ABSTRACT

HARJUNIEMI, ALIISA VILHELMIINA. Effects of Intensive Management in the Restoration of Northeastern Atlantic Forest in Brazil; Survival, Growth and Carbon Sequestered 8 Years after Planting. (Under the direction of Jose Luiz Stape, Bronson Bullock and Gary Blank.)

The Atlantic forest on the Eastern coast of Brazil is one of the world's most endangered biotopes. Less than 12 % of the original forest remains due to agricultural and pasture expansion. In addition, many Atlantic forest restoration projects in the past have failed, largely because of inadequate silvicultural practices. Meanwhile, the growth rates of Eucalyptus and pine plantations have been increased 3 to 4 fold in Brazil over the last four decades by utilizing intensive silvicultural methods such as site preparation, fertilization, and weed control which in turn increases resource supply (nutrient, water and light). This study determines the effects of these same intensive silvicultural methods on Atlantic forest restoration regarding initial growth and carbon sequestering.

Two parallel research sites were established in 2004 on latitudes 11°S and 23°S on the Eastern coast of Brazil to determine the effects of intensive silviculture, planting density and species composition on the development of 20 native tree species. This research focused on the Northern site (200 km North of Salvador, Bahia State) which has a typical tropical climate and soil type. The project has a 23 factorial design totalling 8 treatments, with the following factors: i) intensive and traditional treatments; ii) initial planting densities (3333 trees ha-1 and 1667 trees ha-1); and iii) species composition proportion (50:50 and 67:33 ratio of pioneer vs. late successional species). After 8 years from planting, survival and development of each species, aboveground biomass and leaf area index (LAI) were determined for all the treatments to compare the effects of the different factors.

In summary, the main findings of this study are: 1) The more intensive management methods improved survival and the initial growth of tree species 2) Lower stand density (1667 trees ha-1) had the best response to the intensive management for LAI, stemwood production, and above ground carbon sequestration 3) Out of 20 species, 19 had significantly higher growth with intensive management, indicating that both pioneer and late successional species are constrained by the original site conditions. 4) Intensive management was essential, especially for non-pioneer species. 5) Under low intensity silviculture, the 67:33 ratio pioneers vs. non-pioneers with higher planting density (3333 trees ha-1) was the best option to obtain the highest stemwood volumes 8 years from planting, while the 50:50 ratio pioneers vs. non-pioneers with lower planting density (1667 tree ha-1) could be recommended under intensive silviculture.

Conclusion: Intensive management methods have the potential to increase early restoration success by increasing biodiversity through enhancing survival and growth of non-pioneer species and accelerating the canopy closure. Intensive management methods increased the above ground carbon sequestered in 8 years, remarkably, up to 3-fold compared to traditional management, making it an attractive management option for carbon offsets.

Effects of Intensive Management in the Restoration of Northeastern Atlantic Forest in Brazil; Survival, Growth and Carbon Sequestered 8 Years after Planting

by
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DEDICATION

I dedicate this work to my husband, Josh Kotheimer and my parents, Eeva and Pekka Harjuniemi. Without their unlimited support and patience, this work would not have been possible.

BIOGRAPHY

Aliisa Harjuniemi was born January 7, 1988 in Kemi, Northern Finland. She has three younger siblings, Ossi, Oona and Roope. While growing up, she enjoyed outdoor sports such as orienteering and skiing. She has always been interested in nature, especially forests, and after high school she decided to study Forest Ecology at the University of Helsinki. She graduated with her Bachelor’s degree in Forest sciences in 2010. While studying in Helsinki, she worked as a research assistant for the Metapopulation research group doing field work in Åland, where she mapped the Glanville Fritillary populations. In Åland she developed strong interest in environmental research. Because of her interest in international issues, she decided to apply to the Atlantis program (Transatlantic Master's Degree in Forest Resources). Within this program, she studied one semester in Helsinki, one semester in Alnarp, Sweden (SLU) and one year at NCSU. She met her husband, Joshua Kotheimer, in a wetland delineation class at NCSU. After studying one year at NCSU, in the summer of 2012, she traveled to Brazil to collect her Master’s thesis data and, also, to learn about Brazilian culture. In the future, Aliisa is hoping to work with forest biomass estimation, wetland delineation and eventually pursue her PhD.

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1. Introduction

1.1 Atlantic forest and its restoration

Atlantic forests along the Eastern coast of Brazil are important for ecosystem functions and are valuable biodiversity hotspots. The species in the Atlantic forest biome represent 2.7% of the planet's total number of species and a high proportion of them are endemic (Myers et al. 2000). Furthermore, tropical forest restoration has potential to mitigate climate change through carbon sequestration, which reduces atmospheric CO_2, the most significant greenhouse gas (Bonan 2008). In addition, Atlantic forests provide valuable ecosystem services like water purification and food production (Viana 1996).

However, despite its large diversity and valuable ecosystem services, only 11.7 % of Atlantic forest's original cover remains and it is mostly fragmented in small patches that are less than 50 ha (Ribeiro et al. 2009). This is mainly because of agricultural/pasture expansion and urban sprawl (Geist and Lambin 2002, Brooks et al. 2002). Restoration methods that can accelerate the reforestation process are needed in order to effectively increase the forest cover along the eastern coast of Brazil. Degraded areas on the eastern coast of Brazil that no longer fit for agriculture are frequently used for low productive pasture (Lira et al. 2012).

Restoration projects take place in areas where the ecosystem has lost its ability to recover from disturbance due to the lack of seed bank and intense environmental stress (Cook et al. 2014). The goal of forest restoration is to have the forest reach a state where it sustains itself (Fonseca et al. 2009). This self-sustainment can happen with the successful establishment of a large enough proportion of long-lived (non-pioneer) species.

Restoration of the Atlantic forest biome has a long history, but many of the projects have failed to create a self-perpetuating forest due to the lack of long-living species and the invasion of tropical grasses after pioneers species mortality (Rodrigues et al. 2009). Atlantic forest restoration projects in the past have failed largely because of the lack of adequate silvicultural practices that reduce the environmental constraints to survival and initial growth (Campoe et al. 2014). Exotic tropical grasses, like *Brachiaria* spp, not only compete for the natural resources, but also, bring recurrent fire events as a negative feedback for natural regeneration (Ribeiro et al. 2009). Another reason for failures to restore Atlantic forest was planting mainly fast growing pioneer species that soon died (Kageyama et al. 2000, Wuethrich 2007). Fast growing pioneers and grasses can suppress and eliminate the slower growing non-pioneers and early mortality of the pioneer species can cause a problem if the understory is not developed enough (Parrotta and Knowles 2001, Parrotta and Knowles 2002, Luu 2012).

The growth rates of Eucalyptus and Pine plantations have been increased 3 to 4 fold in Brazil over the last four decades by utilizing intensive silvicultural methods such as site preparation, fertilization, and weed control, which in turn increases resource supply (Eldridge et al. 1994, Santana et al. 2000, Stape et al. 2001). Determining the effect of these intensive silvicultural methods on native species, both pioneers and non-pioneers, is important to establish protocol to increase forest restoration success. However, some research suggests that fertilizing native trees is inadequate and that more intensive management might have an overall negative effect on the restoration process (Carpenter et al. 2004, Sampaio et al. 2007).

Previous studies from South Brazil have shown that canopy closure can be accelerated and carbon accumulation can be significantly increased in the Atlantic forest restoration with intensive management methods that were originally developed for *Eucalyptus* plantations (Campoe et al. 2010, Ferez 2011). These intensive management methods aimed to eliminate stress caused by the weed competition and nutrient limitations and thereby enhance the initial survival and tree growth. Growth of woody compartments increased up to 250 % with intensive management methods in South Brazil during 6 years of study (Ferez 2011) showing their significant potential to accelerate the restoration process and improve its success. However, forest growth and responses to different management methods can vary greatly between regions due to distinct geology and soils, rainfall patterns, temperatures and tree species, among other factors (Silver et al. 2000, Toledo et al. 2011). Furthermore, most of the published data from Atlantic forest is from the South part of Brazil while this study focuses on a much less studied area in the North part of Brazil, which is one of the hottest hotspots regarding biodiversity (Myers et al. 2000).

1.2 Atlantic forest restoration project

Two parallel research sites were established in 2004 on latitudes 11°S and 23°S on the Eastern coast of Brazil to determine effects of intensive management, initial stand density and tree species composition on Atlantic Forest restoration development (Stape et al. 2006). For each site, 20 local species were chosen. So far, just the South subtropical site was analyzed regarding tree growth and carbon sequestration (Campoe et al. 2010, Ferez 2011). This thesis will focus on the effects of intensive management methods on survival, growth and carbon accumulation on the Northern site (200 km North of Salvador, Bahia State) which has a typical tropical climate.

This project has a 2^3 factorial design, with the following factors and levels: i) Management: intensive and usual managements; ii) Density: initial planting densities of 3333 trees ha^{-1} (3m x 1m spacing) and 1667 trees ha^{-1} (3m x 2m spacing); and iii) Composition: species composition proportions of 50:50 and 67:33 ratio of pioneer : non-pioneer species, leading to eight treatments total. Twenty local native tree species were carefully chosen and produced locally for this study. An additional control-baseline treatment was used to capture the natural regeneration of the site.

For the management factor, the intensively managed plots received heavier site preparation, more fertilization and weed control. For the density factor, the higher planting density was considered to determine if they could accelerate the canopy closure and facilitate establishment of long-lived non-pioneers (Campoe et al. 2010). For the species composition factor, the higher proportion of pioneer species is expected to accelerate the stand establishment and initial growth.

A biomass inventory was conducted and survival, growth and carbon stocks were estimated for each of the 8 treatments and each of the 20 tree species at 8 years after planting. Naturally regenerated tree species were also measured and identified. In addition, leaf area index was determined in order to better evaluate the restoration process and to get estimates of the current growth potential for different treatments.

First, these issues were addressed by reviewing some restoration methods and strategies used in Atlantic forest restoration and rehabilitation in the literature review section. The potential of the forest restoration projects in climate change mitigation by the carbon sequestering was briefly discussed. In the practice section, the site-specific questions from the dataset collected in July-September 2012 were discussed. Generalized information in a broader geographic area can be used more reliably in forming guidelines for reforestation projects as well as in determining the carbon sequestering potential.

Soil carbon content and changes in biomass allocation between below- and aboveground woody compartments were excluded from this study but they will be determined in near future (planned for year 12 of the study). Nevertheless, it is expected that the reforestation would enhance the below ground carbon accumulation compared to the areas where reforestation procedures were not applied and that with more intensive methods the trees would allocate more biomass to the aboveground compartments following similar patterns than the Southern site (Ferez 2011).

The main hypotheses of this research are:

1) The intensive management practices will increase the survival, LAI, stemwood biomass and aboveground carbon stocks for an Atlantic Forest restoration in Northeastern Brazil when compared with the usual practices due to the minimization of environmental stresses;

2) The denser stands will have higher stemwood biomass, mainly on the usual silvicultural practices, due to an early canopy closure and providing a better competing vegetation control;

3) The 67:33 pioneer : non-pioneer model will have a higher stemwood biomass due to the fast growth rates of the pioneers;

4) Both pioneers and non-pioneers species will positively respond to more intensive management due to the minimization of environmental stresses;

We also compared the tropical (this study) and subtropical Atlantic restorations (Campoe et al. 2010, Ferez 2011, Campoe et al. 2014) looking for general- and specific-patterns regarding their responses to the silvicultural systems.

2. Literature review of Atlantic forest restoration

Planting of native tree species on degraded tropical landscapes can facilitate the reforestation process by improving soil conditions and providing shade for the late successional species. In this chapter, the concepts and terms relevant to the Atlantic forest restoration project are presented. First, the distribution and characteristics of the Atlantic forest biome is described, together with some of the methods and models that are used, or could be used, in the restoration and rehabilitation projects. Focus will be on the active restoration methods that rely on planting mixtures of native tree species. The factors that influence tropical forests behavior as well as a review of the carbon sequestering potential of the reforestation projects are described. This chapter remarks the successes of the previous studies and methods used for tropical forest restoration.

2.1 *Distribution and characteristics*

The Atlantic forest (“Mata Atlântica” in Portuguese) was once one of the largest rainforests of the Americas, occupying 14% of Brazil landscape (Lira et al. 2012). Today it is also a home for over 110 million Brazilians living in the area. The largest Brazilian city, Sao Paulo, with 16 million people, has its total water supply originated inside Atlantic Forest landscapes. Originally, before the European colonization, the Atlantic forest covered approximately 150 million hectares (Ribeiro et al. 2009). It extends into tropical and subtropical regions along the Eastern coast of Brazil, its latitudinal range being nearly 29^{o}S (4^{o} to 32^{o}S). In addition to the long latitudinal range, the wide longitudinal range (35^{o} to 60^{o}W) produces differences in forest composition because of the decreased rainfall when moving away from the coasts and decrease in temperature as move South (Lira et al. 2012). Coastal areas on Atlantic forest regime receive large amounts of rain year-round (up to 4000 mm/year), while forests further from the coast receive only around 1000 mm/year (Câmara et al. 2003).

Due to these geographical characteristics and large altitudinal range (0-2900 m above sea level), the Atlantic forest is extremely heterogeneous and its species diversity is one of the world's biggest. Its flora and fauna may include 1–8% of the world's total species, many of which are still not identified scientifically (Myers et al. 2000, Silva et al. 2003). The amount of vascular plants in the Atlantic forest is over 20,000 and approximately 40 % of its species are endemic, meaning that they cannot be found anywhere else in the world (Myers et al. 2000, Silva et al. 2003). The Atlantic forest consists mostly of evergreen to semi-deciduous forests but it also includes some mangroves, deciduous forests, "restingas" (lowland forest on sandy soils near the coast), mixed *Araucaria* forests and swamps as well (Oliveira-Filho and Fontes 2000, Morellato and Haddad 2000, Ribeiro et al. 2011). The stable climatic conditions have especially made the forests in the Northeastern part one of the species richest spots in the Atlantic forest biome, and the forests in the state of Bahia are described as a hotspot in hotspots for biodiversity (Myers et al. 2000, Carnaval 2009).

However, the Atlantic forest is one of the world's most endangered ecosystems (Ranta et al. 1998, Myers et al. 2000) and is nowadays fragmented (Ribeiro et al. 2009). This estimated area is larger than the usual total given for the Atlantic Forest (7-10%), probably because of inclusion of small fragments and secondary forests (Câmara et al. 2003, Ribeiro et al. 2009). Nevertheless, agricultural expansion, fires, illegal logging and urban sprawl continue to lead to a rapid deforestation rate, which is estimated to be 0.5% per year (Brooks et al. 2002, Geist and Lambin 2002). Protected areas cover only 1.62% of the Atlantic forest area (Ribeiro et al. 2009), and restoration actions are crucial for its recovery

In addition to the loss in forest area, the biodiversity is also threatened. Because of proliferation of native and exotic pioneer species, tree flora of the Atlantic forest of North Brazil has experienced serious taxonomic homogenization (Lôbo et al. 2011, Tabarelli et al. 2012). That is a response to habitat loss and fragmentation, which have created favorable conditions for light demanding pioneer species and lianas at the expense of the climax species (Swaine and Whitmore 1988, Lôbo et al. 2011). In addition, the forest fragmentation associate with exotic tropical grasses, like *Brachiaria*, has created a negative feed-back related with fire that tends to degraded the forest borders (Martins and Engel 2007). Additionally, the average age of the Atlantic forest is predicted to decrease in the near future because of the delay in biological response combined with high rates of deforestation and fast forest regeneration (Metzger et al. 2009).

Remaining fragments still provide important ecosystem services that millions of people living in the area depend on. Ecosystem services provided by Atlantic forest vary greatly and include, but are not limited to: biodiversity preservation, watershed protection, soil seed bank conservation landscape connectivity, food production and cultural benefits (Viana 1996, Baider et al. 2001, Ribeiro et al. 2009). Protecting these remaining fragments and restoring degraded areas is an important part in climate change mitigation and, especially in the biodiversity conservation.

2.2 *Restoration in Brazil; Review of concepts and methods*

Forest restoration and rehabilitation in the Atlantic forest; concepts and background

Ecological restoration is based on the idea that the restored site should be self-sustaining and that the reconstructed ecosystem resembles the original regarding the species compositions and functions (Jackson et al. 1995). The two main types of restoration are active and passive restoration. Active restoration reintroduces natural processes with direct actions such as planting trees. Passive restoration aims to cease activities that disturb the natural recovery process (e.g. fire and cattle grazing). Other important terms are rehabilitation, afforestation and reforestation. In rehabilitation, the interest is in restoring the functions of the ecosystem, but not necessarily the original species composition. Afforestation means establishment of a forest or stand of trees in an area where there was no forest or that has been deforested for a long time (generally speaking 50 years). Reforestation means reestablishment of the forest cover, either naturally (seed rain, soil seed banks, coppice) or artificially (direct seeding or planting). Restoration and rehabilitation projects take place on areas where the ecosystem has lost its ability to recover from disturbance. Restoring complex and high diversity ecosystems and creating self-sustaining ecosystems has been described as an ultimate test for the ecological theory (Ewel 1987).

Large-scale restoration requires a well-defined plan to succeed. Active restoration projects promoting high diversity are needed, especially in the human-modified tropical landscapes, in order to improve ecosystem services and biodiversity conservation (Rodrigues et al. 2011). Overall, the amount of restoration projects are increasing worldwide and restoring Atlantic forest is an issue that has been discussed more and more during the last decade (Rodrigues et al. 2009). This increase in the number and size of restoration projects is catalyzed by international market mechanisms like the payments for ecosystem services (PES), Clean Development Mechanism (CDM), certification systems and the implementation of the Kyoto protocol (Ferretti and De Britez 2006, Rodrigues et al. 2009). However, the studies published of the Atlantic forest focus largely on the Southern part of Brazil, while North Brazil, where the most valuable biodiversity hotspots are situated, remains understudied (Martini et al. 2007, Carnaval 2009). Additionally, most of the published restoration research is focused on species composition (altering the proportion of pioneer: non-pioneer species and biodiversity), usually not taking into account the potential benefits of more intensive silvicultural methods to mitigate the environmental stresses (Rodrigues et al. 2009, Campoe et al. 2010).

Atlantic forest restoration has a long history, which includes both failures and successes (Rodrigues et al. 2009, Suding 2011). However, the amount of failures seem shockingly large. Only 2 of 98 publicly funded reforestation projects in Brazil were evaluated as successful (Wuethrich 2007). Failures to create self-sustaining ecosystems are valuable learning experiences on how the ecosystem works (Ewel 1987). Some of the researchers considered restoration of the Atlantic Forest as an impossible task because of its huge original functional diversity and regional sociopolitical issues (Dean 1997). However, today the Atlantic forest restoration projects have ambitious goals where they aim to conserve biodiversity while contributing to the mitigation of global climate change by protecting and restoring threatened tracts of the Atlantic Forest (Tiepolo et al. 2002).

According to a long-term study, Atlantic forest needs about one to three hundred years to reach the adequate proportions of animal-dispersed species, non-pioneer species and understory species found in mature forests (Liebsch et al. 2008). Even more time is needed to reach the endemism level found in mature forests, which is approximately 40% of the total species in the Atlantic forest, and the estimated time span varies between one and four thousand years (Myers et al. 2000, Liebsch et al. 2008). Nevertheless, forests can almost fully recover over time spans of hundreds of years (Liebsch et al. 2008).

Intensive silvicultural methods on restoration projects

More intensive silvicultural methods (heavier site preparation, weed control and fertilization) have not shown consistently positive outcomes in the all tropical reforestation studies. For example, one previous study even concluded that fertilizing native tree species might be waste of money and resources (Carpenter et al. 2004). A fertilization regime without adequate competing vegetation control can favor the weed development and indirectly reduce tree growth. However, previous studies from South Brazil have shown that canopy closure can be accelerated and wood growth and carbon accumulation significantly increased in the Atlantic forest restoration by using more intensive management methods (Campoe et al. 2010, Ferez 2011). Those intensive management methods were originally developed for *Eucalyptus* plantations and have increased the growth rates of *Eucalyptus* and Pine plantations 3- to 4-fold in the last 4 decades (Eldridge et al. 1994, Santana et al. 2000, Stape et al. 2001). The intensive management aimed to eliminate all grass competition by spraying 5L ha^{-1} of glyphosate to the entire plot. The amount of fertilizer applied was more than three times that of the usual approach for native restoration (Campoe et al. 2010). Intensive management increased the woody biomass growth 250%, from 1.85 Mg ha^{-1} yr^{-1} to 6.45 Mg ha^{-1} yr^{-1}, compared to usual methods over a six-year period in South Brazil (Ferez 2011). Sites with larger spacing especially benefitted from intensive silvicultural methods regarding the tree growth and light use efficiency (Campoe et al. 2010). Effective weed control, especially when carried out chemically, has proven to be crucial to secure the initial restoration process on the degraded lands where the weed competition is high (Florentine and

Westbrooke 2004, de Souza and Batista 2004, Campoe et al. 2010). However, Sampaio et al. (2007) suggested that a restoration that is too intensive might actually have an overall negative effect on the restoration process over a long period.

According to their results, early succession of seasonal deciduous forest in pastures in Central Brazil does not need to be stimulated once the perturbation is stopped and that the intensive restoration efforts may actually slow recovery (Sampaio et al. 2007). This might be because of the intensified competition between trees for resources and because the fast-developed dense canopy effectively prevents the light from reaching the forest floor and therefore prevents seed germination and growth. In addition, restoration designs with intensively managed plantations of native tree species often do not create authentic looking rainforest and maintain their plantation appearance for a long time (de Souza and Batista 2004, Rodrigues et al. 2009). Intensive methods might only have benefits in the early stage. For example, results from one study showed that the fertilized treatment showed advantages in growth up to the third year, but did not do better than the non-fertilized treatment after 8 years (Carpenter et al. 2004). In addition, more intensive management methods might increase the growth of fast growing pioneers which could lead to suppression and elimination of slow growing non-pioneers (Luu 2012).

Site preparation

Adequate site preparation practices reduce weed competition, facilitate rooting, modify soil moisture conditions and improve nutrient availability for the cultivated seeds and seedlings. Site preparation methods have shown their importance in determining both future productivity and biodiversity of the restored forests and the magnitude of the importance of the site preparation has been described as overwhelming (Parrotta and Knowles 2001). Subsoiling, where the aim is to break up the compacted soil layer has shown clear positive effect on tree growth and stand development by facilitating rooting (Morris and Lowery 1988). For commercial species, like *Eucalyptus* and pine, in general, the more intense site preparation, the better root development that affects positively not only the initial growth but also the long-term growth, especially in areas where recurrent dry seasons occurs (Stape et al. 2010).

Planting densities

High planting densities have been used and are even recommended in some projects and trials to increase biomass stocks and accelerate the canopy closure. Faster canopy closure is expected to secure the success of restoration by reducing erosion and competition from grasses (de Souza and Batista 2004, Campoe et al. 2010). Downsides to this are higher planting expenses, increased competition between planted trees, and a decrease in light use efficiency (Campoe et al. 2010) during droughts. Light use efficiency (LUE) means the ratio of net primary productivity to photosynthetically active radiation absorbed by green vegetation. Larger spacing reduces competition between trees, but increases weed competition, especially, if adequate weed control is not applied (Campoe et al. 2010). The most common planting spacing in Atlantic forest restoration projects is 3m x 2m (1667 trees ha^{-1}) (Rodrigues et al. 2009, Campoe et al. 2010, Ferez 2011).

Tree species compositions (pioneer vs. non-pioneers)

Failures of past Atlantic forest restoration projects can most often be attributed to the planting of predominantly fast growing pioneer species which soon died (Rodrigues et al. 2009). Pioneer and early successional species are first to colonize the ecosystem after a disturbance, are light demanding and usually shade intolerant (Budowski 1964). Late successional and climax species (non-pioneers) develop in the understory and are generally shade tolerant and long-lived (Swaine and Whitmore 1988). Early restoration attempts were not very successful long term because there was no consideration for the basic principles of secondary succession (Kageyama et al. 2000). Early mortality of the pioneer species can cause a problem if the understory is not well developed (Parrotta and Knowles 2001, Parrotta and Knowles 2002). Kageyama et al. (2000) proposed a restoration model with tree species composition proportions of 50:50 pioneers to later successional species, in order to increase the proportion of long living climax and later secondary species as well as increase the biodiversity. This sound like a reasonable suggestion since the fast growing pioneers, that are already endangering the diversity by homogenization, will likely regenerate to the restored sites more readily than the non-pioneers (Lôbo et al. 2011). In addition, increasing pioneer species in previous studies has resulted in a decrease in the growth of neighboring seedlings (Massad et al. 2011, Lôbo et al. 2011). The inclusion of late successional species is important for long-term restoration in order to restore the original structure, functions and species composition, especially if natural recovery processes are limited (Parrotta and Knowles 2001).

However, a previous study in South Brazil (de Souza and Batista 2004) implied that the restoration design regarding the species composition did not influence forest structure and dynamics, at least at the developmental stage studied (5-10 year old forest stands). This merely suggests that the long-term effects of the pioneer species proportions regarding the sustainability in many restoration projects are yet to be seen because the ecological processes occur over a long time scale (de Souza and Batista 2004, Campoe et al. 2010, Massad et al. 2011).

Higher biodiversity increases growth

Restored sites where larger amounts of tree species were planted have been generally more productive and self-sustaining with higher growth and increased natural regeneration (de Siqueira 2002, Barbosa et al. 2003, Florentine and Westbrooke 2004). On average, forests in polycultures have shown 23.7 % higher productivity than in monocultures (Zhang et al. 2012). Positive correlation between biodiversity and productivity has been proposed to be caused by higher functional diversity. However, in a previous study, there was a significant positive association between local tree species richness and wood production while functional species richness did not remain significant (Vilà et al. 2007). This suggests that trees species have different niches within the functional groups. In addition, tree interactions, both competitive and positive, are not fully determined. Overall, the nitrogen fixing trees (*Leguminosae* or *Fabaceae*) can have a positive effect on the development of different tree species by improving the nitrogen availability for non-legume species through litterfall, falling fruits and woody debris from roots (Roskoski 1981, Peoples et al. 1995, Franco and De Faria 1997).

Regeneration

In order for a forest ecosystem to be self-sustaining, it should create adequate circumstances for the next woody species generation. However, different regeneration strategies should be taken into account as well when choosing species to be planted. Germinating seeds is the main source of new plants to regenerate forests and sprouting as a reproductive strategy is rare but it is an important strategy in order to recover after damage (Simões and Marques 2007). Fruit availability for frugivores is positively correlated with the density of stems in the understory, meaning that features such as seed dispersal mechanisms and fruiting season should be considered when selecting species to be planted (Sansevero et al. 2011). In addition, external seed rain is important for high diversity regeneration and therefore the structural connectivity between regenerating fragments and mature forest stands should be carefully considered (Groeneveld et al. 2009).

Tree plantations can facilitate the recruitment of seedlings

The Framework method, which relies on planting mixtures of 20-30 carefully selected fast growing pioneer and climax species to secure the canopy closure, has been proven effective in many cases in Thailand and Australia (Goosem and Tucker 1995, Elliott et al. 2003). Characteristics of framework species are (1) high survival and growth rates in open degraded sites, (2) spreading, dense crowns that shade out herbaceous weeds and (3) provision of resources that attract seed-dispersing wildlife (fruits, nectar, nesting sites) at an early age (Goosem and Tucker 1995).

Benefits of this method are high survival and low costs due to only having a single planting event. Plantations of fast growing species, either commercial or native framework species, can facilitate forest succession in their understories by modifying light, temperature and soil moisture conditions, enabling germination and growth of seeds (Goosem and Tucker 1995, Parrotta et al. 1997, Otsamo 2000).

Nucleation method

Applied nucleation is a cost effective strategy that has potential implications in the Atlantic forest restoration. The initial colonists are called ‘‘nuclei’’, and the process of cluster development and expansion ‘‘nucleation’’ (Yarranton and Morrison 1974). Natural forest recovery has been observed to follow a discrete pattern where pioneer species, which are first to colonize the open habitats, create clusters of vegetation around which other species establish. Applied nucleation mimics natural successional processes and expedites the recolonization of the woody plants and, has potential to restore deforested habitats into areas of diverse species composition (Yarranton and Morrison 1974, Corbin and Holl 2012). In applied nucleation, trees are planted in small patches as focal areas and these patches attract dispersers and facilitate recruitment. Forested area will expand from these patches over time (Figure 1). Because of the reduced planting expenses, applied nucleation is much cheaper than designs relying on the plantation like structure, but it can still accelerate forest recovery to a similar degree as plantation-style restoration (Zahawi et al. 2013). In general, the bigger the nuclei are, the better they facilitate the recolonization process by enchasing seedling establishment (Zahawi and Augspurger 2006, Cole et al. 2010, Zahawi et al. 2013). Zahawi et al (2013) concluded that appropriate size for a nuclei “island” would be approximately 100 m^2.

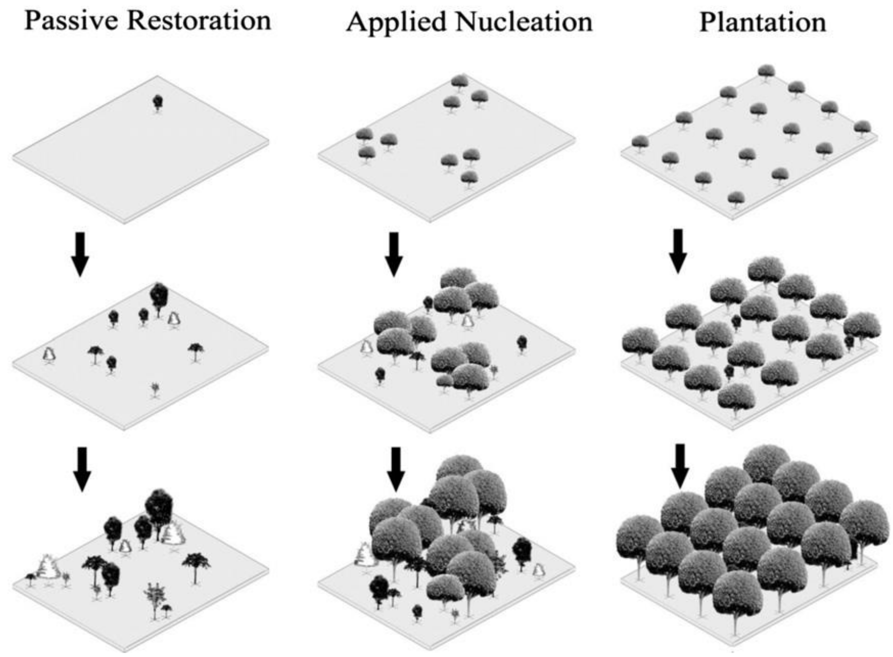

Figure 1. Applied nucleation. In the passive restoration, no planting is made and the recruitment is completely relying on the natural regeneration creating a heterogeneous but sparse canopy. In the applied nucleation, trees are planted in patches, or "nuclei", from where the woody species start to colonize and expand the forest area creating a heterogeneous canopy with much faster canopy closure than without active restoration. In a plantation design, where trees are planted in rows, the canopy closure is fast, but the natural regeneration and species diversity are low. (Corbin and Holl 2012).

Monitoring the restoration project

Determining whether or not a forest restoration project was successful and created a self-sustaining ecosystem might not be easy given the short time frames that are reserved for the restoration projects. Monitoring activities often include evaluating the biomass of planted and regenerated individuals, species richness and composition and leaf area index (de Souza and Batista 2004, Rodrigues et al. 2009, Campoe et al. 2010). To assess the success of the ecological restoration, (Ruiz-Jaen and Mitchell Aide 2005) recommended restoration projects to include attributes (for example diversity, vegetation structure, and ecological processes) that are clearly related to ecosystem functioning and at least two reference sites to capture the variation that exist in ecosystems.

Leaf area index

Leaf Area Index (LAI) is defined as the one sided green leaf area per unit ground area in broadleaf canopies (Chen and Black 1992). Leaf area index is a key structural characteristic of forest ecosystems; green leaves are controlling many biological and physical processes in plant canopies and trees grow faster and develop a denser canopy in favorable environments (Waring 1983a). LAI also controls the amount of light that reaches the forest floor, interfering with grasses and woody understory development. Therefore, in forest restoration projects, LAI is a key variable to indicate the success of different restoration techniques. The denser the canopy is, the higher is the LAI value. LAI usually ranges from 0 (no canopy cover) to 6 (very dense forest) (Olivas et al. 2013). In the study published from the Southern site, LAI was a good predictor of the stemwood growth with r^2=0.93 and ranged from 2 to 4 (Campoe et al. 2010).

Forest floor and understory

Forest floor (FF), which consists of the litterfall, dead branches on the ground, and understory are sometimes sampled to obtain more accurate estimate of carbon stocks. Significant amounts of carbon, even 20% of total aboveground carbon stocks, can be allocated in the forest floor and understory compartment and therefore they are recommended to be included in the biomass inventories (Richter et al. 1999, Keller et al. 2001, Tiepolo et al. 2002). However, some studies from tropical plantations where FF carbon was less than 1% of the system recommend to re-evaluate the importance of forest floor in forest carbon stock balance (Usuga et al. 2010).

2.3 *Potential of the restoration projects in the carbon sequestering initiatives*

Forests regulate the climate and local weather by participating in the hydrologic cycle, providing physical shelters and modifying the atmospheric composition (Bonan 2008). The biophysical properties of the forests, like reflectivity (albedo) and evaporation, also influence the climate (Jackson et al. 2008, Bonan 2008). Industrialization and burning of fossil fuels has increased the amount of greenhouse gasses in the atmosphere, resulting in an increase in the earth's surface temperature of 0.8 ^{o}C over the past 150 years (Houghton et al. 2001, Seinfeld 2011). Greenhouse gasses (mainly carbon dioxide, methane and nitrogen oxide) are causing this rise in temperature by remaining in the atmosphere and absorbing the radiation being reflected from the earth, creating a greenhouse effect (Baumert et al. 2005, Lacis et al. 2010). It is estimated that 15-20 % of greenhouse gas emissions are caused by deforestation and land use changes in forests of tropical countries (Bernstein et al. 2007, Bala et al. 2007). Carbon dioxide is a major greenhouse gas that is fixed during photosynthesis. Consequently, restoration of tropical forests have the potential to mitigate the climate change by sequestering carbon from the atmosphere into the above- and belowground biomass and soil (Cook et al. 2013). Tropical forests are significant carbon sinks, currently storing nearly 30% of the planet’s terrestrial carbon (Phillips 1998, Bonan 2008, Houghton et al. 2009). In addition to the carbon sequestration, tropical forests mitigate warming through the process of evaporative cooling (Bonan 2008).

Estimating how much effect forests have on the global carbon balance is challenging. The carbon stocks vary greatly by forest type causing uncertainty in the estimates, the protocol for estimating C stocks is not well standardized and direct biomass measurements on the ground are expensive and time consuming (Silver et al. 2000, Chave et al. 2005). Nevertheless, estimating how much influence a tropical forest has on climate change requires reliable estimates of the forest biomass (Houghton 2005). The biomass density, expressed as dry weight per unit of area, is a good estimate of the carbon stocks. Approximately 50 % of the dry biomass is carbon and the use of a conversion factor of 0.5, from biomass to carbon, has been widely used in the literature (Silver et al. 2000). The carbon stocks of the forests represent the amount of carbon that could be added to the atmosphere if the forest is for example burned. It could also be used to estimate the carbon dioxide removed from the atmosphere by growing forest (Cook et al. 2013).

Studies have shown that tropical forests with rapid growth rates are good options to mitigate CO_2 emissions through carbon sequestration (Montagnini and Porras 1998, Silver et al. 2000). Forest restoration projects are attractive options in the global carbon credit markets; they have potential to sequester carbon for up to 80 years, possibly even longer, and obtaining precise estimates of the carbon sequestered is feasible (Silver et al. 2000). Aboveground biomass in the restored forests increases on average at a rate of 6.2 Mg ha^{-1} yr^{-1} during the first 20 years and 2.9 Mg ha^{-1} yr^{-1} over 80 years of regrowth yielding much higher increments than without the restoration efforts (Silver et al. 2000). Initial carbon sequestration potential measured during the six years from planting in the South Atlantic forest using intensive methods can be as high as 4.22 MgC ha^{-1} yr^{-1} (Ferez 2011). Over a 50-year period, approximately 50 to 100 Pg. of carbon could be sequestered globally with restoration and afforestation projects (Winjum et al. 1992).

The United Nations Framework Convention on Climate Change (UNFCCC) Clean development Mechanism (CDM) aims to mitigate climate change by reducing greenhouse gasses from the atmosphere. In CDM, industrialized countries are able to purchase “Certified Emission Reductions” (CERs) in order to meet their emission reduction obligations under the terms of Kyoto protocol because actual emissions’ reductions are much more costly (Wara and Victor 2008). However, only four CDM projects out of 1600 are afforestation or reforestation projects (A/R) even though they have a lot of potential to obtain CERs relatively cost-efficiently, especially in the tropical regions (Zomer et al. 2008, Thomas et al. 2010). Obtaining more accurate estimates of the existing carbon stocks in reforestation/afforestation projects, engaging the stakeholders with diverse backgrounds, increasing the flexibly of the CDM scheme and emphasizing the multiple-use benefits of forests are important in order to make A/R more attractive options for CDM projects (Thomas et al. 2010). Besides that, the lack of efficient silvicultural protocols that assure the restoration success in the short- and long-term is also considered a reason for the low number of existing CDM projects.

2.4 *Remarks from the previous studies*

The most effective restoration method might not be necessarily the most ecological one (Rodrigues et al. 2009). Intensive management methods accelerated the canopy closure and increased the stemwood growth and light use efficiency (Campoe et al. 2010, Ferez 2011). However, it may not create an authentic looking tropical forest in the near future and management methods that are too intensive might actually delay natural regeneration or just be a waste of resources (Carpenter et al. 2004, Sampaio et al. 2007). However, many studies showed that plantation establishment with suitable fast-growing tree species facilitate recruitment of a variety of native tree species (Goosem and Tucker 1995, Parrotta et al. 1997, Otsamo 2000). The recruitment of new native species seems to be highly dependent on the intensity of management, tree species planted and site specific conditions. Nevertheless, on seriously degraded land where competition with invasive weeds is high, more intensive methods, like chemical control of grasses and the use of fertilizers, are needed in order to establish a closed canopy (Aide et al. 1995, Chazdon 2003, de Souza and Batista 2004, Campoe et al. 2010). Additionally, it is often enough to rehabilitate the forest functions (e.g. water and wood production, carbon sequestration) without aiming to restore the "original" forest.

Site preparation, higher biodiversity and a higher proportion of nitrogen fixing species showed clear positive effects in the forest development and growth in the literature (Martins and Engel 2007, Siddique et al. 2008, Rodrigues et al. 2009). In addition, the importance of planting local native species should not be underestimated (McKay et al. 2005). The previous research seemed to advocate for a larger non-pioneer species proportion to increase the biodiversity. Larger proportions of pioneer species increase the initial growth and are generally less expensive, but too high proportions might lead to insufficient amounts of non-pioneer in the long-run. This paper focused on the active restoration methods relying mainly on the planting of the native species, but the animal grazing should also be taken into account since it can effectively prevent development of non-grazing tolerant species and negatively affect species diversity (Stern et al. 2002). In addition, it is necessary to remember that all studied sites in the reviewed papers are unique regarding the site-specific conditions that should be taken into account when planning restoration strategies. As de Souza and Batista (2004) concluded, complex models may be theoretically ideal, but practically unfeasible. Adequate monitoring programs are needed to develop restoration techniques. More data from older reforestation projects is necessary to analyze in order to better evaluate the success of the restoration process and obtain accurate estimates of the long-term carbon sequestering. Previous studies showed that reforestation projects have significant potential to sequester carbon even though their magnitude right now might be rather modest in the global scale (Silver et al. 2000, Thomson et al. 2008). However, when the restoration projects only focus on the carbon sequestering seeking for carbon offset, fast growing pioneer species or even exotic tree species and trees that allocate their biomass aboveground might be favored at the expense of the biodiversity and functionality.

3. Materials and methods

In this chapter, the experimental design and methods used in the data collection and analysis are described. Survival, leaf area index (LAI), stemwood biomass and accumulated carbon at year 8 after planting were determined for all treatments in July/August of 2012.

3.1 *Site descriptions*

Study site: The Northern site

The Northern site trial was installed in Bahia State, Brazil, 200 kilometer North from Salvador and 20 km from the coast (11° 48' 26" S, 37° 45' 19" W) (Figure 2). The soil type is Yellow Argisols (Brazilian System of Soil Classification) or clayey Hapludults (USA - Soil Taxonomy). The main characteristics of the soil are the presence of textural B horizon, high bulk density (> 1.4 g cm^{-3}), low porosity, low nutrient availability and a hardpan between 60 to 70 cm deep. The climate is a typical tropical climate; annual mean temperature is 25.6°C (23.2°C to 27.5°C monthly range) and annual rainfall 1635 mm with a rainy season from April to September (70% of the total rain). The natural vegetation type is evergreen rainforest and previous land use of the site was *Pinus caribeae* var *hondurensis* plantation. Caribbean pines were planted in the end of 1980’s and harvested 2 years before the establishment of the trial.

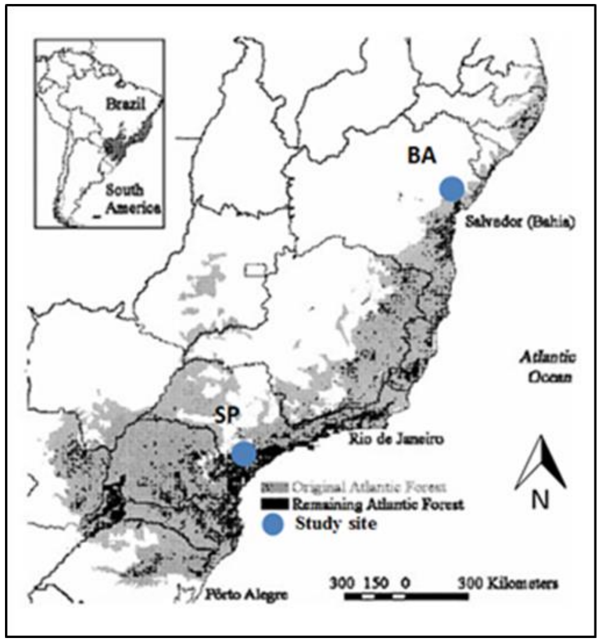

Figure 2. Study site locations. BA = Northern site in Bahia and SP = Southern site in Sao Paulo. Original Atlantic forest (grey) and present distribution (black). (Harris and Pimm 2004).

Comparison: The Southern site

As a comparison, the Southern site trial, installed at the University of São Paulo's Anhembi Forest Research Station (22º43'22''S, 48º10'32''W), has a mean annual temperature of 19.1°C and mean annual rainfall of 1170 mm (75% concentrated from October to March). The soil in São Paulo's trial is a sandy Typic Hapludox. Land at this site was previously used as a pasture. A full description of the site can be found in Campoe et al. (2010) and (2014) and Ferez (2011).

3.2 *Experimental design and treatments*

The project has a 2^3 factorial structure, where intensive and usual management (factor 1) were applied to 20 native species, for two initial stand densities (factor 2, 3333 trees ha^{-1} and 1667 trees ha^{-1}) and two species composition proportion (factor 3, 50:50 and 67:33 ratio of pioneer : late successional species) (Table 1).

Table 1. The treatment descriptions and labels for the Atlantic Forest restoration study.

Factor	Label	Description
Species composition	A	50/50 pioneer : late successional
	B	67/33 pioneer : late successional
Stand density	1	3333 trees ha^{-1}
	2	1667 trees ha^{-1}
Management method	X	Intensive
	U	Traditional
	T	Baseline; no planting of trees
	D	Destructive sampling plots

The trial has 40 plots in total; four replications for each of the 8 treatments totaling 32, 4 plots kept as control, or baseline, where no planting or management operations were made, and 4 plots planted to allow destructive sampling. Each plot is 39 m x 30 m (0.12 ha) with a measurable interior plot of 32 m x 24 m (768 m^2). All trees were planted in August of 2004 and treatments were applied randomly to the experimental units (plots) (Figure 3). More about the study design is explained in the statistical analysis (Chapter 3.7).

1 A1X 1	1 B1X 2	1 A1U 3	1 T 4	1 A2U 5	1 B1U 6	1 B2X 7	2 T 8
2 B1U 9	1 A2X 10	1 B1U 11	2 B2X 12	2 B1U 13	2 A1X 14	1 D 15	2 A2X 16
3 T 17	3 B2X 18	3 A2X 19	3 A1X 20	2 B1X 21	2 D 22	2 A1U 23	3 B1X 24
3 B1U 25	3 D 26	4 D 27	4 B1X 28	3 A1U 29	4 A2X 30	2 A2U 31	3 B1U 32
3 A2U 33	4 A1U 34	4 B1U 35	4 A2U 36	4 T 37	4 B2X 38	4 B1U 39	4 A1X 40

Figure 3. Experimental layout of the Northern site. A stands for species composition 50:50 ratio of pioneer and later successional species, B for 67:33 pioneer and later successional species, 1 = stand density 3333 trees/ha and 2 = 1667 trees/ha, X stands for intensive (maximum) treatment and U for usual (usual) treatment. D plots are for destructive sampling and T plots are for control. The number in the right upper corner is repetition.

A higher proportion of pioneer species (B) to later successional species (A) was expected to increase seeding survival while higher planting density (1) was expected to accelerate canopy closure. Intensive management (X) aimed to reduce the stress caused by water and nutrient limitations and weed competition (especially Brazilian satintail, *Imperata brasiliensis*) in the early stage.

Management Factor: Usual and intensive management procedures

Different soil preparation methods, fertilization and weed control were used for the usual and intensive managements.

Usual or Traditional System (U): One month before soil preparation, a manual slashing was performed and all wood debris larger than 3 cm diameter were removed from the area. Soil preparation consisted of manually pitting the soil for the seedlings, in the same manner the people in the region do usually for native forest restoration, opening a 20cm x 20cm x 20cm hole using blade tools.

August (2004) the planting took place using containerized seedlings followed by watering (3 L/tree) and fertilization with NPK 6:30:6 (230 kg/ha) in the pit. The total amount of N:P:K:Ca:B applied, in kg/ha, were 15, 30, 12, 0 and 0 respectively. Weed competition was controlled manually in bands over the rows in September (2004) and November (2004). Leaf cutter ant (*Atta* spp) control was done once before planting and three times after planting, approximately once a month. First, K-Otrine (Deltametrine, powder, 10 g per m^2 of ant's nest) was used due to rainfall at the time. Later, baits of Mirex-S (Sulfluramide, 5 g per hole of the ant's nest) were used.

Intensive or Maximum Silviculture System (X): A manual slashing was performed over the total area a month in advance to site preparation, removing all wood debris larger than 3 cm diameter. Site preparation consisted of a 90 cm deep subsoiling using a high power D8 Caterpillar tractor followed by typical harrowing in the future planting rows (Figure 4). Planting of the containerized seedlings occurred in August (2004) followed by watering (3 L/tree). Lime (2000 kg/ha) and Rock Phosphate (300 kg/ha) was broadcasted in the plots. Fertilization consisted of a starter at the planting pit (NPK 6:30:6, 230 kg/ha) followed by broadcasting of 200 kg/ha of NPK 11:6:24 at 9 months, 150 kg/ha of FTEBr12 (micronutrients) at 15 months and 500 kg/ha of NPK 13:6:17 at 24 months old. The total amount of N:P:K:Ca:B applied, in kg/ha, were 100, 60, 120, 500 and 2 respectively. Complete weed control using ghyphoste (5 L/ha, 0.2%) was used prior to planting and manual weeding was done in total area, whenever necessary, to keep the plots free of competing vegetation up to 2 years-old (in general every 3 months). Leaf cutter ant control was done as prior described for the usual system.

Figure 4. Above: D8 Caterpillar tractor was used for site preparation in the intensively managed plot. Site preparation consisted of a 90-cm deep subsoiling. Below is the picture of the ground after subsoiling.

Density factor: High and low planting densities

The design uses two distinct spacing, both with a 3 m inter-row distance, which is need for wheel-tractor movement inside the stands. The larger, normal, spacing is the 3m x 2m (coded here as “2”) which is the most common spacing for commercial plantation in Brazil (Stape et al. 2001) and even in many restoration projects (Rodrigues et al. 2009), leading to a stand density of 1667 trees ha^{-1}. The other level of this factor was chosen to be 3m x 1m (coded here as “1”), doubling the population to 3333 trees ha^{-1}, with the rational to reach a sooner canopy closure, increase growth and reduce grass competition.

Composition Factor: Tree species planted and their successional groups

Twenty different native species from 12 botanical families were planted in this trial (Table 2). Three are fast growing pioneer species, seven belong to earlier successional groups, seven are from later successional groups and three are considered climax species, based on the Brazilian literature (Campoe et al. 2014). The first two groups were referred as “pioneer species” and the last two as “non-pioneers” in this study. In the species composition coded as “A” a 50:50 pioneer: non-pioneer species were planted in the rows, rotating between the two types at every planting (Figure 5, Figure 6). In the species composition coded as “B” a 67:33 pioneer: non-pioneer proportion were used and two pioneer species were followed by one non-pioneer species. Trees were classified in successional groups according (Budowski 1964). The trees for this trial were produced locally in plastic containers using local collect seeds from Atlantic Forest areas.

Many of these climax species has huge commercial values and illegal logging is still a problem in Brazil (Lira et al. 2012). Among these, the species that give the name to the country itself is represented: *Caesalpinia echinata* (Brazilwood), which was explored by the Portuguese from the 14th to 19th century.

Table 2. Scientific name, common name, family, successional group and abbreviation of the 20 tree species used in the North Atlantic Forest restoration project. Pioneer and Early successional comprehends Pioneer group, and Late successional and climax are referred as Non-pioneers.

Scientific name	**Family**	**Common name**	**Abbreviation**	**Successional status**
Cecropia pachystachya	Urticaceae	Embaúba	EB	Pioneer
Gochnatia oligocephala	Asteraceae	Candeia	CA	Pioneer
Schinus terebinthifolius	Anacardiaceae	Aroeira-Pimenteira	AP	Pioneer
Anadenanthera sp	Fabaceae	Angico-branco	AN	Early successional
Eriottheca macrophylla	Malvaceae	Embiruçu	EM	Early successional
Inga thibaudiana	Fabaceae	Ingá	IN	Early successional
Micrandra elata	Euphorbiaceae	Mamoninha	MA	Early successional
Tabebuia sp	Bignoniaceae	Ipê-amarelo	IA	Early successional
Tapirira guianensis	Anacardiaceae	Pau-pombo	PO	Early successional
Xylopia frutescens	Annonaceae	Pindaíba	PI	Early successional
Eschweilera ovata	Lecythidaceae	Biriba	BI	Late successional
Myracrodruon urundeuva	Anacardiaceae	Aroeira-do-sertão	AS	Late successional
Cariniana estrellensis	Lecythidaceae	Jequitibá-Branco	JB	Late successional
Protium heptaphyllum	Burseraceae	Amescla	AM	Late successional
Simarouba amara	Simaroubaceae	Pau-paraíba	PA	Late successional
Tabebuia sp	Bignoniaceae	Ipê-roxo	IR	Late successional
Dialium guianense	Fabaceae	Jitaí-Preto	JP	Late successional
Caesalpinia echinata	Fabaceae	Pau-brasil	PB	Climax
Genipa americana	Rubiaceae	Jenipapo	JE	Climax
Hymenaea courbaril	Fabaceae	Jatobá	JT	Climax

Pioneers | Early successional | Late successional | Climax

A2

JE	CA	JB	MA	BI	EM	AS	AN	JT	PO	AM	EB	PA	IN	PB	PI	IR	EB	PA
IA	AM	EB	PA	IA	BI	PI	JP	CA	AM	PO	JB	AP	AS	PO	BI	AP	AS	AN
IR	IN	PB	AN	AS	CA	JT	EM	JE	MA	JP	EM	PB	AN	JE	IN	JB	PI	JE
CA	JP	CA	AM	PI	JB	AN	AM	PO	IR	EB	JP	IA	AM	CA	PA	MA	PB	MA
PA	PI	BI	MA	JT	MA	PA	IA	JT	IN	JT	CA	AS	PO	AM	EM	IR	PO	BI
IN	PB	AP	PA	IN	JP	AP	BI	PO	BI	PI	PA	PI	AS	EB	JB	MA	JE	PI
BI	AP	JB	EM	JE	EB	JE	IA	AS	IA	IR	EB	JT	AN	PB	IN	JP	CA	JT
EM	AM	IA	IR	IN	AM	EM	JB	EB	JT	AN	JB	MA	JP	EM	BI	IA	PA	IA
JT	MA	PB	AP	JE	PI	PB	AN	IR	PO	JT	IN	JE	AP	PB	PI	IR	AP	JP
IA	AS	EB	JP	AN	AM	IN	AS	MA	IR	AP	BI	EM	JE	CA	JB	AN	PB	CA

B2

CA	PO	JE	IA	AP	JB	AN	IN	AS	MA	EB	PB	AP	EM	BI	AN	IA	JP	MA
PI	BI	EB	PO	PB	EM	EB	PA	IN	EM	JE	IA	AN	JB	EB	CA	AM	PI	CA
JB	MA	IA	JB	IN	MA	JT	IA	CA	BI	PO	EB	IR	IA	IN	PB	AP	MA	JT
EB	PO	JP	EB	PI	AM	PI	EM	BI	EB	AN	AS	AP	MA	IR	CA	IA	PA	IA
IN	IR	AN	CA	AS	MA	CA	IR	AN	PI	PB	IN	MA	JP	EM	PO	BI	AN	EM
JT	CA	PO	JE	AP	IA	JB	CA	PO	PA	AP	IA	AM	CA	IN	JE	PI	AP	AS
EB	PI	AM	IN	EM	JT	AP	PI	JP	MA	IN	BI	PO	PI	AS	EM	PO	JP	IN
PO	AS	EM	CA	JE	PO	PI	JB	EM	CA	IR	PO	AP	JT	AN	MA	JP	EB	EM
AM	EM	MA	AS	IA	AN	JT	AP	IN	PB	AN	PI	PA	EB	IA	JT	AP	MA	PA
AP	MA	PB	AN	CA	BI	IA	EM	AM	MA	IN	PA	AP	IN	IR	IA	EB	JP	AN

Figure 5. Above 50:50 proportions of pioneer vs. non-pioneer species where every other tree planted in a row is a non-pioneer tree species and below 67:33 proportions where every non-pioneer is followed by two pioneer species. Layouts are with stand density 1667 trees ha^{-1} (3m x 2m spacing layouts).

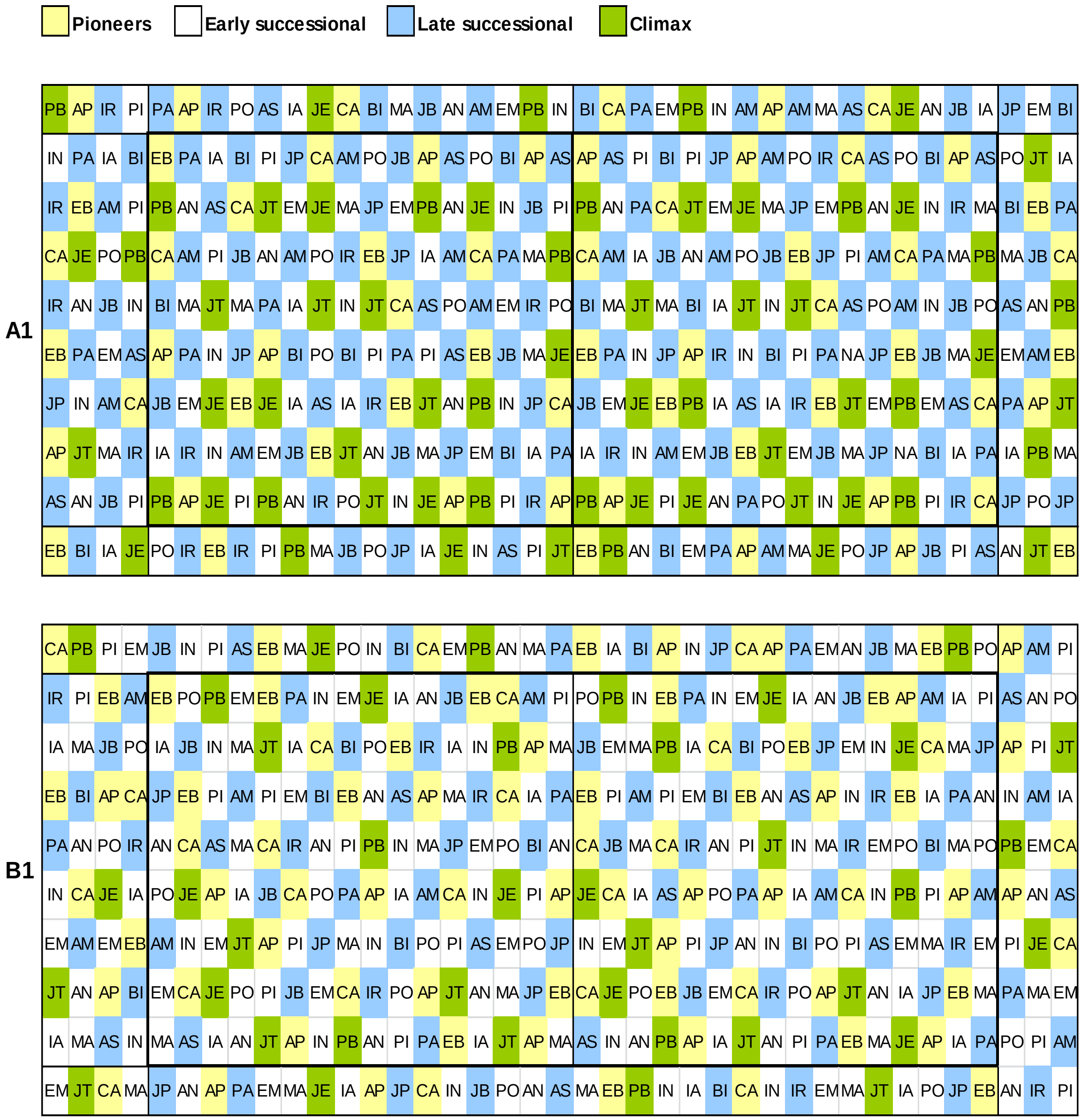

Figure 6. Above 50:50 proportions of pioneer vs. non-pioneer species where every other tree planted in a row is a non-pioneer tree species and below 67:33 proportions where every non-pioneer species is followed by two pioneer tree species. Layouts are with stand density 3333 trees ha^{-1} (3m x 1m spacing layouts).

3.3 *Species survival*

Trees that were missing or dead were marked as F (failure: missing tree) or M (dead tree). All trees were planted in predetermined order (Figure 5 and Figure 6) so it was possible to keep track of the species of missing trees. Mortality was calculated for each plot as a percentage of the dead and missing trees 8 years after planting compared to the trees planted initially (for the 3m x 2m spacing: (F+M)/128 and for the 3m x 1m spacing: (F+M)/256. Survival was 100 - mortality %. Additionally, survival for each species was calculated with intensive and usual management.

3.4 *Leaf area index*

Leaf area index (LAI) was determined for each plot with an indirect method using the Beer-Lambert Law (Equation 1). The Beer-Lambert law assumes that foliage is randomly distributed in space and that there is a spherical distribution among leaf inclination angles (Jarvis and Leverenz 1983).

$$LAI = -\frac{\ln\left(\frac{I}{Io}\right)}{k} \qquad (1)$$

Where

LAI is the leaf area index, in m^2 leaves per m^2 of ground;

I is the photosynthetically active radiation below the canopy, in µmol m^{-2} s^{-1};

I_0 is the photosynthetically active radiation above the canopy, in µmol m^{-2} s^{-1}; and

k is the light extinction coefficient.

Values of k varies generally between 0.4 and 0.7 (Campoe et al. 2010). When k is unknown, 0.5 is a good estimate (Law and Waring 1994, Maass et al. 1995, Saitoh et al. 2012).

Photosynthetically active radiation (PAR) was measured with an AccuPAR ceptometer (Decagon Devices Inc.) within 2 hours from solar noon from all plots where treatments were applied during September of 2012, in the end of the rainy season. Measurements were taken in a diagonal direction (from one corner to the opposite corner on the other site) in 12 stops. From each stop, the reading was taken four times. PAR was measured under full sunlight (I_0

radiation above canopy) before each plot and the calibration measurement closest in time to the sampling stops was used in LAI calculations.

Leaf area index correction (Equation 2) was made in order to minimize the errors caused by different angels of the sun in different times when using a constant k (0.5).

$$LAIc = LAI.\cos(\frac{\pi\theta}{180}) \quad (2)$$

Where:

LAIc is the corrected LAI;

θ is the zenith angle of the sun for given latitude, longitude, date and time of day calculated with PSA algorithm (Blanco-Muriel et al. 2001). Because the site is located at 11°S, and the readings were done between 11am and 1pm, the correction was minor.

3.5 *Stemwood biomass measurements*

Aboveground dry biomass estimates for each plot and trees 8 years after planting were obtained based on the measurements of tree height and diameter, bark and wood densities and estimates of the canopy biomass. In addition, estimates for forest floor and understory dry masses were determined.

Diameter and Height measurements for planted and regenerated trees

To determine the current stemwood biomass stocks (8 years after planting), diameter at 30 cm (D30) was measured along with height for all trees in the measurable plots. Diameter at 30 cm was measured instead of DBH (diameter at 1.3) because of the structure of native species (Brown 1997) and is the Brazilian standard for restoration studies (Durigan et al. 2012).

Heights were measured with Suunto hypsometers. For trees with two or more stems, the height for the highest tree was measured and all stems with D30 greater than 3cm were measured. In addition, all regenerated trees with a D30 greater than 3 cm, were measured (D30 and Height) and, if possible, identified. Regenerated tree species were identified when measured by local people with the local name and scientific names and families were searched later from the lists of native species from the area.

Wood and bark densities

The wood density (ρ_w), in g cm^{-3}, was determined for both wood and bark in order to obtain estimates for the stemwood biomass.

To determine the wood density (ρ_w), wood disc samples were taken from the destructive sampling "D" plots by cutting four stems per species, totalizing 80 trees (4x20). Typical trees chosen for destructive sampling were those that represented their species well for the developmental stage they are in during the trial. Stems were cut from D30 and DBH (1.3m) and a 2.5 cm thick wood disc was taken from each of them. Wood discs were taken instead of using an increment borer in order to eliminate the errors caused by variation in densities from pith to bark.

The diameter with and without bark was measured at D30 for all trees that were felled and bark samples were collected to obtain the bark specific gravity. Averages of tree diameters without bark at D30 were used to estimate the actual diameter of the wood. Because wood and bark may have relatively big differences in their densities and bark can be relatively thick in some tropical species, biomass for wood and bark were calculated separately.

Wood density was determined by dividing the dry weight by the wet volume and then dividing that by the density of the water (Chave et al. 2006, Williamson and Wiemann 2010). Wood discs were dried in an oven at 105 ^{o}C (+/-3 ^{o}C) for 36 hours. After that, the dry weight was measured and the wood discs were put into a water container for two weeks to fully saturate and achieve the wet volume. The same procedure was done for the bark samples. For regenerated tree species, the average densities of all known tree species from the site were used.

Canopy biomass

Canopy biomass was estimated for each of the 4 trees harvested per species. The entire canopies from the felled trees were harvested and weighed. The leaf dry matter content, defined as the oven-dry mass of a leaf divided by its water-saturated fresh mass, was estimated from subsamples (Cornelissen et al. 2003). The canopy biomass for all trees was calculated as a proportion of canopy dry mass to stemwood dry mass (Kauppi et al. 1995).

Forest floor and understory

To determine the biomass of forest floor and understory, samples were collected from all plots using a 1m x 1m quadrat (1 m^2) (Figure 7). Four inter-rows were sampled in a diagonal pre-determined design. The samples were dried at 60 ^{o}C in an oven for at least 36 hours, depending on the sample size and dry weight that was obtained from them.

Figure 7. Forest floor and understory sampling using 1m x 1m quadrat.

3.6 *Stemwood biomass calculations*

Tree biomass and allometric equations

The biomass for each tree was calculated based on the cross sectional areas at 30 cm, heights of trees and wood densities (Equation 3). The tree stem biomass should in general follow the relationship:

$$w = \frac{\pi . D30^2 . H . f . \rho}{4} \qquad (3)$$

Where:

w is the tree stem wood biomass, in kg;

D30 is the diameter at 30 cm from the ground, in cm;

H is the total height, in m;

f is form factor, considered to be 0.5, no unit;

ρ is the wood density, in g cm^{-3};

All tree trunks were assumed to be rotated parabolas and therefore 0.5 was used as a form factor to gain estimates of stem wood biomass for each tree. Campoe et al. (2010) for the South study showed that the factor 0.5 was a good estimate for the Atlantic Forest trees. Because only the highest stem of the tree was measured, the heights for additional stems were calculated based on species-specific D30/height relationships that were obtained from the regression equation from the scatter plots.

The biomass values for wood and bark were calculated separately. Estimates for D30 without bark were obtained using measurements made in the 4 tree felled for each species. Canopy dry masses were added to each tree.

For a comparison and to determine if accurate biomass estimates would be possible to obtain without destructive sampling, the dry forest type equation (Equation 4) presented by (Chave et al. 2005), was used to calculate the aboveground biomass. This equation was also used to calculate the total aboveground dry biomass for the regenerated and identified native tree species.

$$w = 0.112\ (\rho D^2 H)^{0.916} \qquad (4)$$

where:

w is the tree stem wood biomass, in kg;

ρ is the wood density, in g cm^{-3};

D is the tree DBH

H is the total height, in m;

Tropical forests that have an annual rainfall of less than 1500 mm per year are usually categorized as dry tropical forests (Brown 1997). Estimates for DBH were calculated based on the 4 measurements made from destructive sampling by dividing DBH with D30 per species and getting the "DBH" factor that was used to estimate the DBH for the allometric equations. However, the problem with applying biomass equations found in literature to this dataset is that they are not valid in small or large trees. It is recommended not using the Equation (4) for trees with a DBH of less than 5 cm (Chave et al. 2005).

Plot biomass and carbon

To obtain the total aboveground biomass for the plots, all trees, forest floor and understory biomasses were summed. Total biomass of the plots was calculated in Mg of dry mass per hectare. To obtain the aboveground carbon stock estimates for each plot the dry masses were multiplied by 0.5, which is a generic estimate for the carbon content of the dry biomass (Brown 1997, Lamlom and Savidge 2003).

3.7 *Statistical analysis*

This study is a 2^3 factorial experiment with 4 replicates (Table 3). Analyses of variance (ANOVA) was used to analyze the main effects and interactions of different factors. A structure that contains independent variables representing combinations of the levels of 2 or more categorical predictors is called factorial structure. In this study, categorical predictors (factors) were species composition, stand density and management, each of which had two levels. Factorial structures provide more information about the relationships between categorical predictor variables and responses on the dependent variables. Factors are said to interact if the difference in mean responses for two levels of one factor is not constant across levels of the second factor. In other words, parallel lines in the interaction plot do not indicate interaction. The greater the departure from the parallel state the greater the degree of interaction. This study has a full factorial structure containing all possible combinations of the levels of the categorical predictors. All treatments are randomly assigned within the trial. Collected data was typed in excel spreadsheets and the data was organized and summarized using pivot tables. All analyses (ANOVA GLM, unpaired t-test) were made with Minitab 16 software.

Table 3. 2^3 factorial experiment. Factors are species composition, stand density and management.

Species composition							
A: 50/50 pioneer : late successional				B: 67/33 pioneer : late successional			
Stand density				**Stand density**			
1: 3333 trees ha^{-1}		2: 1667 trees ha^{-1}		1: 3333 trees ha^{-1}		2: 1667 trees ha^{-1}	
Management		**Management**		**Management**		**Management**	
X: Intensive	U: Traditional	X: Intensive	U: Traditional	X: Intensive	U: Traditional	X: Intensive	U: Traditional
1: A1X	2: A1U	3: A2X	4: A2U	5: B1X	6: B1U	7: B2X	8: B2U

3.7.1 Anova GLM

The survival, LAI, stemwood biomass stocks and total aboveground carbon stocks were analyzed with ANOVA GLM procedure (general linear modeling) to determine the significant independent effects and interactions. The general linear model allows linear transformations or linear combinations of multiple dependent variables. Tukey's Studentized Range (HSD) tests (α=0.05) was used to test for significant differences to avoid error type 1 (rejecting null hypothesis incorrectly therefore getting a false positive result). This means that the null hypothesis is rejected incorrectly in this study with a 5 % probability.

Assumptions: (1) The distribution of the response is normally distributed. (2) The variance for each treatment is identical. (3) The samples are independent.

Hypothesis for ANOVA GLM procedure:

H_0: No differences between the effects of the different factors

H_1: Differences between different factors

Under the null hypothesis, there would be no interaction among the factors.

3.7.2 Unpaired t-test

An unpaired t-test was conducted for each planted tree species regarding the statistical differences in the stemwood biomass between the two management methods (usual and intensive management) at significance level (α=0.05). Unpaired t-test was also conducted to test if there were differences in wood density between successional groups. Unpaired t-tests compare the means of two populations when samples are independent.

Assumptions for the unpaired t-test: (1) The distribution of the response is normally distributed. (2) The variance for both treatments is identical. (3) The samples are independent.

Hypothesis for unpaired t-test; Stemwood biomass stocks for different tree species with two management methods

H_0: No differences between the effects of the usual and intensive management

H_1: Differences in stemwood biomass stocks between usual and intensive management

Hypothesis for unpaired t-test; Wood density and successional groups

H_0: No differences in density between the successional groups

H_1: Differences in density between the successional groups

4. Results

4.1 *Species survival*

Survivals were calculated for plots and tree species after the third supplement planting. Intensive management increased survival from 83% to 93% ($p < 0.001$) (Table 4). Neither the proportion of pioneer vs. later successional species nor stand density had any statistically significant effects on survival rates ($p = 0.345$ and 0.801) (Table 4). There was no interaction between species composition and planting density ($p = 0.391$), between management and planting density ($p = 0.137$) or between composition and management ($p = 0.835$).

Table 4. Survival (%) of the planted trees per factor 8 years from planting and their standard deviations of the means (sd), N = 4. Compositions: 50:50 pioneers vs. non-pioneers, 67:33 pioneers vs. non-pioneers.

Factors	Levels	Survival %	sd	p-values
Composition	50 : 50	86.6	3.72	0.345
	67 : 33	88.7	4.15	
Stand density	3333 trees ha^{-1}	87.3	3.2	0.801
	1667 trees ha^{-1}	87.9	4.7	
Management	Intensive	92.6	2.7	< 0.001
	Traditional	82.7	3.4	

Survival for different tree species

Intensive management methods significantly increased survival for most of the tree species planted (Table 5). *Eschweilera ovate* (BI) had the lowest survival in usually managed plots; as much as 65 % of the planted trees were missing or dead with the usual management, but intensive management methods reduced mortality down to 24 %. Survival was low with non-pioneers species, but the intensive management methods reduced the mortality of them more than pioneer species (33 vs. 7 % increase in survival) (Table 5). Figure 8 illustrates the differences between usual and intensive management. Table 6 synthetizes the survival results of the intensive silvicultural treatment (pionees vs. non-pioneers) for the other two factors (species composition and stand density).

Table 5. Survival (%) for intensive (X) and usual (U) management methods per tree species 8 years after planting and the change after intensive management (Δ%). Summary of all trees planted on the trial.

Scientific name	Species code	Survival % U	Survival % X	Δ%*
Inga thibaudiana	IN	97.19	98.53	1.38
Schinus terebinthifolius	AP	96.43	97.38	0.99
Cecropia pachystachya	EB	88.01	96.75	9.94
Tapirira guianensis	PO	89.74	96.38	7.39
Protium heptaphyllum	AM	92.36	96.15	4.11
Myracrodruon urundeuva	AS	98.08	96.15	-1.96
Eriottheca macrophylla	EM	85.16	95.64	12.31
Anadenanthera sp	AN	73.96	95.31	28.87
Caesalpinia echinata	PB	96.38	95.31	-1.11
Gochnatia oligocephala	CA	97.34	94.88	-2.53
Genipa americana	JE	80.62	94.16	16.80
Tabebuia sp	IA	89.55	91.35	2.01
Hymenaea courbaril	JT	81.39	90.05	10.64
Micrandra elata	MA	77.70	88.97	14.50
Dialium guianense	JP	83.49	88.16	5.59
Cariniana estrellensis	JB	71.45	86.84	21.53
Xylopia frutescens	PI	87.99	85.62	-2.70
Tabebuia sp	IR	39.64	82.68	108.59
Eschweilera ovata	BI	34.64	76.04	119.55
Simarouba amara	PA	49.49	69.30	40.02
	Pioneers	88.31	94.08	7.22
	Non- pioneers	72.75	87.48	32.38

* Effect of the intensive management method on survival

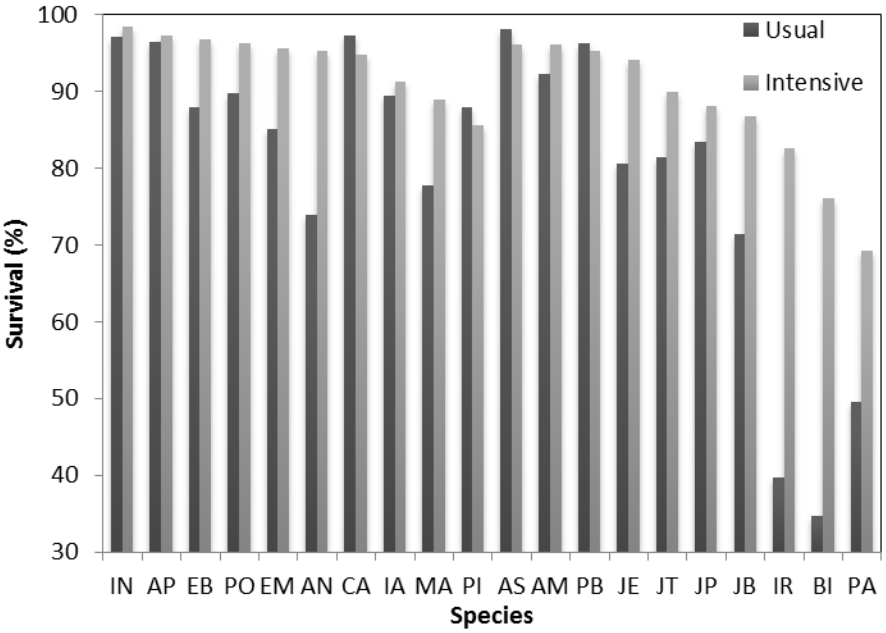

Figure 8. Survival for planted tree species for intensive and usual management 8 years after planting. Survival was generally higher with early successional species (IN-PI) than later successional species (AS-PA). AM = *Protium heptaphyllum*, AN = *Anadenanthera* sp, AP = *Schinus terebinthifolius,* AS = *Myracrodruon urundeuva,* BI = *Eschweilera ovate,* CA = *Gochnatia oligocephala,* EB = *Cecropia pachystachya,* EM = *Eriottheca macrophylla,* IA = *Tabebuia* sp, IN = *Inga thibaudiana,* IR = *Tabebuia* sp, JB = *Cariniana estrellensis,* JE = *Genipa Americana,* JP = *Dialium guianense,* JT = *Hymenaea courbaril,* MA = *Micrandra elata,* PA = *Simarouba amara,* PB = *Caesalpinia echinata,* PI = *Xylopia frutescens,* PO = *Tapirira guianensis.*

Table 6. Survival for pioneer and non-pioneer species in different treatments 8 years after planting. Δ % = change after intensive management, U = usual management, X = intensive management, A = 50:50 pioneers vs. non-pioneers, B= 67:33 pioneers vs. non-pioneers.

Composition	Stand density	Successional status	Survival % U	Survival % X	Δ %
A (50:50)	3333 trees ha^{-1}	Non-pioneer	68.8	83.7	22
		Pioneer	89.7	92.2	3
	1667 trees ha^{-1}	Non-pioneer	70.8	86.4	22
		Pioneer	85.5	87.3	2
B (67:33)	3333 trees ha^{-1}	Non-pioneer	73.7	82.9	12
		Pioneer	90.0	94.2	5
	1667 trees ha^{-1}	Non-pioneer	63.4	87.5	38
		Pioneer	80.8	95.1	18

4.2 *Leaf area index*

LAI was significantly higher in the denser stands (4.7 vs. 3.6, p = 0.011) (Table 7). LAI did not significantly respond to different species compositions even though the model B averaged slightly higher LAI. Statistically significant differences were not observed between intensive and usual management methods when not separating the two different stand densities (p = 0.467). However, intensive management increased LAI (40 %) with larger spacing (p = 0.050), showing the potential management x spacing interaction. Statistical analysis showed no significant interactions between species composition and spacing.

Figure **9** illustrates LAI values for different treatments and values and their standard deviations are presented in Table 8.

Table 7. Leaf area index (LAI) per factor 8 years after planting and their standard deviations of the means (sd), N = 4. Compositions: 50:50 pioneers vs. non-pioneers, 67:33 pioneers vs. non-pioneers.

Factors	Levels	LAI m^2/m^2	sd	p-values
Composition	50 : 50	4.02	0.76	0.094
	66 : 33	4.33	0.52	
Stand density	3333 trees ha^{-1}	4.75	0.63	0.011
	1667 trees ha^{-1}	3.60	0.54	
Management	Intensive	4.54	0.36	0.467
	Traditional	3.82	0.81	

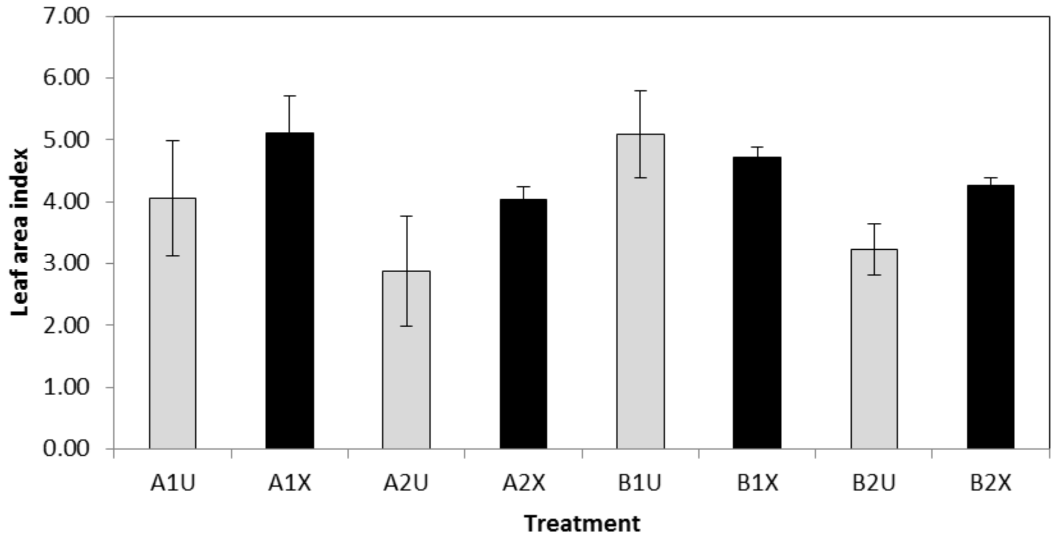

Figure 9. Leaf area index for different treatments 8 years after planting. Error bars represent standard deviations of the means (N = 4). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2= 3333 trees ha^{-1}, X = intensive management (black), U = usual management (grey).

Table 8. Leaf area index (LAI) for treatments 8 years after planting and their standard deviations of the means (sd). N = 4, Δ% = effect of the intensive management, U = usual management, X = intensive management, A = 50:50 pioneers vs. non-pioneers, B= 67:33 pioneers vs. non-pioneers.

Composition	Stand density	LAI m^2/m^2 U	sd_U	LAI m^2/m^2 X	sd_X	Δ%
A (50:50)	3333 trees ha^{-1}	4.06	0.93	5.12	0.59	26
	1667 trees ha^{-1}	2.88	0.89	4.04	0.19	40
B (67:33)	3333 trees ha^{-1}	5.09	0.70	4.72	0.16	-7
	1667 trees ha^{-1}	3.23	0.41	4.27	0.11	32

4.3 *Stemwood and bark densities*

Wood densities varied from 313 to 802 kgm^{-3} (Table 9) and is in line with the values obtained by Ferez (2011) in the Southern site. In general, fast growing species (EB, EM, PA) had lower wood density than slow growing species (IR, MA, AN). Wood density did not vary remarkably between successional groups (t-test: pioneer vs. non-pioneer, p=0.203).

Bark density was on average only 61 % of the wood density and varied from 190 to 440 kg m^{-3}. Figure 10 illustrates that there is no obvious trend between bark and wood densities (i.e. bark density does not increase when wood density increases) and those two variables should be calculated separately to obtain accurate estimates.

Table 9. Wood and bark densities at DBH for the 20 planted native species used in the North Atlantic Forest restoration study 8 years after planting.

Scientific name	Family	Abbr.	D_{wood}	sd_{wood}	D_{bark}	sd_{bark}	D_{bark}/D_{wood}	Successional status
Eriottheca macrophylla	Malvaceae	EM	313	29	269	16	0.86	Early successional
Cecropia pachystachya	Urticaceae	EB	341	11	344	5	1.01	Pioneer
Simarouba amara	Simaroubaceae	PA	352	7	309	14	0.88	Late successional
Tapirira guianensis	Anacardiaceae	PO	408	19	347	6	0.85	Early successional
Cariniana estrellensis	Lecythidaceae	JB	481	5	263	18	0.55	Late successional
Xylopia frutescens	Annonaceae	PI	538	4	380	12	0.71	Early successional
Hymenaea courbaril	Fabaceae	JT	549	13	408	30	0.74	Climax
Dialium guianense	Fabaceae	JP	553	17	331	11	0.60	Late successional
Schinus terebinthifolius	Anacardiaceae	AP	564	12	336	9	0.60	Pioneer
Inga thibaudiana	Fabaceae	IN	573	38	440	17	0.77	Early successional
Genipa americana	Rubiaceae	JE	612	17	305	19	0.50	Climax
Eschweilera ovata	Lecythidaceae	BI	627	7	290	9	0.46	Late successional
Protium heptaphyllum	Burseraceae	AM	637	18	429	30	0.67	Late successional
Tabebuia sp	Bignoniaceae	IA	648	18	190	9	0.29	Early successional
Tabebuia sp	Bignoniaceae	IR	707	15	210	10	0.30	Late successional
Anadenanthera sp	Fabaceae	AN	719	5	431	13	0.60	Early successional
Micrandra elata	Euphorbiaceae	MA	739	10	349	22	0.47	Early successional
Myracrodruon urundeuva	Anacardiaceae	AS	754	11	374	29	0.50	Late successional
Caesalpinia echinata	Fabaceae	PB	798	10	361	17	0.45	Climax
Gochnatia oligocephala	Asteraceae	CA	802	7	392	34	0.49	Pioneer

Abbr.=abbreviation, D_{wood}= wood density kgm^{-3}, D_{bark} = bark density kgm^{-3}, sd = standard deviation of the mean

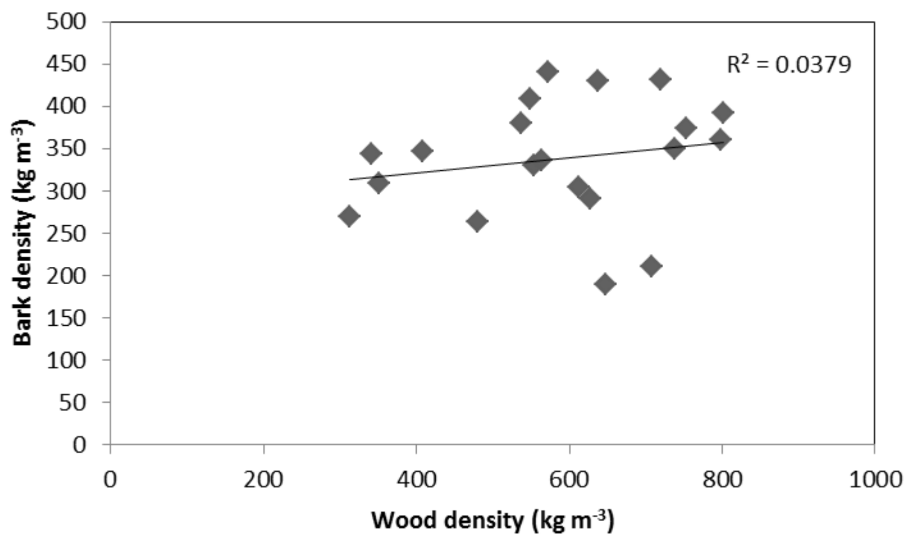

Figure 10. Relationship between wood density and bark density for 20 tropical tree species.

4.4 *Stemwood biomass stocks*

Intensive management methods increased stemwood biomass stocks on average 170 % compared to usual methods 8 years from planting. The denser spacing increased stemwood biomass by 43 %, but the stemwood biomass of different species compositions were approximately the same. Stemwood biomass stocks per factor and p-values are presented in Table 10. Stemwood biomass for all treatments are illustrated in Figure 11 and their average values with standard deviations are presented in Table 11. Intensive management and spacing were the most significant factors explaining differences in the stemwood biomass stocks and accordingly treatments B1X (65.9 Mg ha^{-1}) and A1X (63.8 Mg ha^{-1}) had the highest stemwood biomass followed by A2X (52.5 Mg ha^{-1}) and B2X (51.6 Mg ha^{-1}). No statistically significant interactions were found when comparing the stemwood biomass stocks. Contrary to the Southern trial, there was no statistically significant interactions between management methods and spacing (p = 0.72).

Table 10. Biomass stocks in stemwood per factor (Mg ha^{-1}) and mean annual increments for Northeastern Atlantic Forest restoration study 8 years after planting. Sd = standard deviation of the means, N = 4. , Compositions: 50:50 pioneers vs. non-pioneers, 67:33 pioneers vs. non-pioneers.

Factors	Levels	Mg ha^{-1}	sd	MAI	p-values
Composition	50 : 50	38.4	11.3	4.8	0.299
	67 : 33	41.8	10.7	5.2	
Stand density	3333 trees ha^{-1}	47.0	10.9	5.9	< 0.001
	1667 trees ha^{-1}	33.1	10.0	4.1	
Management	Intensive	58.4	5.1	7.3	< 0.001
	Traditional	21.8	6.2	2.7	

MAI = Mean annual increment Mg ha^{-1} yr^{-1}

Table 11. Biomass stocks in stemwood averages per treatment (Mgha^{-1}) 8 years after planting for Northeastern Atlantic Forest restoration study. U = usual management, X = intensive management, A = 50:50 pioneers vs. non-pioneers, B= 67:33 pioneers vs. non-pioneers, sd = standard deviation of the means (N = 4). Δ% = effect of intensive management.

Composition	Stand density	U (Mgha^{-1})	sd_U	X (Mgha^{-1})	sd_X	Δ%
A (50:50)	3333 trees ha^{-1}	22.79	6.36	63.75	5.91	180
	1667 trees ha^{-1}	14.67	3.22	52.45	0.77	258
B (67:33)	3333 trees ha^{-1}	35.76	5.38	65.86	5.96	84
	1667 trees ha^{-1}	13.82	2.53	51.61	1.24	273

Control (T): 8.5 Mgha^{-1}, sd 0.7

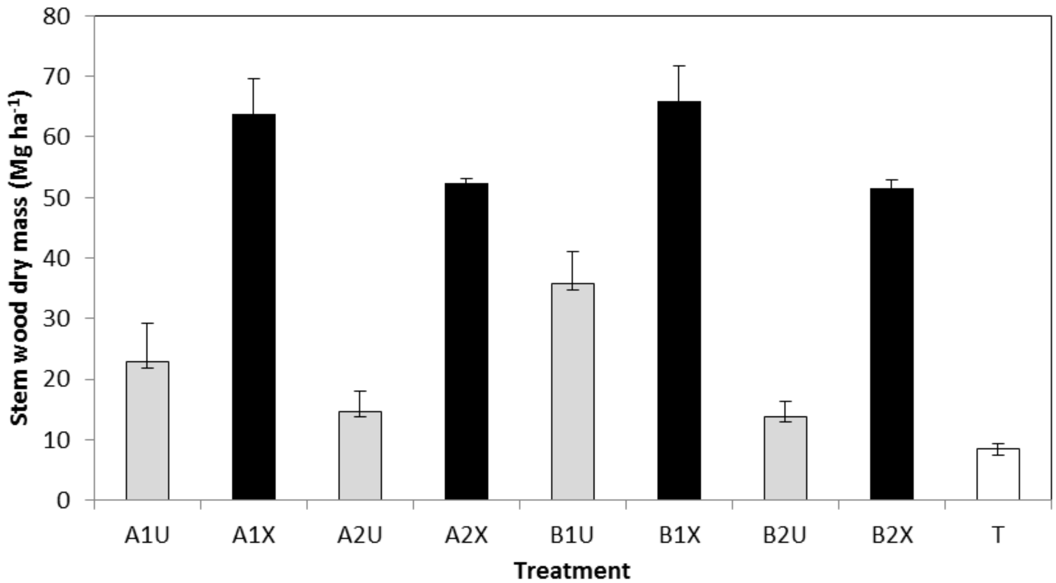

Figure 11. Stemwood biomass stocks per treatment 8 years after planting. Intensive management plots (X) had the biggest stemwood biomass stocks. The spacing was the second most significant factor to explain the differences in stemwood biomass stocks. Error bars represent standard deviations of the means (N = 4). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management (black), U = usual management (grey), T = baseline (white).

4.4.1 Species response to silviculture

Almost all tree species had clear positive responses to the intensive management method ($p < 0.001$). Relative increases in stemwood biomass averages with intensive management were highest with slow growing species like *Eschweilera ovata* (BI, 1095 % increase) and *Genipa americana* (JE, 794 % increase) (Table 12). Only one tree species, *Gochnatia oligocephala* (CA), did not show a clear positive response to the intensive management (p-value = 0.823). Early successional species generally had higher stemwood biomass averages than later successional species per tree in both usual (12.9 vs. 4.8 kg) and intensive management (32.9 vs.14.0 kg) methods. However, non-pioneers had a 40 % higher response to the intensive management methods (Table 12). A legume species *Inga thibaudiana* (IN) had the highest absolute increase in stemwood biomass per tree (64.19 kg) with intensive management compared to usual management. Figure 12 illustrates the effect of different management methods for each plated tree species.

Table 12. Stemwood biomass averages 8 years after planting, standard deviations of the means (sd) and average change (% and kg) for different tree species after intensive management methods. Pioneers: pioneer and early successional species together, Non-pioneers: Late successional and climax species together.

Scientific name	Abbr.	Traditional (kg)	Intensive (kg)	$sd_{traditional}$	$sd_{intensive}$	Δ%	Δkg	Successional status
Gochnatia oligocephala	CA	10.1	10.4	4.3	4.3	2.4	0.2	Pioneer
Schinus terebinthifolius	AP	7.7	26.1	6.1	6.1	237.9	18.4	Pioneer
Cecropia pachystachya	EB	19.5	72.9	18.4	18.4	273.7	53.4	Pioneer
Anadenanthera sp	AN	1.2	7.6	2.7	2.7	535.4	6.4	Early successional
Micrandra elata	MA	3.6	8.8	2.5	2.5	144.0	5.2	Early successional
Tabebuia sp	IA	3.2	11.0	3.0	3.0	245.3	7.8	Early successional
Xylopia frutescens	PI	7.5	16.7	4.9	4.9	123.6	9.2	Early successional
Eriottheca macrophylla	EM	6.5	17.1	4.5	4.5	162.0	10.6	Early successional
Tapirira guianensis	PO	10.4	35.3	11.5	11.5	240.0	24.9	Early successional
Inga thibaudiana	IN	59.2	123.4	29.4	29.4	108.5	64.2	Early successional
Eschweilera ovata	BI	0.0	0.6	0.7	0.7	1094.6	0.5	Late successional
Tabebuia sp	IR	0.3	1.5	0.5	0.5	390.6	1.2	Late successional
Protium heptaphyllum	AM	1.3	4.7	1.5	1.5	257.7	3.4	Late successional
Cariniana estrellensis	JB	0.9	6.5	2.8	2.8	589.4	5.6	Late successional
Dialium guianense	JP	11.9	25.5	8.0	8.0	113.9	13.6	Late successional
Myracrodruon urundeuva	AS	9.2	32.5	7.8	7.8	253.7	23.3	Late successional
Simarouba amara	PA	20.5	48.8	19.2	19.2	137.6	28.3	Late successional
Hymenaea courbaril	JT	0.6	3.3	2.3	2.3	429.5	2.7	Climax
Genipa americana	JE	0.8	6.9	2.5	2.5	794.3	6.1	Climax
Caesalpinia echinata	PB	2.0	10.0	3.2	3.2	395.0	8.0	Climax
Pioneers		12.9	32.9	8.7	8.7	155.4	20.0	
Non-pioneers		4.8	14.0	4.8	4.8	194.3	9.3	

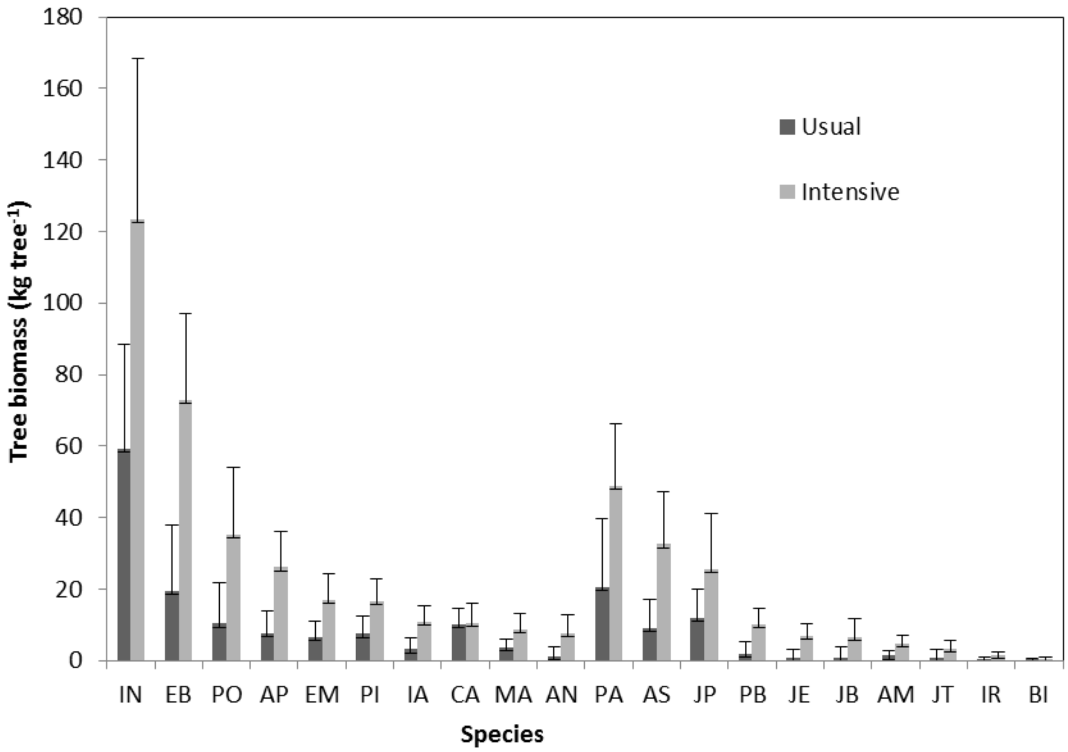

Figure 12. Effects of management strategy on different species 8 years after planting. Species presented from left to right from pioneers (IN – AN) to non-pioneers (PA – BI). Missing and dead trees are excluded (survival of the planted trees is examined in chapter 4.1. Error bars represent standard deviations of the means calculated from all trees per species under different management. AM = *Protium heptaphyllum*, AN = *Anadenanthera* sp, AP = *Schinus terebinthifolius,* AS = *Myracrodruon urundeuva,* BI = *Eschweilera ovate,* CA = *Gochnatia oligocephala,* EB = *Cecropia pachystachya*, EM = *Eriottheca macrophylla*, IA = *Tabebuia* sp, IN = *Inga thibaudiana,* IR = *Tabebuia* sp, JB = *Cariniana estrellensis,* JE = *Genipa Americana*, JP = *Dialium guianense,* JT = *Hymenaea courbaril,* MA = *Micrandra elata,* PA = *Simarouba amara*, PB = *Caesalpinia echinata*, PI = *Xylopia frutescens*, PO = *Tapirira guianensis*.

4.4.2 Pioneers vs. non-pioneers responses

Stemwood biomass proportion of non-pioneer species was highest with treatment A2X (41 %) and lowest with B2U (8 %) (Table 13). Intensive management increased the proportions of non-pioneer species biomass when combined with larger spacing (Table 13).

Table 13. Stemwood biomass and proportions of total plot biomass averages for pioneer and non-pioneer species in different treatments 8 years after planting. U = Usual management, X = Intensive management, A = 50:50 pioneers vs. non-pioneers, B= 67:33 pioneers vs. non-pioneers, sd = standard deviation of the means, N = 4. % (U) stemwood biomass proportion in usual management, % (X) stemwood biomass proportion in intensive management, Δ% effect of intensive management on biomass.

Composition	Stand density	Successional status	U ($Mgha^{-1}$)	sd_U	X ($Mgha^{-1}$)	sd_X	% (U)	% (X)	Δ%
A (50:50)	3333 trees ha^{-1}	Non-pioneer	5.45	1.99	15.70	1.54	32.41	32.67	188
		Pioneer	16.82	4.67	48.06	4.82	67.59	67.33	186
	1667 trees ha^{-1}	Non-pioneer	2.96	0.70	15.12	1.80	28.21	**40.54**	410
		Pioneer	10.50	2.43	37.30	1.31	71.79	59.46	255
B (67:33)	3333 trees ha^{-1}	Non-pioneer	4.73	1.12	10.03	2.11	15.28	17.96	112
		Pioneer	30.93	4.48	55.83	4.51	84.72	82.04	80
	1667 trees ha^{-1}	Non-pioneer	0.99	0.17	7.29	0.37	**8.12**	16.57	638
		Pioneer	12.17	2.18	44.03	0.95	91.88	83.43	262

4.4.3 Aboveground biomass; comparison of two equations

The biomass estimates according to the dry forest type equation (TAGB 2) that used only diameter, height and wood specific gravity (Chave et al. 2005) were very close to the calculations based on separate field measurements (TAGB 1) (stem wood biomass, bark and canopy biomass calculated separately) (Figure 13). In all treatments, the difference, when compared to the TAGB 2, was less than 6 % (Table 14). On average, the difference between the biomasses obtained from the two equations was less than 1 %. TAGB 2 gave higher estimates than TAGB 1 with usual management and smaller with intensive management. Figure 13 illustrates the similar results with the two different calculation methods.

Table 14. Total aboveground biomass (TAGB) per treatments with two different calculation methods 8 years after planting. TAGB 1 = stemwood, bark biomass and canopy are calculated separately and then summed. TAGB 2 = dry forest type equation presented by Chave (2005). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management, U = usual management, T = baseline. Sd = standard deviations of the means, N = 4.

	TAGB 1 $Mgha^{-1}$	sd_{TAGB1}	TAGB 2 $Mgha^{-1}$	sd_{TAGB2}	Difference %
A1U	24.86	7.08	25.27	6.42	1.66
A1X	69.28	6.62	67.20	5.48	-3.01
A2U	15.99	3.48	16.62	3.18	3.94
A2X	57.36	0.67	54.25	0.91	-5.43
B1U	38.65	5.59	38.51	4.72	-0.36
B1X	71.06	6.27	68.44	4.97	-3.69
B2U	14.95	2.80	15.33	2.76	2.57
B2X	55.94	1.36	53.29	1.41	-4.74
T	9.46	0.70	9.56	1.04	1.13

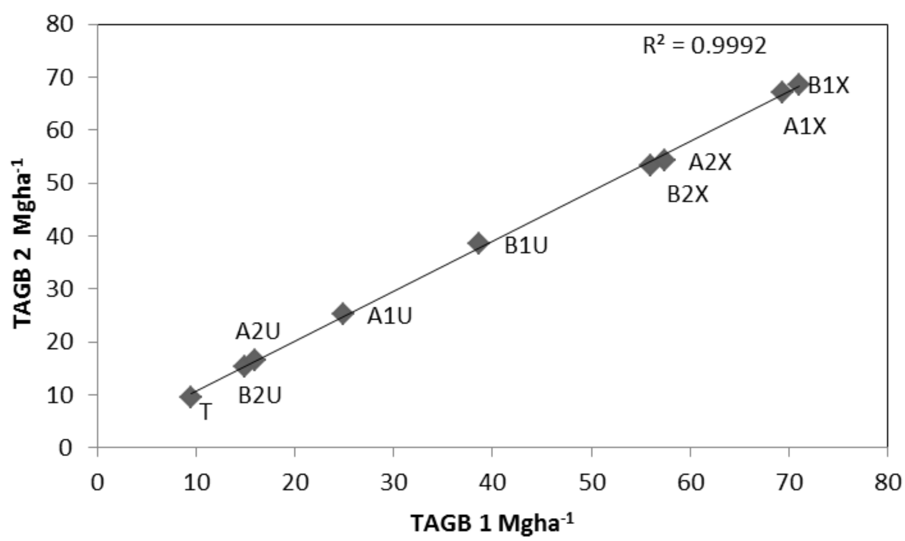

Figure 13. Total aboveground biomass (TAGB) per treatments with two different calculation methods and their relationship. TAGB 1 = stemwood, bark biomass and canopy are calculated separately and then summed. TAGB 2 = dry forest type equation presented by Chave (2005). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management, U = usual management, T = baseline.

4.4.4 Productivity and LAI

Increase in LAI predicted increase also in stemwood biomass productivity ($r^2 = 0.57$) (**Figure 14**). Productivity was calculated as a mean annual increment during the 8 years. The intensive management had not only higher LAI than the usual management, but also a higher light use efficiency (**Figure 15**) which means that trees can grow more with the same LAI. Using the average productivity of the 8-year period as the growth value, the estimates for light use efficiency are: 0.74 Mg ha^{-1} yr^{-1} per unit of LAI for the usual management, and 1.61 Mg ha^{-1} yr^{-1} per unit of LAI for the intensive management (118% increase).

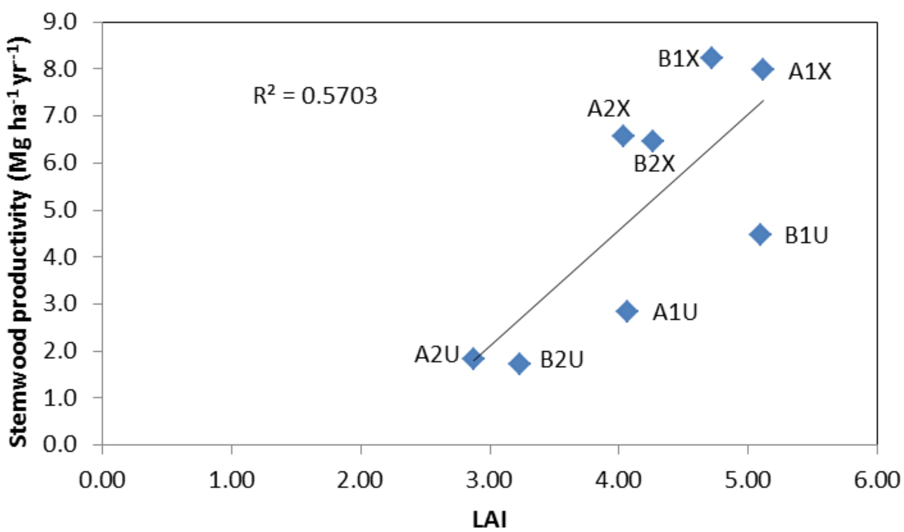

Figure 14. Relationship between stemwood productivity and leaf area index. Productivity presents the mean annual increment during the 8-year period. $R^2 = 0.57$. A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2= 3333 trees ha^{-1}, X = intensive management, U = usual management.

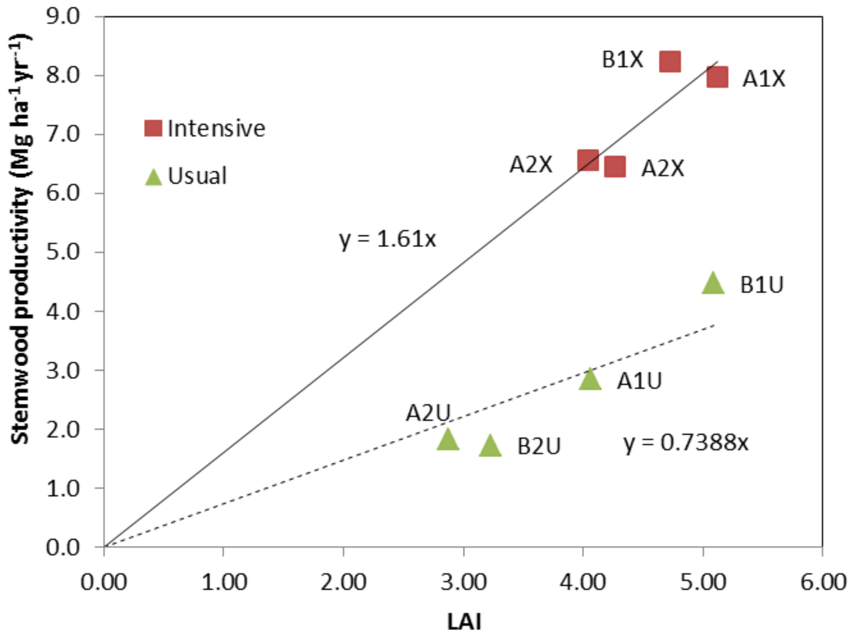

Figure 15. Light use efficiency estimates for intensive and usual management methods. Productivity presents the mean annual increment during the 8-year period. Higher values in productivity with intensive management than expected based on LAI indicate increase in light use efficiency. A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2= 3333 trees ha^{-1}, X = intensive management, U = usual management.

4.5 *Aboveground forest carbon*

4.5.1 Stemwood carbon

Mean annual stemwood carbon increment varied within treatments between 0.86 (B2U) and 4.12 MgC ha^{-1} yr^{-1} (B1X). Stemwood carbon mean annual increment with the intensive management was on average 7 fold bigger than baseline during 8 years (3.7 vs 0.5 MgC ha^{-1} yr^{-1}) (Table 15).

Table 15. Stemwood carbon accumulated during 8 years and mean annual increment (MAI = MgC ha^{-1} yr^{-1}). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2= 3333 trees ha^{-1}, X = intensive management, U = usual management, T = baseline, sd = standard deviation of the means (N = 4).

Treatment	MgC ha^{-1}	sd	MAI
A1U	11.39	3.18	1.42
A1X	31.88	2.96	3.98
A2U	7.33	1.61	0.92
A2X	26.23	0.38	3.28
B1U	17.88	2.69	2.23
B1X	32.93	2.98	4.12
B2U	6.91	1.27	0.86
B2X	25.81	0.62	3.23
T	4.26	0.35	0.53

4.5.2 Forest floor and understory

Understory carbon stocks were higher in the usually managed plots (0.37 Mg C ha^{-1} vs. 0.08 Mg C ha^{-1}, p = 0.002, unpaired t-test) while the forest floor carbon was relatively constant across the different treatments (1.19 Mg C ha^{-1} for intensive management and 1.34 Mg C ha^{-1} for usual management, unpaired t-test p = 0.142). Averages for all treatments are presented in Table 16 and illustrated in Figure 16.

Table 16. Forest floor and understory carbon stocks 8 years after planting with standard deviations (sd) of the means, N = 4. A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management, U = usual management, T = baseline.

Treatment	Mg C/ha US	sd_{US}	Mg C/ha FF	sd_{FF}
A1U	0.31	0.11	1.66	0.24
A1X	0.03	0.02	1.12	0.08
A2U	0.57	0.20	1.12	0.08
A2X	0.16	0.06	1.09	0.06
B1U	0.12	0.04	1.54	0.06
B1X	0.05	0.04	1.35	0.12
B2U	0.46	0.14	1.05	0.11
B2X	0.08	0.03	1.19	0.08
T	0.23	0.04	0.90	0.11

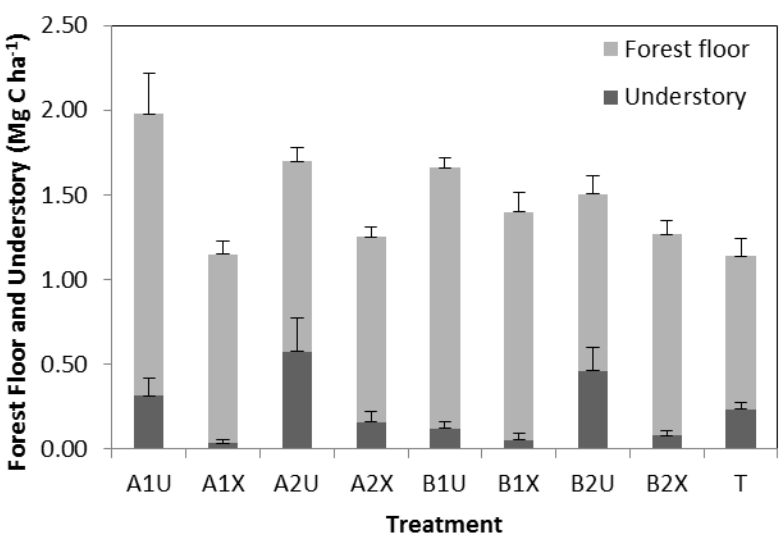

Figure 16. Forest floor and understory biomasses for treatments 8 years after planting. Error bars represent standard deviations of the means (N = 4). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management, U = usual management, T = baseline.

4.5.3 Aboveground carbon

Intensive management methods increased total aboveground carbon stocks 144 % (13.5 vs. 33.0 Mg C ha^{-1}) (Table 17). Carbon stocks on the control plots were less than 6 Mg C ha^{-1} on average. Values for different treatments are presented in Table 18 and Figure 17 illustrates carbon stocks per treatment. Pictures of the intensive and usual managements are shown in Figure 18.

Table 17. Forest carbon stocks per factor 8 years after planting and p-values from GLM test for Atlantic forest restoration trial. Sd = standard deviations for the means, N = 4. Compositions: 50:50 pioneers vs. non-pioneers, 67:33 pioneers vs. non-pioneers.

Factors	Levels	Mg C ha^{-1}	sd	p-values
Composition	50 : 50	22.5	6.0	0.374
	67 : 33	24.0	5.7	
Stand density	3333 trees ha^{-1}	27.0	5.8	< 0.001
	1667 trees ha^{-1}	19.5	5.3	
Management	Intensive	33.0	2.7	< 0.001
	Traditional	13.5	3.4	
Baseline		5.9	0.5	

Table 18. Carbon stocks per treatment 8 years after planting. U = Usual management, X = Intensive management, A = 50:50 pioneers vs. non-pioneers, B= 67:33 pioneers vs. non-pioneers, sd = standard deviations of the means (N = 4), Δ% = effect of intensive management.

Composition	Stand density	U ($MgCha^{-1}$)	sd_U	X ($MgCha^{-1}$)	sd_X	Δ%
A (50:50)	3333 trees ha^{-1}	14.41	3.66	35.79	3.37	148
	1667 trees ha^{-1}	9.69	1.61	29.93	0.35	209
B (67:33)	3333 trees ha^{-1}	20.99	2.78	36.93	3.20	76
	1667 trees ha^{-1}	8.98	1.62	29.24	0.64	226

Baseline (T): 5.9 $MgCha^{-1}$, sd 0.5

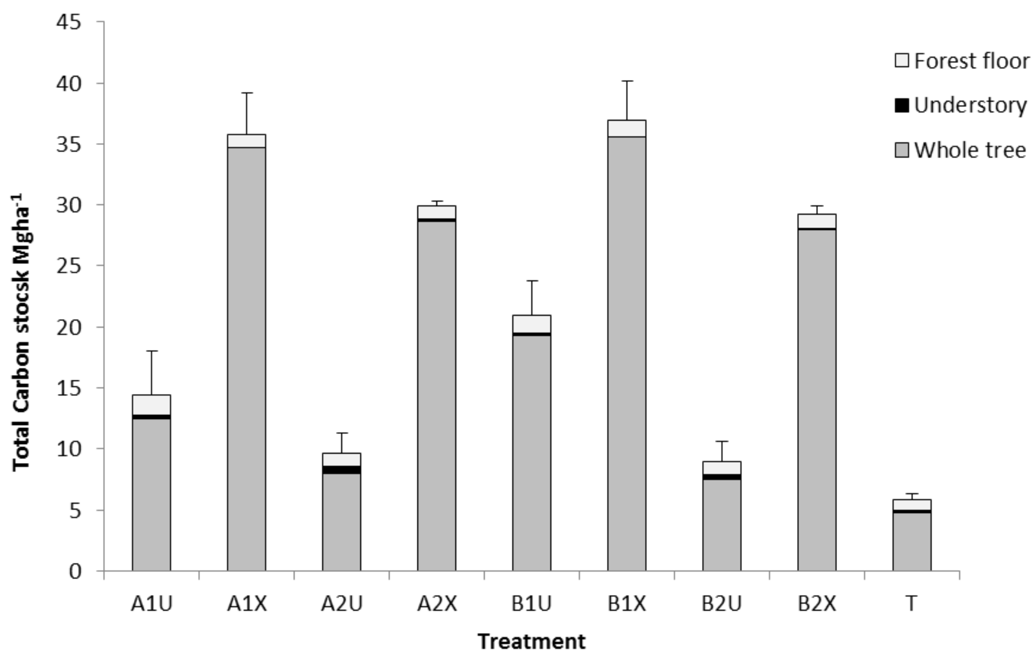

Figure 17. Carbon stocks per treatment in ($Mgha^{-1}$) 8 years after planting. Error bars represent standard deviations for total stemwood biomass stocks (N = 4). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management, U = usual management, T = baseline.

Figure 18. The top picture is from intensively managed plot (A2X). The bottom picture is from a plot where usual management methods were applied (B1U). The intensively managed plot had a plantation like look with no understory. More pictures of different treatments can be seen in Appendix.

4.6 *Regeneration*

To get a better picture about the site development, the regenerated trees were also measured and if possible identified. Many of the species remained unidentified because of the huge diversity of tree species. In the control plots, several species that were not among the 20 species initially planted were present. Out of the 20 planted species, five are already regenerating individuals. Pioneer and early successional species *Cecropia pachystachya* (EB) and *Inga thibaudiana* (IN) had the highest number of regenerated trees (Table 19).

Table 19. Number of regenerated trees of the tree species planted on the trial 8 years after planting.

Scientific name	Family	Abbreviation	Successional status	Regenerated trees
Cecropia pachystachya	Urticaceae	EB	Pioneer	33
Inga thibaudiana	Fabaceae	IN	Early successional	21
Schinus terebinthifolius	Anacardiaceae	AP	Pioneer	2
Gochnatia oligocephala	Asteraceae	CA	Pioneer	2
Tapirira guianensis	Anacardiaceae	PO	Early successional	1

4.6.1 *Tree regeneration per treatment*

The tree regeneration was minimal in intensively managed plots (averaging 0.08 $Mgha^{-1}$) but higher in usually managed plots (averaging 0.68 $Mgha^{-1}$) (Table 20). Most of the intensively managed plots had no regeneration at all (see pictures in Figure 18). Only three intensively managed plots out of 16 had any regenerated individuals. The highest amount of biomass from regenerated trees were from usually managed plots with lower stand density (B2U and A2U) (Figure 19).

Table 20. Tree regeneration (Mg ha^{-1}) in different treatments 8 years after planting. (Average of all repetitions). A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management, U = usual management.

Treatment	Mg ha^{-1}
A2U	1.30
B2U	0.72
A1U	0.59
B2X	0.29
B1U	0.11
A2X	0.03
A1X	0.00
B1X	0.00

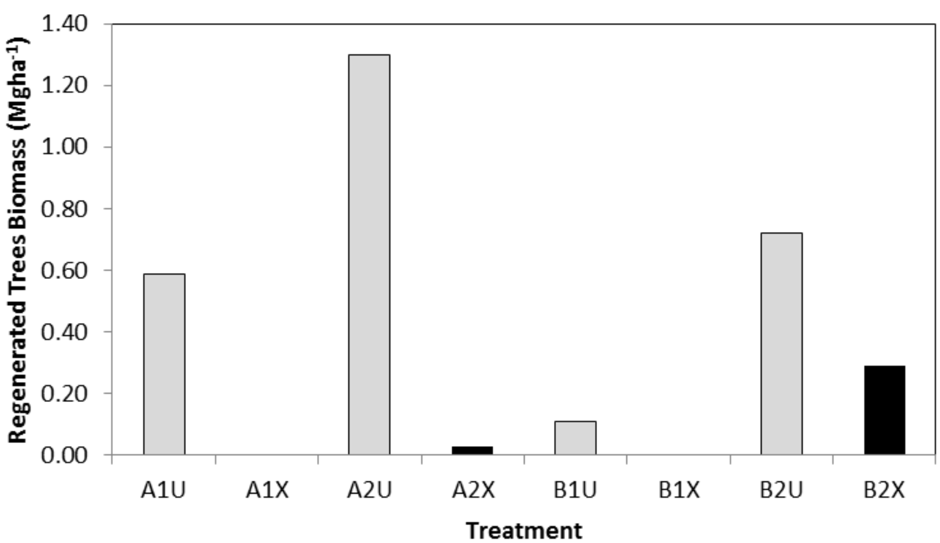

Figure 19. Dry mass Mgha^{-1} of the regenerated tree species per treatment 8 years after planting. A = 50:50 pioneer : non-pioneer species, B = 67:33 pioneer : non-pioneer species, 1 = 1667 tees ha^{-1}, 2 = 3333 trees ha^{-1}, X = intensive management (black), U = usual management (grey).

4.6.2 Regenerated native tree species

In addition to the tree species planted, 27 regenerated species were identified from the study site (Table 21). Most of them were native tree species found from surrounding Atlantic forest fragments, except Caribbean pine (*Pinus caribeae* var *hondurensis*), which used to be planted on the site before the restoration trial was established.

Table 21. Identified regenerated tree species and the number of individuals in the trial with average and total drymass per species. Dry mass was calculated based on dry forest type equation for total aboveground biomass (Chave et al. 2005)**.**

Scientific name	Family	No. of trees	kg tree^{-1}	kg ha^{-1}
Pinus caribeae	Pinaceaea	10	42.37	423.68
Byrsonina sp	Malpighiaceae	12	11.10	66.61
Zanthoxylum rhoifolium	Rutaceae	16	3.05	48.83
Dinizias sp	Mimosaceae	6	14.89	44.67
Vismia ferruginea Kunth	Hypericaceae	19	4.36	41.44
Peltogyne cf. subsessilis	Caesalpiniaceae	5	7.44	37.20
Ficus insipida	Moraceae	5	59.38	37.11
Bowdichia spp	Fabaceae	4	27.30	36.40
Manilkara bidentata	Sapotaceae	5	6.08	30.42
Manihot esculenta	Euphorbiaceae	14	10.62	21.24
Anadenanthera sp	Fabaceae	1	6.45	6.45
Mauritia flexuosa	Arecaceae	5	5.92	5.92
Erythrina sp.	Fabaceae	1	5.27	5.27
Luehea divaricata Mart.	Tiliaceae	1	10.73	4.37
Albizia hasslerii	Mimosaceae	1	10.57	3.52
Machaerium hirtum	Fabaceae	2	6.60	2.64
Tabebuia impetiginosa	Bignoniaceae	12	2.56	1.62
Celtis iguanaea	Cannabaceae	2	2.39	1.19
Annona sp	Annonaceae	3	6.60	1.10
Pogonophora schomburgkiana	Euphorbiaceae	1	0.84	0.84
Quinna sp	Rubiaceae	2	1.34	0.67
Psidium cattleianum	Myrtaceae	6	1.62	0.61
Tovomita mangle	Clusiaceae	2	6.26	0.54
Myrcia multiflora	Myrtaceae	1	1.07	0.53
Tibouchina sp	Melastomataceae	3	0.83	0.50
Dicksonia sellowiana	Dicksoniaceae	1	5.98	0.27
Patagonula americana	Boraginaceae	1	0.56	0.19

5. Discussion

5.1 *Intensive management increased growth on the Northern site*

Intensive management increased the survival, LAI, stemwood biomass and carbon accumulated during 8 years on the Northern site of the Atlantic forest restoration project with 20 local native species. Across treatment, the stemwood biomass with the intensive management was almost 3- fold larger than under usual management showing that the site severely suffered from soil physical and fertility limitations plus weed competition that was a constraint on growth.

Annual productivity averaged 2.7 Mg ha^{-1} yr^{-1} in usual and 7.3 Mg ha^{-1} yr^{-1} in the intensive management during the first 8 years. This compares well to the development in the productivity of the *Eucalyptus* plantations in Brazil; *Eucalyptus* productivity is nowadays 18 Mg ha^{-1} yr^{-1} and was 8 Mg ha^{-1} yr^{-1} in the 1960's (Stape et al. 2001, Gonçalves, Jose Leonardo de Moraes et al. 2004). The results of this study were in line with the results of the Southern site presented by Ferez (2011) and Campoe et al. (2010) and show, that unlike previous claims, the productivity of native species can benefit from the intensive management methods just as much as commercial *Eucalyptus* plantations, at least regarding the initial growth (Carpenter et al. 2004).

These results showed that the site has a high level of environmental stresses that constrain tree root and aboveground growth. The intensive management effects on soil physical conditions, fertility and suppression of competing vegetation was essential to allow the species to express their potential growth fully. This combined effect of site preparation, fertilization and competing vegetation control is increasing the availability of the three required natural resources for tree growth: water, nutrients and light. In naturally high-density soil sites, *Eucalyptus* production increases dramatically as a function of site preparation, water supply and competing vegetation control (Gonçalves et al. 2004, (Stape et al. 2004). In addition, previous studies have shown that fertilization and therefore improved nutrient availability can improve water use efficiency and allow more allocation to stemwood biomass in trees (Ewers et al. 1999, Albaugh et al. 2004, Stape et al. 2008).

On this tropical system, it is relevant to mention that control of leaf cutter ants (*Atta* spp) is crucial for establishment and early growth, and on this site, the ant management was kept under control up to year 6 for all treatments. From now on, the differential behavior of the ants, which are already present at the site, will provide more information if they have any preferential feeding under intensive and usual management.

5.1.1 Leaf area index and light use efficiency

Observed leaf area index values in this study were also higher with intensive management, as reported also from the Southern site (Campoe et al. 2010). Since LAI correlated well with stemwood growth at the Southern site ($r^2 = 0.93$), LAI was assumed to be a good variable to estimate the stemwood growth rates at the Northern site as well (Campoe et al. 2010). Indeed, stemwood productivity increased with higher LAI values ($r^2 = 0.57$). Higher LAI indicates denser canopies and higher photosynthetic capacity that often lead increase in productivity (Waring 1983b, Olivas et al. 2013). The weaker relation between growth and LAI at the Northern site when compared with the Southern site is most likely because in this study the annual productivity is calculated as an average of 8 years, while in the Southern site the current annual growth was used.

Intensively managed plots had higher productivity with the same LAI values than the plots under usual management, indicating an increase in light use efficiency (LUE) as observed also on the Southern site (Campoe et al. 2010). Higher light use efficiency indicates in turn better use of resources. Higher LUE is due to a better nutrition and water status of the intensive management canopy, allowing large rates of photosynthesis per unit of LAI (e.g., more chlorophyll and stomata open longer).

Interestingly, under intensive management, significantly higher LAI values were observed only with lower stand density (36 % increase in LAI). The results suggest that LAI alone is not enough to explain the growth, and LUE will need to be further evaluated in future studies. For instance, the same LAI can provide distinct growth rates pending on the nutrient status of the leaves, like nitrogen content, and the water stress of the leaves. For instance, Campoe et al. (2010) noticed in the Southern site that closer spacing was water stressed during the dry season and presented a lower LUE. That same behavior is expected in the Northern site, which also suffers a pronounced drought period. Annual or bi-annual inventories are necessary to obtain accurate estimates of stemwood productivity and light use efficiency.

5.1.2 Intensive nuclei system

Both sites are located on typical Atlantic forest domain regions. Observing these initial positive and similar silvicultural practice effects on tree survival and growth (3-fold increases) forces us to reflect upon best strategies to accomplish forest restoration in Brazil, using the same amount of financial resources. Two choices exist: (i) current system - larger continuous areas with a lower per/area cost; or (ii) a nuclei system - an island of more intensive silvicultural practices, with higher per/area costs.

Planting 100 m^2 nuclei as recommended by Zahawi et al (2013) to every 300 m^2 and managing them intensively would still lead to higher carbon sequestration than uniform plantation with usual management. For example, an extensive plantation under usual management could sequester on average 0.9 MgC ha^{-1} yr^{-1}, and an intensive nuclei system with one third of the planted area 1.5 MgC ha^{-1} yr^{-1} in stemwood during the first 8 years. Comparing the productivity per unit of area restored and amount of money investment would be needed to evaluate the cost efficiency between these two strategies. Anyway, it seems likely that nuclei studies would be needed on larger landscapes to further evaluate their efficacy.

Additionally, it has been remarked that the intensively managed plantations do not create authentic looking rainforest (de Souza and Batista 2004, Rodrigues et al. 2009). Indeed, the appearance of the intensively managed plots in this trial, at this age, was plantation like and very far from the natural rainforest appearance. This however can be considered as a minor trade off compared to, for example, the benefits gained from better non-pioneers species survival and carbon sequestering. When it comes to achieving more natural looking forest, combining intensive management and the nuclei method would both reduce planting expenses and create more diverse and authentic looking restored forests (Corbin and Holl 2012, Zahawi et al. 2013).

5.1.3 Intensive management; Long-term research needed

The results from this study and from the Southern site show strong evidence that unlike previously claimed (Carpenter et al. 2004, Sampaio et al. 2007), intensive management of restored forests can cause dramatic improvements in the early restoration success. However, the results Carpenter et al. (2004) presented are not fully comparable with this study since they inter-planted only two to three tree species per plot. In addition, the positive outcomes from fertilization might only emerge with adequate weed control. Effective weed control has proven to be crucial in many studies to secure the initial restoration process on degraded lands (Florentine and Westbrooke 2004, de Souza and Batista 2004, Campoe et al. 2010). Exotic C_4 grasses are known to compete strongly for water and nutrients with the planted trees and increase fires events. Not eliminating those grasses leads to increased mortality and suppressed growth (Eyles et al. 2012). Without adequate weed control, fertilization might actually have negative feedback because of increased competition from weeds that benefit from the nutrients more than the trees.

The trial in this study might not be old enough to reach the point where the intensive management methods would not show additional increase in stemwood growth or slower forest recovery that Sampaio et al. (2007) suggested. However, in the Southern site, Luu (2012) reported that annual increases were actually lower in intensively managed plots 7 years after planting. He hypothesized that this could be size related competition since the tree sizes were already bigger on intensively managed plots. Nevertheless, in tropical system establishment, the fast canopy closure is essential to the short- and long-term restoration success, and that is clearly increased by the intensive management. In order to determine, if annual increments are decreasing with the intensive management more than with usual management, biomass inventories should be conducted regularly during the study period.

This study is a long term study where inventories are planned for years ahead so the development of the size related competition will be documented in the Northern site as well in the future (Stape et al. 2006).

Both Northern and Southern sites showed benefits of intensive management on the survival and initial stemwood growth. However, this leads to a question; which out of site preparation, fertilization or weed control was the most important factor? All of these have different costs and their individual significance needs to be determined for future restoration designs. A new study with a factorial design with 2 levels of site preparation x 2 levels of fertilization x 2 level of weed control could be proposed to better evaluate the main effects, interactions and cost efficiency of different silvicultural procedures for Atlantic forest restoration.

5.2 *Effects of the planting densities*

The denser stands were expected to have higher stemwood biomass, especially under the usual silvicultural practices, due to an early canopy closure that provides a better competing vegetation control. Higher density of trees reduces erosion and provides better shading, which prevents the competition for resources from exotic grasses (de Souza and Batista 2004, Campoe et al. 2010). This hypothesis got supporting evidence from the results. Under usual management, higher density is necessary to secure the development of closed canopy to reduce competition from highly competitive exotic C_4 grasses. This effect is still easily visualized in the plots at year 8 (Figure 18).

When examining only the lower stand density (1667 trees ha^{-1}), the stemwood biomass from intensive management was almost 4-fold larger than the usual management. Therefore, under the intensive management method, the lower stand density should be recommended to decrease inter-specific tree competition in the future and to reduce planting expenses.

The differences in stemwood biomass and carbon stocks between the two different stand densities were much smaller than differences between management methods. This indicates the level of stress that the tree species were under caused by soil physical and fertility attributes and weed competition. When these stresses were minimized or removed, the tropical species were able to reach their potential, and each tree was able to use more natural resources (more m^2 area), requiring fewer trees per area. If we extend this thinking, the intensive management methods to native species should be tested with even larger spacing, for example 3.0 m X 2.5 m (1333 trees ha^{-1}) or 3.0 m x 3.0 m (1111 trees ha^{-1}). This would be interesting to better understand tropical tree growth under lower stockings and high silviculture practices and how the competition among neighboring trees develops (Campoe et al. 2010, Luu 2012). However, different from commercial species, like *Eucalyptus* and Pine, native species enjoy no tree-improvement and present high levels of tree-by-tree variability for each single species, and that fact, *per se* will constrain larger spacing.

5.3 *Pioneers vs. non-pioneers*

Larger proportion of pioneers suppresses the growth of non-pioneers?

Earlier it was hypothesized that the 67:33 pioneer vs. non-pioneer ratio would have higher stemwood biomass due to the fast growth rates of the pioneers. This however was not observed in this trial at the age of 8 years. As Campoe et al. (2010) hypothesized, native tree species in plantation design might behave differently than when natural succession takes place. Pioneer tree species usually colonize the gaps in forests after disturbance, initial conditions being very different than those in the plantations. In addition, another reason for not observing differences in stemwood biomass might be that at the age of 8 years, the fast growing but short-lived pioneers are reaching their maturity and having slower growth rates.

The goal in reforestation is to establish self-sustaining forest with sufficient amounts of long-lived non-pioneers to maintain the closed canopy. As discussed in the literature review, a too large proportion of the pioneer species could have a negative effect on the long-term sustainability of the restoration site with their early mortality and by outcompeting non-pioneers (Kageyama et al. 2000, Rodrigues et al. 2009). Short-lived pioneer species might not maintain the closed canopy and thereby protect from the weed competition long enough for the suppressed non-pioneers to achieve sufficient size.

Since the higher proportion of pioneer species showed no evidence of increased stemwood biomass, a 50:50 ratio of pioneer to non-pioneer species should be recommended to increase biodiversity and provide a large enough proportion of long-lived non-pioneers. This recommendation of limiting the amount of pioneer species is in line with those presented by Kageyama et al. (2000) and Campoe et al. (2010). For long-term sustainability, this model is expected to be more favorable in order to attain large enough amounts of long living non-pioneers that would maintain the closed canopy. Limiting pioneer species might also reduce the stress from herbivores since later successional species allocate more resources for defenses against herbivory (Massad et al. 2011).

In fact, it would be appropriate to ask the question about using even lower proportions of pioneer vs. non-pioneer species, like for instance 40:60, 30:70, 25:75 or 20:80. Knowing the minimum requirement of pioneer species will have two important consequences: (i) a larger amount of non-pioneer long-living species (sustainability); (ii) More planting spots for more non-pioneer species per site (biodiversity).

The increased biodiversity could lead to higher productivity for the long term (de Siqueira 2002, Barbosa et al. 2003, Florentine and Westbrooke 2004). Considering that the Atlantic forest has more than 2,000 tree species (Myers et al. 2000, Silva et al. 2003), our sample of 20 species is still very limited and we do recommend the test of more species to silvicultural treatment. To determine the effects of biodiversity on the productivity under intensive management, a parallel trial was established on the Southern and Northern sites with 20, 60 and 120 tree species (Stape et al. 2006).

Non-pioneers benefitted more from intensive management

Both pioneer and non-pioneer species were expected to respond positively to more intensive management due to the minimization of environmental stresses. The results of this study support this expectation; 19 out of 20 tree species had clear positive response to intensive management regarding survival and stemwood growth. The results also indicate that the non-pioneers were almost outcompeted in the usually managed plots. Non-pioneers seemed to be more sensitive to the environmental constraints on this trial than were the pioneers. This might be because non-pioneers normally germinate in shade, under closed canopies and do not usually face that strong competition from grasses (Swaine and Whitmore 1988). Without intensive management methods, the slow growing non-pioneers in this trial suffered more from weed competition than pioneers and were more often suppressed and eliminated at the early stage. Intensive management also increased relative abundance of non-pioneers compared to usual management in both Northern and Southern sites by increasing the survival of non-pioneers proportionally more.

However, Luu (2012) reported that intensive management might actually increase the competitive pressure that pioneers cause, especially to non-legume non-pioneers, even more than usual management methods. Even though the results in this study would suggest the positive result when combining intensive management with lower stand density and equal ratios of successional groups, more time is needed to determine if the competition with more intensive management methods will actually cause an overall negative effect on the restoration process in the long run. This design, both in the South and Northern sites, was established to allow spatial dependent growth models to be developed and the exact position of each species, and neighbors, will need to be addressed across time, because completion and facilitation indices may change with stand development (Stape et al. 2006).

Southern and Northern sites showing similar patterns

Both study sites showed similar improvements in tree survival and biomass development with intensive management. For example, in the Southern site for the layout with 50:50 ratio of pioneers vs. non-pioneer and lower density (1667 tress ha^{-1}), the wood productivity during the first 6 years averaged 0.8 MgC ha^{-1} yr^{-1} with usual management and 2.7 MgC ha^{-1} yr^{-1} with intensive management. In the Northern site, wood growth averaged 0.9 MgC ha^{-1} yr^{-1} under usual management and 3.3 MgC ha^{-1} yr^{-1} with the intensive management during the first 8 years. This demonstrates the potential of intensive management methods across a broad geographical gradient on degraded land regarding the initial carbon sequestration potential (Campoe et al. 2010, Ferez 2011, Campoe et al. 2014).

5.4 *Natural regeneration*

Regenerated tree species in this trial have potentially three main origins: i) soil seed banks; ii) from those few narrow fragmented Atlantic forest areas close to the restoration trial; and iii) seeds from the restored plots themselves.

Five pioneer species out of the 20 planted species were already regeneration and 27 species not among those species that were planted were recorded. Those new species most likely come from the surrounding area and are carried by birds. Most of the regenerated species were native pioneer species, expect Caribbean pine, which is still present in adjacent plantations. To better assess the regeneration dynamics, assessment of wildlife factors should be initiated.

Plantations of fast growing pioneer species are expected to facilitate the recruitment of native species (Goosem and Tucker 1995, Parrotta et al. 1997, Otsamo 2000). However, in this trial, the natural regeneration, especially in the intensively managed plots, was very low, probably due to the young age of the trial and to strong shading. Intensive management might therefore at some extent delay natural regeneration as suggested by Sampaio et al. (2007). However, this does not yet indicate if it would diminish long-term sustainability.

Additionally, biomass in the control (baseline) site was very low compared to the plots where usual or intensive management was applied (8.5 control *vs* 21.8 and 58.4 Mg ha^{-1}, respectively). This supports previous research that soil seed banks have a diminishing role in restoration on heavily disturbed sites and active restoration procedures are required (Baider et al. 2001). However, determining those regenerated tree species that largely originate from soil seed banks provides valuable information about the species compositions and functions of the original Atlantic forest.

5.5 *Considerations for Atlantic forest reforestation*

Carbon stocks in the baseline plots (control) were on average only one sixth of those with the intensive management (5.9 vs. 33.0 Mg C ha^{-1}) 8 years after planting. Natural forest stemwood carbon varies in this Northern site area from 80 to 120 Mg C ha^{-1} (Stape et al. 2006). Therefore, as an index of efficacy of the treatments, we can speculate how many years it would take to reach such levels if the 8-year accretion values for stemwood C stay constant.

For that purpose, we selected the 50:50 proportions of pioneer : non-pioneer species in the lower spacing (1667 trees ha^{-1}), and the following time periods were estimated:

i) Natural regeneration: 150 to 230 years
ii) Usual management: 90 to 130 years.
iii) Intensive management: 24 to 37 years.

To further motivate reforestation projects, the potential use of these native tree species for commercial uses should be also addressed. For example, Brazilwood is one of the most valuable woods in the world, and could provide incomes from restoration projects.

5.6 *Limitations of the work*

It is expected that there are at least some measurement errors and typing errors while writing the measurements. Errors however are assumed to be randomly distributed and somewhat canceling each other out. Using non species-specific form factor (0.5) when calculating biomass might have provided slightly imprecise biomass estimates for different tree species. In addition, rough estimate for carbon (0.5) might have been an underestimate, at least for some species (Lamlom and Savidge 2003). These three issues mentioned above should not matter too much when comparing different treatments, but they might reduce the comparability between different tree species.

In addition, Northern and Southern site trials are not completely comparable because their intensive and usual management methods were different between the two sites. There are also some differences between the methods in how biomasses were determined between the two sites (allometric equations for each tree species vs. generic form factor 0.5).

6. Conclusions

The study of 8-year development of the restoration techniques for Atlantic forest ecosystem in Northeastern Brazil, using 20 local species and a factorial design with two levels of Management (usual and intensive), Density (1667 and 3333 trees ha^{-1}) and Species Composition (50:50 and 67:33 pioneer : non-pioneer species ratio) allow us to conclude the following:

- Management methods: The intensive management methods have some potential to increase restoration success by i) increasing biodiversity through increasing survival and growth of non-pioneer species, ii) speeding up canopy closure and growth rates between 3- and 4-fold when compared to usual practices;
- Density: The planting density depends on the silvicultural level. Better silvicultural practices allow lower density;
- Composition: A 50% proportion of non-pioneer species was shown to be as productive as the one with 33%, and should be pursued as the best option to improve long-run sustainability;
- Leaf area index and light use efficiency: intensive management increased the growth as a combined function of increasing LAI and LUE simultaneously;
- Carbon Sequestering: the intensive management method increased the carbon accumulated over 3-fold when compared to the usual management, and over 5-fold when compared with natural regeneration, making it a potential management option when seeking carbon offsets.
- Northern site and Southern site share many similar results, indicating that native tropical species can grow faster and speed up the restoration process when environmental limitations are adequately addressed;

- Overall, the results indicate that the trade-off of restoration area vs silviculture intensity strategies should be revised for Brazilian Atlantic forest restoration (extensive restoration versus intensive nuclei system).

Out of the four hypotheses, three were supported:

1) The intensive management practices will increase the survival, LAI, stemwood biomass and aboveground carbon stocks for an Atlantic Forest restoration in Northerneastern Brazil when compared with the usual practices due to the minimization of environmental stresses; ***Supported***

2) The denser stands will have higher stemwood biomass, mainly on the usual silvicultural practices, due to an early canopy closure and providing a better competing vegetation control; ***Supported***

3) The 67:33 pioneers : non-pioneers model will have a higher stemwood biomass due to the fast growth rates of the pioneers; ***Not supported***

4) Both pioneer and non-pioneer species will positively respond to more intensive management due to the minimizati-n of environmental stresses; ***Supported***

Based on the result of this study, it can be suggested that;

1) The model with lower proportions of pioneers (50%) , lower density (1667 trees ha^{-1}) with intensive management can be recommended to combine the potential benefits of increased biodiversity and resilience, higher initial growth, reduced planting expenses and competition between trees, when enough financial resources are available for restoration
2) The model with higher proportion of pioneers (67%), larger planting density (3333 trees ha^{-1}) is the recommend option when the usual management (U) is the only available option

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APPENDIX

Appendix

Pictures of the Atlantic forest restoration trial (July 2012).

Figure 20. Eucalyptus plantations (left) and the restoration trial (right)

Intensive (X) treatment:

Figure 21. Treatment B1X: Higher proportion of pioneer species (67:33 pioneers vs. non-pioneers) and higher planting density (3333 trees ha^{-1}) with intensive management.

Figure 22. Treatment A2X: 50:50 proportions of pioneer vs. non-pioneers and smaller planting density (1667 trees ha^{-1}) with intensive management.

Figure 23. Treatment B2X: Higher proportion of pioneer species (67:33 pioneers vs. non-pioneers) and smaller planting density (1667 trees ha^{-1}) with intensive management.

Figure 24. Treatment A1X: 50:50 proportions of pioneer vs. non-pioneers and higher planting density (1667 trees ha^{-1}) with intensive management.

Usual (U) treatment:

Figure 25. Treatment B1U: Higher proportion of pioneer species (67:33 pioneers vs. non-pioneers) and higher planting density (3333 trees ha^{-1}) with usual management.

Figure 26. Treatment A1U: 50:50 proportions of pioneer vs. non-pioneers and higher planting density (3333 trees ha^{-1}) with usual management.

Figure 27. Treatment B2U: Higher proportion of pioneer species (67:33 pioneers vs. non-pioneers) and smaller planting density (1667 trees ha^{-1}) with usual management.

Figure 28. Treatment A2U: 50:50 proportions of pioneer vs. non-pioneers and smaller planting density (1667 trees ha^{-1}) with usual management.

Control (T)

Figure 29. Control plot. The biggest trees on this picture are Caribbean pines, which used to be cultivated on the site before the establishment of the reforestation trial.

Figure 30. Control plot. Weed competition is high and prevents the development of long-lived non-pioneer tree species.