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Restoration of seasonal semideciduous forests in Brazil: influence of age and restoration design on forest structure

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Abstract

With the high rates of deforestation in tropical regions, the restoration of degraded lands has become an important way for maintaining the diversity of plant communities and for creating wildlife habitats. Evaluating the success of restored areas is essential for improving restoration designs and for successfully restoring such complex ecosystems. In this study, the development of restoration forests with respect to age (5, 9 and 10 years old) and the restoration models used (proportion of pioneer trees) was assessed along the margins of Companhia Energética do Estado de São Paulo (CESP) reservoirs, located in the region of Pontal do Pananapanema, in São Paulo state, southeastern Brazil. The overstory (trees $\geq$4.8 cm in DBH) was assessed in nine 900 m^2 permanent plots, and all woody understory regenerating plants (>50 cm in height and <4.8 cm in DBH) were counted and identified in 54 1 m radius subplots. Canopy and grass cover were assessed in the wet and dry seasons. All of the parameters were recorded again 1 year later to evaluate the development of the forests. In general, neither the restoration design nor age appeared to influence forest structure and dynamics, at least at the developmental stage studied here. The floristic complexity and density of regenerating individuals were still fairly low compared to natural forests. The arrival of propagules from other forest remnants was insignificant, suggesting that inadequate seed dispersal and faunal colonization limited species enrichment within the restoration sites. Although the overall results suggested that these restoration forests had reached a structural complexity sufficient to give a start to secondary succession, some factors such as the dominance of pioneer trees in the overstory, the small number of colonizing individuals arriving from external seed sources, the persistence of weedy grasses in the understory and the degree of isolation of the restoration sites may endanger the sustainability of these forests in the long term. Although neither the age nor restoration model produced differences in the forest structure and dynamics, it is possible that such differences may require a longer time to develop. Monitoring restoration sites is essential for understanding a forest's trajectory and for guiding management and intervention practices.

Keywords: Forest restoration; Natural regeneration; Canopy; Ecological guilds; Grasses; Succession

1. Introduction

Restoration ecology is a subject of increasing interest and research in Brazil. Despite the very high rates of fragmentation in the Atlantic forest and its associated ecosystems (ISA, 2001), few attempts have

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been made at restoring deforested areas on a large scale.

In the state of São Paulo, southeastern Brazil, where remaining forest areas are estimated to be about only 9% of the original native vegetation (ISA, 2001), restoration practices received their greatest initiative in the early 1980s, mainly by hydroelectric companies seeking to compensate for the extensive flooding of areas of natural vegetation during the construction of dams and reservoirs. However, these first attempts were not very successful, since the restorations were based on the simple random planting of species, with no consideration for the basic principles of secondary succession (Kageyama and Gandara, 2000). At the end of 1980s, when the applicability of succession concepts to ecological restoration began to be discussed in Brazil (Kageyama and Castro, 1989), a new restoration model was proposed in which trees would be planted in a pioneer:non-pioneer proportion of 1:1 (''pioneer'', in this case, including pioneer and early secondary species, and ''non-pioneer'' including late secondary and climax species, as defined by Budowski, 1965). Thus, the shade of pioneer species would provide adequate conditions for non-pioneer species to develop (Kageyama and Gandara, 2000).

The lower cost of seedlings is an important advantage when using a high proportion of pioneer species to restore degraded lands. The greater production and availability of pioneer seeds during the year and the faster growth of its saplings allow a greater annual production of pioneer seedlings per area compared to non-pioneer species (V.L. Engel, personal communication).

In addition to lower costs and greater availability of seedlings, the quick re-covering of the soil proportioned by fast growing pioneer versus non-pioneer species reinforces the tendency to use a greater proportion of pioneer species in restoration models. Current restoration plantings frequently use a greater proportion of pioneer species than the 1:1 model initially proposed. However, it is still unclear to what extent the early death of this larger proportion of pioneer trees may produce an open canopy before the understory is able to develop successfully without any intervention.

Most studies done up to the end of the 1990s were restricted to analyzing the growth and development of only a few species in pure or mixed plantations, particularly during the first few years of forest establishment. Even today, adequate monitoring programs are still lacking, with few studies having been done in other Brazilian ecosystems (see the papers by J.A. Parrotta on the Brazilian Amazon) or are still in the initial stages (e.g. Engel and Parrotta, 2001). Similarly, there is little information on restoration forests development in Latin America (Leopold et al., 2001).

The assessment and monitoring of restored forests are essential for improving restoration techniques, especially in tropical and subtropical ecosystems in which the high diversity and complexity of interactions between organisms make restoration challenging. In recent years, the growing demand for restoration practices as a result of changes in the Brazilian environmental legislation has made it more urgent to determine the most appropriate techniques for successfully restoring such complex ecosystems.

This study describes the restoration of seasonal semideciduous forests in Brazil. We focused on assessing similar-aged restoration forests planted using models with different proportions of ecological guilds, and also assessed the restoration of forests planted using the same design at different times. The developmental stages of those restoration forests were recorded to provide the basis for further research and restoration practices.

2. Methods

2.1. Study areas

The two study sites are located along the margins of two Companhia Energética do Estado de São Paulo (CESP) reservoirs, currently belonging to Duke Energy International. One site is located close to the town of Rosana (Rosana reservoir), and the other, close to Itaguajé (Taquaruçu reservoir), in the region of Pontal do Paranapanema, on São Paulo and Paraná state boundaries, in southeastern Brazil (22°15′–23°00′S and 51°30′– 53°00′W, 300 m asl, Fig. 1). The restoration sites receive a mean annual rainfall of 1100–1300 mm. The mean annual temperature is 21 °C, with monthly maximum average (32 °C) occurring between January and March and a minimum (13 °C) between May and August (Diegues, 1990).

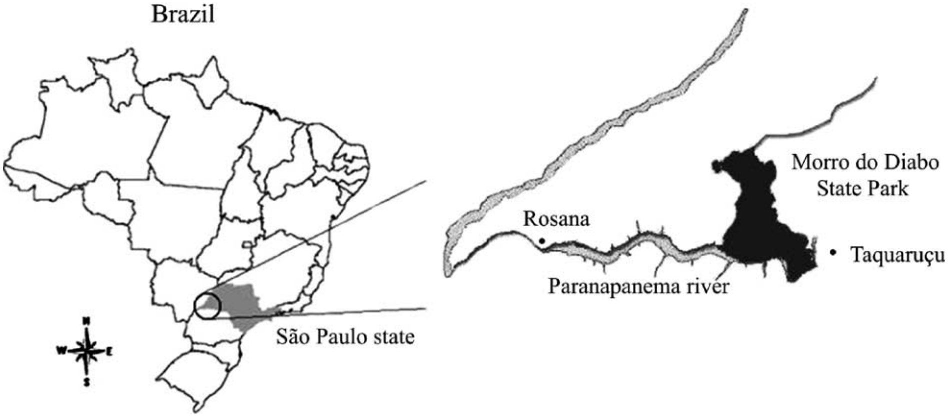

Fig. 1. Location of the study sites (Rosana and Taquaruçu reservoirs) and the largest remnant of seasonal semideciduous forest in São Paulo state (black filled area) located at Pontal do Paranapanema.

The regional vegetation is seasonal semideciduous forest, in which a portion of the trees defoliates during the dry season (Veloso et al., 1991). As with the rest of São Paulo state, this region is severely fragmented, and the landscape consists mostly of pastures with some forest remnants that cover only about 5% of the 246,000 ha of the original Pontal do Paranapanema Great Reserve, created in 1942 (Ditt, 2002). Despite the high level of fragmentation, the largest (34,000 ha) seasonal semideciduous forest remnant in São Paulo state, known as Morro do Diabo State Park, is located in this region (Fig. 1). In a survey done by Schlittler et al. (1995), 37 families containing 85 genera and 104 tree species were found in Morro do Diabo forest.

2.2. Restoration design

Two restoration areas along the margins of two reservoirs were examined, namely Rosana reservoir, with plantations 7–10 years old (plantings carried out from 1988 to 1991, in an area of 300 ha) and Taquaruçu reservoir, in which the planting was done in 1993 (5 years old at the beginning of this study). No herbicides were used in the plantings.

At the Rosana reservoir, we assessed only 9.4- and 10-year-old restoration forests since other sites had different degradation histories (including soil removal). In this account, the 9.4-year-old stand will be referred simply as 9-year-old. The landscape was similar in both areas, with the restoration sites surrounded by pastures, roads and open water from the reservoir (Fig. 2).

The difference between the restoration designs related mostly to species mixtures, with different proportions of ecological guilds among the trees planted. In model 1 (10 years old), the plantings were done using a pioneer:non-pioneer proportion of approximately 3:2. In model 2 (9 and 5 years old), the pioneer:non-pioneer proportion was around 4:1 (Table 1). The species planted are listed in Appendix A.

The 9- and 10-year-old restoration sites (stands with a similar age but different restoration models) were compared to assess forest development in relation to restoration design. The forest status in relation to age was assessed by comparing 5- and 9-year-old forests, in which the restoration design was essentially the same (model 2).

2.3. Overstory structure and dynamics

In July 1998, the overstory was studied in 900 m^2 permanent plots (30 m × 30 m) randomly located in each study site (4, 2 and 3 plots in 10-, 9- and 5-year-old forests, respectively). Planted and regenerating trees with DBH (stem diameter at breast height, 1.3 m) $\geq$4.8 cm were measured, tagged and identified. Planted and regenerating trees were easily distinguished because the latter were clearly out of the planting lines. Total height was measured with a 12 m tall telescopic pole, and the remaining height

Fig. 2. Aerial view of the reforestation areas at Pontal do Paranapanema, São Paulo, Brazil. *Upper:* Margins of the Rosana reservoir (9- and 10-year-old forests). *Lower:* Margins of the Taquaruçu reservoir (5-year-old forest). Note the predominance of pastures in the surrounding landscape at both sites.

of trees taller than this was estimated visually. Growth, mortality and recruitment were estimated by remeasuring all of the parameters 1 year later (July 1999).

2.4. Understory regeneration

The understory was studied in six 1 m radius subplots (3.14 m^2) randomly located within each plot. All individuals of woody species (trees, shrubs and lianas) greater than 50 cm in height and less than 4.8 cm in DBH were counted, measured (height) and identified. The measurements were recorded again 1 year after the initial survey.

2.5. Canopy and grass cover measurements

Canopy and grass (“capim colonião”—*Panicum maximum* L.) cover was assessed with a vertical densiometer (GRS Densitometer). In each plot (30 m × 30 m), five transects were established such that four of

Table 1
Restoration design of study sites at Pontal do Paranapanema, São Paulo, Brazil

Restoration design	Model 1	Model 2	Model 2
Reservoir	Rosana	Rosana	Taquaruçu
Year of planting	1988	1989	1993
Forest age at the beginning of this study (years)	10	9.4	5
Approximate spacing (m)	3.0 × 1.5	2.0 × 2.0	2.0 × 2.3
Pioneer:non-pioneer proportion	3:2	4:1	4:1
Number of tree species planted	42	39	38

them (approximately 21.2 m each) connected the midpoints of each side of the plot and one (30 m) connected the midpoint of the two opposite sides of the plot. This design was used to avoid measurements along or between planting lines. The measurements were taken at points 1 m apart along transects, in a total of approximately 113 points per plot. These measurements were taken in the wet (March) and dry (July) seasons.

2.6. Data analysis

Unpaired *t*-tests were used to compare basal area, height, tree density, number of species, growth and recruitment between models (9- and 10-year-old forests) and ages within the same model (5- and 9-year-old stands). Mortality percentage data were transformed (arcsin square root) prior to analysis (*t*-test). Spearman Rank correlation analysis was used to investigate the relationships between canopy and grass cover.

3. Results

3.1. Restoration model

3.1.1. Overstory structure and dynamics

Forest structure (basal area, height, tree density and number of species) and dynamics (basal area growth, height growth, mortality and recruitment) were very similar between the restoration models, with no differences among the parameters analyzed (Table 2).

Four species shared the overstory dominance in model 1 forest (*Cecropia pachystachya* Trécul, *Croton floribundus* (L.) Spreng., *Genipa americana* L. and *Peltophorum dubium* (Spreng.) Taub.). These species accounted for 44.5% of the total density of trees sampled, while in model 2 only two tree species (*C. floribundus* and *C. pachystachya*) comprised more than 50% of total tree density.

Overstory regenerating trees (i.e. non-planted trees with DBH $\geq$ 4.8 cm) comprised about 2% of the total tree density in model 2 and 11% in model 1 (Table 2). Nine of 13 regenerating tree species sampled in model 1 were common to planted and regenerating species (Appendix A). Two of these were species remaining before planting (palm trees) and the other two were found exclusively among regenerating trees. In model 2, only four tree species were found regenerating in the overstory (one species common to planting and regeneration, one species remaining before planting and two species found exclusively among regenerating plants, Appendix A).

3.1.2. Understory regeneration

A total of 27 species were found in the model 1 understory, of which 17 were trees, three were shrubs and six were liana species. In the model 2 forest, 14 species were trees and two were shrubs, giving a total of 16 understory species. No lianas were sampled at the model 2 site.

Only one (*Bastardiopsis densiflora* (Hook et Arn.) Hassl.) of the 17 tree species found in model 1 understory appeared to have regenerated from external primary forest sources or to have emerged from the soil seed bank since no adult individuals were planted in the area. All of the other species (including model 2 site) had at least one adult tree sampled in the overstory or were contained in the list of species planted in the restoration area, probably close to the survey plots.

Comparison of the density and number of species from understory regeneration revealed no significant differences between restoration models (Table 2).

3.2. Age

3.2.1. Overstory structure and dynamics

Apart from the basal area, which was lower in the 5-year-old site ($t = -3.59$, d.f. $= 3$, $P < 0.05$, Table 2), the structural parameters did not differ significantly between ages.

Table 2
Characteristics of 10-, 9- and 5-year-old restoration forests at Pontal do Paranapanema, São Paulo, Brazil (mean ± S.E., $n = 4$, 2 and 3 for the 10-, 9- and 5-year-old stands, respectively)

	Model 1 (10)	Model 2 (9)	Model 2 (5)
Overstory			
Total plot area (m^2)	3600	1800	2700
Density (trees/ha)			
Total density	1528 ± 94 a	1661 ± 128 ac	1426 ± 26 c
Planted trees	1361 ± 140 a	1628 ± 181 ac	1426 ± 26 c
Regenerating trees	167 ± 48 a	33 ± 0 a	0
Height (m)	9.1 ± 0.4 a	10.2 ± 0.4 ac	8.6 ± 0.4 c
Basal area (m^2/ha)	22.4 ± 1.5 a	24.9 ± 3.3 ab	14.8 ± 1.0 c
Species richness	23 ± 3.1 a	26 ± 5 ac	17.3 ± 1.3 c
Growth			
Basal area (m^2/ha per year)	1.61 ± 0.17 a	1.72 ± 0.05 ac	1.70 ± 0.26 c
Height (m per year)	0.47 ± 0.02 a	0.30 ± 0.1 ac	0.63 ± 0.07 c
Recruitment (trees/ha)	30.55 ± 5.3 a	55.6 ± 22.2 ac	33.3 ± 12.8 c
Mortality (%)	1.8 ± 0.8 a	1.3 ± 0.1 ac	1.8 ± 0.9 c
Understory regeneration			
Total plot area (m^2)	75.4	37.7	56.5
Density (individuals/ha)*	6499 ± 1992 a	3448 ± 796 a	0
	6631 ± 2602 a	4244 ± 0 ab	707 ± 707 c
Species richness*	5.2 ± 1.4 a	5 ± 1 a	0
	6.5 ± 1 a	5 ± 1 ac	1 ± 1 c

The numbers in parenthesis are the forest ages (years). Letter "a" indicates that means were not significantly different between models (10- and 9-year-old forests) ($P > 0.05$, t-test); letter "b" indicates significant differences between ages within the same model (5- and 9-year-old forests) ($P < 0.05$, t-test); letter "c" indicates that means were not significantly different between ages within the same model (5- and 9-year-old forests) ($P > 0.05$, t-test).

* The first row refers to the first survey and the second row, to the second survey (1 year later).

At both sites, the overstory was dominated by two pioneer species. *C. floribundus* and *C. pachystachya* accounted for 50.2% of the relative density in the 9-year-old forest, and *Trema micrantha* (L.) Blume and *Guazuma ulmifolia* Lam. dominated the 5-year-old site with 57.4% of the total density. Regenerating trees (DBH $\geq$ 4.8 cm) were absent from the 5-year-old overstory plots, and accounted for about 2% of total tree density in the 9-year-old forest (Table 2).

Basal area and height growth were not significantly different between ages. Similarly, there were no differences in mortality and recruitment (Table 2).

3.2.2. Understory regeneration

No woody regenerating plant was sampled in the 5-year-old understory in the first survey. In the second survey, the number of woody individuals sampled was significantly greater in the 9-year-old stand than in the 5-year-old forest ($t = -3.859$, d.f. $= 3$, $P < 0.05$, Table 2). However, the mean number of species did not differ between ages (Table 2, Appendix B).

3.3. Canopy and grass cover

Despite the age of the forests and the shade provided by the trees, there was a marked presence of weedy herbs and grasses at all sites (Fig. 3). The dominant species was the grass *P. maximum* (capim colonião), which is widely used for cattle grazing and was also predominant in neighboring areas.

There was a negative correlation between the canopy cover in the dry season and grass cover (Spearman's coefficient: $r = -0.8167$, $P = 0.0196$), and a

Fig. 3. A 10-year-old plot understory dominated by the grass *Panicum maximum* in the dry season (the foreground pole is 1.5 m tall).

positive correlation between the decrease in canopy cover (expressed as the difference between canopy cover in the wet and dry seasons) and the abundance of grasses ($r = 0.9333$, $P = 0.0089$).

4. Discussion

The parameters used to describe forest structure (height, density of adult and regenerating trees), and the number of species in restoration forests were apparently unaffected by the restoration models or by forest age, at least up to the stage of development examined in this study. The lack of difference in forest height between the 5- and 9-year-old forests, in contrast to the minor basal area of the 5-year-old site, showed the potential of pioneer species to allocate resources and energy to height instead of diameter growth, thereby promoting fast soil coverage and canopy formation.

Mortality and recruitment values were similar to those in tropical primary forests (Swaine et al., 1987; Philips and Gentry, 1994), suggesting that these restoration forests had reached a structure sufficiently complex to start secondary succession.

Although statistical tests revealed no significant differences in the overstory and understory regenerating trees between models, a subtle difference was observed in these forest physiognomies. The model 1 forest had started to loose its typical ‘‘plantation’’ appearance, which was being masked by the presence of regenerating individuals. On the other hand, the limited and scattered regeneration at the model 2 site left planting lines still clearly visible.

The great heterogeneity in species dominance may have masked possible differences between treatments. This variability in species number and composition is commonly observed in restoration plantings in Brazil, and reflects the inability of nurseries to supply seedlings of all species throughout the year. Seedling production is very seasonal (Barbosa et al., 2003) and depends on seed availability and an understanding of seedling formation.

The low number of species available on nurseries at certain times represents one of the greatest bottlenecks of forest restoration in Brazil. On average, projects involve only 35 tree species (Barbosa et al., 2003), whereas recommended richness is around 100–120 species (Rodrigues, 1999).

The floristic diversity and complexity of the ‘‘original’’ forest are still far from being reached at all restoration sites (see Schlittler et al., 1995). The very small fraction of regenerating species originating from external seed sources suggests that seed dispersal may limit species enrichment within restoration areas. In this case, the proportion of guilds (models) used in restoration appears to be of minor importance compared to the ‘‘quality’’ of the surrounding matrix. The presence of seed sources in the neighborhood, which is often limited in degraded landscapes (Wunderle, 1997) may be decisive for re-establishing species-rich communities. As reported by others (Tucker and Murphy, 1997; Carnevale and Montagnini, 2002; Holl, 2002), the floristic diversity of reforested sites may be greatly enhanced through propagules arriving from primary and secondary forest remnants (McClanahan, 1986; Robinson and Handel, 1993; Wijdeven and Kuzee, 2000) or even isolated trees (Carriere et al.,

2002) in the surroundings. However, this was not observed at the sites studied here. The degree of isolation may endanger the ongoing development of floristic diversity in restoration forests.

In this context, it is possible to infer that the low number of species present in restoration forests is closely related to the low number of species initially planted. The high diversity of tropical forests is usually associated with a large number of species occurring at low densities (Hartshorn, 1980), and has already been discussed and tested experimentally in some restorations projects in Brazil (Kageyama et al., 1994). These authors suggested that 'rare' and common species should be planted in the same proportion as they occur in nature. However, if dispersal is limited, species may be unable to find new populations in the restoration area, and this could be a problem for rare species conservation (Strykstra et al., 1998).

Not only the number of species but also the density of woody regenerating individuals was extremely low at the sites studied compared to natural and other restoration forests in Brazil. Durigan and Dias (1990) found 140,650 individuals/ha (plants greater than 5 cm in height and up to 2 m tall) in a 17-year-old, mixed species reforestation site. This is a very high value compared to the 28,800 individuals/ha (up to 2 m tall) reported by Parrotta et al. (1997a) in a 10-year-old restored site in the state of Pará. Although this area had been highly degraded by bauxite mining, the whole site was surrounded by primary forest, which contributed significantly to regeneration processes (Parrotta et al., 1997a). Grombone-Guaratini (1999) reported a density of 27,500 individuals/ha (individuals greater than 50 cm in height and up to 4 m tall but less than 4.8 cm in DBH) in a seasonal semideciduous forest site in São Paulo state.

Despite some differences in the sampling criteria and age of the forests that were compared, the low density of regenerating individuals suggested the presence of factors affecting the mechanisms of regeneration in the restoration forests studied. Among the several factors determining regeneration patterns in a forest, the land-use history (Guariguata et al., 1995), the seed bank (Brown, 1992; Guariguata et al., 1995; Strykstra et al., 1998; Bakker and Berendese, 1999; Zhang et al., 2001), the distance from external seed sources (McClanahan, 1986; Robinson and Handel, 1993; Parrotta et al., 1997a,b) and propagule dispersal (McDonnell and Stiles, 1983; Bakker et al., 1996; McClanahan, 1986; McClanahan and Wolfe, 1993; Robinson and Handel, 1993; Strykstra et al., 1998) were the most important features acting at the study sites.

The abundance of grasses (*P. maximum*) at all restoration sites was surprising, given the advanced age of the forests. The shade provided by forest canopy was not enough to suppress understory competing herbs, differently than the results reported by Parrotta et al. (1997a), expected by Haggar et al. (1997) and the observations taken by practitioners. Our results suggest that the persistence of this vegetation may be related to canopy opening in the dry season, a very common event in seasonal semideciduous forests. Although it has been not quantified, it is possible that the number of deciduous trees used in the plantations was excessive for forest restoration purposes. Thus, the massive leaf shedding in the dry season probably opened the canopy enough to provide grasses the light needed for survival.

Plant spacing was another factor that may have contributed to the establishment and persistence of grasses. Although the spacing used in the plantings studied (2000–2500 trees/ha) was lower than usual for plantings in São Paulo state (1667 trees/ha), it was still wide enough to allow abundant sunlight to reach the forest floor (J. A. Parrotta, personal communication). The main consequence for forest development is that this herbaceous layer may delay or even prevent the success of regeneration establishment in the understory (Parrotta, 1993; Guariguata et al., 1995; Parrotta et al., 1997a; Holl and Kappelle, 1999).

The dominance of a few pioneer species, especially at the 9- and 5-year-old restoration sites, associated with the abundance of grasses and a small number of regenerating individuals in the understory, may arrest the continuity of successional processes. The mortality of these short-lived species and the following reopening of the canopy can increase the chances of weedy species becoming established at these sites, and result in forest decline.

Despite the short period of assessment, some of the parameters (low species richness, abundance of grasses, predominance of pioneer trees and low density of regenerating individuals) suggested that the forests studied may not be self-sustainable. Taking into account that the abundance and diversity of colonizing species are influenced by the distance from external seed sources (van Ruremonde and Kalkhoven, 1991; Parrotta et al., 1997a), and that seed dispersal and colonization by new species are important factors affecting the restoration of biodiversity (Wunderle, 1997), the isolation of studied sites may endanger the long-term maintenance of diversity and successional processes.

As pointed out by Ewel (1987), the forest physiognomy and dominance of species alone are insufficient to measure the success of a restoration since in the long run the apparently successful community may disintegrate. The assessment of other important features such as wildlife colonization (Tucker and Murphy, 1997; Block et al., 2001), soil components and dynamics (Jackson et al., 1995), biotic interactions (Ewel, 1987), seed dispersal (Parrotta et al., 1997a) and many others (e.g. Bentham et al., 1992; Jackson et al., 1995; Aronson and Le Floc'h, 1996; Andersen and Sparling, 1997; Ehrenfeld and Toth, 1997; Jansen, 1997; Majer and Nichols, 1998) has been proposed to evaluate the success of restoration and may be useful for improving our comprehension of how restoration models work.

5. Guidelines for restoration

At the stage of development studied here, both models produced very similar forest structures and dynamics, although this does not necessarily mean that the different restoration models will result in identical forests. Ecological processes occur over a long time scale, and it is probable that the expected differences in forest structure will take longer to appear.

Planning a restoration within a landscape approach may facilitate the recruitment of fauna and flora (Bell et al., 1997) and can play a decisive role in the success of the restoration. External sources of propagules may be vital for maintaining and improving forest diversity throughout the years. If such sources are available, restorationists may rely on them to ensure the restoration of some ecological processes, such as dispersal, faunal colonization and regeneration.

Thus, if it is correct that the early mortality of pioneer species can be prejudicial to forest development when the understory is not developed enough to keep secondary succession going (Parrotta and Knowles, 1999, 2001), then it is reasonable to suppose that model 1 is preferable over model 2 where the landscape matrix does not provide sources of plant and wildlife species that can contribute to the enrichment of restored areas. On the other hand, if there are conserved forest remnants close to the restoration sites, planting a higher proportion of pioneer trees (as in model 2) could be a good strategy to provide a faster soil coverage and forest structure formation. Continuous monitoring (Tucker and Murphy, 1997; Lindenmayer et al., 2002) is strongly recommended to assess the trajectory of restoration forests, and is crucial for defining the most appropriate strategies for restoration and for guiding intervention practices in the restored ecosystems.

Grasses and other weedy invaders are still a problem in restoration forests, especially during the first years of establishment, and maintenance practices increase restoration costs. According to our results, the deciduousness of planted species seemed to favor the persistence of grasses, which is why the proportion of deciduous trees also deserves attention in restoration projects. In addition, alternative techniques should be used to prevent the survival of these grasses and other weedy invaders, such as reducing the spacing between trees and incorporating evergreen tree and shrub species in plantation designs (J.A. Parrotta, personal communication).

There appears to be a gap between research and restoration practice. Although restoration designs have been tested and discussed in recent years, often what is proposed in the projects is not exactly what is done in the field, mainly because of operational difficulties. Researchers need to appreciate that models must combine theoretical and practical features, to allow the restoration to be easily put into practice. Proposing spatially detailed or complex models may be theoretically ideal, but practically unfeasible, especially on a large scale.

Numerous factors must be considered in a restoration project and should be continuously evaluated. As pointed out by Bradshaw (1987), if any components that are critical for good functioning of the ecosystem are omitted, then it will function improperly.

In the tropics, where rates of deforestation are particularly high (Wilson, 1988; Young, 2000) and where formerly forested landscapes were converted into mosaics of small patches of forest remnants, forest restoration has become an increasingly important tool for species preservation (Jordan et al., 1988; Young, 2000) and for maintaining the diversity of tropical forest communities (Bawa and Seidler, 1998). A multi-disciplinary approach provides one of the best ways of working in restoration ecology in order to better understand and recreate such complex ecosystems.

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Appendix A

Tree species sampled in 5-, 9- and 10-year-old restoration forests at Pontal do Paranapanema, São Paulo, Brazil. M1: model 1; M2: model 2; G: tree species with DBH $\geq$ 4.8 cm; L: tree species with DBH $<$ 4.8 cm; p: planted; r: regenerating species. The numbers in parenthesis are the forest ages (years).

Family/Scientific name	M1(10)		M2 (9)		M2 (5)	
	G	L	G	L	G	L
Anacardiaceae						
Astronium graveolens Jacq.	p					p
Myracrodruon urundeuva Allemão	p					
Spondias lutea L.		p				
Annonaceae						
Annona cacans Warm.				p		
Duguetia lanceolata A. St.-Hil.		p				
Xylopia aromatica (Lam.) Mart.			p			
Apocynaceae						
Aspidosperma cylindrocarpon Müll. Arg.		p		p		
Aspidosperma polyneuron Müll. Arg.		p		p		
Tabernaemontana hystrix Steud.	r					
Araliaceae						
Didymopanax morototonii (Aubl.) Decne. Et Planch.	p					
Arecaceae						
Acrocomia aculeata (Jacq.) Lodd. ex Mart.	r					
Syagrus romanzoffiana (Cham.) Glassman			r			
Bignoniaceae						
Sparattosperma leucanthum (Vell.) Schum.	p					
Tabebuia heptaphylla (Vell.) Toledo	p				p	
Tabebuia impetiginosa (Mart. ex DC.) Standl.	pr				p	
Bombacaceae						
Chorisia speciosa A. St.-Hil.	p		p			

Appendix A. (*Continued*)

Family/Scientific name	M1(10)		M2 (9)		M2 (5)	
	G	L	G	L	G	L
Boraginaceae						
Cordia ecalyculata Vell.					p	
Cordia trichotoma (Vell.) Arrab. ex Steud.					p	
Patagonula americana L.				p	p	
Caesalpiniaceae						
Holocalyx balansae Micheli		p				
Hymenaea courbaril L.	p		p			p
Peltophorum dubium (Spreng.) Taub.	p		p		p	
Pterogyne nitens Tul.	p		p		p	
Caricaceae						
Jacaratia spinosa (Aubl.) A. DC.	p		p			
Cecropiaceae						
Cecropia pachystachya Trécul	pr		p		p	
Euphorbiaceae						
Croton floribundus (L.) Spreng.	p		p		p	
Croton urucurana Baill.	p		p		p	
Mabea fistulifera Mart.						p
Sapium glandulatum (Vell.) Pax			r			
Savia dictyocarpa Müll. Arg.	p					
Fabaceae						
Centrolobium tomentosum Guill. ex Benth.			p			
Lonchocarpus cultratus (Vell.) A.M.G. Azevedo and H.C. Lima			p		p	
Lonchocarpus muehlbergianus Hassl.	pr			p	p	
Machaerium aculeatum Raddi	pr		p			
Machaerium stipitatum Vogel	pr			p		
Myroxylon peruiferum L. f.	p		p			p
Platypodium elegans Vogel					p	
Poecilanthe parviflora Benth.	pr		p			
Pterocarpus rohrii Vahl	r			p		p
Lauraceae						
Nectandra megapotamica (Spreng.) Mez			p			
Lecythidaceae						
Cariniana estrellensis (Raddi) Kuntze			p			p
Meliaceae						
Cabralea canjerana (Vell.) Mart.					p	
Mimosaceae						
Acacia polyphylla DC.					p	
Albizia hasslerii (Chodat) Burkart	pr		pr			

Appendix A. (*Continued*)

Family/Scientific name	M1(10)		M2 (9)		M2 (5)	
	G	L	G	L	G	L
Anadenanthera macrocarpa (Benth.) Brenan			p		p	
Anadenanthera peregrina (L.) Speg.	p		p			
Enterolobium contortisiliquum (Vell.) Morong	p		p		p	
Inga cylindrica Willd.			p			
Inga laurina (Sw.) Willd.		p			p	
Inga vera subsp. *affinis* (DC.) T.D. Penn	p		p		p	
Parapiptadenia rigida (Benth.) Brenan		p			p	
Moraceae						
Chlorophora tinctoria (L.) Gaudich.	p					
Ficus sp.	p			p		
Myrtaceae						
Eugenia uniflora L.		p				p
Plinia rivularis (Cambess.) Rotman		p		p		p
Psidium guajava L.	p					
Phytolaccaceae						
Gallesia integrifolia (Spreng.) Harms			p		p	
Seguieria floribunda Benth.	r					
Polygonaceae						
Triplaris brasiliana Cham.			p		p	
Rubiaceae						
Genipa americana L.	p		p		p	
Randia armata (Sw.) DC.	r					
Rutaceae						
Balfourodendron riedelianum (Engl.) Engl.	p			p		p
Citrus aurantium			r			
Sapindaceae						
Diatenopteryx sorbifolia Radlk.						p
Sapotaceae						
Chrysophyllum gonocarpum (Mart. et Eichler) Engl.	p					
Sterculiaceae						
Guazuma ulmifolia Lam.	p		p		p	
Tiliaceae						
Luehea candicans Mart.			p			p
Ulmaceae						
Trema micrantha (L.) Blume					p	
Verbenaceae						
Citharexylum myrianthum Cham.	pr		p		p	
Vitex montevidensis Cham.	p		p		p	

Appendix B

Species of woody individuals (trees, shrubs and lianas) $\geq$50 cm in height and <4.8 cm in DBH sampled on the understory of 5-, 9- and 10-year-old restoration forests at Pontal do Paranapanema, São Paulo, Brazil. M1: model 1; M2: model 2. The numbers in parenthesis are the forest ages (years).

Family/Scientific name	Life form	M1 (10)	M2 (9)	M2 (5)
Amaranthaceae				
Hebanthe paniculata Mart. O. Kuntze	Liana	×		
Annonaceae				
Xylopia aromatica (Lam.) Mart.	Tree		×	
Apocynaceae				
Aspidosperma cylindrocarpon Müll. Arg.	Tree		×	
Prestonia cf. *riedelii* (Müll-Arg.) Mgf.	Liana	×		
Tabernaemontana hystrix Steud.	Tree	×		
Araliaceae				
Didymopanax morototonii (Aubl.) Decne. et Planch.	Tree	×		
Arecaceae				
Acrocomia aculeata (Jacq.) Lodd. ex Mart.	Palm tree	×		
Syagrus romanzoffiana (Cham.) Glassman	Palm tree		×	
Bignoniaceae				
Macfadyena unguis-cati (L.) A. Gentry	Liana	×		
Tabebuia heptaphylla (Vell.) Toledo	Tree		×	×
Boraginaceae				
Cordia trichotoma (Vell.) Arrab. ex Steud.	Tree	×		
Caesalpiniaceae				
Peltophorum dubium (Spreng.) Taub.	Tree	×	×	
Euphorbiaceae				
Croton floribundus (L.) Spreng.	Tree	×	×	
Sapium glandulatum (Vell.) Pax	Tree			×
Fabaceae				
Centrolobium tomentosum Guill. ex Benth.	Tree		×	
Lonchocarpus cultratus (Vell.) A.M.G. Azevedo and H.C. Lima	Tree			×
Lonchocarpus muehlbergianus Hassl.	Tree	×	×	
Machaerium stipitatum Vogel	Tree	×		
Poecilanthe parviflora Benth.	Tree	×		
Flacourtiaceae				
Casearia gossypiosperma Briq.	Tree	×	×	
Lauraceae				
Nectandra megapotamica (Spreng.) Mez	Tree		×	
Malpighiaceae				
Malpighiaceae sp.	Liana	×		

Appendix B. (*Continued*)

Family/Scientific name	Life form	M1 (10)	M2 (9)	M2 (5)
Malvaceae				
Bastardiopsis densiflora (Hook et Arn.) Hassl.	Tree	×		
Mimosaceae				
Inga cylindrica Willd.	Tree	×		
Mimosaceae sp.	Tree	×		
Moraceae				
Ficus sp.	Tree	×		
Myrtaceae				
Eugenia uniflora L.	Tree	×	×	
Phytolaccaceae				
Seguieria floribunda Benth.	Tree	×		
Rubiaceae				
Randia armata (Sw.) DC.	Shrub	×		
Sapotaceae				
Pouteria sp.	Tree	×		
Solanaceae				
Cestrum calycinum Willd.	Shrub	×	×	
Sterculiaceae				
Guazuma ulmifolia Lam.	Tree		×	
Teophrastaceae				
Clavija nutans (Vell.) Stohl	Shrub	×	×	
Tiliaceae				
Luehea candicans Mart.	Tree		×	
Verbenaceae				
Citharexylum myrianthum Cham.	Tree		×	
Unidentified				
Unidentified 1		×		
Unidentified 2	Liana	×		
Unidentified 3	Liana	×		

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