Invasion of the Gran Canaria ravines ecosystems (Canary Islands) by the exotic species *Acacia farnesiana*

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Abstract The main objective of this study was to analyze if *Acacia farnesiana*, an introduced shrub from North and Central American tropics is spreading from areas in which it was introduced, and also to determine which animals operate as vectors for the shrub dispersion in the study area. The study site was located in southern Gran Canaria, one of the islands of the Canary Island archipelago. We selected six ravines in an area where approximately 40 plants of *A. farnesiana* were planted around 40 years ago. We analyzed the size structure of the population of *A. farnesiana* (density and biovolume) in the ravines, as well as its spatial distribution, in order to assess the degree of aggressiveness of this shrub in terms of spreading and expansion. Our results suggest that this shrub is spreading to new areas very quickly due to dispersion of seed by rabbits and an enhanced germination by the action of *Mimoseste mimosae*, a bruchid that decreases the germination period of the seed once it has separated from the fruit. Based on the results obtained in this study, we strongly suggest that managers of this area consider a management program to control further spread.

Keywords Invasive species · *Mimosestes mimosae* · Neyman–Scott process · Plant–animal interaction · Plant size distribution

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Introduction

The spread of invasive alien species can have a detrimental economic impact on human enterprises such as agriculture, livestock production, forestry (Shackleton et al. 2007), and tourist activities (Arévalo et al. 2005). Invasive species have been identified as one of the major threats to ecosystems and biodiversity (D'Antonio and Kark 2002), as well as human well-being (Pimentel 2002). At the present time, non-native plants are important components of plant communities in many parts of the world (Mack et al. 2000) and have become a global problem that requires an analysis and evaluation of the status of the threaten ecosystem. These analyses will help to adopt strategies for the eradication and control of invasive plants, where necessary.

Acacia farnesiana has been identified in some areas as an invasive species in pastures and shrublands, competing effectively with native species and significantly reducing productivity of invaded

ecosystems (Carmona et al. 2001). This shrub was introduced in Gran Canaria in the eighteenth century as an ornamental plant. Although it does not appear in some checklists of exotic plants species of the Canarian archipelago (Sanz-Elorza et al. 2005; Kunkel 1976), it is clearly identified as an invasive species in others (Izquierdo et al. 2004). Dispersion of *A. farnesiana* is directly affected by two introduced animal species that consume its fruits and seeds, the rabbit (*Oryctolagus cuniculus*) and a bruchid beetle (*Mimosestes mimosae*).

Spatial patterns are important characteristics of plant communities (Vacek and Lepš 1996) and can be used to analyze tree species replacement (Horn 1975; Woods 1979; Busing 1996), species regeneration (Condit et al. 1992), changes in plant communities after disturbance (Haase et al. 1997; Vacek and Lepš 1996; Richardson and Higgins 1998), adult juvenile relationships (He et al. 1997), and biological relationships among species, such as competition (Hatton 1989; Duncan 1991) and dispersion (Collins and Klahr 1991). Consequently, knowledge of the spatial distribution of plant species is critical for the management of natural areas (Moeur 1993) and for an understanding of the biological relationships between native and exotic plant species (Arévalo et al. 2005).

Islands have been considered areas with an intrinsically low resistance to invasions, as shown in Hawaii, Galápagos or Azores (Carlquist 1974; Silva and Smith 2004). Some conspicuous species such as *Pennisetum setaceum* (Pérez et al. 1999), *Agave americana*, *Oxalis pes-caprae*, or *Opuntia ficus-indica* have had a great invasion success in the Canaries, getting established in disturbed and natural areas. On the other hand, many introduced species remain in a restricted area, including tree species like *Pinus radiata*, *P. pinea*, or *Eucalyptus globulus*, planted in other world areas as well (Arévalo and Fernández-Palacios 2005; Arévalo et al. 2005).

The aim of this study was to evaluate the invasive status of *A. farnesiana* in the Gran Canaria Island, as well as to evaluate its interactions with animals that consume and disperse the seeds of this shrub in the area. We tested the hypothesis that *A. farnesiana* is invading new areas with similar environmental conditions to those where it is currently distributed, and its expansion is being assisted by exotic animal vectors.

Understanding the ecological circumstances under which particular factors promote invasions, and the interactions among factors favoring the invasion, can help predict future spreading and the potential impact of the invasion on native vegetation (Sanders et al. 2007). These results will help to implement control plans, if necessary, to contain this shrub within the already invaded areas.

Materials and methods

Study site

The study site was located in the southern part of Gran Canaria Island (Fig. 1) and was on the property of the company Costa Canaria de Veneguera, S.A. which limits access to the area in order to maintain a very low inflow of vehicles and people. In 1987, the main road was built, and some 40 individuals of *A. farnesiana* were planted and watered during the construction process.

The dominant bed rocks in the area are ignimbrites and riolític flows, sienites and traquites (ITGE 1990). The soils are petric calcisoil and, in the bottom of the ravines, litic leptosoils with a high rock component and poor presence of organic matter. In all the cases soils are less than 30–40 cm deep (Sánchez et al. 1995).

The climate of the area is considered EdB'3a' following the Thornthwaite classification, with a potential evapotranspiration of 850–1,000 mm year^{-1}, a long periodic drought and no humid season. Precipitation in the area is less than 300 mm.

The potential vegetation is a xerophyllic shrubland dominated by *Euforbia balsamifera* in the plateaus and *Plocama pendula*, *Schizogyne glaberrina*, *Klenia neriifolia*, *Launaea arborescens*, and *Lycium intricatum* along the bottoms and sides of the ravines. In areas with a higher presence of rock, it is possible to find very large *Euphorbia canariensis* individuals. Although the plant community in the area is under water stress, it is fairly well conserved and extensive; invasive species and some human disturbance are the main threats.

The species

The variety in the study site, native to Central America (Traveset 1991), is *A. farnesiana*, a medium-sized shrub with many spreading branches and basal stems. The leaves are alternate, bipinnately

Fig. 1 a Canary Islands archipelago and **b** map of Gran Canaria indicating the location of the study site where the study was developed. Also, potential invasible ravines similar in soil (rocky and poorly developed), climate (with an annual precipitation 50–250 mm, and high temperature, with an annual average >19°C), high insolation and intermittent flooding locate during the winter period are indicated in the map
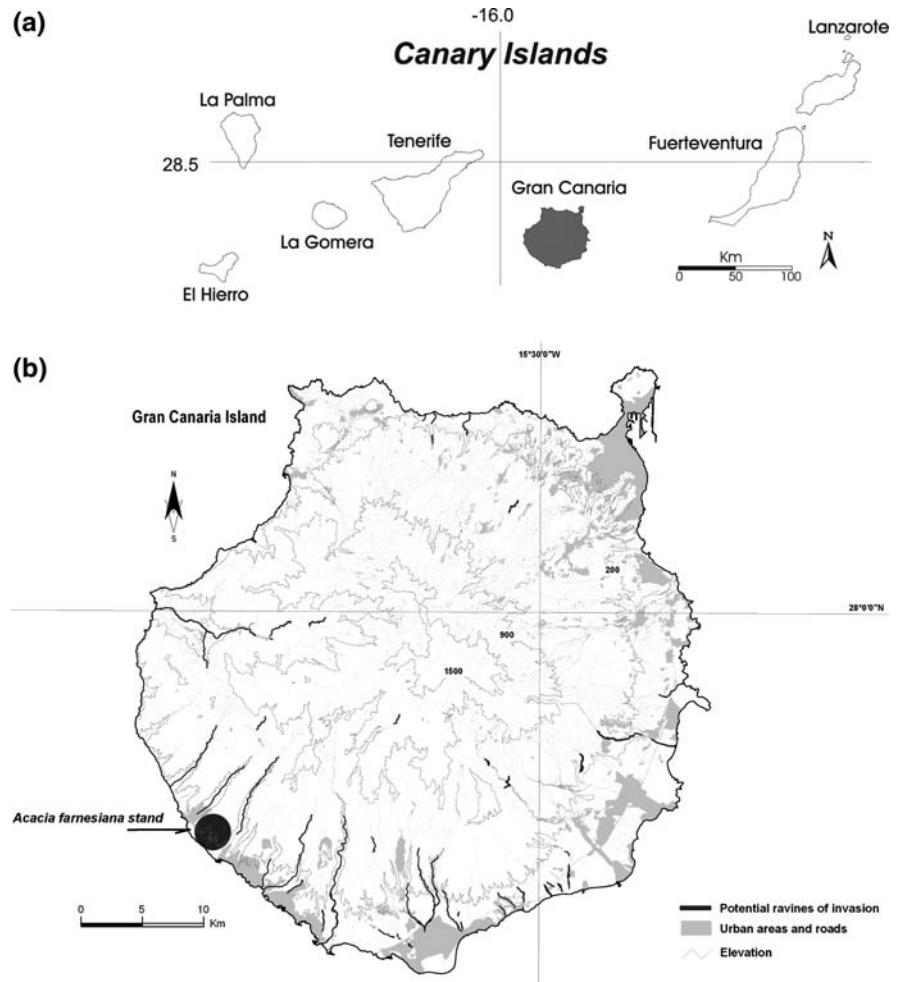
compound with two to six pairs of pinnae, each with 10–25 pairs of narrow leaflets 3–5 mm in length. The slightly zigzag twigs are dark brown with light-colored dots (lenticels) and paired spines, 3–20 mm in length at the nodes (Parrotta 1992). The older bark is also dark brown and smooth. Its bright yellow or orange flowers are produced over a period of 2–4 months, depending on locality. In Gran Canaria, the production of flowers is continuous during the year. The pods are attached to the plant for 3–4 weeks once they reach maturity, but we found that in our study site they can be attached for more than a year.

Design of the experiment

During 2004, we sampled the Acacia populations in six ravines in the study area. We counted all individuals found and classified them as adults (taller

than 50 cm, when they can produce fruits, pers. observation) or saplings (under 50 cm).

For adult shrubs, we recorded height and measured two diameters from the top of the plant: firstly, the longest one in terms of width and then the diameter lying perpendicular to the first. We also recorded the number of stems and the basal area of these stems at 10 cm from the base in each individual. For each individual we also recorded the presence of flowers or fruits (classified as mature, green or dried).

We mapped all the individuals using polar coordinates (distances and polar angle) starting, for the first one, at the top of the ravine, for which UTM coordinates were recorded. At the end, we extrapolated all the UTM coordinates for all individuals.

We collected mature pods from the shrubs (selecting randomly 15 individual's aprox.) in the six ravines over a 3-month period and recorded the

number of seeds attacked by the bruchid beetle *Mimosestes mimosae*, as well as the number of seeds per pod. We collected and opened, in the laboratory, a total of 270 pods (from 100 shrubs) to estimate infestation rate and number of seeds per pod.

In order to obtain preliminary information about germination rates of seeds infected by *Mimosestes mimosae* versus those not infested, we planted 50 seeds (25 infested and 25 clean) in a tray with sterilized soil and kept them in a greenhouse with constant humidity levels for 2 months.

Statistical analysis

We correlated the densities of individuals at each size class (based on biovolume; Fernández-Palacios and de los Santos 1996) in the six ravines in order to determine the relationship between the size class distribution between the ravines, using the Pearson correlation coefficient, for $P < 0.05$ and $n = 8$, (the eight size classes being: saplings, <5, 5–10, 10–50, 50–100, 100–200, 200–300, >300 m^3). We also correlated the total biovolume in each size class between the ravines using the Pearson correlation coefficient, for $P < 0.05$ and $n = 7$, because the biovolume of saplings was not estimated.

Our prediction that the spatial distribution of the individuals of *A. farnesiana* would follow a homogeneous Neyman–Scott process in the six ravines lead us to estimate the intensity of parent events (ρ) and the number of offspring per parents (σ) according to a Poisson distribution (Diggle 1983). If the number of offspring follows a Poisson distribution and the location of offspring, n relation to the parent individual, has a bivariate, Gaussian distribution, the offspring should follow a Neyman–Scott process. The *K*-function and the paired correlation function $g(r)$ for the Neyman–Scott process are given by:

$$K(r, \sigma, \rho) = \pi r^2 + \frac{1 - \exp\left(-\frac{r^2}{4\sigma^2}\right)}{\rho}$$

$$g(r, \sigma, p) = 1 + \frac{\exp\left(-\frac{r^2}{4\sigma^2}\right)}{4\pi\sigma^2\rho}$$

with r representing the distance at which the calculations are made. After several approximations, ρ and σ are estimated from the data and with these parameters a null model is constructed with the software Programita

(Weigand 2004). The 95% confidence envelopes were constructed with 19 iterations of the null model.

The $K_1(t)$ function is transformed into a function $L_1(t)$ as follows:

$$L_1(t) = \left(k_1(t)/\pi\right)^{0.5}$$

Transforming $K_1(t)$ to W_k allowed us to graphically present the results by displaying the pattern across multiple scales. The indicator W_k is used to estimate the departure of individuals' distributions from that of a homogenous Neyman–Scott process (Szwagrzyk 1990).

$$W_k = \frac{\max_t(L_1(t) - t)}{S} * \text{sign}(L_1(t) - t)$$

$\max_t(L_1(t) - t) = $ maximal deviation of the $L_1(t)$

function from CSR

$S = $ width of the confidence interval; sign $(L_1(t) - t) = -1.0$, if $L_1(t) - t < 0$; sign $(L_1(t) - t) = 1.0$, if $L_1(t) - t > 0$.

Values of $W_k < -1$ indicate regular uniform distribution of the individuals, while values >1 indicate clumped distribution or aggregation relative to the homogenous Neyman–Scott value. When the spatial distribution of stems at a given distance does not differ from a random distribution (using a Monte Carlo test with 19 iterations of randomly generated x, y coordinates), the value of the curve at that distance is zero. When significant aggregation was detected, the values of the curve were positive and higher than 1 (the value increased depending on the differences among the expected value and observed value). Negative values $W_k < -1$ indicate significant repulsion at the given distance.

In the analysis, we used 1 m of grid for the calculations and we evaluated spatial distribution up to 50 m (except for ravine 3, which was narrower than the others and was only analyzed up to 33 m).

Basic statistical methods followed Zar (1984) and were implemented using the SPSS statistical package (SPSS 1986).

Results

The number of individuals ranged from 346 in ravine 2 to 24 in ravine 6. The number of saplings were

much more abundant in ravines 1 and 2 (37 and 110) than in the rest, with less than 24 in all of them (0 saplings in ravine 6).

Ravines revealed strong differences with respect the structure and the size (calculated using the longest measurement as that of the two longest sides of the rectangle, and then taking the longest perpendicular measurement as the shorter two sides). Ravines 4 and 5 were the largest in size and length, while ravines 2 and 6 were the smallest (Table 1). The average number of seeds per pod was 16.1, with 25% of pods infested with *Mimosestes mimosae*. Of these pods, 25% of the seeds were infected and we were able to visually detect the insect in 8% of pods.

The germination experiment revealed that after 1 month of seeding, only 4% of the seeds without infestation germinated, whereas 75% of the infested seeds germinated. One of the factors that made some of the infested seeds unable to germinate was the destruction of embryos when beetles consumed the seed.

We detected seeds and saplings of *A. farnesiana* in areas where rabbits use for deposition of feces, which contained seeds of this shrub, revealing the ingestion of seeds of *A. farnesiana* by rabbits. Ten of the seeds found in feces were placed in a greenhouse to germinate; this trial revealed the role of rabbits as a expansion vector for *A. farnesiana*.

The percentage of individuals in the different biovolume classes showed differences between ravines, revealing that only ravines 2 and 5 had individuals over 300 m^3 that acted as a source of individuals. Ravines 4 and 6 did not have individuals over 200 m^3, and ravine 6 did not have any saplings (Fig. 2a). When the density of individuals in each ravine was normalized to the size of the ravine, the results showed that ravines 1 and 2 (and to a lesser extent ravine 5) acted as sources, based on the higher densities of the sapling class present (Fig. 2b).

The percentage of total biovolume in each biovolume class did not differ from the percentage of the densities in the classes, offering a similar structure of the distribution of individuals in the biovolume classes (Fig. 2c). When these values of biovolume were normalized according to the ravin's area, it was again found that ravines 1 and 2 (and to a lesser extent 5) had the highest population of *A. farnesiana*, and thus an important source of propagules. This is probably due to the fact that ravines 1 and 2 are closer to the area where *A. farnesiana* was first planted in the island. When we correlated the amount of biovolume in different size classes for the different ravines, there was a significant relationship in ravines 4 and 6 (ravins most recently invaded by *A. farnesiana*) while the others acted as sources of propagules (Table 2).

All the spatial distributions of the individuals in the ravines followed a homogenous Neyman–Scott process up to 20–25 m. Based on that, the number of parent events in each ravine varied from 6 to 11 (and the number of individuals in each ravin ranged between 7 and 16), and was smaller in the ravines where abundance of *A. farnesiana* was lower (Table 1). Although we detected some rejections regarding the null Neyman–Scott model, these were not significant. In ravine 4, this rejection was significant over 40 m, and may be related to the structure of the ravine, which is very broad and permits wide dispersion of all biovolume groups. In ravine 6, there was no departure from the spatial distribution model at any distance (Fig. 3).

Discussion

In general, *A. farnesiana* can be considered a fast colonizing invasive shrub of degraded areas in southern Gran Canaria. Although this shrub is barochory disperse along the ravines from where it was planted, an introduced dispersion vector, the

Table 1 Number of parents identify by the Neyman–Scott model distribution and the size in individuals created by the group

Ravine	Number	Size in	Dimensions		
	Parents	Individuals	(m)		Area (ha)
1	11	10	240	151	3.62
2	9	7	114	68	0.77
3	5	7	220	211	4.64
4	8	15	590	395	23.3
5	11	16	477	367	17.5
6	6	6	214	123	2.63

Also, dimensions of the ravines are indicated reflecting the minimum rectangle that encloses the ravine and the total area estimate

(a) % Density

(b) Density/ha

(c) % Biovolume
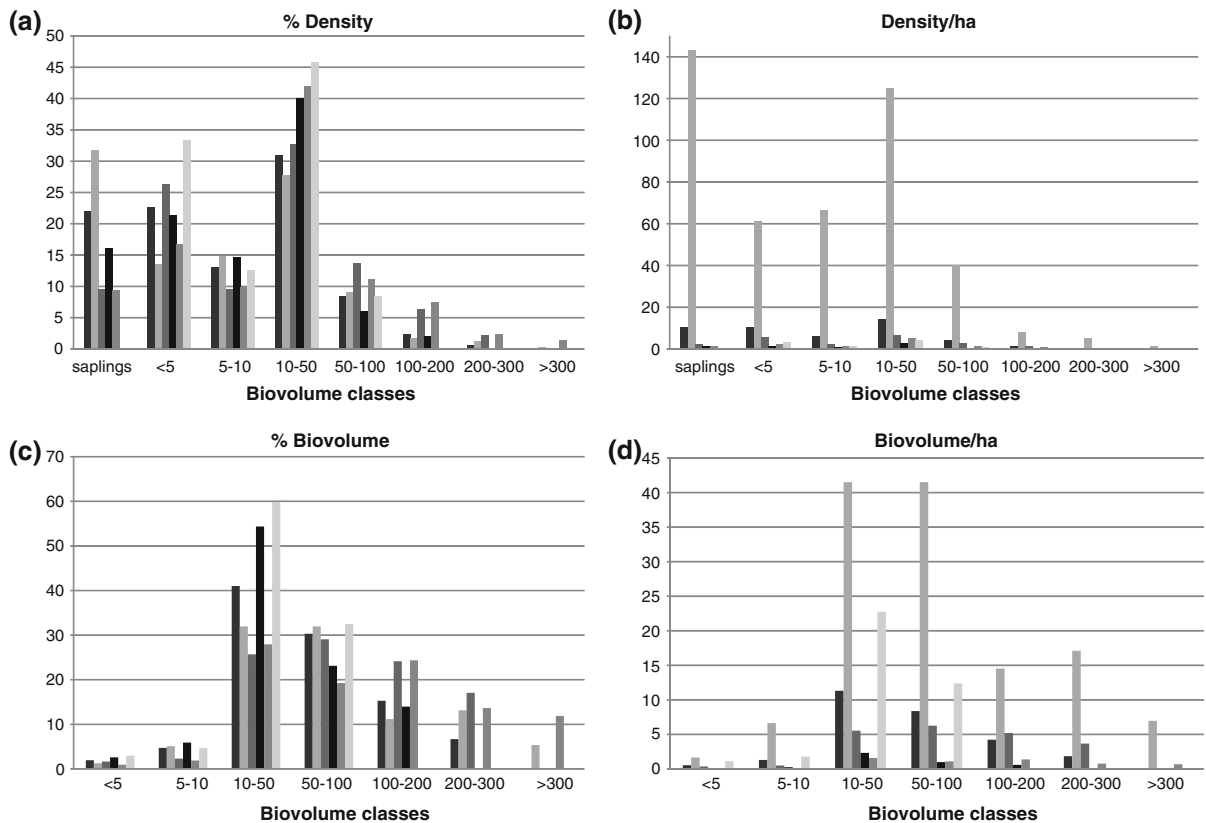
(d) Biovolume/ha

Fig. 2 **a** Percentage of the number of individuals in each biovolume class for the six ravines; **b** Number of individuals in each biovolume class normalize to the size of the ravine (assimilate to a rectangle of minimum side enclosing the total);

c Percentage of the total biovolume in each biovolume class and **d** biovolume in each class normalize to the size of the ravine

Table 2 Spearman correlation coefficient (r1: ravine 1, r2: ravine 2…) for biovolume in different size classes used among ravines

	r1	r2	r3	r4	r5	r6
r1	1	**	**	**	**	ns
r2	0.847	1	**	ns	**	ns
r3	0.964	0.847	1	*	*	ns
r4	0.847	0.564	0.811	1	ns	*
r5	0.857	0.865	0.786	0.667	1	ns
r6	0.593	0.393	0.556	0.785	0.259	1

rabbit, is favoring the colonization of other areas that have similar topographic conditions. The colonization of woody plants favored by the introduction of another vector is well known. Examples include *Pinus pinea* and squirrels in South Africa (Grotkopp et al. 2004) and livestock and Acacia species in Chile

(Fuentes et al. 1989), making exotic herbivores common drivers of plant invasion (Holmgren 2002).

An additional factor favoring the fast invasion of *A. farnesiana* is the presence of *Mimosestes mimosae* in the pods, whose presence increased germination of seed by 60%. This is in sharp contrast with other bruchid beetles and Acacia species, where infestation negatively affects the reproduction of these shrubs, and causes economic losses in plantations of these plants (Miller 1996; Sanders et al. 2007). In the case of *A. farneasiana* and *M. mimosae*, we found a synergistic relationship among these species. In areas where *A. farnesiana* is considered native, infestations by *M. nubigens* and *M. mimosae* negatively affects the reproduction of shrubs, but the parasite populations are controlled by other parasitoids or unfavorable weather conditions (Traveset 1991). We do not discard the presence of parasitoids of the bruchids in our study area, but we detected a high infestation

Acacia farnesiana (m)
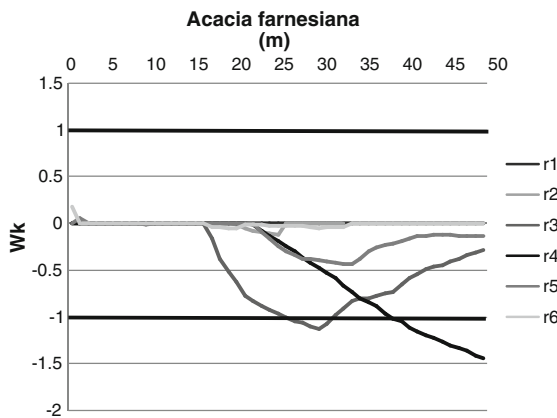
Fig. 3 Values of the W_k index for de individuals of *Acacia farnesiana* in the six ravines. $W_k > 1$ indicates a significantly clumped distribution with respect a Neyman–Scott distribution function, $W_k > 1$ and < 0.333 indicates an intermediate state between clumping and Neyman–Scott process. $W_k < 1$ indicates a significantly aggregated distribution, and $W_k > -1$ and < -0.333 indicates and intermediate stage between aggregation and Neyman–Scott process

level of *M. mimosae* in the shrubs, together with a regular distribution of these beetles (around 25% of all pods). As long as the weather remains mild in the area, we do not expect bruchid control by heat or desiccation to occur.

In other oceanic islands, *A. farnesiana* has been identified as a major threat to native biodiversity (Tassin et al. 2006) based on the level of its impact on the landscape and its ability to rapidly become a pest.

An important effect of invasive *Acacias* is their ability to fix N which has a large impact on soil nutrient cycling (e.g., *A. saligna*, Yelenik et al. (2007); *A. cyclops*, Stock et al. (1995). Our results can suggest a significant change in the N cycle as long as plant traits are good predictors of ecosystem level effects of plant species as in other studies (Eviner and Chapin 2003; Eviner 2004).

Given that a large portion of Gran Canaria has similar environmental conditions to those of the invaded area (e.g., soil, precipitation) and also that propagule formation is very high, according to our results, we foresee that the invasion of this shrub will continue at a very fast rate. It is worth nothing that in the ravines studied there are currently over 1,000 Acacia plants from about 40 individuals planted in the late 1960s along a road that crossed ravine one and two.

We currently have a situation in which the source of propagules in several ravines is multiplying, and this is one of the most important factors that determine the success of an invasion (Rejmanek et al. 2005). Also, *A. farnesiana* have not historically been present in the Canary Islands and legumes are not found in the xeric and low altitude areas of the island. These historical facts hint toward another important factor that facilitates the success of an invasion: the lack of general or specific herbivores (Lockwood et al. 2001). The presence of a vertebrate disperser (in this case the rabbit) can also increase the success of invaders in disturbed as well as in undisturbed habitats (Strasberg 1995; Richardson et al. 2000). The reduction of the germination period is very important for legumes, lasting for a long time in many species of this family, and it is an important factor in the colonization of new areas.

The presence of *Mimosestes mimosae* can be considered fundamental in the success of this plant as an invader of Gran Canaria. The Z* function, a function base in the mass seed values and vertebrates dispersals (Rejmanek and Richardson 1996) offered values above zero, and given the presence of vertebrate dispersers, the prediction of this species following this function is: "very likely to be invasive". We strongly suggest that the land manager of this area consider a monitoring procedure of this species with experts in plant biology and ecology and a control program to limit the spread of *A. farnesiana*. Its presence is currently very low in many ravines, but it is spreading rapidly.

Control of this species will require ecological restoration in order to favor the establishment of native vegetation that has been largely disturbed by past management practices (Suárez 1987; Súarez 1997). These recommendations, based on the ecology and biology of the species, are also in accordance with the laws of the Canarian Government, which require exotic species to be eliminated.

References

Arévalo JR, Fernández-Palacios JM (2005) Gradient analysis of exotic *Pinus radiata* plantations and potential restoration of natural vegetation in Tenerife, Canary Islands (Spain). Acta Oecologica 27:1–8

Arévalo JR, Naranjo A, Salas M (2005) Regeneration in a mixed stand of native *Pinus canariensis* and introduce *Pinus pinea* species. Acta Oecologica 28:87–94

Batista JLF, Maguire DA (1998) Modeling the spatial structure of tropical forests. For Ecol Manag 110:293–314

Busing RT (1996) Estimation of tree replacement in an Appalachian Picea-Abies forest. J Veg Sci 7:685–694

Carlquist S (1974) Island biology. Columbia University Press, Columbia

Carmona R, Carvalho BS, Carlvalho R (2001) Controle de *Acacia farnesiana* e de *Mimosa pteridofita* em pastagem. Pesq Agrope Bras 36:1301–1307

Collins SL, Klahr SC (1991) Tree dispersion in oak-dominated forest along an environmental gradient. Oecologia 86:471–477

Condit R, Hubbell SP, Foster RB (1992) Recruitment near conspecific adults and the maintenance of tree and shrub diversity in a neotropical forest. Am Nat 140:261–286

D'Antonio CM, Kark S (2002) Impacts and extent of biotic invasions in terrestrial ecosystems. Trends Ecol Evol 17:202–204

Diggle PJ (1983) The statistical analysis of spatial point patterns. Academic Press, New York

Duncan RD (1991) Competition and the coexistence of species in a mixed podocarp stand. J Ecol 79:1073–1084

Eviner VT (2004) Plant traits that influence ecosystem processes vary independently among species. Ecology 85:2215–2229

Eviner VT, Chapin FSIII (2003) Biogeochemical interactions and biodiversity. In: Melillo JM, Field CB, Moldan M (eds) Element interactions: rapid assessment project of SCOPE. Island Press, Washington, pp 151–173

Fernández-Palacios JM, de los Santos A (1996) Ecología de las Islas Canarias. Muestreo y análisis de poblaciones y comunidades. Sociedad La Cosmológica, Santa Cruz de Tenerife

Fuentes ER, Avilés R, Segura A (1989) Landscape change under indirect effects of human use: the Savanna of Central Chile. Landsc Ecol 2:73–80

Grotkopp E, Rejmánek M, Sanderson MJ, Rost TL (2004) Evolution of genome size in pines (Pinus) and its life-

history correlates: super-tree analyses. Evol Int J Org Evol 58:1705–1729

Haase P, Pugnaire FI, Clark SC, Incoll LD (1997) Spatial pattern in *Anthyllis cytisoides* shrubland on abandoned land in southeastern Spain. J Veg Sci 8:627–634

Hatton TJ (1989) Spatial patterning of sweet briar (*Rosa rubignosa*) by vertebrate species. Aust J Ecol 14:199–205

He F, Legendre P, LaFrankie JV (1997) Distribution patterns of tree species in a Malaysian tropical rain forest. J Veg Sci 8:105–114

Holmgren M (2002) Exotic herbivores as drivers of plant invasions and switch to ecosystem alternative states. Biol Invasions 4:25–33

Horn HS (1975) Markovian processes of forest succession. In: Cody M, Diamond J (eds) Ecology and evolution of communities. Belknap Press, Cambridge, Massachusetts, pp 196–211

ITGE (1990) Mapa geológico de Gran Canaria E. 1:100.000. ITGE, Madrid

Izquierdo I, Martín JL, Zurita N, Arechavaleta M (2004) Lista de especies silvestres de Canarias (hongos, plantas y animales terrestres). Consejería de Política Territorial y Medio Ambiente Gobierno de Canarias

Kunkel G (1976) Gran Canaria, Tenerife y La Gomera: notas florísticas y adiciones. Cuad Bot Canar 26(27):75–81

Lockwood JK, Simberloff D, McKinney MK, von Holle B (2001) How many and which, plants will invade natural areas? Biol Invasions 3:1–8

Mack RN, Simberloff D, Lonsdale WM, Evans H, Clout M, Bazzaz FA (2000) Biotic invasions: causes, epidemiology, global consequences, and control. Ecol Appl 10:689–710

Miller MF (1996) Acacia seed predation by bruchids in an African savanna ecosystem. J Appl Ecol 33:1137–1144

Moeur M (1993) Characterizing spatial patterns of trees using stem-mapped data. For Sci 39:756–775

Parrotta JA (1992) *Acacia farnesiana* (L.) Willd. Aroma, huisache. SO-ITF-SM-49. U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station, New Orleans, LA

Pérez PL, García A, Heene A (1999) Control y erradicación del "rabo gato" (*Pennisetum setaceum*). Excmo. Cabildo de La Palma. S/C de La Palma

Pimentel D (2002) Biological invasions: economic and environmental costs of alien plant, animal, and microbe species. CRC Press, Boca Raton

Rejmanek M, Richardson DM (1996) What attributes make some plant species more invasive? Ecology 77:1655–1661

Rejmanek M, Richardson DM, Higgins SI, Pitcairn MJ, Groktopp E (2005) Ecology of the invasive plants. State of the art. In: Mooney HA, Mack RN, McNeely JA, Neville LE, Schei PJ, Waage JK (eds) Invasion alien species: a new synthesis. Scope 63. Island Press, Washington, pp 104–161

Richardson DM, Higgins SI (1998) Pines as invaders in the southern hemisphere. In: Richardson DM (ed) Ecology and biogeography of Pinus. Cambridge University Press, Cambridge, pp 450–473

Richardson DM, Allsopp N, D'Antonio CM, Milton SJ, Rejmanek M (2000) Plant invasions—the role of mutualisms. Biol Rev 75:65–93

Sánchez J, Ríos C, Pérez-Chacón E, Suárez C (1995) Cartografía Potencial del Medio Natural de Gran Canaria. Cabildo Insular de Gran Canaria, Universidad de Valencia, Universidad de Las Palmas de Gran Canaria

Sanders NJ, Weltzin JF, Crutsinger GM, Fitzpatrick MC, Nuñez MA, Oswalt CM, Lane KE (2007) Insects mediate the effects of propagule supply and resource availability on a plant invasion. Ecology 88:2383–2391

Sanz-Elorza M, Dana ED, Sobrino E (2005) Aproximación al listado de plantas vasculares alóctonas invasoras reales y potenciales en las islas Canarias. Lazaroa 26:55–66

Shackleton CM, Shackleton SE, Buiten E, Bird N (2007) The importance of dry woodlands and forests in rural livelihoods and poverty alleviation in South Africa. For Policy Econ 9:558–577

Silva L, Smith CW (2004) A characterization of the non-indigenous flora of the Azores Archipelago. Biol Invasions 6:193–204

SPSS (1986) SPSS/PC + V.6.0. *Base manual*. SPSS Inc., Chicago, IL

Stock WD, Wienand KT, Baker AC (1995) Impacts of invading N_2 fixing Acacia species on patterns of nutrient cycling in two Cape ecosystems: evidence from soil incubation studies and ^{15}N natural abundance values. Oecologia 101:375–382

Strasberg D (1995) Processus d'invasion par les plantes introduites à la Réunion et dynamique de la végétation sur les coulées volcaniques. Ecologie 26:169–180

Suárez V (1987) La propiedad pública, vinculada y eclesiástica en Gran Canaria, en la crisis del Antiguo Régimen. Cabildo Insular de Gran Canaria, Las Palmas de Gran Canaria

Súarez F (1997) Mogán. De pueblo aislado a cosmopolita. Ilustre Ayuntamiento de Mogán, Madrid

Szwagrzyk J (1990) Regeneration of forest related to the spatial structure of trees: a study of two forest communities in Western Carpathians, southern Poland. Vegetatio 89:11–22

Tassin J, Revière JN, Cazanove M, Bruzzese E (2006) Ranking of invasive woody plant species for management on Réunion Island. Weed Res 46:388–403

Traveset A (1991) Pre-dispersal seed predation in Central American *Acacia farnesiana*: factors affecting the abundance of co-ocurring bruchid beetles. Oecologia 87:570–576

Vacek S, Lepš J (1996) Spatial dynamics of forest decline: the role of neighboring trees. J Veg Sci 7:789–798

Walters C (1986) Adaptive management of removable resources. Macmillan, New York

Weigand T (2004) Introduction to point pattern analysis with Ripley's L and O-ring statistic using the Programita Software. Department of Ecological Modelling, UFZ—Centre of Environmental Research, Leipzig

Woods KD (1979) Reciprocal replacement and the maintenance of codominance in a beech-maple forest. Oikos 33:31–39

Yelenik SG, Stock WD, Richardson DM (2007) Functional group identity does not predict invader impacts: differential effects of nitrogen fixing exotic plants on ecosystem function. Biol Invasions 9:117–125

Zar JH (1984) Biostatistical analysis, 2nd edn. Prentice-Hall, Englewood Cliffs, NJ