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The role of fencing in the success of threatened plant species translocation

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Abstract Plant translocation has become a widely used tool to improve the conservation status of threatened plants. *Dianthus morisianus* (Caryophyllaceae) is a narrow endemic plant which only grows on the Portixeddu coastal dune (South-West Sardinia). Its natural habitat has been strongly modified, and it is currently considered one of the most threatened plants of Sardinia. In a conservation effort, a translocation of reproductive plants was planned. Plants were obtained from seeds collected in the natural population and cultivated at the Botanic Gardens of Cagliari University. The following two suitable areas near the natural population were identified: the first is located in a fenced site which is managed by public administration, and the second is located in an unprotected site. In November 2010, 113 plants were reintroduced in site one, and in February 2011, 25 plants were reintroduced in site two; all plants were regularly monitored. The aim was to analyse the effect of different management activities (i.e. the herbivore and human exclusion) on transplanted plants. The following consistent differences between sites with different management types were found: the survival and growth of *D. morisianus* were enhanced by reducing herbivory and human disturbance; in particular, fences positively enhanced the plant's long-term survival, reproductive success and seedling recruitment. This study highlights that management activities (i.e. erection of fences) should be incorporated into translocation design since they contribute to translocation success. Our experience can serve as a model for further translocations of the threatened plants of Sardinia and, more widely, of the Mediterranean islands.

Keywords Coastal plant · *Dianthus morisianus* Vals. · Endemic plant · Mediterranean islands · Plant conservation · Sardinia

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Introduction

Plant translocation (the controlled placement of plant material into a natural or managed area; Godefroid et al. 2011) is a relatively recent development and a potentially important tool for conservation. The rationale for translocation (population reinforcement, reintroduction and introduction) is the establishment or augmentation of new or existing populations to increase the survival prospects of a species by increasing population size and genetic diversity, or by representing specific demographic groups or stages (Pavlik 1996; Godefroid et al. 2011). In particular, translocation of endemic and endangered plant species to their natural habitat is one of the emerging tools of biodiversity management. Consequently, to prevent the extinction risk of threatened plant species and to improve their conservation status, translocations have become increasingly important in plant management worldwide (e.g. Maunder 1992; Falk et al. 1996a; Godefroid et al. 2011).

However, many restrictions remain in the implementation of these conservation actions, such as the high economic and time costs, the availability of the optimal site, the difficulties in the implementation of these actions in private areas and the high uncertainty of success principally connected to natural stochastic events (Maunder 1992; Gorbunov et al. 2008; Fenu et al. 2015a). Translocation is commonly recognized as a relatively high-risk and high-cost activity (i.e. Maunder 1992; Gorbunov et al. 2008), and thus, it is necessary to properly evaluate the risks and costs before starting such projects. From this perspective, disseminating the results of previous experiences (successful and unsuccessful) is important to provide examples and case studies that will allow the development of common standards and methodologies (Godefroid et al. 2011). There are many cases in the literature where translocations did not reach the desired objectives due to several reasons, including lack of knowledge about the species' biology and ecology, difficulties in transplanting numerous plant species and poor financial support (Fiedler and Laven 1996; Menges 2008; Godefroid et al. 2011 and references therein). However, there is a strong inclination not to publish negative or discouraging experimental results, even though the publication of such results may consistently increase the opportunities for new scientific applications (Godefroid et al. 2011; Dalrymple et al. 2012; Drayton and Primack 2012).

Any translocation ideally requires a thorough understanding of the biology of the species involved, (e.g. its growth form, mating system, fertility, germination) including its genetic and ecological characteristics (e.g. the amount and distribution of genetic variation, habitat requirements, mechanisms of dispersion, symbiotic relationships, pests and diseases) and the conservation status of wild populations (Falk et al. 1996b; Guerrant and Pavlik 1998). Several authors emphasise the importance of the knowledge of the biology and ecology of a species, the selection of the ecologically suitable area, the origin and type of material, the expertise in ex situ multiplication and cultivation procedures, and the method's approach to monitoring the undertaken action (Guerrant and Pavlik 1998; Maunder et al. 2004; Guerrant and Kaye 2007; Colas et al. 2008; Aguraiuja 2011; Godefroid et al. 2011; Reckinger et al. 2010; Abeli et al. 2012; Rita and Cursach 2013; Abeli et al. 2014).

While long-term monitoring of experiments has been seen as a key part of any translocation effort (Falk et al. 1996a, b; Guerrant and Kaye 2007; Godefroid et al. 2011; Drayton and Primack 2012), there are strong considerations which tend to keep the length of post-introduction monitoring to a minimum of a few years at most (average duration of 3–4 years; Godefroid et al. 2011; Dalrymple et al. 2012; Drayton and Primack 2012; Rita and Cursach 2013). However, success as measured only by the survival of transplants for a few years may be followed by the death of these plants in subsequent years. Drayton and Primack (2012) found that early plant performance (5 years after planting) may not necessarily reflect longer-term performance; in fact, the majority of the plants were absent in a new period of monitoring conducted 10 years later. Besides this case, published reports of long studies examining translocation efforts are uncommon (Menges 2008; Godefroid et al. 2011; Drayton and Primack 2012).

Among the potential determinants of successful translocations which have not received much attention in conservation biology, it is critical to consider post-translocation management activities, including the erection of fences and reduction in competition (i.e. Godefroid et al. 2011; Bontrager et al. 2014; Daws and Koch 2015). In particular, Godefroid et al. (2011) reported that management of the out-planting site through either preparation for planting (e.g. fencing) or post-planting management (e.g. reduced competition) positively impacted the short-term establishment and survival of plant reintroductions and increased the probability of translocation success. Comparable results were also obtained in other studies (Bontrager et al. 2014; Daws and Koch 2015). However, as far as we know, this aspect as well as the economic evaluation of this high-cost activity also remains unexamined. The economic evaluation becomes relevant when working in areas rich in threatened plant species, such as the Mediterranean islands, and when the economic resources are often limited (Fenu et al. 2015a).

This study was focused on the case study of *Dianthus morisianus* Vals. (Caryophyllaceae), an endemic and threatened perennial herb which only grows on the Portixeddu coastal dune system (Buggerru, South-West Sardinia; Bacchetta et al. 2010; Fenu et al. 2013). The natural habitat of *D. morisianus* has been strongly modified by human activities, causing habitat loss and fragmentation: there are several settlements in the species' habitat, and since 1950, much of the dune system has been afforested to stabilize the dunes and halt the movement of sand inland. In addition, the high level of unregulated grazing by domestic (i.e. sheep, goats, horses) and feral (i.e. deer, boars, rabbits) animals, the small size of the population and limited seedling recruitment have made this plant potentially prone to extinction. For all these reasons, *D. morisianus* is considered one of the most threatened plants on the island (Bacchetta et al. 2012), and it is categorized as critically endangered on the Global Red List (Fenu et al. 2013).

The Autonomous Region of Sardinia funded a conservation project for several threatened endemic plants comprising in situ and ex situ research studies and experimental projects, such as the construction of protective fences and/or experimental translocations (Fenu et al. 2012, 2015a). In a conservation effort to reduce the extinction risk for *D. morisianus*, a translocation program with the support of a public institution (Ente Foreste della Sardegna) was planned (see Cogoni et al. 2013 for details). Preliminary surveys focused on the ecology of *D. morisianus*, and the level of human disturbance in its habitat facilitated the identification of two suitable areas near the natural population. These two sites are subjected to different management situations. The first one is located in a fenced area (protected area, hereafter) and managed by a public institution, and the second one is located in an open area (unprotected area, hereafter).

The aim of this study was to analyse the effect of different management activities on transplanted plants; specifically, we wanted to examine the effects of herbivore and human exclusion (in particular, trampling and off-road activities) on plants in terms of survival, reproductive capacity (i.e. reproductive stems and number of fruits per plant) and seedling recruitment between the two different sites with different management levels. An additional aim was to evaluate the economic costs of these activities comparing the relative weight of the various activities performed in relation to the results obtained.

Materials and methods

Study species and site

Dianthus morisianus (Caryophyllaceae) is a perennial suffrutex characterized by numerous woody stocks and erect stems long 20–45 cm, and by a basal rosette with thin and linear leaves, 1–15 cm long (Valsecchi 1985; Bacchetta et al. 2010). The stems bear terminal multi-flowered heads (normally 2–18 flowers/head); the calyx has lanceolate teeth; the colour of the corolla is normally pink (Valsecchi 1985; Bacchetta et al. 2010). The flowering season lasts from early May to late June, whereas ripe fruits can be found during June and July (Valsecchi 1985; Cogoni et al. 2012).

This plant is the only psammophilous *taxon* belonging to the *D. sylvestris* Wulf. complex (Bacchetta et al. 2010); it is a narrow endemic plant of Sardinia, currently restricted only to the Portixeddu coastal dune system (Buggerru, South-West Sardinia; Fig. 1), where it grows on stabilized dunes at an altitude of 20–90 m a.s.l. and on slopes with varied incline and aspect. Although some plants are found in different ecological situations (e.g. at the edge of *Juniperus* and *Pinus* spp. micro-forests, on the open discontinuities, within the scrubs dominated by *Cistus* spp.), *D. morisianus* preferentially grows in areas colonized by several Mediterranean species, such as *Cistus salviifolius, C. monspeliensis, C. creticus* ssp. *eriocephalus, Helichrysum microphyllum* ssp. *tyrrhenicum, Lavandula stoechas*, *Lotus dorycnium* and *Rosmarinus officinalis*.

Geologically, the dune system, which spreads up to ca. 3 km inland, mainly consists of Holocene sandstones and aeolian sands which present irregular heights, ranging from 10 to 190 m a.s.l. (Cesaraccio et al. 1986). Available climate data (obtained from Fluminimaggiore weather station at 45 m a.s.l.) indicates a typical Mediterranean annual pattern of temperatures (mean annual = 16.7 °C; *T* mean

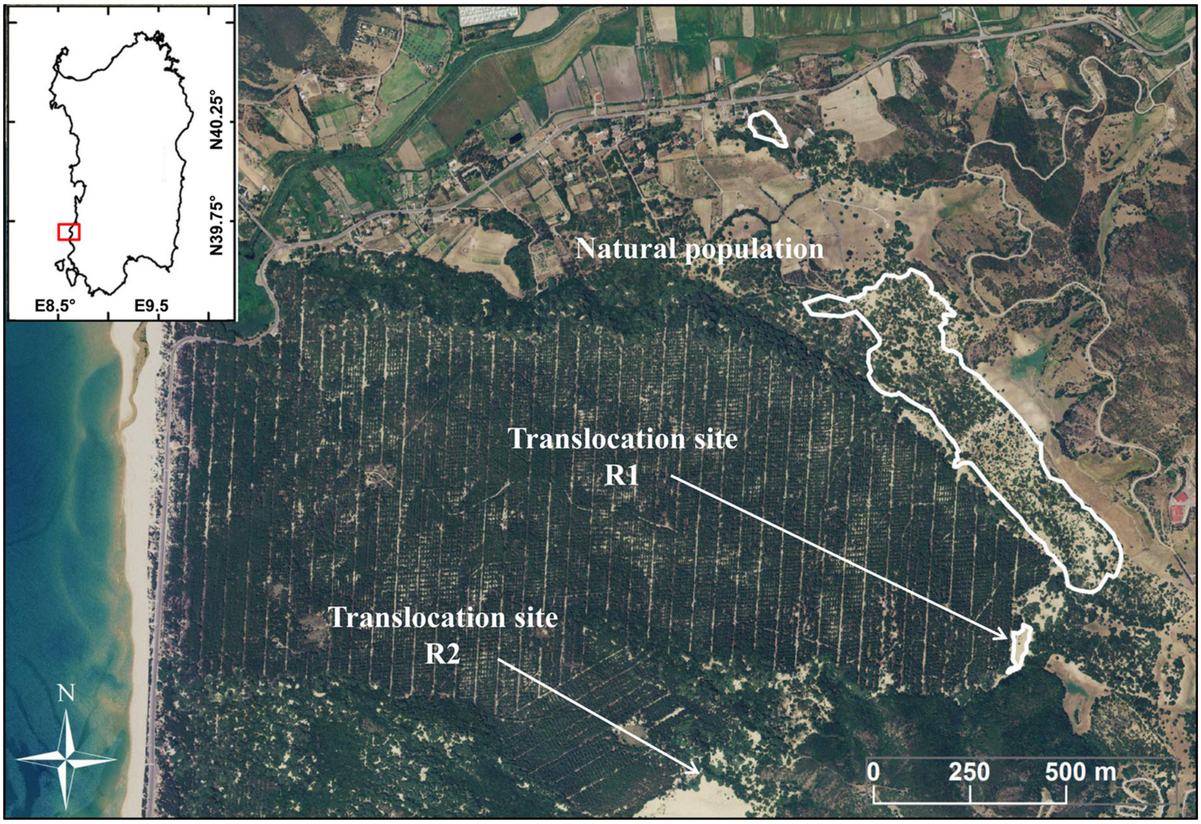


Fig. 1 Locations of the natural population and the reintroduction sites (R1 and R2) for the threatened *Dianthus morisianus* on the Portixeddu dune system; the *rectangle* on the *inset* indicates the location of the main map in Buggerru, South-West Sardinia (Italy)

max = 22.3 °C; T mean min = 10.8 °C) and precipitations (mean annual: 733 mm), with a prolonged dry summer (>2 months).

The Portixeddu coastal dune system is characterized by a high number of domestic and wild herbivores, as well as by the many human recreational activities (i.e. trampling and off-road activities) that determine relevant changes in natural vegetation and, in particular, among the endemic flora (including *D. morisianus* plants).

Experimental design

In a conservation effort to reduce the extinction risk, we planned a translocation program with the support of a public institution (Ente Foreste della Sardegna). Considering that this species is not seed-limited and that seedling emergence and establishment are the most critical stage for the plant's long-term persistence (Cogoni et al. 2012), we scheduled a reintroduction program based only on adult plants (Cogoni et al. 2013).

Fruit collection was carried out in the natural population in July 2009, and seeds were sowed ex situ at the Botanic Gardens of Cagliari University. After germination, all seedlings were placed in pots with a substratum composed of sand collected in the natural population site; no horticultural precautions were adopted (see Cogoni et al. 2013 for details). Preliminary research focused on the ecology of *D. morisianus* and the level of human disturbance in its habitat. These surveys facilitated the identification of two potential suitable areas near the natural population (Fig. 1): the first is located in a protected, fenced site and is managed by public administration, and the second is located in an open and unprotected site adjacent to the area managed by the Ente Foreste della Sardegna.

In November 2010, 113 plants were reintroduced to the first site; the plants were placed in nine groups at a mean distance of ca. 15 m from each other, depending on the availability of suitable microhabitats (see Cogoni et al. 2013 for details). This area was surrounded by using chestnut poles and metal fences with variable mesh (7 cm large at the bottom and 15 cm at the top) in order to exclude (or substantially reduce) the access of humans and herbivores of different sizes. In February 2012, 25 additional plants were reintroduced to the unprotected site following exactly the same procedure adopted in the first translocation; depending on the availability of suitable microhabitats, the plants were placed in two groups at a distance of ca. 10 m from each other.

During the first year after transplantation, all plants were monitored monthly and, subsequently, they were regularly monitored twice a year: the first monitoring was carried out in February, which corresponds to the seedling appearance period, and the second was conducted in June when ripe fruits can be found (Cogoni et al. 2012). All plants (including seedlings) were counted, marked with wooden poles, and surveyed in February and June; during each monitoring, survival (assessed by using a presence/absence binary scale), and reproductive traits (total number of reproductive stems, number of damaged stems by grazing and/or trampling, and fruits per plant) were recorded for every plant.

Statistical analysis

The non-parametric Kaplan–Meier product-limit method was used to estimate the survival function directly from the survival times in our dataset. Afterward, Cox's Proportional Hazard Models were used, and the log-rank test was performed to analyse the differences in the survival functions between the reintroduction sites.

Two-way ANOVA with repeated measures was applied to test the effects of the presence/absence of fences (used as category) and year (as a repeated variable) on the cumulative proportion of stems per plant and on the cumulative proportion of fruits per plant. The test compared the average stems per plant and the average number of fruits per plant at multiple time periods for single groups of plants. All these analyses were performed with Statistica 8.0 (StatSoft, Inc, Tulsa, OK, USA) software.

In order to analyse the theoretical economic costs incurred to conduct the two transplantations and the subsequent monitoring activities, a simple analysis was carried out, and the following parameters were considered: (1) the costs of the transplantation project, including transport, materials, and workman costs (official data from Ente Foreste della Sardegna) and (2) the costs related to the monitoring activities, including personnel and travel costs (data obtained from Fenu et al. 2015b).

Results

We found that survival, reproduction, and seedling recruitment were enhanced with a sensible reduction of herbivory and human disturbance. In the first year after transplanting, the plant survival rate was high and did not differ between the protected and unprotected sites, with few dead plants (Fig. 2). Subsequently, a significantly different pattern was observed: in the fenced site, after 5 years, 91.15 % of the transplanted plants survived, whereas in the unprotected fenced site only, 32.0 % of the plants transplanted survived after only 3 years. Indeed, the survival rate remained high for the protected plants and showed a considerable decrease for the unprotected ones, with important mortality events in the second and fourth years

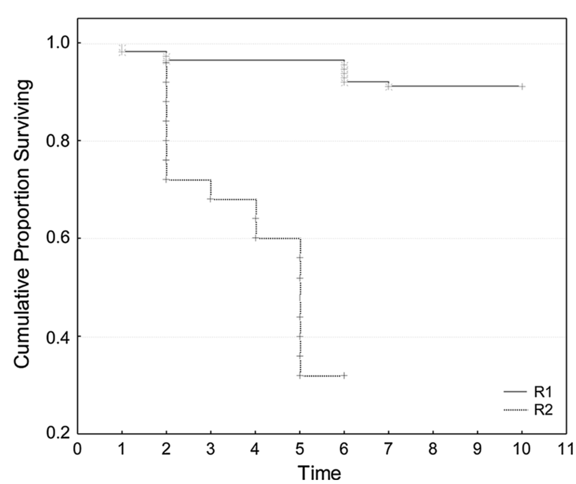
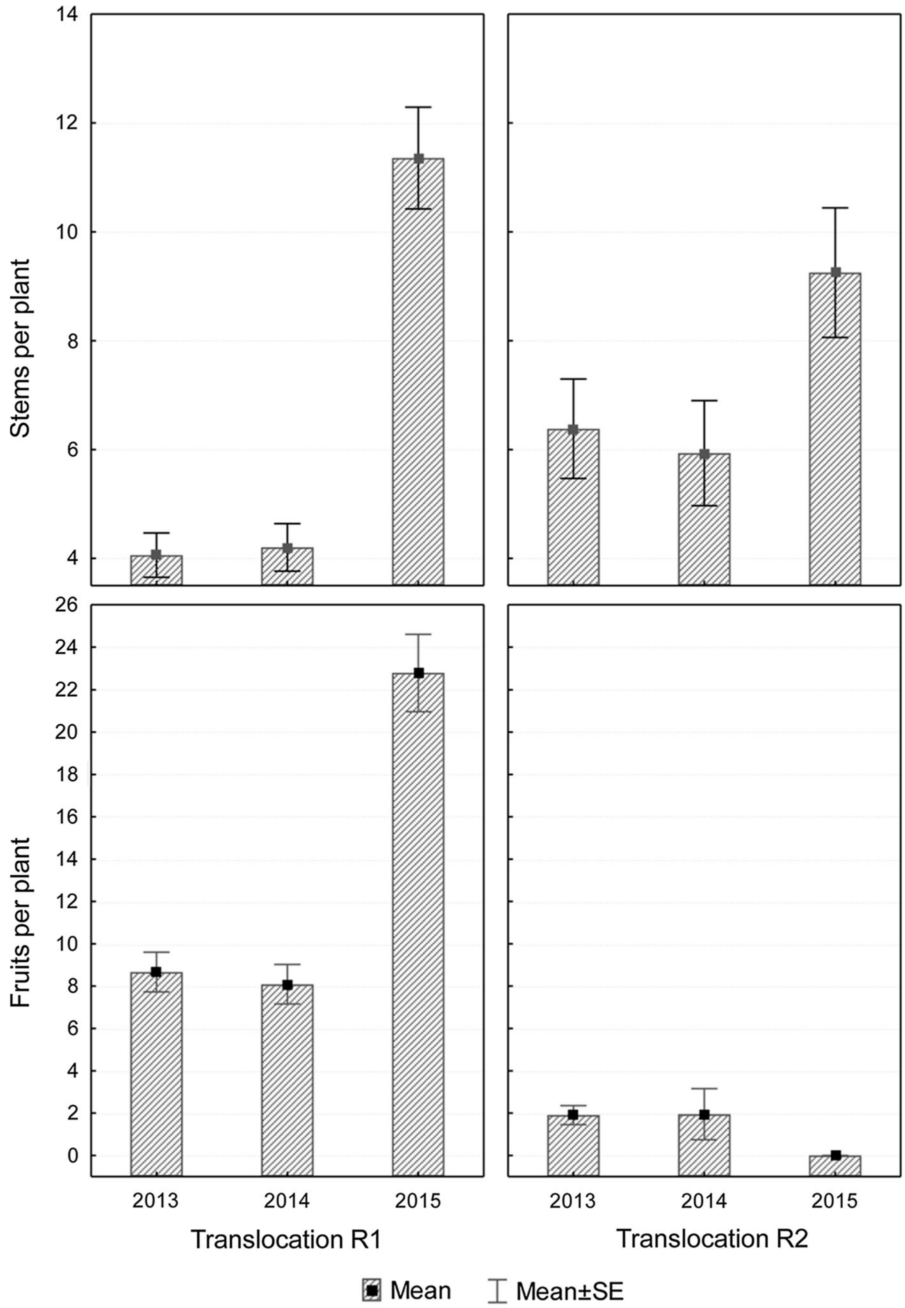


Fig. 2 Kaplan–Meier survival curves representing plants survival for the two different translocations investigated in this study. Trends differed significantly at the p value <0.001 level in the log-rank test. Each step on the x-axis represents an interval of ±6 months

(Fig. 2). The shape of the survival curves significantly differed between sites (log-rank test = 3.84, $df = 1$, p value <0.001).

All surviving plants, with some minor exceptions (6.2 % for the protected site and 12 % for the unprotected site), became reproductive and produced reproductive stems. The number of reproductive stems per plant (mean ± SE) was comparable between sites and it varied according to the year (Fig. 3). In the protected site, the mean number of reproductive stems per plant was 6.2 ± 0.41; the lowest value was found in 2013 (3.68 ± 0.39 stems per plant), while the highest mean value was recorded in 2015 (11.34 ± 0.94 stems per plant). The mean

Fig. 3 Change in patterns of the average measure of dependent variables (number of stems and fruits per plant) at three time periods, both for the protected (R1) and unprotected (R2) plants



number of reproductive stems per plant (±SE) for unprotected plants was 8.40 ± 0.71, ranging from 6.07 ± 1.03 (2014) to 9.25 ± 1.19 (2015). However, a different pattern at the stem level was observed between protected and unprotected plants: in the fenced site, only a minimal percentage of reproductive stems were damaged (mean value 9.23 %, ranging from 12.7 to 6.8 % in 2013 and 2015, respectively), while, in the unprotected area, the majority of the stems were damaged; a mean value of 86.6 % of the total stems (ranging from 83.8 to 100 % in 2013 and 2015, respectively) were predated by herbivores or damaged by human disturbance.

The results of the two-way ANOVA to test the effects of reintroduction type (protected and unprotected) on the number of stems per plant are reported in Table 1. The effect of protection on the number of stems per plant was not significant (F value <1; p value >0.05), whereas the effect of 'year' was significant (F value >1; p value >0.001). Likewise, the interaction between 'category' and 'year' was not significant (F value >1; p value >0.05; Table 1). The average measure of the dependent variables over three time periods showed a less clear trend among the unprotected plants than the protected ones (Fig. 3). The pattern of the average number of stems per plant over three time periods reflects the absence of significant effects of reintroduction type observed in the two-way ANOVA test, and the average number of stems per plant was often higher in the unprotected reintroduction (Fig. 3). Conversely, there was a clear trend in the average number of stems per plant during the monitoring time (both for the unprotected and the protected plants), and this indicates that the parameter 'year' did not have a significant effect on this variable. Moreover, the time courses of both patterns are rather similar (Fig. 3), so no interaction seems to exist between the factors 'category' and 'year' (Table 1).

The number of fruits per plant (mean ± SE) greatly varied between sites (Fig. 3). In the protected site, the mean number of fruits per plant was 12.33 ± 0.82; the lowest value was found in 2014 (7.33 ± 0.88 fruits per plant), while the highest mean value was recorded in 2015 (22.79 ± 1.86 fruits per plant). The mean number of reproductive stems per plant (± SE) for unprotected plants was 1.59 ± 0.51, ranging from zero (2015) to 1.90 ± 0.45 (2013).

The results of the two-way ANOVA to test the effects of reintroduction type on the cumulative number of fruits per plants are reported in Table 1. The effect of protection was significant with respect to the cumulative number of fruits per plants (F value >1; p value <0.001). Likewise, the parameter 'year' showed a significant effect on the number of fruits per plants (F value >1; p value <0.05); the interaction between category and year was also significant (F value >1; p value <0.01). Overall, the cumulative proportion of fruits per plant showed an important increase in the protected plants and a correspondent decrease in the unprotected plants over time (Fig. 3). The number of fruits per plant among protected plants increased from 2013 to 2015, with an important effect in the last year, whereas an opposite trend was observed for unprotected plants. However, the development over time of both patterns is rather different and suggests that no interaction exists

Table 1 Summary of results of the two-way ANOVA with repeated measures to test the effects of fence protection (category) and year (repeated variable) on number of stems and number of fruits per plant

Effect	SS	*df*	MS	*F* test	*p* value
No. stems per plant					
Intercept	6285.66	1	6285.656	164.0719	0.000000
Category	14.03	1	14.031	0.3662	0.545474
Year	632.01	2	316.003	8.2485	0.000319
Year × category	106.41	2	53.206	1.3888	0.250813
Error	12,757.35	333	38.310		
No. fruits per plant					
Intercept	7265.26	1	7265.264	50.05135	0.000000
Category	4915.42	1	4915.421	33.86298	0.000000
Year	939.02	2	469.510	3.23452	0.040620
Year × category	1606.75	2	803.373	5.53454	0.004320
Error	48,337.01	333	145.156		

SS sum of squares, *df* degree of freedom, *MS* mean-square effect between the factors 'category' and 'year'. Consistent differences between sites were observed in terms of seedling recruitment (Table 2). In the protected site, a higher number of seedlings were produced and 137 new plants became reproductive after 4 years. In particular, after 5 years, the number of new plants in the protected site was higher than that of individuals transplanted, and the total size population in terms of reproductive plants had doubled. An opposite trend was found in the unprotected site: the cumulative number of seedlings was very low and no reproductive plants were recruited.

Economic analysis of management activities

The rapid cost analysis showed that in smaller transplantation projects like this, the main cost is represented by the fence erection (ca. 80 % of the total cost during 5 years of activities); the other activities incurred lower costs, though these are prolonged over time, such as the monitoring surveys (Table 3).

Conversely, in the translocations without fence erection, the costs are unbalanced in the monitoring activities (ca. 50 %), apart from an initial cost related to the plant positioning (Table 3); however, in perspective, the weight of the monitoring activities is expected to become prevalent (>60 % after 5 years of monitoring activities).

Discussion

A key measure of the success of a plant reintroduction is the survival rate, the transplants' ability to flower and set fruit and the recruitment of new individuals (IUCN 1998; Morgan 2000; Menges 2008; Godefroid et al. 2011). We found consistent differences between sites with different management types: the survival and growth of our study species were enhanced by reducing herbivory and human disturbance and, consequently, these disturbances were the key factors in enhanced plant outcomes.

Table 2 Cumulative number of new plants recorded in the two sites

	2012	2013	2014	2015
Fenced site				
Seedlings	92	236	349	516
Established plants		27	82	473
Reproductive plants			16	137
Unprotected site				
Seedlings		0	6	6
Established plants				0
Reproductive plants				

The high survival rate and the mean number of reproductive stems per plant recorded in the first year post-transplantation confirm the ecological suitability of the selected sites as a key feature for successful plant reintroduction (Colas et al. 2008; Menges 2008; Reckinger et al. 2010) as well as the positive use of older and structured plants that generally improve the success of reintroductions (Maschinski and Duquesnel 2007; Reckinger et al. 2010; Godefroid et al. 2011). These aspects also highlight that the differences in the performance of protected and unprotected plants in the subsequent years were mostly linked to the management conditions. As already demonstrated (Godefroid et al. 2011; Bontrager et al. 2014), the management actions post-transplantation are thus very important, and their relevance in terms of plant outcome increases with time.

Our results confirm that fences positively impacted the long-term survival, reproductive success and seedling recruitment of plant translocations (Godefroid et al. 2011; Daws and Koch 2015). Herbivore and human exclusion clearly improved the survival of the transplanted plants and enhanced the plant responses in terms of reproductive success and the spread of reintroduced plants.

Previous studies reveal that survival, flowering and fruiting rates in reintroduction projects are generally low and sometimes show a downward trend with time (Godefroid et al. 2011 and references therein). In our case, the survival and fruiting rate remain high over time and the mean number of fruits per plant was higher in the protected site than in the natural population (Cogoni et al. 2013). While the mean number of reproductive stems increased with time in both sites (i.e. protected and unprotected), the mean number of fruits per plant showed the opposite pattern depending on the site; the mean number of fruits per plant increased considerably in the last year in the protected site and decreased in the unprotected one during the same year (2015). In the unprotected site, where the majority of reproductive stems (consequently fruits and seeds) were predated by herbivores and/or damaged by human disturbance, only a few

Table 3 Summary of economic resources (in euros) for the implementation translocations as well as the subsequent monitoring activities; the relative importance of each activity as indicated as a percentage

Activity	Standard/costs (€)	Transplantation 1		Transplantation 2	
		Costs (€)	(%)	Costs (€)	(%)
Transport of plants from Cagliari, including the driver (centner)	115.10	390.19	1.31	86.32	4.51
Workman costs/day	115.00	1725.00	5.78	575.00	30.06
Pit opening (deep 30 cm; each one)	4.85	548.05	1.84	121.25	6.34
Plant placement (each one)	7.40	836.20	2.80	185.00	9.67
Metal fence with chestnut poles (meter)	98.50	23,640.00	79.16	0.00	0.00
Monitoring activities	460.00	2300.00	7.70	690.00	36.08
Travels costs per year (mean value)	85.00	425.00	1.42	255.00	13.33
Total		29,864.44	100.00	1912.57	100.00

The economic costs involved in the plant productions were not considered in this analysis

seeds reached the soil surface; the few surviving seedlings were killed by animals, human trampling and off-road recreational activities. These changes in reproductive patterns over time indicate the importance of the respective census year for information about the translocation situation of the protected and unprotected plants. These changes thus confirm the need for long-term monitoring to understand the status of transplanted plant populations, as previously documented (e.g. Maunder 1992; Godefroid et al. 2011; Drayton and Primack 2012).

Few studies have included data on seedling recruitment, which would allow more direct assessment of population viability. Recruitment is considered the highest measure of success (Pavlik 1996; Sutter 1996) because it indicates that the population is self-sustaining through the development of successive generations (Primack 1996; Kaye 2008), which is the primary goal of a typical translocation project. In our study, the seedling recruitment, as well as the probability that new plants became reproductive, was favoured by herbivore and human exclusion. The number of seedlings produced by the reintroduced plants increased every year, and after 5 years, the total number of reproductive plants had doubled. Although the ultimate success of a translocation project, i.e. autonomous maintenance of populations, can be determined only after many years of follow-up (from 10 years to several decades depending on the generation time for the species; Maunder 1992; Pavlik 1996; Milton et al. 1999), in our protected translocation, this last aspect is the most important: the size population increased over time, indicating that it is self-sustaining through the development of successive generations. Nevertheless, according to Guerrant and Kaye (2007), the translocation success may best be viewed not as a summary conclusion (or final result) but in terms of progress or status reports at one or more times after out-planting; accordingly, the monitoring activities for *D. morisianus* translocations have been planned for a long-time period (Fenu et al. 2015b).

Our study highlights that conservation of threatened plants is more practicable on legally protected sites managed by public administration (Ente Foreste della Sardegna in our case), as previously reported by Godefroid et al. (2011). However, there are a large number of endangered plant species growing in private areas (outside of the protected areas), as in our case in Sardinia, and therefore policies that include single citizens in these projects must be developed by the public institutions.

Bearing all constraints in mind, including the high costs to build the fences, it is evident that management efforts (fence erection, grazing exclusion, etc.) should be incorporated into reintroduction design since they contribute to the higher success rate of a translocation project.

Translocation is generally a relatively high-risk and high-cost activity (Gorbunov et al. 2008). The translocation of *D. morisianus*, however, is an example of a low-cost project, excluding the costs of fence erection:

the involvement of researchers, public authorities and local stakeholders was voluntary, and site management was not intensive. In particular, our results highlight the pivotal role of the basic practical precautions (such as fence erection) necessary in order to promote a translocation program.

In conclusion, we are aware that the translocation activities are a straightforward instance of how good purposes seldom bump into the reality: for example, the reasons which drive the choice of the optimal location are commonly influenced by external causes (e.g. private areas, local interests) which sometimes can even reverse the decision away from the site with the optimal ecological conditions (Fenu et al. 2015a). Although proactive actions in the field are the best way to conserve natural plant populations, very little has been done compared to what is necessary to prevent the risk of extinction of many plant species. However, taking into account the limited available funds and human resources, the implementation of the active conservation measures, such as the translocation of threatened plants, must be one of the first priorities at the regional level in order to also achieve the targets established by the international conventions (Fenu et al. 2015a). We believe that this translocation project can serve as a model for further translocations of other threatened species of Sardinia and, more widely, of the Mediterranean islands.

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