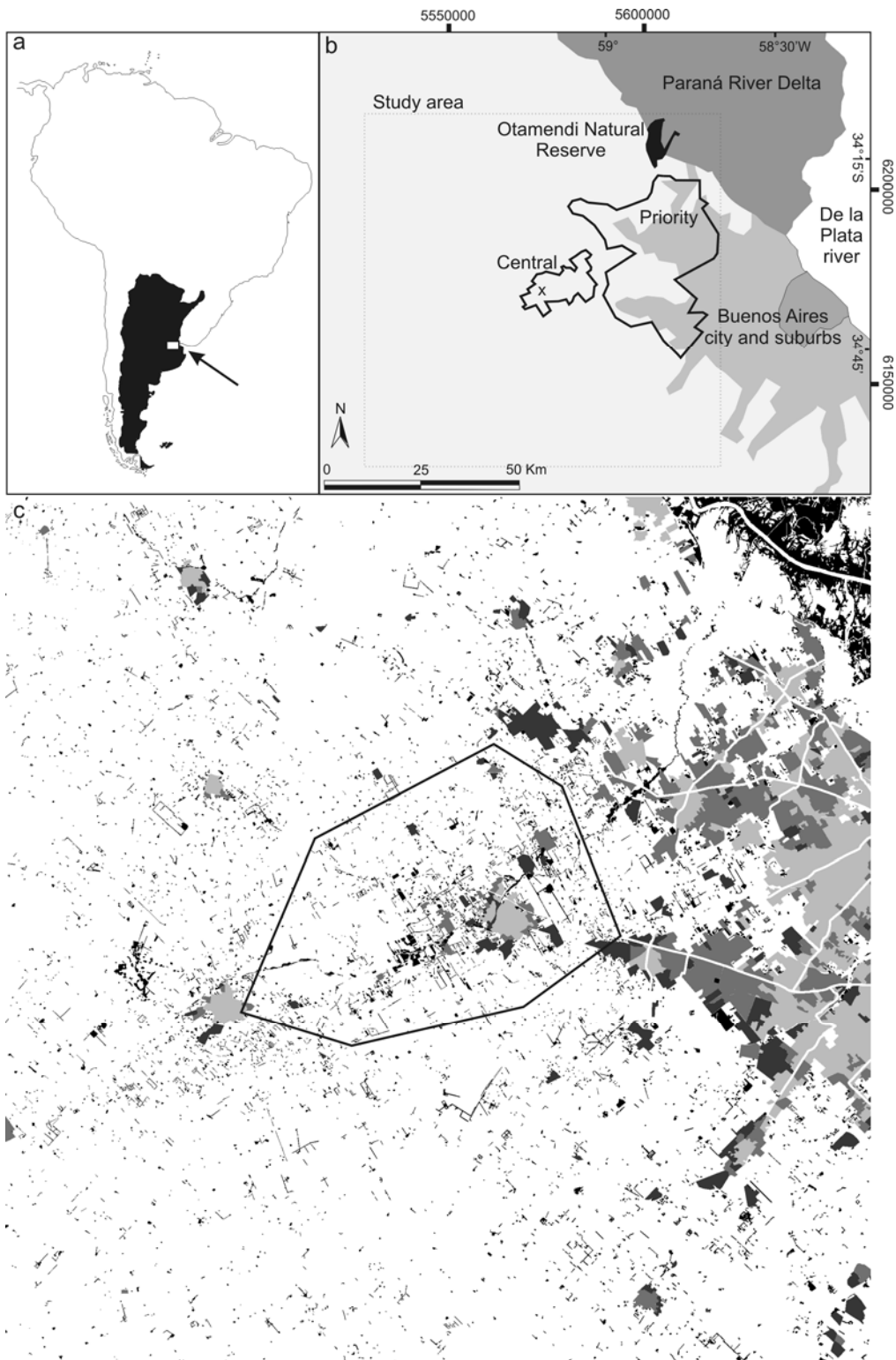




Figure 2 – Guichón & Doncaster

a



b

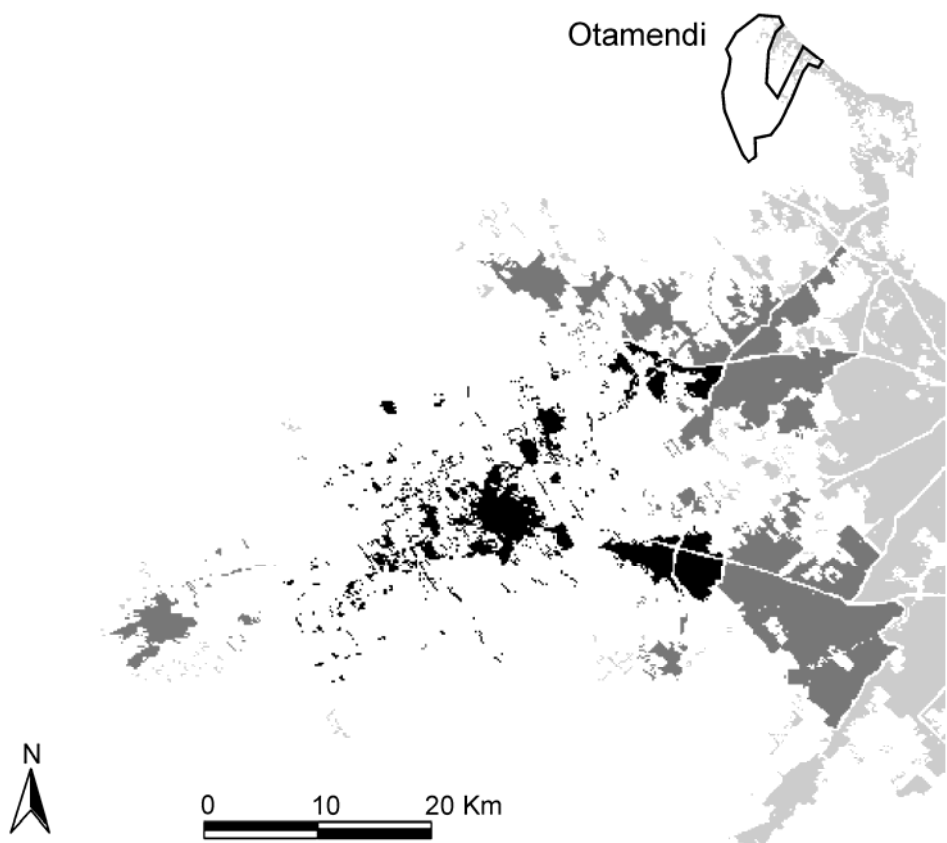


Figure 3 – Guichón & Doncaster

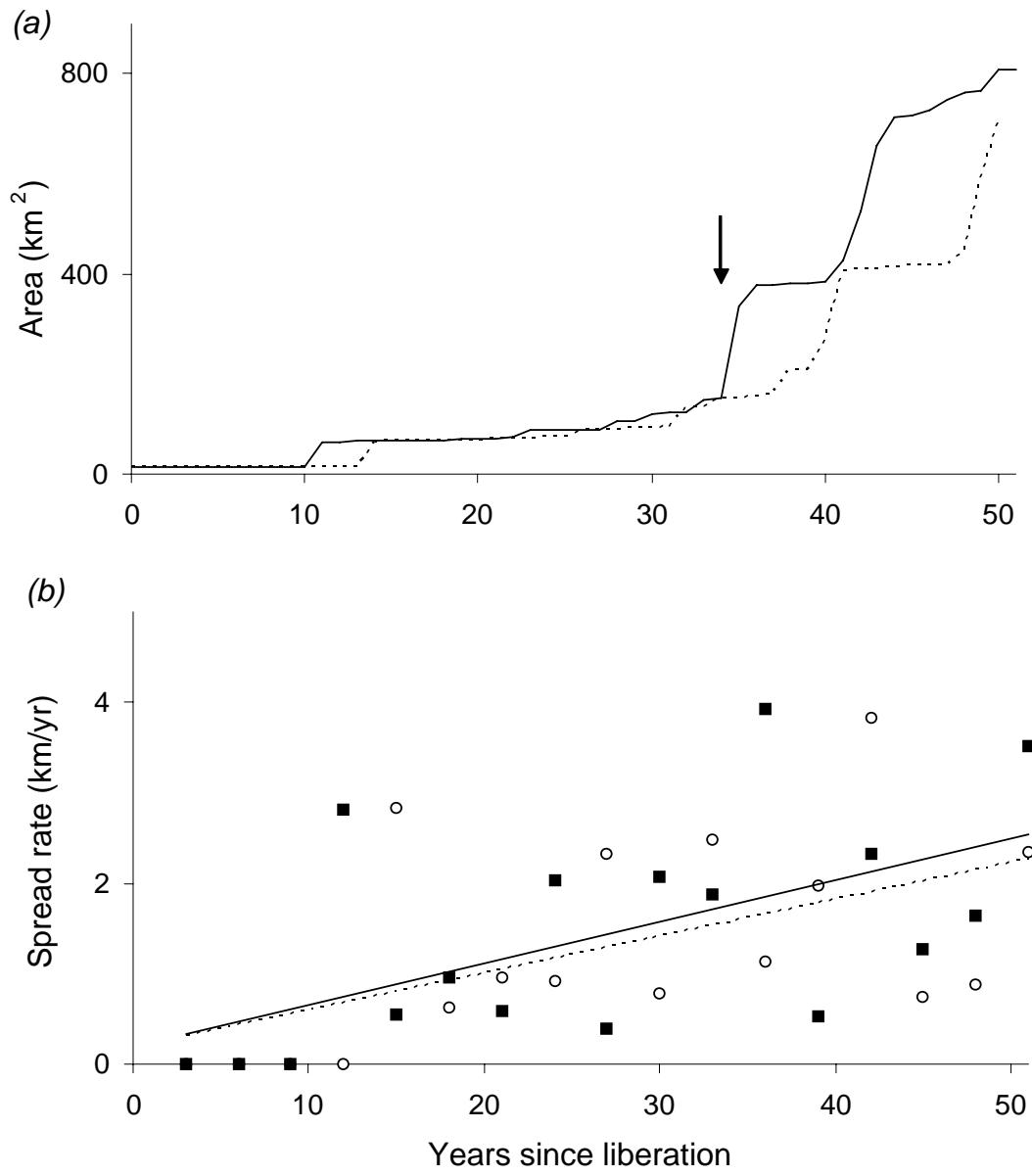


Figure 4 – Guichón & Doncaster

