

Chapter 11

Recruitment of Dry Forest Tree Species in Central Brazil Pastures

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11.1 Introduction

Despite its importance as a wealthy source of unique biodiversity (Janzen 1988) and that it covers 42% of the tropical vegetation worldwide (Murphy and Lugo 1995), the dry forest is the most endangered terrestrial ecosystem and one of the least protected (Scariot and Sevilha 2005). In Central America for example, less than 1% of the original 500,000 km² is preserved in conservation units (Janzen 1988, Sanchez-Azofeifa et al. 2003). In Brazil, the seasonal deciduous forests—a type of dry forest—covers 27,367,815 ha (3.21%) of the territory, from which 1,072,946 ha (3.9%) are protected by conservation units. Among conserved areas of deciduous dry forest, only 117,980 ha (0.43%) are in conservation units of restricted use as national parks and biological reserves. Most of the conservation units (402,456 ha or 1.47%) allow limited economic use of the land and natural resources. There is no information of the conservation unit assigned in official database for 552,509 ha (2.02%) of dry forests (Sevilha et al. 2004).

Dry forests occur in many soil classes. In Brazil they occur in at least 13 soil classes of the Brazilian Soil Classification (Scariot and Sevilha 2005), but deciduous dry forests occur predominantly in rich soils. There are many valuable trees used for timber. These forests are the main targets of farmers, who log the trees and sell the timber to acquire financial resources to establish farms. In Central Brazil, where some of the most important remnants of dry forests of the country occur (Fig. 11.1), most of the area originally occupied by dry forests has been replaced by pasture composed by exotic grass in extensive farmlands on rich soils.

Recommendations have been made for the creation of new conservation units, but given the high degree of landscape fragmentation any new conservation unit will contain significant proportion of its area covered by pasture and forest fragments with different levels of disturbance caused by logging, cattle grazing, fire and invasive species. Thus, in this scenario it is fundamental to understand the mechanisms of tree regeneration in pasture fields to address vegetation recovery and use this information to facilitate forest regeneration.

There are four to five times more studies on the natural regeneration of rainforests than on dry forests, and even less on restoration (Vieira and Scariot 2006a). Taking into account the threat level, the lack of conservation units, the lack of information

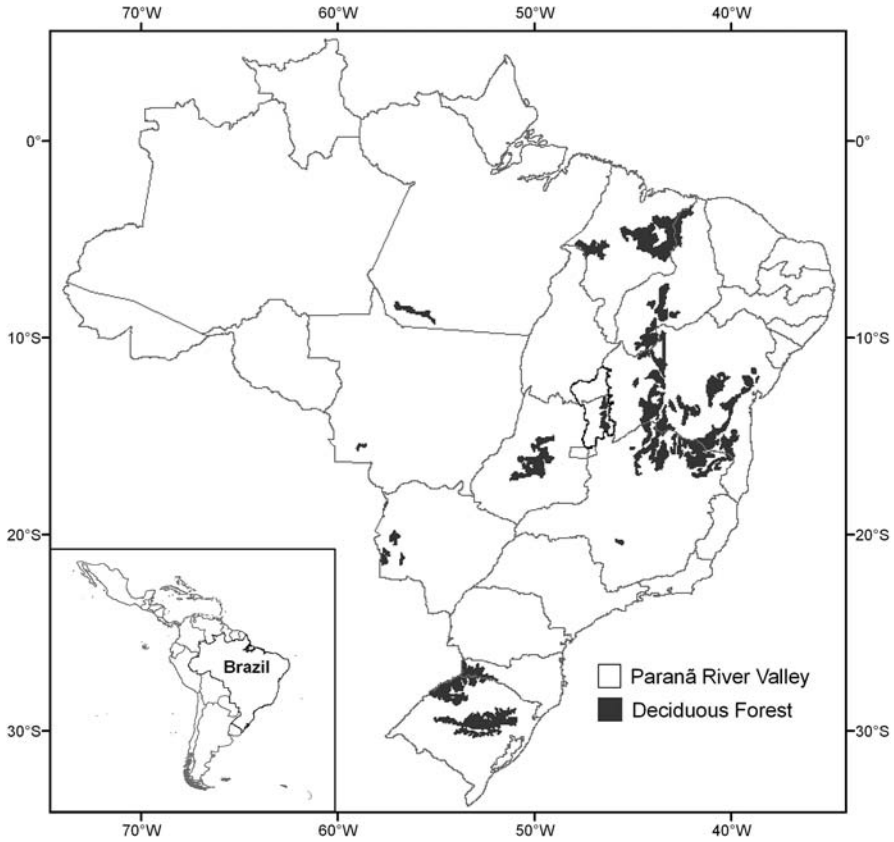


Fig. 11.1 Map of Brazil showing the distribution of the deciduous forests (shaded areas) and the location of the Paranã River Valley in Central Brazil

on regeneration mechanisms, and understanding how regeneration can be enhanced are crucial concerns for the conservation, restoration and management of the dry forests.

11.2 The Case of Paranã River Valley in Central Brazil

The Paranã River valley has an area of 59,403 km² in the Cerrado biome. It is embedded in a transition zone between the wet Amazon region and the semi-arid Caatinga region. It has a Tropical climate with two well defined seasons (AW of Köppen) varying to Cwa (Altitude Tropical Climate) (IBGE 1995).

This region presents a mosaic of vegetation types, and has some of the last remnants of seasonally deciduous forests in Brazil. These occur from 400–600 m above sea level, with annual precipitation of 1,236 mm ± 255 (SD, data from 1969 to 1994). 89% of the rains fall between October and March. The mean annual

temperature is 23°C, fairly constant throughout the year. The landscape is flat and consists of limestone geology, with occasional limestone outcrops containing a moderately distinct tree flora (Silva and Scariot 2003). Nitosoil is the main soil class upon flat to lightly rugged (Krejci et al. 1982). Rocky limestone outcrops are common in the region.

This basin is between two well known vascular plant distribution patterns in Brazil. One of them is formed by forest species with different levels of leaf shedding that depend mainly on the presence of fertile soils within the Cerrado domain and occurs mainly in the northeast-southeast arch, connecting the Caatinga to the Chaco border. The other pattern is formed by species from the Amazon and Atlantic rainforests, crossing the Cerrado biome through riparian forests (Oliveira-Filho and Ratter 1995).

On the flat lands, the forest canopy is 17–23m height with a basal area of 23–28m²/ha (Scariot and Sevilha 2005). The authors found 128 tree species in 90 genera and 41 families. Most species are commonly found in large Brazilian biomes such as Cerrado, Caatinga, Amazonia and Atlantic Forest (Scariot and Sevilha 2005). Cattle farming, logging and incipient agriculture are the main economic activities of the region. Wood has been harvested mainly for corral and fence building, for the production of woodchip, and for civil construction. The main exploited species are *Astronium fraxinifolium* Schott, *Myracrodruon urundeuva* Fr. All., *Schinopsis brasiliensis* Engl. and *Tabebuia impetiginosa* (Mart. ex DC.) Standl., which have been commercialized locally, regionally and also in the Southern Brazilian markets (IBGE 1995).

This region has been acknowledged for biodiversity conservation efforts by Brazilian researchers and the Government (Brasil 2002). However, despite its importance, the region is dominated by cattle farms and forest fragments rarely exceeding 250 ha (Andahur 2001). There are three conservation units, but only the smallest one, with less than 2,000 ha, is fully implemented.

11.3 Seed Germination in Pastures

Seed dispersal is the main barrier for plant recruitment on abandoned pastures (Holl 1999, Holl et al. 2000, Wijdeven and Kuzee 2000), but it can be overcome either by direct seeding (Engel and Parrotta 2001, Camargo et al. 2002, Doust et al. 2006) or by installation of bird perching (Melo et al. 1997, Holl 1998, Duncan and Chapman 1999, Melo et al. 2000). However, after sowing germination can also be highly limited by (1) seed predation (Nepstad et al. 1996, Holl and Lulow 1997, Jones et al. 2003, Myster 2003, 2004a,b); (2) competition with exotic grasses (Desteven 1991, Holl 1999, Holl et al. 2000) and (3) water shortage (Lieberman and Li 1992, Ray and Brown 1995), and (4) pathogens (Myster 1997, 2004a). The relative importance of each factor varies widely among species and studied areas (Holl 1999).

Although studies are scarce, seed predation has been found to have high impact on seed germination of dry forest species in pastures in Central Brazil (Vieira and Scariot 2006b), similarly to findings from other tropical regions (Nepstad et al. 1996,

Camargo et al. 2002, Myster 2004a,b). The intensity of seed predation is highly dependent on the seed species (Osunkoya 1994, Holl and Lullow 1997). Medium sized seeds (0.2–4 g) are usually more susceptible to rodents, since they are easier to find and they are easier to manipulate than larger seeds (Price and Jenkins 1986, Nepstad et al. 1990, Osunkoya 1994, but see Hammond 1995, Holl and Lullow 1997 for different results). Less rodent abundance in pastures may result in less medium-size seed predation than in forests (Jones et al. 2003, Meiners and LoGiudice 2003). Small seeds are eaten mainly by insects, such as ants (Nepstad et al. 1990, 1996, Osunkoya 1994), resulting in higher predation of small seeds in pastures (Nepstad et al. 1990, 1996, Duncan and Duncan 2000, Jones et al. 2003). From 6 tree species studied in active pastures of Central Brazil dry forests, the small seeded *Tabebuia impetiginosa* (0.11g) and *Astronium fraxinifolium* Schott ex Spreng. (0.02g) were highly preyed or removed, mostly by ants (*Tabebuia*: 50% and *Astronium*: 28%), when compared to seeds dispersed in forest remnants. In recently abandoned pastures, small seeds are preyed and removed at rates varying from 46–73% under grass cover, although one species, *Guazuma ulmifolia* experiences only 1% of seed loss (Guarino 2004). The large seeded (>7.5g) *Cavanillesia arborea* (Willdenow) K. Schum. and *Swartzia multijuga* Vogel, had over 90% predation by cattle in pastures (Vieira and Scariot 2006b).

Harsh climatic conditions during the dry season and dry spells in the wet season can be critical for seed survival in dry forest regions (Gerhardt 1994, Ray and Brown 1995, McLaren and McDonald 2003). Desiccation killed ca. 75% of the thin-coated and high water-containing seeds of *Swartzia multijuga* and *Eugenia dysenterica* DC. in the pasture (Vieira and Scariot 2006b). A small amount of shade in the forests, however, was enough to prevent seed desiccation. Small seeds with low water content and small mass can survive the harsh environmental conditions of pastures (Vieira and Scariot 2006b). In the wet season, grasses maintain soil moisture to the same levels found under deciduous dry forests canopy (22–23%) thus contributing to a significant increase in seed germination in 6 out of 12 tree species in Central Brazil pastures (Guarino 2004). Three species (*Cedrela fissilis*, *Amburana cearensis* and *Sterculia striata*) showed increases in seed germination by 2 to 10 fold under grass cover (Fig. 11.2).

Pathogen attack was also a significant source of seed mortality, mostly for *Aspidosperma pyrifolium*. *A. pyrifolium* and *Tabebuia impetiginosa* experienced high pathogen attack in greenhouse seedlings, mainly in most shaded conditions (Vieira 2006). Even though this is well known for rain forest species (e.g. Augspurger 1983, Dalling et al. 1998), seed mortality by pathogens has been recently reported for dry forest species as well (Grogan and Galvão 2006, Vieira and Scariot 2006b). Despite the lack of information, it has been reported that *Fusarium* and *Colletotrichum* fungi are the second most important seed mortality agents in abandoned pastures and early succession areas in Puerto Rico and Ecuador (Myster 1997, 2004a). However, fungi from *Phoma* and *Phyllosticta* genera and *Colletotrichum gloeosporioides* although abundant on *Inga* and *Cecropia schreberiana* respectively, resulted in modest seedling leaf loss (Myster 2002).

The use of direct seeding in pastures must consider that adequate species choice and seeding time are fundamental for restoration success. In the Parana River Valley

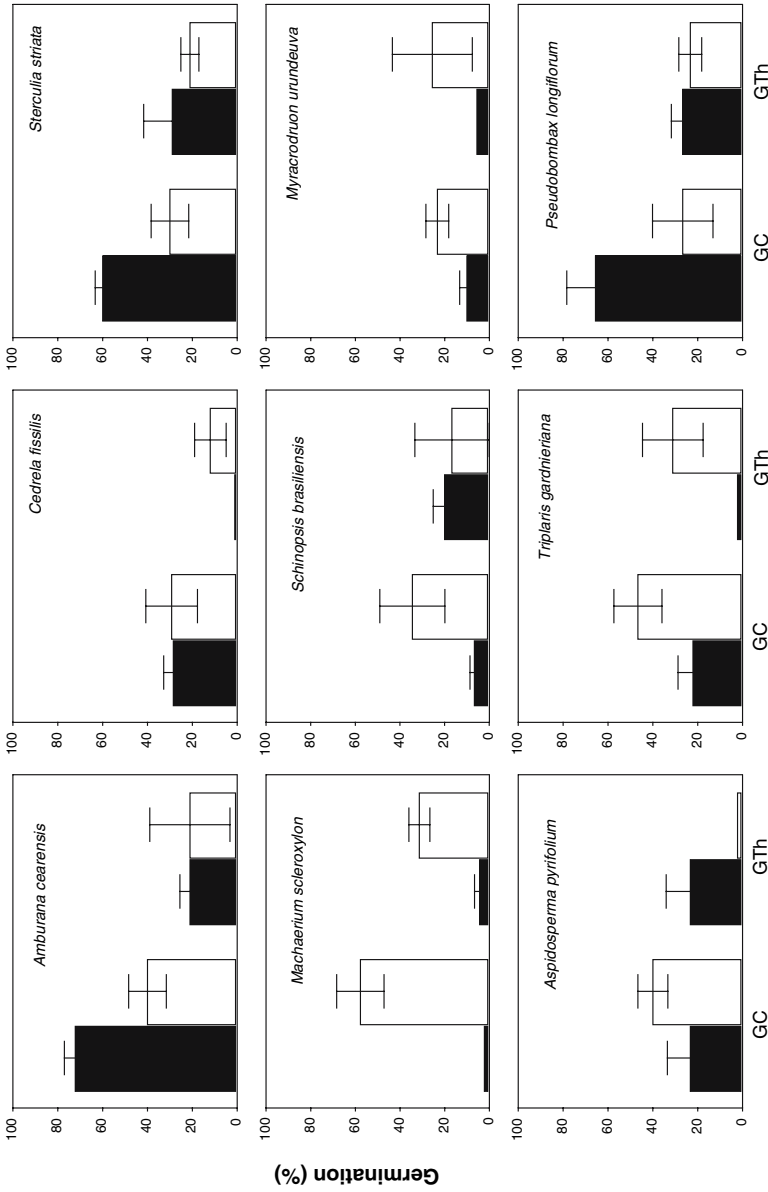


Fig. 11.2 Seed germination (mean \pm 1 standard error) of nine dry forest tree species in abandoned pastures in the Paran River Valley, Central Brazil. Shaded bars indicate buried seeds and opens bars unburied seeds (Grass Cover (GC), Grass thinning (GTh))

dry forests, priority must be given to species with seeds with small mass and low water content that can tolerate the harsh environmental conditions found in early successional stages and are less affected by predation and removal.

The positive effect of grass cover on seed germination must be further explored to better understand seedling establishment and growth. Burying seeds instead of disposing them on the soil surface, seems to be a good strategy to decrease mortality by predation (Woods and Elliott 2004) and desiccation (Negreros-Castillo et al. 2003) in agricultural lands previously covered by dry forests and degraded soils (Doust et al. 2006). However, Guarino (2004) found contradictory results, highlighting that this issue deserves further investigation.

11.4 Seedling Establishment

Several factors have been indicated to restrict the establishment of forest tree seedlings in pastures. These factors can act differently in germination and seedling establishment processes (Schupp 1995). For instance, some species can germinate under grass cover but will not develop due to grass competition (Zimmerman et al. 2000, Holl 2002). Herbivory by wild (Nepstad et al. 1996, Holl and Quiros-Nietzen 1999, Sweeney et al. 2002) or domestic animals (Posada et al. 2000) have been considered to affect seedling establishment in pastures. Other factors include limited water and nutrient availability (Aide and Cavelier 1994, Nepstad et al. 1996), competition with exotic or invasive plants (Holl 2002, Sweeney et al. 2002, Hau and Corlett 2003), high light availability, high air and soil temperatures (Nepstad et al. 1996, Loik and Holl 1999, Alvarez-Aquino et al. 2004) and soil compaction (Nepstad et al. 1996). Among these, grass competition is considered to be the most limiting, especially in tropical rain forest regions (Holl et al. 2000). Compared to forest environments, the establishment of seedlings in pastures is considerably reduced (Aide and Cavelier 1994, Nepstad et al. 1996, Alvarez-Aquino et al. 2004).

In two abandoned pastures originally covered by seasonal deciduous forests in Central Brazil, tree seedling survival was low but apparently was not related to grass competition. Despite the low survival, thinning the grass cover of *Andropogon gayanus* Kunth. did not interfere on the survival of 6 out of 7 species of planted tree seedlings after 14 months (Fig. 11.3). Some grass species which develop in tussocks, such as *A. gayanus*, leave bare soil patches, covered by a canopy of grass leaves, in between the grass tussocks. In the water deficient ecosystem of dry forests, grass canopy improves the microclimate allowing germination and establishment of tree seedlings in between the grass tussocks (Aide and Cavelier 1994, Guarino 2004). Pasture temperature is higher in plots where the grass cover is cleared (Gerhardt 1996, Zimmerman et al. 2000). Therefore the cooler and moister environment created by the grass cover seems to balance the negative effects of grass competition (Gerhardt 1996).

The benefits of grass cover might be even higher than the possible negative effects of grass competition. One of the 7 planted species in abandoned pastures in Central Brazil, *Talisia esculenta* (A. St.-Hil.) Radlk., had lower survival for grass thinning

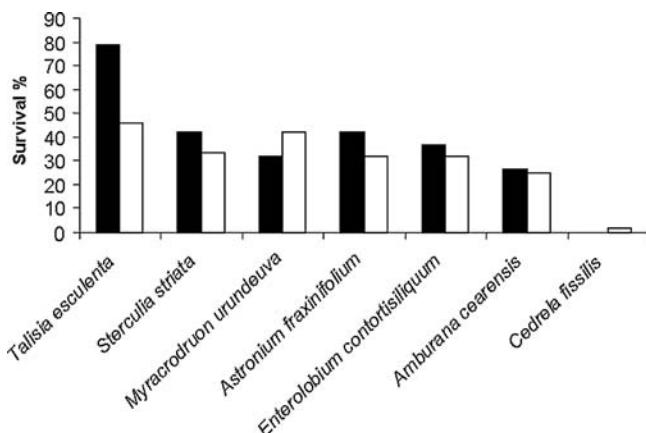


Fig. 11.3 Percentage survival after 14 months of planting of tree seedlings in two pastures in areas originally covered by Dry Forests (data from both pastures aggregated). Dark bars indicate no grass thinning ($N = 19$ seedlings per species) and open bars indicate grass thinning ($N = 57$ seedlings per species; grass thinning once, twice or four times). The only species that showed significant effect of grass thinning was *Talisia esculenta* (Pearson Chi-square = 6.4, $df = 1$, $p = 0.01$)

treatment (Fig. 11.3). *Talisia esculenta* seems to survive less in water stressed conditions of bare soil during the dry season and seems to be able to survive under low light levels created by the grass cover. The soil tends to be moister under the grass canopy (Hooper et al. 2002), allowing higher survival and growth of some tree species than in areas with less grass cover (Aide and Cavelier 1994). In dry forest regions water might be the most limiting resource (Gerhardt 1996, Cabin et al. 2002, Marod et al. 2002) and the grass cover in pastures might ameliorate the water deficit for tree regeneration (Aide and Cavelier 1994, Gerhardt 1996). The removal of dominating shrubs in a deforested seasonal moist evergreen forest in Uganda affected tree regeneration in a negative way, indicating possible facilitation interactions between trees and shrubs (Duncan and Chapman 2003). This was similar to the interactions for tree seedlings survival and grass cover found in our studies in Central Brazil. The interactions among grasses and tree regeneration in harsh environments such as pastures in dry forests tend to be more a facilitation relationship rather than competition (Callaway and Walker 1997).

Despite of the apparent positive effect of grass cover on tree seedling survival, seedling growth can be reduced by grass competition (Holl 1998, Hooper et al. 2002, Schaller et al. 2003). A study carried out by Sampaio et al. 2007 in the same region in Central Brazil dry forests that cleared out the grass cover leaving only the naturally established tree regeneration found a higher average increase in height compared to control plots, indicating competition between natural tree regeneration (mostly root-suckers) and grasses. The growth of dry forest tree seedlings planted in greenhouses under different light conditions (10, 40 and 72% of full sun photon flux density) seems to be higher in the most elevated light levels for the majority of tested species, however seedling survival did not follow a consistent pattern according to the light gradient (Vieira 2006). Grass root system also inhibits tree root

development resulting in low growth due to grass competition (Putz and Canham 1992, Schaller et al. 2003). In areas with pronounced dry season, grass root competition may have decreased seedling survival and growth. However, clearing out the grass cover (i.e. mowing treatment) can transform pastures in even harsher environments, as detrimental to seedling establishment as grass competition (Hooper et al. 2002).

Light seems to promote seedling growth, and grass cover might out compete tree seedlings creating shadow, inhibiting tree root development and therefore reducing growth. In contrast, low light levels as those created by grass cover may not promote tree seedling survival and some species regeneration may even be facilitated by grass cover. This may indicate water deficit reduction of bare soil of dry forest regions. Therefore, light and water interacting with grass cover seem to be promoting tree seedling establishment at the studied pasture sites in the dry forest regions. These results suggest that seedlings can be introduced in pastures among grass tussocks for pasture restoration (as in *A. gayanus* pastures in Central Brazil, Sampaio et al. 2007). However, grass cover should be cleared just around the seedlings to improve tree seedling development (Vieira and Scariot 2006a). This will avoid excessive exposure to the plants under harsh conditions.

11.5 Resprouting Ability

Root-suckers from trees, shrubs and lianas are found in recently established pastures in tropical forest areas (Uhl et al. 1988, de Rouw 1993, Nepstad et al. 1996). The density of these shoots can be high enough to allow forest regeneration if not eliminated. Actually, shoot elimination by clipping or by use of selective herbicides and root collection after plowing, are common management practices in Central Brazil dry forests region. These practices respond to a large proportion of annual money and labor expenditures in pasture management. Even though farmers battle forest plants exhaustively, many dry forest woody plants persist resprouting many years after pasture establishment (Vieira et al. 2006c, Sampaio et al. in press).

The real contribution of root-suckers relative to seedling establishment for forest regeneration in pastures along tropical forest areas is neglected. To some degree this is due to the laborious work need to detect resprouting which involves digging the superficial soil. The other reason is simply because root-suckers are depleted by sequential cutting, fire, and intensive tractor use (Uhl et al. 1988, de Rouw 1993, Sampaio et al. 1993). Much more attention has been given to evaluate limiting factors and how to improve seed dispersal and survival and seedling survival and growth (reviewed in Holl 2002, Myster 2004b).

Plowing the soil in the middle of the dry season and seeding grass at the beginning of the rainy season is a common practice among farmers of Central Brazil to restore grass cover in pastures. Plowing completely eliminates plant cover, allowing plants to start resprouting soon. Vieira et al. (2006c) investigated the regeneration from root-suckers after plowing in a 10 year-old pasture, a 25 year-old pasture and an early successional forest recently clear cut in Central Brazil dry forests. They

found that species richness and density were extremely high compared to other disturbed tropical sites with comparable history of land use (e.g. studies by Uhl et al. 1988, Nepstad et al. 1996, Zahawi and Augspurger 1999, Holl et al. 2000). They found 14–182 individuals per 100 m² of 30–42 species (6, 5–17, 6 species per 100 m²) among the three sites, representing 80% of the species richness found in intact forest remnants (Vieira et al. 2006c). In another investigation on tree regeneration in active pastures in dry forests of Central Brazil, stem density, regardless of the origin, did not varied with pasture age (from ≤ 6 up to 40 years old, n=25), while richness decreased significantly with age (Sampaio et al. in press). Despite the long time since forest removal, pastures still maintained a significant number of dry forest species. The 40 year-old pastures still had 12 dry forest tree species regenerating in relatively high densities (19–58 stems/100 m², 2nd and 3rd quartiles). Species composition was quite similar among pasture ages and pastures and forest remnants (Vieira et al. 2006c, Sampaio et al. in press), indicating the potential of these areas for dry forest return.

These optimistic results corroborate others findings that dry forest species are strong resprouters and contribute to forest regeneration after disturbance. In other tropical dry forests, studies show a high frequency of resprouting after disturbance (Kennard et al. 2002, McLaren and McDonald 2003). The reasons why resprouting is a particularly important recovery mechanism in dry forests are not certain. Some probable reasons for this trait are (1) seeds have a lower probability of establishment, so the survivorship by resprouting was favored in this forest ecosystem (Ewel 1980); (2) trunk bases are less prone to decay, having more chance to resprout (Ewel 1980); and (3) dry forest plants are adapted for above ground mortality or drought, so they usually resprout (Sampaio et al. 1993, Bond and Midgley 2001).

Although some species are strong resprouters others are not and resprouting ability will depend on the frequency, intensity and duration of the disturbance (Sampaio et al. 1993, Kammesheidt 1999, McLaren and McDonald 2003). Information available from actual literature does not allow one to predict what species are able to resprout in a forest community (Bond and Midgley 2001, Vesk and Westoby 2004). Despite this limitation on the current knowledge it is essential to understand the traits or identities of strong resprouters to be able to (1) help to understand present and future community composition (Kennard 2002, Saha and Howe 2003); (2) use root and branch cuttings of these species as nurse trees in restoration plans; and (3) invest in other restoration mechanisms for weak or non-sprouting species in early successional forests dominated by resprouters.

In dry forests of Central Brazil, species show high density in forest remnants, but that does not happen in ploughed sites (see above in this section; Vieira et al. 2006c). Species with low-density wood such as *Cavanillesia arborea* and *Chorisia pubiflora* (A. St.-Hil.) G. Dawson. *Spondias mombin* L. and *Pseudobombax tomentosum* (C. Martius and Zuccarini) Robyns, and soft wood species were found only in forest fragments and early successional sites. Species with low-density wood could lose their resprouting ability faster and sometimes did not resprout at all because of decay. In a chronosequence of pasture implementation, most species appear to be able to continue resprouting for up to 11–16 years after disturbance (Sampaio et al. in press). However, in 40 year-old pastures *Eugenia dysenterica* dominated the

community, which was poorer in tree species than younger pastures. *E. dysenterica* occurs in Central Brazil dry forests, but occurs in high frequency in the surrounding savanna vegetation. Species from the Brazilian savannas tend to resprout more than cogenetic species typical from riparian forests in the Cerrado biome, where root systems is not as developed as well as in the dry soils of savannas (Hoffmann et al. 2003). Therefore, savanna species like *E. dysenterica* that develop deep root system to survive in dry soils of savannas and dry forests might be able to survive longer and even increase its density via vegetative reproduction in pastures. The composition of the secondary vegetation developed in abandoned pastures will be dominated by strong resprouters, probably hard wood and savanna-related species.

Dry forests of Central Brazil show a high degree of resilience after pasture implementation due to root resprouting. Thus, planting seedlings to enhance early succession in abandoned pastures may not be desirable, particularly if standard planting techniques are used. Soil and vegetation disturbances, such as plowing or mechanical hole digging, reduce the density of naturally regenerating trees and consequently slow recovery (Sampaio et al. 2007). Non-resprouting species can be efficiently introduced by cautiously planting nursery-grown seedlings, minimizing damage to natural regeneration. Considering that resprouting ability is positively related to branch cutting success (Itoh et al. 2002), using branch cuttings to improve dry forest recovery should be tested. The rooting ability of branch cuttings was tested, with relative success, for tropical rain forests (Itoh et al. 2002, Zahawi 2005), but we have not found published studies for dry forests. Root cuttings taken from fallen trees from deforested areas may also be tested to enhance dry forest restoration, since planting them is equivalent to root-suckering after plowing.

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