Restoration of a Restinga Sandy Coastal Plain in Brazil: Survival and Growth of Planted Woody Species

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Abstract

In this article we report the results of an experiment introducing 17 native shrub and tree species into a Brazilian restinga (i.e., coastal sandy plain vegetation). Restingas have been affected by human impact for about 8,000 years, and human occupation for housing, tourism, and land speculation has recently increased in such a way that there is a need for conservation of remnant patches and restoration of degraded areas throughout the coast to protect biodiversity. Our study site is a remnant located in Rio de Janeiro, the second largest city in the country, and has been subjected in the past to deforestation, man-made fire, and sand extraction. Although trees and shrubs predominantly compose natural restinga vegetation, local vegetation after impact was replaced by an exotic grass cover, which meant a drastic reduction in species richness. Thus, in this experiment we removed the grass cover, introduced shrub and tree species, and monitored survival and growth of 20 plants per species for 2 years. Despite the adversities imposed by the nutrient-poor sandy soil, 70% of the species showed high survival percentage and considerable growth. This report on restoration initiatives in the restingas points out the viability of shrub and tree plantation following exotic grass removal as a strategy to restore Brazilian coastal vegetation.

Key words: **conservation, restingas, restoration, seedling growth, seedling survival.**

Introduction

Forestation of coastal barriers and dunes is commonly undertaken successfully in many countries of the southern hemisphere. Species of *Pinus*, *Casuarina*, and *Eucalyptus* are often used as exotic trees with economic purposes, even in originally treeless coastal habitats of these regions (e.g., Ndiaye et al. 1993; Smith et al. 1994). This successful background has led Orellano and Isla (2004) to call for a "land use change from unproductive dunes to forested woods" in temperate coastal South America (Argentina, Uruguay, and south Brazil) aiming at a considerable increase in carbon sequestration by the trees compared to previous ecosystems. These authors, however, acknowledge that environment impact studies should be undertaken to prevent unforeseen environmental problems.

There are also positive examples of restoration of native coastal plant communities and associated ecosystems, such as in the cases of coastal dune forest restoration following heavy mining in Australia and South Africa (see Lubke & Avis 1998, for a review of the literature). This reduces the impacts of biodiversity loss and restores ecosystem functions. This strategy is likely to be more appropriate to degraded coastal habitats of countries with naturally large forested areas and high biodiversity under threat. Brazil, for instance, has one of the largest forested areas of the world, thus consisting of a natural carbon sink (Fearnside 2001; de Mattos & Scarano 2002), and is home to one of the highest biodiversities on the planet, currently threatened by high rates of deforestation and habitat destruction (WCMC 1992).

The past two decades have seen an increase in biodiversity and ecosystem restoration initiatives in Brazil (for review, see Kageyama et al. 2003). Although this has resulted in much scientific and technological advancement, some knowledge gaps still exist. For instance, the vegetation covering the sandy coastal plains along circa 5,000 km of the Atlantic coast, the so-called restinga, has received little attention. Restingas differ from dunes, in that they are sand marine deposits, whereas dunes are wind deposits. Previous research on restoration efforts on coastal sandy ecosystems in Brazil has involved case studies conducted on dunes, such as an assessment of the potential of 14 species (two shrubs and 12 herbs) for dune fixation (Freire 1983) and dune restoration with creeping plants and grasses after mining activities (Carvalho & Oliveira-Filho 1993; Miranda et al. 1997).

Restinga is both a geomorphological and a botanical term. It applies equally to the sandy plains dating from the Quaternary, mostly from the Holocene, and to the

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vegetation covering these plains. The restinga vegetation is a mosaic of plant communities ranging from creeping types to open scrubs and even forests (Lacerda et al. 1993; Martin et al. 1993). These areas have been affected by human impacts for about 8,000 years (Kneip 1987). Human occupation has recently increased to such an extent that there is a need for conservation of remnant patches and restoration of degraded areas. Restinga vegetation has suffered considerable habitat destruction because most Brazilian big cities are on the coast. However, these systems are often not treated as a conservation priority because they have few endemic species (Barbosa et al. 2004). Most plants and animals inhabiting the restingas originated in the neighboring Atlantic rainforest and successfully migrated and colonized the geologically younger sandy plains (Rizzini 1979; Araujo 2000). The restinga ecosystem is therefore unique because it comprises a pool of species with high ecological plasticity, since, despite their rainforest origin, they colonized, survive and grow in the dry, resource-poor restingas. This characteristic may be of key relevance in a global change scenario (Scarano 2002).

Restinga vegetation is often species rich, although plants are subjected to the various constraints imposed by drought, nutrient-poor sandy substrate, wind, salinity, and high soil and air temperatures (Reinert et al. 1997). Paradoxically, it has been shown that few restinga plants are capable of establishing via seeds on bare sand and, therefore, the structure and function of open restinga vegetation relies on a few pioneer nurse-plants that facilitate the entry and establishment of a number of other species (Scarano 2002; Dias et al. 2005).

The city of Rio de Janeiro once had a vegetation complex consisting of Atlantic forest, mangroves, and restingas. The restinga vegetation covered an extensive area on the west side of the city; however, only 0.63% (770.65 ha) of the total area of the municipality is still occupied by this vegetation type, which represents a loss of 30% of the restinga cover in the period from 1984 to 1999 (PCRJ 2000*a*). Despite this massive reduction in area, the restingas of Rio de Janeiro provide important habitat for endemic species of plants, insects, fish, and lizards, which are threatened with extinction (Vanzolini & Ab'Saber 1968; Araujo & Maciel 1998; PCRJ 2000*b*).

In 1993, the government of Rio de Janeiro city launched the project "Flora do Litoral" (coastal flora) that aimed to restore degraded areas within the conservation units of the municipality. Thus, in 1994 a nursery was constructed, which, during the next 10 years, produced circa 650,000 seedlings of 130 native restinga species and provided 375,000 seedlings of 67 species for planting in 68 ha of degraded restingas. Two previous papers have described nursery activities, including seed germination tests and seedling production (Zamith & Dalmaso 2000; Zamith & Scarano 2004). This article discusses the first results from this restoration effort as regards survival and growth of planted shrubs and trees.

We performed a pioneer experiment by introducing restinga shrubs and trees into a degraded sandy coastal plain to assess the feasibility of native plant reintroduction, within the scope of the above project. We monitored the survival and growth of 17 restinga species for 2 years after planting. We hypothesized that introduced seedlings would have a high overall success rate, based on the expectation that the critical germination phase was overcome in the nursery and seedlings would be more resistant to the harsh restinga conditions than seeds. Further, on a local level, we aimed to provide information on the best choices of trees and shrubs for planting in such areas.

Methods

Study Site

The experiment was conducted in the city of Rio de Janeiro, southeast Brazil, neighborhood of Recreio dos Bandeirantes (lat 23º00′S, long 43º26″W) in the Natural Municipal Park of Marapendi, circa 250 m from the border of the Lake Marapendi (Fig. 1). The area was subjected to deforestation and fire in the past in order to clear the area for a housing project. The site was chosen because since 1991, it is protected by the Natural Municipal Park of Marapendi and it had not suffered substantial changes in soil physical properties and thus remains sandy. The scenario shown in Figure 1 is typical of many restingas along the coast of southeast Brazil.

Invasive plants occupied the area before the onset of the experiment, such as the exotic grasses *Panicum maximum* Jacq., *Imperata brasiliensis* Trin., and *Melinis minutiflora* P. Beauv, which are locally common throughout urban areas, and the exotic tree *Casuarina equisetifolia* L. (Casuarinaceae), which is frequently planted in coastal zones in Brazil as a wind shield. A few remnants of the native vegetation still occurred, such as *Cupania emarginata* Cambess. (Sapindaceae), *Eugenia ovalifolia* Cambess. (Myrtaceae), *E. rotundifolia* Casar. (Myrtaceae), *Inga maritima* Benth. (Leguminosae–Mimosoideae), *Myrciaria floribunda* (H. West ex Willd.) Legrand (Myrtaceae), and *Ocotea* sp. (Lauraceae). These species provide a strong indication that the original plant community prior to disturbance was that described as a "Myrtaceae thicket" (Lacerda et al. 1993).

Average annual rainfall for the period from 1998 to 2000 was 1,190 mm. Mean maximal and mean minimal annual air temperatures were 28 and 21ºC, respectively. However, daily maximal values may reach up to 42ºC in the peak of the summer (December–February). Moreover, temperatures at the sandy surface on hot days have been reported to reach 70ºC (Scarano 2002). Although a previous study indicated that this area shows no period of water deficit during the year (Araujo & Henriques 1984), the low water retention and low nutrient availability of the sandy soils suggest that plants may undergo intermittent water and nutrient stress (Reinert et al. 1997).

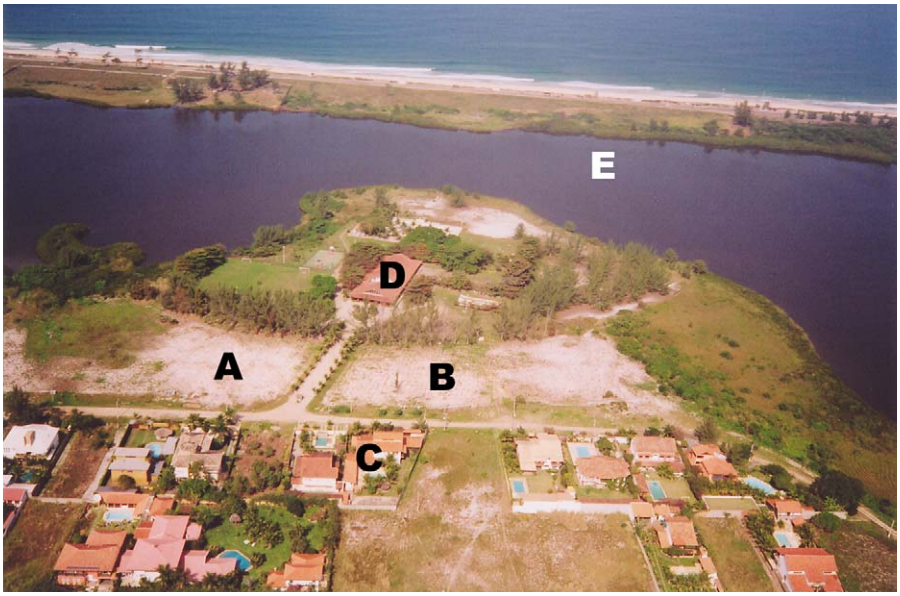

Figure 1. Aerial view of the plantation site at the Wildlife Conservation Zone of the Municipal Natural Park of Marapendi, in July 1998: (A) area soon after introduction of shrubs and trees; (B) shrubs and trees were also introduced here, but they were not included in this experiment; (C) and (D) human occupation with houses and club; and (E) Marapendi lake. The park area shown in the photograph is only in between the road and the *Casuarina* plantation, that is, A and B.

Management Procedures before Planting

Individuals of *C. equisetifolia* were removed with a chainsaw and invasive grasses were mechanically removed and buried circa 20 cm deep into the soil. The planting holes had dimensions of 30 × 30 × 30 cm and were separated from each other by 2.5 m along a given line, and lines were separated from each other also by 2.5 m. Between lines, holes were dug at 5-m distance from each other and at 1.25-m distance of the lines. Although native vegetation in such sites is a closed thicket, we have chosen this uniform, spaced plant distribution pattern as an attempt to reduce the effect of competition between neighbors. This would also favor the maintenance procedures described below. The planting holes each received 20 L of an organic compound produced from urban garbage.

Species Selection and Planting

Species were chosen based on two criteria: first, the occurrence of the species in the Myrtaceae thicket of the restingas on the west side of Rio de Janeiro (Araujo & Henriques 1984) and second, the seedling availability. Seventeen species of trees and shrubs were chosen belonging to 10 botanical families—Anacardiaceae: *Tapirira guianensis* Aubl.; Bignoniaceae: *Tabebuia chrysotricha* (Mart. ex DC.) Standl.; Bombacaceae: *Pseudobombax grandiflorum* (Cav.) A. Robyns; Erythroxylaceae: *Erythroxylum ovalifolium* Peyr.; Euphorbiaceae: *Pera glabrata* Baill.; Leguminosae–Caesalpinioideae: *Chamaecrista ensiformis* (Vell.) H.S. Irwin & Barneby, *Senna australis* (Vell.) H.S. Irwin & Barneby, and *S. pendula* (Humb. & Bonpl. ex Willd.) H.S. Irwin & Barneby; Myrtaceae: *Eugenia ovalifolia* Cambess., *E. rotundifolia* Casar., *E. sulcata* Spring. ex Mart., *E. uniflora* L., and *Myrcia* cf. *multiflora* (Lam.) DC.; Ochnaceae: *Ouratea cuspidata* (A. St.-Hil.) Engl.; Rubiaceae: *Tocoyena bullata* Mart.; and Sapindaceae: *Allophylus puberulus* Radlk. and *Cupania emarginata* Cambess.

During the rainy season between 23 and 30 March 1998, 4,700 seedlings belonging to the 17 chosen species were planted on the site. Once planted, seedlings were individually tied to bamboo sticks to protect them from wind damage and facilitate visibility, so as to avoid damage during maintenance activities. During the first 30 days after planting, seedlings were irrigated to field capacity on all days when no rain occurred. Invasive grasses were removed with a backpack brushcutter six times from March 1998 to June 1999. Logistics did not allow further control of invasive grasses. Surviving plants were counted 130 days after planting. Survival counts were undertaken at this stage to assess if the replacement of seedlings was necessary.

Survival and Growth Monitoring

To allow acclimation, monitoring was started 90 days after planting. A total of 337 plants were monitored: 20 individuals of each species, which were selected randomly and labeled with numbered aluminum tags, except for *P. glabrata*, which had only 17 surviving individuals. Plants were considered dead when they were leafless and the stem was visibly dry. We monitored height (distance from the soil to the apical bud of the plant), basal diameter of the stem (bd), and canopy area. These parameters were chosen because they allow for an estimate of growth and also of vegetation cover of the soil. Canopy area was calculated according to the formula for the area of an ellipse: $(D \times d \times \pi)/4$, where D = largest diameter and d = largest transversal diameter to D. Measurements were done bimonthly up to 14 months after planting and then at the 24th month, that is, there was an interval of 20 months between first and last measurement.

The monthly relative growth rates (RGR) were obtained for each plant and averaged per species. RGR was calculated for each of the parameters measured to minimize the effect of size at the moment of planting, according to the formula:

$$\mathrm{RGR} = \left\{ \frac{(\mathrm{Mf} / \mathrm{Mi})}{\Delta t} - 0.05 \right\} 100$$

where Mf is the final measurement, Mi is the initial measurement, and Δt is the difference in months between the two measurements (adapted from Melo et al. 2000). Because data distribution was not normal (as tested by Kolmogorov–Smirnov; Zar 1996), data comparison was done by a nonparametric analysis of variance, using Kruskal–Wallis and Dunn tests for multiple comparisons (Sokal & Rohlf 1995; Zar 1996). Wilcoxon tests were used to verify if the growth results obtained for each species differed significantly from zero. All calculations and graphs were done using GraphPad Prism (GraphPad Software, Inc. 1994–1995, San Diego, CA, U.S.A.).

We created two indexes, obtained from survival and growth data, to make interspecific comparisons regarding efficiency of performance as a restoration species: a growth index and a use viability index (UVI). The growth index ($\mathrm{GI}_{(i)}$) is given by:

$$\mathrm{GI}_{(i)} = \frac{\mathrm{Ph}_{(i)} + \mathrm{Pbd}_{(i)} + \mathrm{Pca}_{(i)}}{\mathrm{Ph}_{(max)} + \mathrm{Pbd}_{(max)} + \mathrm{Pca}_{(max)}}$$

where $\mathrm{Ph}_{(i)}$, $\mathrm{Pbd}_{(i)}$, and $\mathrm{Pca}_{(i)}$ are points attributed (from lower to higher growth) for each species according to statistical differences in RGR in height, basal diameter, and canopy area, respectively. The points attributed ranged from 1 to 7 in the case of height and basal diameter and 1 to 9 in the case of canopy area. $\mathrm{Ph}_{(max)}$, $\mathrm{Pbd}_{(max)}$, and $\mathrm{Pca}_{(max)}$ are the maximal possible values for each of the three parameters; therefore, $\mathrm{Ph}_{(max)} = \mathrm{Pbd}_{(max)} = 7$ and $\mathrm{Pca}_{(max)} = 9$. Thus, $\mathrm{GI}_{(i)} = (\mathrm{Ph}_{(i)} + \mathrm{Pbd}_{(i)} + \mathrm{Pca}_{(i)})/23$. In the case of *E. rotundifolia*, *E. sulcata*, *E. uniflora*, and *S. australis*, basal diameter was not measured because they form multiple branches at ground level, and thus, $\mathrm{GI}_{(i)} = (\mathrm{Ph}_{(i)} + \mathrm{Pca}_{(i)})/16$.

The UVI is given by $\mathrm{UVI}_{(i)} = \mathrm{GI}_{(i)} \times \mathrm{SP}_{(i)}$, where $\mathrm{SP}_{(i)}$ is the survival percentage for each species at the end of the experiment.

Results

Survival

There was low mortality after planting, confirming our initial expectations. Only 5.2% of the 4,700 plants introduced had died 130 days after planting. Thus, no replacement of monitored plants was necessary. After 2 years, at the final monitoring, there was 81.9% survival of the 337 plants monitored. Almost 50% of the species introduced had 100% survival at the end of the experiment (Table 1).

Nine out of 17 species had some mortality during the 2 years of monitoring (Table 1). Only one individual of each of *Chamaecrista ensiformis*, *Tabebuia chrysotricha,* and *Tapirira guianensis* died, and in all cases, death occurred in the summer, between October 1998 and February 1999. *Erythroxylum ovalifolium*'s mortality was concentrated between June 1998 and February 1999. Mortality also occurred under the high summer temperatures for *Eugenia ovalifolia* and *E. rotundifolia*, always between October 1998 and April 1999, except for one *E. ovalifolia* plant that died in the following summer.

Senna pendula showed a distinct pattern: mortality was zero until April 1999, but, from then on, 40% of the plants died and the remaining plants showed signs of senescence at the end of the experiment. High mortality was also found for *E. sulcata*, which showed a steady increase in the number of dead individuals from June 1998 to October 1998 (50%), and by the end of the experiment in March 2000, this species showed 65% mortality. Similarly, *Myrcia* cf. *multiflora* had 65% mortality by March 2000, with mortality always being higher during the summer periods.

Growth

Nine of the 17 species studied (*Allophylus puberulus*, *C. ensiformis*, *E. ovalifolium*, *E. ovalifolia*, *E. sulcata*, *E. uniflora*, *M.* cf. *multiflora*, *S. pendula*, and *T. chrysotricha*) showed no significant increase in height. Individual negative height increments were common. We collected no data on wind, but we believe it could be partly responsible for this pattern because shoots of many seedlings were broken, although plants were tied to bamboo sticks. Damage to shoots was often followed by resprouting. Height increase was largest for *Pseudobombax grandiflorum*, *Cupania emarginata*, *S. australis*, and *Tocoyena bullata* (Fig. 2a). Averaging the values of increment in height of the eight species that showed statistically significant positive values (*C. emarginata*, *E. rotundifolia*, *Ouratea cuspidata*, *Pera glabrata*, *P. grandiflorum*, *S. australis*, *T. guianensis*, and *T. bullata*) resulted in a mean annual RGR for height of 11.32 cm/yr.

All species monitored showed a significant increase in basal diameter, except for *M.* cf. *multiflora*. Again, *P. grandiflorum* showed the highest increment in basal diameter, along with *T. guianensis* (Fig. 2b).

Canopy area growth was highest for *S. australis* and lowest for *C. emarginata* (Fig. 2c). The latter, along with

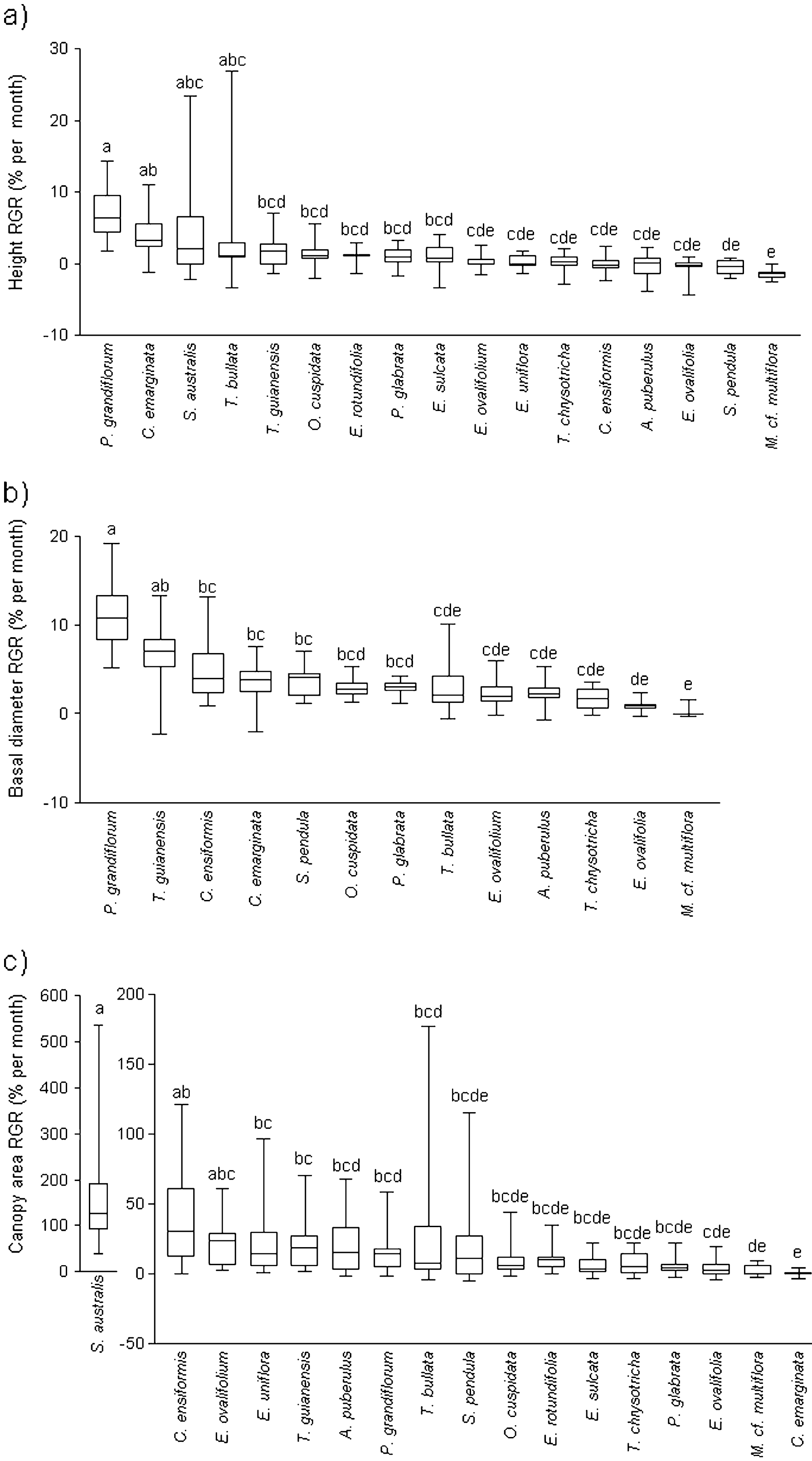

Figure 2. Monthly average increments for restinga shrub and tree species after 24 months of planting: (a) height increment for 17 species; (b) basal diameter increment for 13 species; and (c) canopy area increment for 17 species. Differences were tested by Kruskal–Wallis test followed by Dunn test. Different letters indicate significant differences between species at p less than 0.05.

Table 1. Survival percentage of 17 shrub and tree restinga species, 24 months after planting ($n = 20$; except for *Pera glabrata*, $n = 17$).

Survival (%)	*Species*
100	*Allophylus puberulus*, *Cupania emarginata*, *Eugenia uniflora*, *Ouratea cuspidata*, *Pera glabrata*, *Pseudobombax grandiflorum*, *Senna australis*, *Tocoyena bullata*
95	*Chamaecrista ensiformis*, *Tabebuia chrysotricha*, *Tapirira guianensis*
85	*Erythroxylum ovalifolium*
65	*Eugenia ovalifolia*, *Eugenia rotundifolia*
60	*Senna pendula*
35	*Eugenia sulcata*, *Myrcia* cf. *multiflora*

M. cf. *multiflora*, *E. sulcata*, and *S. pendula*, showed no significant increase in canopy area. Decreases in canopy area were usually due to temporary deciduousness or breaking of branches.

Senna australis showed the best growth performance overall (Table 2). The species canopy area growth was the highest, increasing on average 1.6 times per month. *Pseudobombax grandiflorum* had the second best performance due to high height and basal diameter increment, whereas *M.* cf. *multiflora* and *E. sulcata* had the worst performance. They had no significant growth in any of the parameters analyzed over the 2 years of the experiment.

Use Viability in Restoration Initiatives

A UVI was used to integrate the growth index and survival rate as a single value (Table 2). *Senna australis*, *P. grandiflorum*, *C. ensiformis*, and *T. guianensis* showed the best results and we recommend their use in open restinga areas subjected to wind and high temperatures, given some of the restrictions mentioned above. *Eugenia uniflora* and *T. bullata*, *C. emarginata*, *O. cuspidata*, and *P. glabrata* showed excellent survival and an intermediate reasonable growth, as did *E. ovalifolium*, which, however, had 15% mortality. *Allophylus puberulus* and *T. chrysotricha* had reduced growth but high survival and could also be used with relative success in restoration initiatives in similar restingas. The Myrtaceae, *E. rotundifolia*, *E. ovalifolia*, and *E. sulcata*, and particularly *M.* cf. *multiflora*, had a very poor overall performance.

Table 2. Growth index (total sum of points attributed from lower to higher growth for each species according to statistical differences in RGR in height, basal diameter, and canopy area divided by the maximum possible total) and UVI (growth index multiplied by survival percentage) for restoration of degraded restingas of 17 shrub and tree species.

Species	*Growth Index*	*UVI*
1 *Senna australis*	0.87	87
2 *Pseudobombax grandiflorum*	0.82	82
3 *Chamaecrista ensiformis* *Tapirira guianensis*	0.69	65.6
4 *Eugenia uniflora* *Tocoyena bullata*	0.56	56
5 *Cupania emarginata* *Ouratea cuspidata* *Pera glabrata*	0.52	52
6 *Erythroxylum ovalifolium*	0.56	47.6
7 *Allophylus puberulus*	0.47	47
8 *Tabebuia chrysotricha*	0.43	40.9
9 *Eugenia rotundifolia*	0.50	32.5
10 *Senna pendula*	0.47	28.2
11 *Eugenia ovalifolia*	0.34	22.1
12 *Eugenia sulcata*	0.50	17.5
13 *Myrcia* cf. *multiflora*	0.17	6.0

Discussion

Our results indicate that direct planting of tree and shrub seedlings has the potential to improve the restoration of degraded restinga areas: after 24 months, we obtained 81.9% survival of the 337 plants monitored and 12 of the 17 species studied had survival rates higher than 80%. Considering that tree species richness is an important structural component of woody plant communities and directly affects ecosystem function (e.g., Hooper et al. 2005), this result is very exciting indeed. In a long-term rehabilitation program conducted in coastal dune forests in South Africa, tree species richness of 18 was achieved in an average of 8–11 years after the beginning of the project (van Aarde et al. 1996). However, overall growth rate in our site was low, as indicated by the values obtained for RGR for height (11.32 cm/yr) of the eight species that showed significant positive growth values. This value is as small as the ones reported by Bozelli et al. (2000) for plants growing in a bauxite tailing substrate in the Amazon. Slow growth rates are most likely to be related to nutrient-poor soils, high light intensities, and the wind action common to all restingas (Araujo & Oliveira 1988; Scarano et al. 2001).

Senna australis showed the highest UVI of the species studied. The rapid soil cover by the canopy of this species indicates that it might be useful to early avoid or reduce grass invasion because grass invaders in the restingas, such as the ones removed prior to the start of the experiment, are typical sun plants. Moreover, it is likely that this species also plays an important role in nutrient cycling due to an apparently high leaf turnover that results in a thick layer of dry leaves in its understory. However, the use of *S. australis* in restoration programs requires caution because this plant shows a profuse branch formation close to the soil level, which may mechanically hinder seedling establishment underneath and around its canopy. *Pseudobombax grandiflorum* had the second best performance, despite its small canopy cover and deciduousness, which are the disadvantages to be considered in areas where invasive grasses are present. In such cases, it would appear that the use of this species could be in association with a species with higher potential for canopy cover.

It is likely that introduction of larger seedlings, the use of more fertilizers, and more frequent removal of invasive grasses should improve overall survival and growth, but under the conditions of the present experiment, the performances of the Myrtaceae *Eugenia rotundifolia*, *E. ovalifolia*, *E. sulcata*, and particularly of *Myrcia* cf. *multiflora* would not recommend their use on restoration projects. This was unexpected given that the natural vegetation that used to occupy the study site prior to damage is named Myrtaceae thicket. This low performance deserves further investigation, but we believe it could be related to age and size of seedlings and initial planting season.

Although we assessed only a small sample of the broad range of restinga species, the results obtained here provide an optimistic perspective for the possibility of successful restinga restoration, particularly from a biodiversity viewpoint. Recent excitement with the carbon sequestration potential of plantations of exotic trees in temperate coastal South America (Orellano & Isla 2004) should be viewed with caution. This example of successful introduction of native woody plants in Brazilian restingas indicates that this type of restoration may restore both biodiversity and ecosystem processes in tropical coastal habitats of Brazil and elsewhere. In turn, this may prove useful for carbon sequestration as well. Finally, studies on plant colonization and succession in natural restinga areas have often suggested that restingas are a fragile habitat, where plant germination and establishment depend on specific nurse-plants, such as bromeliad and *Clusia* sp. (see Scarano 2002, for review). Man can play ex situ the role played in situ by the nurse-plants, that is, grow restinga plants in nurseries until they can be safely transferred to the field (Eliason & Allen 1997). Future experiments using introduction of nurse-plants should cast further light on the adequacy of restoration strategies for the restingas.

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