Introduction

Over the last century vast regions of arable land have been degraded and abandoned. The Food and Agriculture Organization of the United Nations has estimated that 28% of the world’s soils present some degree of degradation. In Latin America, the estimates are that 27% of all land area is either severely or very severely degraded (Bot et al. 2000). In Brazil alone this amounts to a total of 236 million hectares (Mha) or approximately four times the area dedicated to arable crops.

A large part of the degraded area in Brazil is of run-down planted pasture in the central savanna (Cerrado) region of the country, which represents at least 50 Mha (Zimmer and Euclides 1997, Boddey et al. 2003). Degradation of these pastures, which generally occupy former agricultural areas, occurred mainly due to overgrazing, absence of chemical fertilization and regular burning of vegetation. However, the favorable relief and annual rainfall of >1000 mm of this region allow these areas to be reclaimed for food, timber or sugar-cane production using either conventional or, preferably, no-till techniques along with application of chemical fertilizers.

In contrast to the gently rolling areas of Cerrado, reclamation of the large areas of deforested hillside in the Atlantic coastal region needs more intensive intervention. This region, which stretches from the State of Rio Grande do Norte in the northeast (latitude 5°) to Santa Catarina (latitude 30°), and occupies ~17.4% of Brazilian territory, was originally covered by ~100 Mha of exuberant rainforest (Dean 1995, Boddey et al. 2006). A recent study has estimated that only 11–16% of the original Atlantic Forest cover now remains, and most of it consists of small forest fragments, which are generally isolated,
unprotected and/or severely altered (Ribeiro et al. 2009). This devastation was the result of centuries of non-sustainable agricultural and other land-use activities, described in detail by Dean (1995). As a result, most of the cleared forest areas have evolved into pastures of very low productivity, which in many areas have formed immense gullies due to the intense summer rainfall. These gullies are gradually expanding every year, causing a huge loss of sediments that are deposited in streams and rivers of the watersheds.

The subsoil exposed in gullies has virtually no organic matter, is very poor in plant nutrients and lacks any significant seedbank. Similar situations occur in subsoils exposed by road cuttings, land leveling on construction sites and mining activities. Under these conditions, the revegetation of these areas by native vegetation is almost always extremely limited. Even in the humid tropical conditions of southeastern Brazil, these sub-soils can remain exposed for many years. Moreover, while seeds from native forest species may be deposited by wind, water or animal vectors, and various may germinate, they do not thrive and spontaneous revegetation of the areas normally does not occur. In other situations these areas are invaded by persistent, fire-prone grasses that are known to preclude or severely slow subsequent forest regeneration (Uhl et al. 1988, Aide et al. 1995, Nepstad et al. 1996, Parrota et al. 1997).

The capacity of legume trees to obtain N from the air through symbiosis with N₂-fixing bacteria, collectively called rhizobia, led to the idea that these plants could be useful agents to revegetate such seriously degraded soils and to accelerate the natural succession. Trees from the genera Acacia, Mimoso and Gliricidia, among other N₂-fixing species, were tried initially, and early investigations soon showed that in these subsoils there was an almost complete lack of rhizobium bacteria capable of nodulating these species (Franco and de Faria 1997, Andrade et al. 2000, Costa et al. 2004).

Most legume tree species are also able to form symbioses with arbuscular mycorrhizal fungi (AMF), which are known to improve the capacity of the plant to take up phosphorus and other macro- and micro-nutrients in sub-optimal situations (Barea et al. 1987, Pacovsky 1988). Arbuscular mycorrhizal fungi are also known to mitigate water and salt stresses (Sanchez-Diaz et al. 1990, Evelin et al. 2009) and to interact synergistically with rhizobium, resulting in better plant performance (Ames and Bethlenfalvay 1987, Barea et al. 1987, Pacovsky 1988, Siqueira et al. 1998). However, as for free-living rhizobium bacteria, propagules (spores) of AMF are almost totally absent in subsoils (Franco and de Faria 1997). This led to the development of technologies to produce legume tree seedlings inoculated with selected rhizobium strains and AMF species suitable for the recovery of severely degraded areas.

In this article we describe the results of several studies where N₂-fixing legume tree species were used for recuperation of gullies, deforested hillsides and areas degraded by mining activities, emphasizing the potential of the technique to restore soil organic matter (SOM) levels, ecosystem biodiversity and other environmental functions.

**N₂ fixation in legume trees**

Most tropical legume tree species are able to nodulate with effective (able to fix nitrogen from the atmosphere) rhizobia (de Faria et al. 1984, 1987, 1999, de Faria 1995, Sprent and Parsons 2000). These species are among all three Leguminosae sub-families, which have together ~8000 woody species (out of >19,000 species in Leguminosae) (Lewis et al. 2005). The largest sub-family Papilionoideae (~4000 woody species) also concentrates most of the nodulating species, followed by the Mimosoideae, but only a small proportion of the legumes of the Caesalpinioideae are able to nodulate (de Faria et al. 1989, Sprent and Parsons 2000, Sprent 2009). Further information on the distribution of nodulation and N₂ fixation in the Leguminosae can be found in Sprent and Parsons (2000) and Sprent (2009).

Research on legume tree nodulation started in the mid-1960s (Döbereiner 1967) with strain selection and field response of Mimosa caesalpinifolia, a native species from the Brazilian Caatinga (dry forest biome). This species is certainly one of the most frequently planted legume tree species in Brazil today, and has been widely used for live fencing, landscape stabilization and land reclamation. Study of biological nitrogen fixation (BNF) in tropical legume species in Brazil has intensified since the early 1980s, resulting in the identification of a great number of nodulating species, and of biotic and abiotic factors that limit nodulation and BNF (de Faria et al. 1984, 1987, 1989, Moreira et al. 1992, Franco and de Faria 1997). Most of the research on BNF in legume trees and their use to reclaim degraded lands in Brazil has been made by the team at Embrapa Agrobiologia (Brazilian Agricultural Research Corporation—Agrobiology Centre). The Embrapa Agrobiologia rhizobium collection has >5500 strains isolated from woody legume species that have potential use for revegetation of degraded land. To date, the laboratory has recommended efficient N₂-fixing bacteria for 87 legume species (de Faria et al. 2010).

The selection of rhizobium strains is of fundamental importance because some rhizobia have a strict specificity for leguminous plants. It is common to find bacteria that have a high capacity for BNF when associated with a given legume and are unable to nodulate or fix N₂ with other species (Franco and de Faria 1997, de Faria et al. 1999). The selection of rhizobium strains begins with identification in the field of legume species able to nodulate under natural conditions. Nodules from these plants are then collected and, in the laboratory, bacteria inside the nodules are isolated, purified and stored in culture collections. Following the isolation phase, several tests are performed to identify the most efficient rhizobium strains with respect to
Production of legume seedlings for land reclamation

The successful results that several studies have obtained for the recovery of degraded areas (RDA) in Brazil are mainly due to the selection of fast-growing legume trees (FGLTs) and their specific mycorrhizal and rhizobial symbiotic partners (de Faria et al. 2010). A great deal of experience has been obtained with many legume trees for different climates and for soils with different restrictions (salinity, acidity, etc.). These species, classified by climatic region and soil restriction, can be found in the Technical Bulletin of de Faria et al. (2010).

Production of legume tree seedlings consists of several steps, which start with the harvest of seeds from selected mother plants. Selection of mother plants is important to ensure that seeds originate from healthy plants containing superior phenotypic characteristics, and maximum genetic variability, so they should be collected from a range of individual plants from a given region and not from a single plant.

In general, the inoculum of AMF used during seedling production designated to RDA sites consists of a mixture of Glomus clarum and Gigaspora margarita, which are produced using the pot culture technique in pasteurized soil (Brundrett and Juniper 1995), where a host plant (generally Brachiaria spp.) is used to multiply the fungal propagules (spores and vegetative structures). These two AMF species can efficiently colonize a large number of plant species, and play an important role in the RDA as they have been considered to be able to colonize most plants, which facilitates the succession of vegetation in areas being recuperated (Rocha et al. 2006). In spite of this generalization, recent studies have shown that seedlings of tree species from the Acacia genus (Acacia mangium, Acacia holosericea and Acacia auriculiformis) presented better development when inoculated with the species Acaulospora morrowiae, Scutellospora calospora, Scutellospora gilmorei and Scutellospora heterogama instead of the commonly used G. clarum and G. margarita (Angelini 2008). The AMF inoculant (~1 g) is inserted into the hole made for the seed immediately prior to planting. Inoculation with rhizobium is made at planting by treating the seeds with a standard rhizobium inoculant using sterile peat as a base (Somasegaran and Hoben 1994). Further details about inoculation and production of seedlings suitable for RDA can be found in Resende et al. (2010).

Case studies

Revegetation of erosion gullies

A large proportion of the deforested lands of the Brazilian Atlantic Forest region are mountainous, especially in the southeastern states of Rio de Janeiro, Minas Gerais and Espirito Santo. The hillsides were used mainly for coffee production during the 19th century, and after the decline in this activity these areas were abandoned or used as pasture for extensive cattle production (Boddey et al. 2006). Recolonization of these hillsides by woody species was prevented principally by the presence of grazing animals and frequent burning of the vegetation, which is normally induced by the landowners as a means of stimulating new growth of grasses for forage. As mentioned in the Introduction, these forms of land occupation, coupled with heavy rainfall events during the summer, have resulted in the formation of innumerable gullies.

Although gullies generally occur in poorly structured dispersive soils, such as Cambisols (Ferreira et al. 2007), in southeastern Brazil they also proliferate in well-structured and free-drained Oxisols (Machado et al. 2010). To give an example of the extension of this problem, along a 70-km stretch of the valley of the river Paraiba do Sul in the interior of the State of Rio de Janeiro, 160 erosion gullies were counted of up to 150 m extension and up to 8000 m² area (Machado et al. 2006). According to a study by local watershed committees, this form of erosion is the principal cause of the accelerated silting up of the Paraiba do Sul river, which is the main source of water to nine million people living in the metropolitan area of the city of Rio de Janeiro.

In this case study we report the use of FGLTs to recover a gulley in a rural site in Pinheiral, south of the State of Rio de Janeiro (Figure 1). The gulley had an area of ~1000 m², 10 m depth and a volume of approximately ~10,000 m³ (equivalent to 2000 truckloads of sediment). The intervention was started in 2000 with the construction of terraces at the upper and lower ends of the gulley, and walls of bamboo and tires were positioned in the inner part to trap sediments. Seedlings of several legume trees, inoculated with selected rhizobia and AMF, were planted along the gulley into holes cut into the walls with 2 m × 2 m spacing. The success of the intervention was measured by the growth of the trees and by the amount of sediments collected in sediment tanks.

The species A. mangium, Mimosa artemisiana, M. caesalpinifolia and Pseudosamanea guachapele showed the best survival and development after 170 days. The species A. auriculiformes, Acacia angustissima, Albizia lebbek, Enterolobium contortisiliquum and Samanea saman showed low indices of survival, sometimes because of their lower resistance to drought, or their position in the gulley where water was not retained, or because they suffered from attack by leaf-cutting ants.

The evaluation of the run-off of sediments was performed 6 years after the intervention during a 3-month period over a summer rainy season (December 2005 to March 2006) (details in Machado et al. 2010). Sediments were collected in tanks constructed at the lower end of the gulley. Similar tanks were constructed in two adjacent gullies of similar size: one
that was left without a recovery operation, and one where the recovery operation had started 2 years before. The amount of sediments collected from the non-reclaimed gulley was 195 Mg, and for the gullies reclaimed in 2004 and 2000 totaled 4.5 and 2.7 Mg, respectively, over the evaluation period. Based on the nutrient content in the sediments, it was estimated that just in the non-reclaimed gulley the losses of K and Mg were 944 and 823 kg, respectively.

With respect to the cost of this operation, ~US$20,000 was spent to recover one of the gullies (~US$20.00/m²). The largest proportion of the cost (64%) was for labor followed by the cost of the 4000 seedlings (20%—US$ 1.00 per seedling) and transport; the other costs were for materials such as fencing posts and wire, rock phosphate and fritted trace elements, manure and insecticide.

Reclamation of areas degraded by mining activities

Opencast mining to extract bauxite, iron, cassiterite, manganese and kaolin has caused the destruction of 2000–3000 ha per year of tropical forest in Brazil since the 1960s (Parrota and Knowles 1999). The deforested area is even greater if the large number of areas of sand and clay mining for use in the civil construction and ceramic industries are considered. While these mining activities cause devastation of small areas compared with forest clearance for agriculture or unsustainable logging, the local environmental impact is much greater as the ecosystem suffers drastic alterations. Loss of biodiversity, soil erosion, dust emissions, and siltation and contamination of rivers and other water bodies are among the impacts caused by mining activities.

The general procedure used in opencast mining is the removal of both vegetation and the upper horizons to reach the raw material needed by the industry. After exploration, the mining remains, or ‘overburden’, are used to fill in the pits and to reconstitute the topography of the area. Following this, the area must be revegetated as close to the original botanical composition as possible, as required by Brazilian regulations.

Next, we describe two study cases of mined areas located in different Brazilian biomes, which were successfully revegetated using FGLT species.

Revegetation of iron mining overburden

The ‘overburden’ from iron mining is defined as part of the surface soil and subsoil from the vegetation down to the haematite level. Haematite, the economically exploitable layer, is often mined at depths of 6–10 m, but sometimes even down to 20 m. The overburden material is generally composed of a mixture of laterite and filite (iron and aluminum oxides). This material is removed to give access to the iron ore and is deposited in embankments.

FGLT technology has been applied since 1992 at a haematite mining site in the Mariana–Ouro Preto districts of Minas Gerais State (southeast Brazil) (‘B’ in Figure 1), where ~25 ha per year are revegetated. The original practice of the mining company in this case was to plant the embankments with a mixture of laterite and filite (iron and aluminum oxides). This material is removed to give access to the iron ore and is deposited in embankments.

Fertility analyses of the material at this site revealed that it had extremely low phosphorus content, no detectable organic matter, traces of potassium and pH ranging from 4.0 to 5.5. In the first tests to evaluate native N₂-fixing legume species, only a narrow range of native species were available due to lack of available seeds. So, non-native species were also tested. Fifteen species were tested in this area (Figure 2), including N₂-fixing and non-fixing legume species, and non-legume species specially selected for their potential to attract fauna. The seedlings of all 15 species were prepared in a nursery at the mining site. Seeds of the N₂-fixing species were inoculated both with specific rhizobium strains and with mycorrhizal fungi (G. clarum and G. margarita).

The seedlings were planted in a triangular array with a distance of 1.5 m between them. The strategy was to mix N₂-fixing
species with species attractive to fauna (e.g., birds, bats and rodents) that would ultimately bring in native species not introduced initially. Each planting hole in the iron mining waste had received 100 g lime, 200 g rock phosphate, 10 g fritted trace elements (containing Mo, Zn and B) and 1 l of cattle manure. Control of leaf-cutting ants was performed 1 month before the plantation using sulfluramid granulated baits.

Tree heights were measured 7, 14 and 24 months after planting (Figure 2). *Acacia angustissima*, *M. artemisiana* and *E. contortisiliquum* (all N₂-fixing legumes) reached >2 m height after 2 years. Some species were not adapted to these conditions, and disappeared: *Parkinsonia aculeata* (non-N₂-fixing legume), mulberry (*Morus* sp.—non-legume species) and *Mimosa bimucronata* (a N₂-fixing legume). After this time, the canopy of plants completely covered the embankments of the iron mining waste. Colonization by non-planted species was observed as a result of seed dispersal by wind, birds and bats. After only 2 years these species accounted for ~15% of the total number of species originally planted.

Nowadays, the planting of such non-native species as *A. angustissima* and *A. mangium* is no longer allowed by the Brazilian Institute of the Environment and Renewable Natural Resources in forest reserves (areas of permanent conservation).

**Revegetation of areas degraded by ‘piçarra’ extraction in a semi-arid region**

The Brazilian on-shore production of petroleum is concentrated in the semi-arid areas of northeastern Brazil (Caatinga biome). One of the main environmental impacts of this activity is caused by the extraction of *piçarra*, a subsoil material mainly composed of silt, sand and gravels used in land leveling for the petroleum pump jack stations. The mining process consists of removing the native vegetation and extracting *piçarra* to a depth of 2–10 m. At the end of the mining activity, the topography of the excavated areas is smoothed and the embankments must be revegetated.

In contrast to the humid forest areas of north and southeast Brazil, these semi-arid areas present a long dry season that may extend up to 8 months (annual rainfall amounts to 200–600 mm, and almost all of it is concentrated from February to May), which poses an extra difficulty for recomposing the vegetation. In addition, there is scarce information about species suitable for revegetation of these degraded mining areas, despite a very recent study that evaluated the potential for BNF of a range of legume trees from the Brazilian semi-arid Caatinga (Freitas et al. 2010).

In 2008 Resende et al. (unpublished data) installed trials in five *piçarra* mines to evaluate the development of 10 N₂-fixing legume tree species (N₂-fixing group) and of 10 other non-nodulating legume and non-legume tree species (non-fixing group) planted in areas with and without topsoil replacement (Table 1). The mining areas are located in the State of Rio Grande do Norte, northeastern Brazil (Figure 1), and had been explored for several years by the national petroleum company Petrobras for *piçarra* extraction. Prior to planting operations, half of the experimental plot in each mining site received a layer of 20 cm of topsoil, originating from newly opened mining areas, and the other half was left with the original *piçarra* subsoil. Seedlings were planted in March 2008 using 2 m x 2 m spacing and adding 40 g of soluble phosphate and 10 g of fritted micronutrients to the planting hole. All seedlings had been inoculated with AMF, and the nodulating legume species with selected rhizobia strains.
Evaluation of plant development after 22 months showed a marked difference between the growth of species from the N$_2$-fixing and non-fixing groups. Accordingly, the mean height for the N$_2$-fixing group was close to 2 m, while the non-fixing group did not reach half of this measurement (Table 1). In fact, several species of the non-fixing group barely presented any growth during the 22 months. In addition, N$_2$-fixing species presented survival rates close to or equal to 100% for most species. Among the N$_2$-fixing group, *Mimosa tenuiflora*, *M. caesalpinifolia* and *P. guachapele* (a non-native species) presented the best rates of survival and development (Table 1). Other N$_2$-fixing species that presented high rates of survival and good development were the native *Caesalpinia ferrea* and *Acacia farnesiana*, and the non-native *Gliricidia sepium*. Overall, application of topsoil did not cause significant changes and precipitation for nutrient and water resources caused by the profuse herbaceous vegetation that grew in areas that received the topsoil.

### Table 1. Height and percent of survival of N$_2$-fixing legume trees, and of non-legume and non-nodulating legume trees 22 months after planting in areas degraded by *piçarra* extraction. Planting was performed either on bare subsoil or on a 20-cm layer of topsoil added to the area. Values of height correspond to the mean of survivor plants out of 50 initially planted in five trial plots.

<table>
<thead>
<tr>
<th>Species</th>
<th>Height (cm)</th>
<th>Survival (%)</th>
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<tbody>
<tr>
<td></td>
<td>Topsoil</td>
<td>Subsoil</td>
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<tr>
<td><strong>N$_2$-fixing legume tree species</strong></td>
<td></td>
<td></td>
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<tr>
<td><em>Erythrina velutina</em></td>
<td>130 ± 18</td>
<td>138 ± 12</td>
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<tr>
<td><em>G. sepium</em>¹</td>
<td>181 ± 16</td>
<td>149 ± 17</td>
</tr>
<tr>
<td><em>M. tenuiflora</em></td>
<td>285 ± 16</td>
<td>313 ± 13</td>
</tr>
<tr>
<td><em>P. guachapele</em>¹</td>
<td>262 ± 19</td>
<td>261 ± 21</td>
</tr>
<tr>
<td><em>A. angustissima</em>²</td>
<td>228 ± 20</td>
<td>178 ± 22</td>
</tr>
<tr>
<td><em>M. caesalpinifolia</em></td>
<td>217 ± 15</td>
<td>214 ± 12</td>
</tr>
<tr>
<td><em>Enterolobium timbouva</em></td>
<td>135 ± 10</td>
<td>142 ± 14</td>
</tr>
<tr>
<td><em>C. ferrea</em></td>
<td>172 ± 15</td>
<td>178 ± 24</td>
</tr>
<tr>
<td><em>Calliandra seloi</em>²</td>
<td>156 ± 20</td>
<td>121 ± 10</td>
</tr>
<tr>
<td><em>A. farnesiana</em></td>
<td>172 ± 12</td>
<td>207 ± 15</td>
</tr>
<tr>
<td><strong>Group mean</strong></td>
<td>194 ± 190</td>
<td>79 ± 89</td>
</tr>
<tr>
<td><strong>Non-legume and non-nodulating legume tree species</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Azadirachta indica</em>³</td>
<td>115 ± 16</td>
<td>151 ± 21</td>
</tr>
<tr>
<td><em>Myracrodruon urundeuva</em></td>
<td>64 ± 7</td>
<td>78 ± 13</td>
</tr>
<tr>
<td><em>P. aculate</em></td>
<td>124 ± 9</td>
<td>135 ± 16</td>
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<tr>
<td><em>Tabebuia caraiba</em></td>
<td>69 ± 4</td>
<td>92 ± 13</td>
</tr>
<tr>
<td><em>Adenanthera pavonina</em>³</td>
<td>82 ± 15</td>
<td>97 ± 24</td>
</tr>
<tr>
<td><em>Aspidosperma pyrifolium</em></td>
<td>29 ± 2</td>
<td>29 ± 3</td>
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<tr>
<td><em>Tabeuia impetiginosa</em></td>
<td>73 ± 5</td>
<td>76 ± 10</td>
</tr>
<tr>
<td><em>Schinus terebinthifolius</em></td>
<td>70 ± 9</td>
<td>66 ± 8</td>
</tr>
<tr>
<td><em>Amburanae arense</em></td>
<td>39 ± 2</td>
<td>54 ± 5</td>
</tr>
<tr>
<td><em>Caesalpinia bracteosa</em></td>
<td>54 ± 7</td>
<td>58 ± 12</td>
</tr>
<tr>
<td><strong>Group mean</strong></td>
<td>72</td>
<td>84</td>
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¹Species non-native to the Brazilian Caatinga biome.

**Carbon accumulation in soils reclaimed with legume trees**

This study aimed to evaluate the recovery of nutrient cycling processes and of soil C and N stocks after 13 years of soil rehabilitation using leguminous N$_2$-fixing trees. The area is located in the town of Angra dos Reis, along the western coast of the State of Rio de Janeiro, within the limits of the Atlantic Forest biome (Chada et al. 2004) (Figure 1). Mean annual precipitation is 2300 mm. The area has a slope varying from 45 to 60%. The soil is classified as a Ferralsol (Red Yellow Argisol, according to the Brazilian Soil Classification System 2006).

In 1991, when the area was dominated by grass vegetation, the topsoil was removed from the site and used for the foundations of a shopping mall. Exposure of the soil to rainfall led to severe erosion, which, after a short period of time, resulted in the formation of erosion gullies. The area was restored in 1993 by planting seedlings (at a spacing of 2 m × 3 m) of *A. mangium*, *A. auriculiformis*, *M. tenuiflora*, *E. contortisiliquum*, *G. sepium*, *Leucaena leucocephala*, *M. caesalpinifolia* and *Falcata molucanna*, all of which were inoculated with rhizobia and AMF. To avoid further erosion, when the seedlings were planted, bamboo stems were anchored crosswise in the erosion channels or gullies to slow down the rainwater running down the slope. The recovered area was ~1 ha in size.

In September 2004, soil samples were collected from two reference areas and the area rehabilitated with legume trees. One of these reference areas was a fragment of native forest (Atlantic Forest), with few signs of human presence or disturbance, while the other consisted of 2 ha of deforested land. These three areas are located in close proximity to each other on the same hillside. The deforested area (where no intervention was performed) was spontaneously overgrown with Guinea grass (*Panicum maximum*). The topsoil of this area was not completely removed as in the case of the rehabilitated area. Soil samples were collected from the following depth intervals: 0–5, 5–10, 10–20, 20–30, 30–40 and 40–60 cm. The soil C and N stocks were calculated from the C and N concentrations measured at each depth interval multiplied by the respective bulk density and the thickness values of the corresponding soil layer. To avoid overestimates of C and N in compacted soils, stocks were corrected for differences in soil mass to 60 cm depth using the procedure of Veldkamp (1994). The amount of standing litter on the soil surface was also determined in samples from the recovered and native forest areas collected during the dry (September 2004) and rainy (March 2005) Seasons.

The C and N concentrations in the soil of the recuperated area were higher than in the soil of the deforested area, and similar to C values of the native forest soil (Macedo et al. 2008). The planting of FGLTs promoted an increase in the stock of soil C at 0–30 cm depth from 35.5 to 54.8 Mg ha$^{-1}$ (Table 2), almost the same as the C stock under the native forest (58.3 Mg ha$^{-1}$). However, when C stocks were evaluated
to a depth of 60 cm, the results indicated that the FGLTs promoted an increase in soil C from 65 to 88 Mg C ha\(^{-1}\) but this was still somewhat lower than the C stocks under the native vegetation. Assuming that the soil C stock of the deforested area is equivalent to the C stock of the recovered area prior to planting the legume trees, it may be concluded that the soil C stock increased by 23 Mg ha\(^{-1}\) in 13 years or a mean of >1.7 Mg C ha\(^{-1}\) year\(^{-1}\).

The litter stocks of the recovered and native forest areas were statistically similar for rainy and dry seasons and ranged from 5.0 to 6.7 Mg ha\(^{-1}\) (Table 3), indicating that the net aerial primary productivity of the area planted with legume trees was at least equal to that of the native forest. These results are similar to those reported by Vital et al. (2004) for a steady-state forest (6.2 Mg ha\(^{-1}\)), although somewhat lower than the values observed by Arato et al. (2003) in a 9- to 10-year-old agroforestry system established on degraded land (8.7 Mg ha\(^{-1}\)).

Soil C and N were restored in a short period of time after planting of legume trees in symbiotic association with N\(_2\)-fixing bacteria and AMF. Other studies have shown an increase in soil C and N during forest development (Brown and Lugo 1990, Gleason and Tilman 1990, Feldpausch et al. 2004) but not in a situation where the soil had been decapitated.

### Land reclamation and the process of plant succession

The primary objective of reclaiming severely degraded areas is to promote fast plant colonization of the area in order to protect the soil against erosion, and to input new biomass/carbon to the system. The planting of FGLTs inoculated with selected rhizobium strains and AMF is a strategy that has proved to be very efficient in achieving these objectives. These species can add large quantities of organic matter and N to the soil through litterfall in a relatively short time, improving nutrient cycling and the rehabilitation process. For instance, Costa et al. (2004) showed that litterfall of *M. caesalpinifolia*, *A. auriculiformis* and *G. sepium* after 10 years of planting in a decapitated soil in Seropédica, State of Rio de Janeiro, varied from 5.7 to 11.2 Mg ha\(^{-1}\) year\(^{-1}\) dry matter. These values were not very different from those obtained in a nearby 20-year-old secondary forest (9.2 Mg ha\(^{-1}\) year\(^{-1}\)). The annual nutrient input in kg ha\(^{-1}\) year\(^{-1}\) was in the range 130–170 for N, 4.9–7.9 for P, 24–31 for K, 150–190 for Ca and 29–40 for Mg in the reforested areas, values similar or superior to those observed in the secondary forest site.

Increasing SOM is very important in degraded land rehabilitation projects, since, according to Francis and Read (1994), it enhances the capacity of the system to support a more complex community. Macedo et al. (2008) also showed that the N increase derived from BNF was directly related to C incorporation, as indicated by the strong correlation of soil C and N in all areas in this study (r = 0.78, P < 0.0001, n = 50). Owing to their ability to fix nitrogen, legume species have been used as

### Table 2. C and N stocks (0–60 cm) in the whole soil profile of recovered, native forest and deforested areas in Angra dos Reis, RJ (n = 3). After Macedo et al. (2008).

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<thead>
<tr>
<th>Depth (cm)</th>
<th>Recovered</th>
<th>Native forest</th>
<th>Deforested</th>
</tr>
</thead>
<tbody>
<tr>
<td>C stock (Mg ha(^{-1}))</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0–5ns</td>
<td>10.9 ± 0.2</td>
<td>12.0 ± 1.5</td>
<td>7.1 ± 0.3</td>
</tr>
<tr>
<td>5–10ns</td>
<td>10.6 ± 1.0</td>
<td>10.9 ± 1.0</td>
<td>7.2 ± 0.1</td>
</tr>
<tr>
<td>10–20ns</td>
<td>19.7 ± 1.2</td>
<td>21.7 ± 2.5</td>
<td>13.3 ± 0.5</td>
</tr>
<tr>
<td>20–30ns</td>
<td>15.9 ± 2.4</td>
<td>16.8 ± 4.3</td>
<td>11.0 ± 2.4</td>
</tr>
<tr>
<td>30–40ns</td>
<td>14.4 ± 1.9</td>
<td>17.9 ± 4.8</td>
<td>10.1 ± 3.0</td>
</tr>
<tr>
<td>40–60ns</td>
<td>21.5 ± 2.7</td>
<td>34.1 ± 14.4</td>
<td>19.8 ± 6.7</td>
</tr>
<tr>
<td>0–30ns</td>
<td>54.8 ± 2.1</td>
<td>58.3 ± 7.7</td>
<td>35.4 ± 1.7</td>
</tr>
<tr>
<td>0–60</td>
<td>88.1ab ± 0.4</td>
<td>107.7a ± 22.1</td>
<td>65.1b ± 11.2</td>
</tr>
</tbody>
</table>

N stock (Mg ha\(^{-1}\))

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>Recovered</th>
<th>Native forest</th>
<th>Deforested</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–5ns</td>
<td>0.94 ± 0.08</td>
<td>1.17 ± 0.07</td>
<td>0.59 ± 0.10</td>
</tr>
<tr>
<td>5–10ns</td>
<td>0.92 ± 0.09</td>
<td>1.00 ± 0.06</td>
<td>0.52 ± 0.04</td>
</tr>
<tr>
<td>10–20</td>
<td>1.67a ± 0.15</td>
<td>1.70a ± 0.12</td>
<td>0.99b ± 0.07</td>
</tr>
<tr>
<td>20–30</td>
<td>1.41ab ± 0.14</td>
<td>1.57a ± 0.18</td>
<td>0.94b ± 0.08</td>
</tr>
<tr>
<td>30–40ns</td>
<td>0.98 ± 0.10</td>
<td>1.27 ± 0.21</td>
<td>0.97 ± 0.02</td>
</tr>
<tr>
<td>40–60</td>
<td>1.75b ± 0.12</td>
<td>2.54a ± 0.56</td>
<td>1.96ab ± 0.09</td>
</tr>
<tr>
<td>0–30</td>
<td>5.0a ± 0.4</td>
<td>5.4a ± 0.2</td>
<td>3.0b ± 0.1</td>
</tr>
<tr>
<td>0–60</td>
<td>7.7ab± 0.3</td>
<td>9.1a ± 0.9</td>
<td>6.0b ± 0.1</td>
</tr>
</tbody>
</table>

### Table 3. Dry matter (kg ha\(^{-1}\)) of different litter fractions on native forest, recovered and deforested areas in Angra dos Reis, RJ, Brazil, collected during the dry and rainy seasons (n = 5). After Macedo et al. (2008).

<table>
<thead>
<tr>
<th>Area</th>
<th>Leaves</th>
<th>Stem</th>
<th>Decomposed</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Dry</td>
<td>Rainy</td>
<td>Dry</td>
<td>Rainy</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Native forest</td>
<td>1278Aa</td>
<td>855Aa</td>
<td>1419Aa</td>
<td>1088Aa</td>
</tr>
<tr>
<td>Recovered area</td>
<td>1725Aa</td>
<td>543Ab</td>
<td>1463Aa</td>
<td>1935Aa</td>
</tr>
<tr>
<td>CV (%)</td>
<td>45</td>
<td>12(^{1})</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Different capital letters indicate significant differences between areas in the same season according to the Bonferroni t-test (P < 0.05), while different lowercase letters indicate significant differences between seasons of the same area according to the same test.

\(^{1}\)Data transformed by Ln(x).
an N source in several tropical agroecosystems, including pastures (Fisher et al. 1994, Tarré et al. 2001), no-till fields (Sisti et al. 2004, Boddey et al. 2010), tree plantations (Resh et al. 2002, Balieiro et al. 2008) and agroforestry (Handayanto et al. 1995). In these diverse systems, soil N content and SOM stocks were found to increase. Organic matter is very important in tropical soils since it plays a crucial role in the formation and maintenance of soil structure, fertility, and nutrient and water availability (Bayer et al. 2001, Craswell and Lefroy 2001, Six et al. 2002). It seems that in pasture, forest or arable systems under no-till, where soil is not regularly disturbed by ploughing, etc., N$_2$-fixing legumes can play a very important role in increasing soil carbon (i.e., sequestering atmospheric CO$_2$), especially in degraded areas where C stocks start at a very low level (Boddey et al. 2009).

A secondary but equally important objective in restoring such degraded areas is to stimulate the resilience of the ecosystem by the processes of natural succession. This includes the colonization of the area by native plant species, the restoration of soil health and the return of wildlife at all levels.

Resende et al. (2006) have proposed the thesis that FGLTs can stimulate the autogenic succession in degraded environments, a process where successional changes occur through inner interactions of the ecosystem that promote the flux of energy and nutrients altering the structure and stability of plant communities (Odum 1983). Accordingly, the fast establishment and accumulation of biomass of one or a few species, able to tolerate the harsh environmental conditions of the degraded area, will improve soil quality and provide microclimatic conditions allowing the establishment of other secondary and climax species. The accumulation of litter material by legume trees also promotes enrichment of the soil fauna and the activation of processes of nutrient cycling and SOM formation (Chada et al. 2004, Costa et al. 2004, Banning et al. 2008).

Nevertheless, the use of plants of a single botanical family to accomplish these objectives has been criticized (Kageyama et al. 1994, Reis et al. 1999, Rodrigues et al. 2009). The main allegation is that the original ecosystem must be taken as a model, and so diverse native plant species must be used in the revegetation in order to avoid inhibition of the natural succession. Similar criticism has been leveled at the introduction of non-native species in the revegetation of degraded areas (Reis et al. 1999). Although these arguments sound reasonable, studies of land reclamation developed along the last 25 years have shown that the activation of the processes of natural succession by the introduction of either native or non-native legume tree species in degraded areas is a reality, given the existence of propagule sources in the nearby areas. For instance, Chada et al. (2004) observed, after 7 years of planting N$_2$-fixing legume trees in a degraded hillside in Angra dos Reis, RJ, near a native forest, colonization of the understory of the planted trees by 50 species from 25 botanical families. It was also observed that some of the introduced plant species were in senescence and slowly being replaced by native ones. Nevertheless, it should be considered that when propagule sources are distant, the non-native species may perpetuate for several generations, which may require a new intervention for planting of secondary native species.

Another study conducted in an area degraded by the extraction of soil for construction of an airport in the State of Amazônia, northern Brazil, Campello (1998) observed higher biomass and richness of native plant species regenerating in the understory of plantings of native (Tachigali vulgaris) and non-native (A. mangium) N$_2$-fixing legume tree species.

Figure 3. Total number and Shannon diversity index of native species regenerated in the understory of native and non-native legume and non-legume tree plantations.
primary recolonization presented the highest richness with 11 species of 9 genera. Cecropiaceae (67). The three Leguminosae sub-families also contributed, with Caesalpinioideae (2), Tiliaceae (209), Annonaceae (104) and Leguminosae (Mimosoideae, 315; Papilionoideae, 107; Caesalpinioideae, 2). The authors noted that families with a greater number of individuals were better indicators of the vegetation structure of a sampling area of 0.5 ha showed higher plant regeneration under legume trees compared with areas planted with other non-legume native (Goupi a glabra) or non-native (Eucalyptus pellita and Eucalyptus citriodora) species (Figures 3 and 4). These observations were made 10 years after the intervention, and demonstrated that natural regeneration was benefited by planting N2-fixing legume tree species, independent of the origin of the species. According to the author, the higher plant regeneration under legume trees could be associated with a lower lignin:nitrogen ratio of these plants, which favored soil biota and nutrient cycling. This suggestion was supported by a significant negative correlation (−0.85; P < 0.05) between the dry biomass from the regenerated vegetation and the lignin:nitrogen ratio.

Empirical observations have shown that natural colonization of a severely degraded soil in an Atlantic Forest region in the State of Minas Gerais was predominantly dominated by legume species (Araújo et al. 2006). According to this study, a hilly area of pasture was impacted in 1985 by deposition of overburden derived from kaolin mining. The area, which had nearby forest fragments, was isolated for 20 years, when natural colonization formed a secondary forest. Evaluation of the vegetation structure of a sampling area of 0.5 ha showed that families with a greater number of individuals were Leguminosae (Mimosoideae, 315; Papilionoideae, 107; Caesalpinioideae, 2), Tiliaceae (209), Annonaceae (104) and Cecropiaceae (67). The three Leguminosae sub-families also presented the highest richness with 11 species of 9 genera. Gehring et al. (2005) suggested that primary recolonization by legume species is a natural ecosystem strategy for recuperation of forest areas after slash-and-burn farming in the central Amazon. In this work, they showed that BNF was high throughout the first 25 years of secondary regrowth, as opposed to low BNF in the mature rainforest. This evidence was based on the high vegetation share of legume species capable of BNF in secondary regrowth as opposed to lower shares in mature rainforest.

Final considerations
The use of N2-fixing legume tree species for reclamation of severely degraded lands is a technique that can be applied in several situations. In all but the most arid regions of Brazil, the revegetation of subsoils that have been exposed by mining activities, land leveling/moving for roads and construction sites or erosion, will result in the areas being totally covered by trees in <24 months. However, its success will be conditioned by the correct choice of species and by the presence of propagule sources, pollinators and seed dispersers necessary to the advance of the natural succession of vegetation. The objective of the technical intervention is to accelerate these natural processes of ecosystem recovery.

The results obtained during the last 25 years of research of using native and non-native N2-fixing legume trees for land reclamation in Brazil indicate that the patterns of natural succession are dependent on the ecological function that each species has in the system, instead of the botanical identity or origin of the species. For reclaiming degraded areas, thus, there are no ready formulae, and in each situation the highest possible number of native and non-native species must be tested under the harsh initial conditions for plant establishment. It is necessary, however, to understand the behavior of the non-native species considered for land reclamation regarding their successional stage in their natural environment. This is important to avoid the invasion of other natural areas by these species, or them becoming dominant in the regenerated ecosystem precluding the establishment of other native species.

The environmental services provided by the recovery programs of degraded areas are diverse and highly valuable. The recovery of plant diversity is of both direct and indirect value, in that it necessarily provides habitats for fauna, whether endangered species or not. The elimination or mitigation of the negative effects of erosion, such as the reduction of the sediment load reaching rivers and reservoirs and the increase in infiltration, and reduced flooding potential are also highly beneficial consequences. Results also show that trees themselves not only act as a sink for carbon dioxide, but also promote the accumulation of N and C in the soil as SOM.

References


In Simpósio Sul-Americano de Recuperação de Áreas Degradadas. SOBRADE, Foz do Iguaçu, PA, pp. 569–576.


