Universidade Federal de Uberlândia
Instituto de Biologia
Programa de Pós-Graduação em
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Efeitos de reservatórios artificiais sobre a composição florística e a estrutura de comunidades arbóreas nativas no Triângulo Mineiro

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(Orientador)

Uberlândia - MG

Fevereiro/2012

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Dedico...

Aos integrantes do Laboratório de Ecologia Vegetal: *Ivan, Sérgio, Ana, Jamir, Carol, Maca, Glein* e á minha família francana: *Pai, Mãe, Vi, Vandão e Valéria.*

Tiro o chapéu para...

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Introdução Geral

Barragens são construídas em rios por todo o planeta com diversas finalidades: irrigação, sedimentação, uso doméstico e, principalmente geração de energia elétrica (Baxter 1977; Kaygusuz 2004; Evans et al. 2009). Contudo, geram dois problemas básicos e graves para o ecossistema: à jusante pode causar a redução na vazão de água, devido à retenção das águas do rio, e a montante causa alagamento de uma extensa, literalmente afogando diversos elementos da paisagem (Nilsson & Bergreen 2000; Fearniside 2001). Para organismos sésseis, como as plantas, que sobrevivem ao impacto inicial e passam a se situar as margens da nova condição imposta pela barragem, são esperadas modificações, principalmente devido a completa mudança na disponibilidade hídrica do solo. Mesmo pequenas mudanças na disponibilidade de água afetam o estabelecimento e sobrevivência das espécies de plantas (Nilsson 1996; Munoz-Reinoso 2001; Stromberg 2001) e tem seus efeitos notados por toda a comunidade, uma vez que as plantas são a base produtora para os organismos terrestres (Loreau et al 2001).

Apesar de sua importância, a maioria dos estudos se concentra em ambientes frios, com foco para arbustos, espécies herbáceas e gramíneas (Nilsson et al. 1991; Toner & Keddy 1997; Dynesius et al. 2004). Todavia, os trabalhos sobre os efeitos das barragens artificiais sobre as comunidades arbóreas nos trópicos são incipientes. Tal fato é um paradoxo, uma vez que a maioria das barragens é construída justamente nos ambientes tropicais (Guo et al. 2007; Nilsson et al 2005), onde as árvores detêm a maior biomassa (Dixon 1994) e são o principal componente vegetal da paisagem. Dos poucos estudos existentes sobre esses efeitos, não há uma avaliação temporal das comunidades, mas apenas comparações via cronosequencias de comunidades afetadas por barragens (Nilsson et al 2002, Dynesius et al 2005, Janson et al 2000).

O presente estudo iniciou-se como um dos Planos de Controle Ambiental (PCAs) obrigatórios para o licenciamento das Usinas Hidrelétricas de Capim Branco I e II (atualmente UHEs Amador Aguiar I e II), no rio Araguari. A PCA consistiu no Monitoramento dos Impactos sobre a vegetação e foi implantado a partir do ano de 2004. Cinco áreas de florestas foram escolhidas para o monitoramento: quatro áreas de florestas estacionais, localizadas na futura margem dos reservatórios, e uma área de floresta ciliar do rio Araguari, localizada no futuro Trecho de Vazão Reduzida (TVR), à jusante da barragem da UHE Amador Aguiar I.

Nas áreas de floresta estacional (três deciduais e uma semidecidual) foram alocadas 60 parcelas posicionadas a partir da futura margem dos reservatórios, abrangendo 200m de comprimento paralelo à margem (10 parcelas de 20m x 10m) e as demais 50 parcelas foram alocadas perpendicularmente à primeira faixa, avançando até 60m de distância da futura margem. Na floresta ciliar, as parcelas foram alocadas a partir da margem do rio, avançando até o limite da mata ciliar com as formações abertas ou com a encosta. Em todas as áreas foram posicionados pontos de monitoramento da umidade do solo em três profundidades.

As primeiras medidas sobre a estrutura da comunidade arbórea e da umidade do solo foram feitas antes do enchimento dos reservatórios e constituíram o controle (T0) para o monitoramento. As medidas posteriores na vegetação foram feitas após dois (T2) e quatro anos (T4) do enchimento de cada reservatório e para as variações na umidade do solo o monitoramento foi feito após um ano (T1) e três anos (T3) do enchimento dos reservatórios. Os resultados completos para o T0 encontram-se nos trabalhos de Siqueira et al (2009); Kilca et al. (2009) e Rodrigues et al. (2010).

O presente estudo apresenta o monitoramento para quatro comunidades arbóreas sob impacto recente de represamento: a floresta ciliar localizada no Trecho de Vazão Reduzida, e três florestas estacionais que passaram a ficar próximas ao lago artificial gerado por barragens. Como foco, temos a avaliação destas comunidades ao longo de quatro anos após a construção da barragem, bem como analisar mudanças no nível de espécies. Por fim, buscamos relacionar as alterações ocorridas por meio das análises de grupos de espécies com semelhantes repostas a estes distúrbios e, com isso, buscar indicar o destino destas florestas e as reais conseqüências das modificações ocorridas.

Temos como hipótese central de que as mudanças na umidade do solo, mesmo em um curto espaço de tempo para árvores (quatro anos), causaram drásticas modificações na estrutura e diversidade destas florestas, sendo esperadas altas taxas de dinâmica, sobretudo devido ao desfavorecimento de espécies especialistas (especialistas a saturação hídrica, no caso da floresta ciliar sujeita a uma condição de vazão do rio reduzida, e especialistas de ambientes com déficit hídrico, no caso das florestas estacionais) e favorecimento de espécies generalistas quanto a disponibilidade hídrica.

Ainda assim, cada capítulo apresentou hipóteses ou perguntas específicas que acreditamos auxiliar na compreensão e mitigação dos impactos causados nestas florestas em futuras barragens que afetem, sobretudo, formações florestais.

Chapter 1

Short term effects of reduced water flow in riparian forest community

Resumo: Efeitos no curto prazo da redução do fluxo de água em comunidades florestais ciliares

Florestas ciliares promovem diversos serviços ambientais, mas estão sujeitas a impactos antrópicos. Dentre os mais comuns estão as construções de barragens. Barragens provocam alagamento de áreas a montante da represa; porém, pode reduzir o fluxo de água à jusante, afetando diretamente as formações ciliares no trecho de vazão reduzida (TVR). Desta forma, este estudo buscou evidenciar o quanto a umidade do solo de uma floresta ciliar pode diminuir no TVR e quais as influências causadas pela redução no fluxo de água sobre uma comunidade arbórea de floresta ciliar. Temos como hipótese que poucos anos sob o efeito da redução na vazão de água de um rio são capazes de alterar a estrutura de uma comunidade arbórea, reduzindo sua riqueza e diversidade. Foi realizado um acompanhamento temporal da umidade do solo (a 0-10, 20-30 e 40-50cm de profundidade) e da comunidade arbórea (dinâmica das árvores com diâmetro a altura do peito de 4.77cm) com amostras antes e depois da redução do fluxo de água do rio. Após a construção da barragem a umidade do solo foi reduzida, principalmente na estação seca, a 0-10cm de profundidade, mas a riqueza e diversidade não apresentaram variações. Ainda assim, a estrutura da comunidade foi afetada, com a redução no número de árvores e na área basal, devido à alta mortalidade e queda de troncos de árvores vivas, assim pode ser considerada em “fase de degradação”. A dinâmica da comunidade apresentou taxas muito altas de mortalidade (5.15% ano^{-1})e perda em área basal (5.65% ano^{-1}), demonstrando que a redução do fluxo de água pela represa tem impacto forte e está modificando severamente a comunidade. Esta modificação foi mais intensa no sub-bosque, mais negativamente afetado pela redução na umidade do solo na superfície (0-10cm) onde espécies generalistas estão se estabelecendo melhor. Porém, diferente de outros impactos, a redução do fluxo de água é um distúrbio constante e duradouro, logo a floresta deve continuar sofrendo alterações e modificar sua fisionomia até se assemelhar estruturalmente e floristicamente a floresta estacional. Consideramos assim o efeito da redução do fluxo de água como um impacto em larga escala, capaz de degradar a floresta a ponto de alterar a estrutura e a fisionomia de uma floresta ciliar.

Palavras chave: dinâmica, umidade do solo, mortalidade, rotatividade
Key-words: dynamic, soil moisture, dams, mortality, turnover

Introduction

Riparian forests are associated and influenced by river systems. These sites belong to most diverse, dynamic and complex terrestrial ecosystems in the world (Naiman et al. 1993). A riparian forest influences water quality (Hill 1996; Shabaga & Hill 2010), contributes to regulation and maintenance of biodiversity in landscapes (Dynesius & Nilsson 1994) and provides a lot of other environmental services. Tree roots keep soil cohesive and reduce erosion (Hubble et al. 2010; Kiley & Schneider 2005), sediment runoff into rivers, and protect downstream areas from siltation (Guo et al. 2007)**.** Riparian forests are important pathways for plant dispersal (Naiman & Decamps 1997; Nilsson & Berggren 2000), acting as corridors to fauna movements (Gundersen et al. 2010) and as a refuge to many vertebrate (Bagno & Marinho-Filho 2001a, b; Marinho-Filho & Guimarães 2001; Palmer & Bennett 2006) mainly during dry periods (Naiman & Decamps 1997), not only for watering but also for fruit feeding. Despite its importance, riparian forests are being suppressed by many anthropogenic activities and one of the most devastating is dams' construction.

In the world, at least 900 thousand dams above 15m high (Avakyan & Iakovleva 1998) retains 15% of total annual river runoff (Gornitz 2001) and obstruct 60% of fresh water that flows to oceans (Nilsson et al. 2005). These dams are important to many human services but are also associated with many environmental problems (Sarkar & Karagoz 1995), especially for riparian systems. Many of these problems are related to water level elevation. Habitat becomes fragmented by water storage (Humborg et al. 1997; Jansson et al. 2000; Nilsson & Berggren 2000) which eventually kills many flora and fauna elements. The hydrological flow patterns of entire rivers change which could alter species richness and composition (Jansson et al. 2000; Nilsson & Grelsson 1995; Nilsson et al. 1997). The diversity of fish, for example, in these reservoirs declines drastically and a few species high abundance dominate the river (Fearnside 2001, 2005; Joy & Death 2001). Another important impact is greenhouse gas emission (mainly methane) released by dead matter to the reservoir surface whose, in some cases, can being similar to those liberated by fossil fuel (Fearnside 2002). Furthermore, the reservoir could increase incidence of diseases dependent of vectors related to stagnant water (Fearnside 2005; Guimarães et al. 1997; Luz 1994).

Thus, many dam problems are well known or have been studied over the years. However, some impacts in such constructions are poorly understood and should be better studied. Most of studies only comprise effects of water increase upstream of dam, neglecting water flow reduction problems downstream, especially immediately after flow reduction. To the dam construction is often necessary to divert the riverbed, reducing water flow channels (Dynesius & Nilsson 1994; Nilsson & Berggren 2000) and depleting water tables (Ward 1998)**.** Nevertheless, there are no studies that show, with empirical data, how much soil moisture is reduced in these areas and implications to riparian systems. Riparian vegetation is the guide to major changes in all other taxa which live in terrestrial habitats associated with river, so monitoring riparian vegetation is extremely important to understand effects of dam on all other organisms. Finally, few studies that evaluate water flow reduction effects on riparian vegetation occurs in cold and low biodiverse environments (Nilsson et al. 1991; Nilsson & Grelsson 1995; Toner & Keddy 1997), but a large number of dams for hydroelectric power production have been built in countries with high species diversity (Nilsson et al. 2005).

Considering the numerous ventures that reduce water available to riparian forests, this work aims to supply some information about downstream effects on riparian forests, focus on arboreal dynamics after flow regulation by a dam in Central Brazil. A spillway controls water flow, and then a riparian forest, before near riverbed, is now 10 to 50 meters away from river water. Whereas, some works show the influence of water discharge on river associated plants (Minshall et al. 1985) and the importance of soil moisture to species richness and riparian cover in these riparian systems (Ehleringer & Dawson 1992; Munoz-Reinoso 2001; Naiman & Decamps 1997; Stromberg et al. 1996), we evaluated the soil moisture reduction in riparian forest after flow reduction and its influence on riparian species and structure.

Our central hypothesis is that four years of water flow reduction are sufficient to alter the structure and floristic composition of an arboreal community. We compartmentalize our central hypothesies into four hypotheses to better understand the final result: 1) The river remoteness, due to water flow reduction, should affect riparian forest soil, significantly reducing the moisture in layers up to 50cm depth; 2) This soil moisture reduction should be more severe in dry season because rain in wet season should help maintain soil moisture; if soil moisture reduction is confirmed, 3) this will have negative effects on some water associated species, reducing richness, alpha diversity and evenness of arboreal riparian flora; 4) thus, net changes in riparian forest will be negative with mortality and outgrowth rates being larger than recruitment and ingrowth rates, and riparian forest would become less dense and supports a smaller basal area.

Material and Methods

This study was conducted in a riparian forest (between 18°47´40”S, 48°08’57”W and 18°47´51”S, 48°08’43”W) located in the Amador Aguiar Dam influence area. For dam construction part of river was diverted by a 27m spillway reducing the flow over a 10km sector from December 2005. This river sector, where water flow was reduced, is called the “Reduced Outflow Stretch” (Trecho de Vazão Reduzida – ROS Figure 1.1). The spillway reduced the water flow from $359m^3.s^{-1}$ to $7m^3.s^{-1}$. Thus, riparian forest near the riverbed in 2005 (Figure 1.2A) is now about 10–50 meters further away from the direct water influence (Figure 1.2B). The average altitude is 595m with a low slope. The climate is Aw (Koppen 1948) with a dry winter (april to september) and a rainy summer (october to march), with an average annual temperature of 22°C and average rainfall of around 1595 mm (Santos & Assunção 2006).

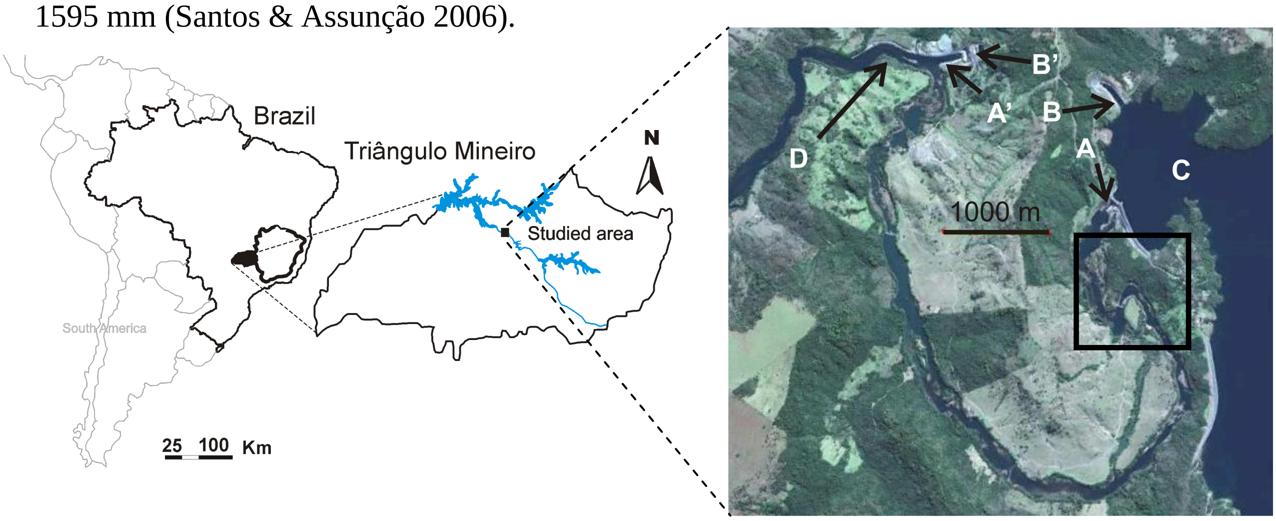

Figure 1.1 - Study area image marked with a square showing studied area. A = Spillway and the beginning of Reduced Outflow Stretch, A’ = end of Reduced Outflow Stretch, B = hydroelectric dam, B’ = end of hydroelectric dam, C = artificial lake created by dam, D = river patch returns to normal flow.

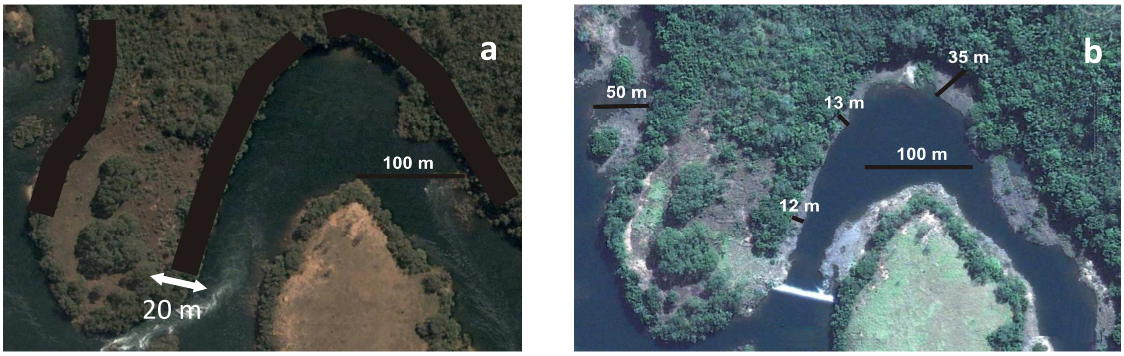

Figure 1.2 - Satellite image of riparian forest on southern Brazil, object of this study, a = 2005 (T0) riparian plots figure and b = 2009 (T4) riparian forest showing flow reduction and the distance between forest with water after spillway construction. Plant sample were plotted along the black belt (a).

Soil - We carried out ten soil collections in riparian forest at three distinct depths: 0–10 cm, 20–30cm and 40–50cm (total of 30 samples) along the ciliar forest. To verify the soil moisture variation we calculate soil moisture based on EMBRAPA methodology (EMBRAPA 1997). This separation is important because we try to substantiate how water flow reduction affects soil moisture at different depths. Then, we repeated soil sampling every three months to cover middle and end of rainy and dry seasons. We also repeated soil moisture collections over three distinct years: before the spillway construction (T0–2005), after (T1–2006) and during the third year of water flow reduction (T3– 2008).

We performed some soil moisture analyses for the three collections times, T0, T1 and T3. First to check the soil data normality we performed Lilliefors test, but soil data did not presented normality. Then we used non-parametric Wilcoxon test with all soil moisture results collected (near and far from shore together) over the years for the three soil depths. To compare effects of flow reduction on the seasons we performed a Kruskall-Wallis (a non parametric test for ANOVA) analysis followed by a post-hoc median test. All these analyses were performed in Systat 10.2 program (Wilkinson 2002).

Plant Sampling - The first inventory (T0) was carried out during the 2005 year after rain season on 110 plots of 10x10m in riparian forest, at 0-10 m and 10-20 m of distance to the river. All trees with a diameter at breast height (DBH) of 4.77cm were tagged with aluminum labels. The diameter of stem was measured at 1.30 m from the ground and in multiple stems all live tillers were also measured at 1.30m. The first inventory of results were published by Rodrigues et al. (2010). Second (T2) and third (T4) inventories were carried out in early 2008 and 2010, respectively two and four years after spillway construction (occurred in the december of 2005). These sampling methods followed the same procedure of first inventory. New individuals that met the inclusion criteria (recruits) were measured and identified. Mortality refers to standing dead trees, fallen trees or individuals which were not found. We prepared tree mortality distributions into classes of diameter using class intervals with exponentially increasing ranges of intervals of T0–T4. All the reproductive botanical material were inserted on *Herbarium Uberlandense*, and the species nomenclature and synonymies follow Missouri Botanical Garden web site (http://www.tropicos.org/).

Diversity and evenness - We calculated two diversity indexes (Shannon-Weaver and Simpson) to measure to changes in diversity over three measurement periods (T0–T2–T4). Use of more than one diversity index is important because both are widely used in scientific studies but are not always published together. Shannon-Weaver is the most popular index and is widely used in phytosociological and dynamic studies. Simpson is a good measure of diversity because it only varies between 0 (minimum diversity) and 1 (maximum diversity) which facilitates comparisons. To compare changes in diversity we performed statistical analyses between the three periods for each index. For Shannon we applied Hutcheson t test (Hutcheson 1970), and to Simpson index we followed the procedures suggested by Brower et al. (1998). We conducted a Wilcoxon test (a non-parametric test equivalent to Student t test) between T0-T2, T2-T4 and T0-T4 on the number of individuals and basal area using plots as samples. For evenness we performed the Pielou evenness indices (Brower et al. 1998).

Dynamics rates - We based community dynamics on mortality, recruitment, outgrowth and ingrowth rates. Annual mortality (m) and recruitment (r) were calculated in terms of annual exponensial rates (Sheil et al. 1995 and Sheil et al. 2000). Outgrowth annual rates (o) refers to basal areas of dead trees plus dead branch basal areas of living trees (decrement) and ingrowth annual rates (i) refers to basal areas of recruits plus growth in basal area of surviving trees (increment). Then we analyze mortality/recruitment rates and ingrowth/outgrowth annual rates of the most representative species (minimum of 10 individuals). Annual rates formulas used were,

$m = 100 \times \{1-[(n_0 - n_m) / n_0]^{1/t}\}$;

$r = [1 - (1 - n_r/n_t)^{1/t}] \times 100\}$;

$o = \{1 - [(BA_0 - BA_m + BA_d)/BA_0]^{1/t}\} \text{ x } 100$ and

$i = \{1- [1 - (BA_r + BA_g)/BA_t]^{1/t}\} \text{ x } 100$

where n_0 is the original number of trees; n_m is number of deaths; n_r is number of recruits; n_t is final number of individuals; BA_0 is original basal area; BA_m is basal area of dead trees; BA_d is basal area of dead stems of living trees; BA_r is basal area of recruits; BA_g is growth basal area; BA_t is final basal area and “t” is the time period between measurements in years. However, to compare community with studies which uses logarithmic model, we performed the logarithmic model to “m” and “r” rates (see formulas in Condit et al. 1999).

To evaluate changes in forest we computed turnover rates for individuals and basal area through mortality-recruitment rates and outgrowth-ingrowth rates (Oliveira-Filho et al. 2007):

$TN = (m + r) \times 2^{-1}$;

$TBA = (o + i) \times 2^{-1}$

where TN is individuals turnover and TBA is basal area turnover. Then we evaluated net change (Korning & Balslev 1994) to individuals and basal area,

$ChN = [(N_t/N_o)^{1/t} - 1] \times 100$;

$ChBA = [(BA_t/BA_o)^{1/t} - 1] \times 100$

where ChN is individuals net change and ChBA is basal area net change and we develop an overall net change based on average of ChN and ChBA rates. All these analyses were conducted on each species with at least 10 individuals. Finally we evaluated the species individuals, basal area, and cover value (an average relative value between the number of individual and basal area) for species with at least 10 individuals, focusing on unstable species). However, due to large number of species, we only reported on those species that represented greatest changes in the community.

Results

Soil Changes - Major significant differences between soil moisture occurred in middle of the dry season at 0-10cm deep (Table 1.1). In this season, soil was moister before flow reduction at 0-10cm, but not at other depths (Table 1.1). At end of dry season a clear reduction in moisture after spillway construction was perceptible at three depths only one year after flow reduction.

Table 1.1. Wilcoxon test results (with "p "and "Z" values) for soil moisture in each season between different soil depths. MR = middle rainy season, ER = end rainy season, MD = middle dry season, ED = end dry season, T0 = before spillway construction, T1 = one year after and T3 = three years after flow reduction in a riparian forest in southern Brazil. In bold $p < 0.05$ and in italic $p < 0.10$.

		0 – 10 cm			**20 – 30 cm**			**40 – 50 cm**		
	p\Z values	**T0**	**T1**	**T3**	**T0**	**T1**	**T3**	**T0**	**T1**	**T3**
MR	**T0**	-	1.478	-0.866	-	0.663	-0.357	-	0.968	-0.153
	T1	0.139	-	-2.090	0.508	-	-0.663	0.333	-	-0.764
	T3	0.386	0.037	-	0.721	0.508	-	0.878	0.445	-
ER	**T0**	-	0.459	1.580	-	-1.682	0.153	-	-0.764	0.968
	T1	0.646	-	1.988	*0.093*	-	2.293	0.445	-	1.784
	T3	0.114	**0.047**	-	0.878	**0.022**	-	0.333	*0.074*	-
MD	**T0**	-	-2.803	-2.599	-	-0.764	-1.682	-	0.153	-0.255
	T1	**0.005**	-	-0.051	0.445	-	-2.497	0.878	-	-0.357
	T3	**0.009**	0.959	-	*0.093*	**0.013**	-	0.799	0.721	-
ED	**T0**	-	-2.599	-1.172	-	-1.955	-1.580	-	-2.073	-1.172
	T1	**0.009**	-	1.682	**0.050**	-	0.969	**0.038**	-	1.244
	T3	0.241	*0.093*	-	0.114	0.333	-	0.241	0.214	-

In general, there were no significant differences between years. In all years, middle and end of rainy season are obviously more moist (Figure 1.3) at all three depths. However, middle of dry season before dam construction was as humid as moister seasons of that year (Figure 1.3). Nevertheless, after dam construction and water flow reduction of middle dry season became as dry as the end of dry season. This effect was best demonstrated at 0-10cm depth (Figure 1.3 A), probably because this depth was more river dependent than deeper soil layers, which may be more subjected to moisture from groundwater. In other layers (20-30cm and 40-50cm) test was not significant; however, demonstrates a moisture reduction tendency after dam construction (Figure 1.3 B, C). These results (Table 1.1) show that the ground surface was more affected by reduction in water flow on river than deeper layers and became drier (Figure 1.3) after flow reduction which was mainly evident in middle of dry season.

Floristic changes - After four years of water flow reduction the richness in riparian forest varied little (92 in T0 and T2, and 93 in T4). Two species were found in T0 and T2 (*Machaerium villosum* and *Rudgea virburnoides*) represented by a single tree, but these individuals were killed throughout T2-T4. Otherwise, three new species were registered with single trees through T2-T4: *Chomelia* *sericea*, *Eugenia involucrata* and *Lonchocarpus cultratus*. As well as richness, diversity seems not to have been affected even four years after flow reduction, neither in number of individuals either for basal area (Table 1.2). Also there was no change in evenness in number of individuals, although a reduction in basal area evenness after four years of damming is perceptible (Table 1.2).

Table 1.2 - Diversity indexes (with respective tests results) and evenness for number of individuals and basal area in a riparian forest in southern Brazil. Different letters indicate significant variation between times.

	Number of individuals			Basal area		
	T0	T2	T4	T0	T2	T4
Absolute values	1405	1369	1288	45.60	44.75	44.21
Shannon Index (H’)	3.659a	3.714a	3.685a	3.253a	3.273a	3.144a
Simpson Index (1-D)	0.955a	0.959a	0.960a	0.955a	0.953a	0.939a
Pielou Evenness	0.807	0.819	0.809	0.717	0.722	0.690

Structure changes - There was a reduction in number of individuals and basal area between the years evaluated (Table 1.3). For number of individuals there was a clear reduction four years after water spillway construction (4.05%) and marginally statically significant difference after first two years (2.56%, Table 1.3). However, to basal area only the periods T0-T2 (loss of 1.86%) and T0-T4 (loss of 3.04%, Table 1.3) presented differences that was marginally statistically significant. These decreases are mainly due to high mortality (268 trees) and lower recruitment (151) along four measurement years (Table 1.4). Despite of the high increment (7.68m^2 in four years), high mortality added to strong loss of stems provided a greater loss in basal area than gain (Table 1.4). Analysis of mortality by diameter classes demonstrates a higher mortality in two first classes (4.77cm until 9.9cm and 10cm until 19.9cm), with 131 and 95 dead trees respectively. The third class (20cm until 39.9cm) and fourth class (higher than 40cm of diameter) only presented 29 and 14 respectively. These results demonstrated that smaller trees (smaller than 20cm of diameter) were most negatively affected by disturbance, representing 84% of dead trees in this riparian forest.

Table 1.3 - Wilcoxon test between basal areas in the plots in T0, T2, T4 in a riparian forest in southern Brazil. In bold $p < 0.05$ and in italic $p < 0.10$

	Number of individuals (p\Z)			Basal área (p\Z)		
Years	T0	T2	T4	T0	T2	T4
T0	-	-3.655	-3.500	-	1.048	1.601
T2	*0.096*	-	-1.664	*0.092*	-	1.873
T4	**0.001**	**0.001**	-	*0.100*	0.295	-

Table 1.4 - Dynamics parameters in four years of water flow reduction in a riparian forest in southern Brazil. N = number of individuals, BA = basal area)

Parameters	**T0-T2**	**T2-T4**	**T0-T4**
Mortality (N)	125	143	268
Recruitment (N)	89	62	151
Mortality (m^2)	2.80	3.26	6.07
Recruitment (m^2)	0.22	0.17	0.39
Decrement (m^2)	1.82	1.57	3.39
Increment (m^2)	3.55	4.13	7.68
Mortality rate (% ano^{-1})	4.55	5.35	5.15
Recruitment rate (% ano^{-1})	3.26	2.44	3.06
Outgrowth rate (% ano^{-1})	5.20	5.55	5.65
Ingrowth rate (% ano^{-1})	4.30	4.99	4.91
Turnover (N) (% ano^{-1})	3.91	3.90	4.11
Turnover (BA) (% ano^{-1})	4.75	5.10	5.28
Individuals net change (N) (% ano^{-1})	-1.29	-3.00	-2.15
Basal área net change (BA) (% ano^{-1})	-0.94	-0.61	-0.77

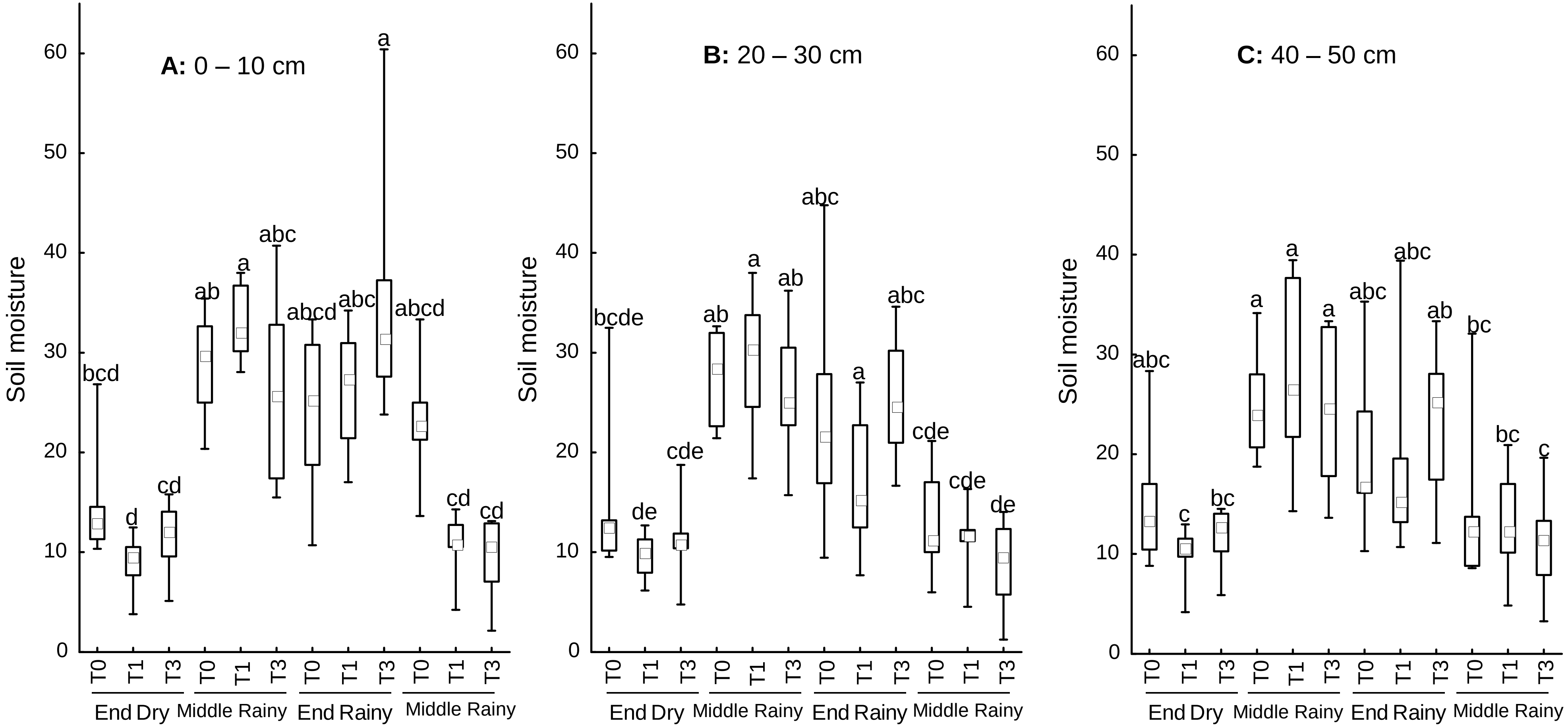

Figure 1.3. Box-Plot representation to Kruskall-Wallis test to distinct depths before spillway construction (T0), one year after (T1) and three years after flow reduction (T3) in a riparian forest in southern Brazil. Dashes = range, Bars = interval between first and third quartile, square = median. Results detail on Appendix 1.

Dynamic rates - Tree dynamics presented higher mortality rates than recruitment (Table 1.4). The same occurred between outgrowth and ingrowth annual rates, especially due low recruits basal areas and due high mortality and basal area loss by trees with dead stems. All these rates reveal a faster tree dynamic community than other riparian and tropical forests, represented by high turnover rates of number of individuals and basal area (Table 1.4). At the same time with a strong negative balance exemplified by a negative net rate on number of individuals and basal area.

While some species showed slow dynamics with low mortality and recruitment rates (lower rates than the entire community – Table 1.4, Figure 1.4A), others have high rates with high recruitment and/or mortality (Figure 1.4A). From those species with fast dynamics, only three species had a positive individual's net change, meanwhile another 14 had a negative individual's net change. Thus, the flow reduction in general affects more species negatively than positively.

The same is seen to ingrowth/outgrowt rates (Figure 4B) because eight species had rates lower than the entire community. Nine species presented ingrowth larger than outgrowth and had a positive basal area net change. However, a further 16 have more outgrowth rates higher than ingrowth, and possessed a negative basal area net change. Hence, negative impacts of water flow reduction were intense for basal area and individuals in this riparian forest and surpassed any positive effects.

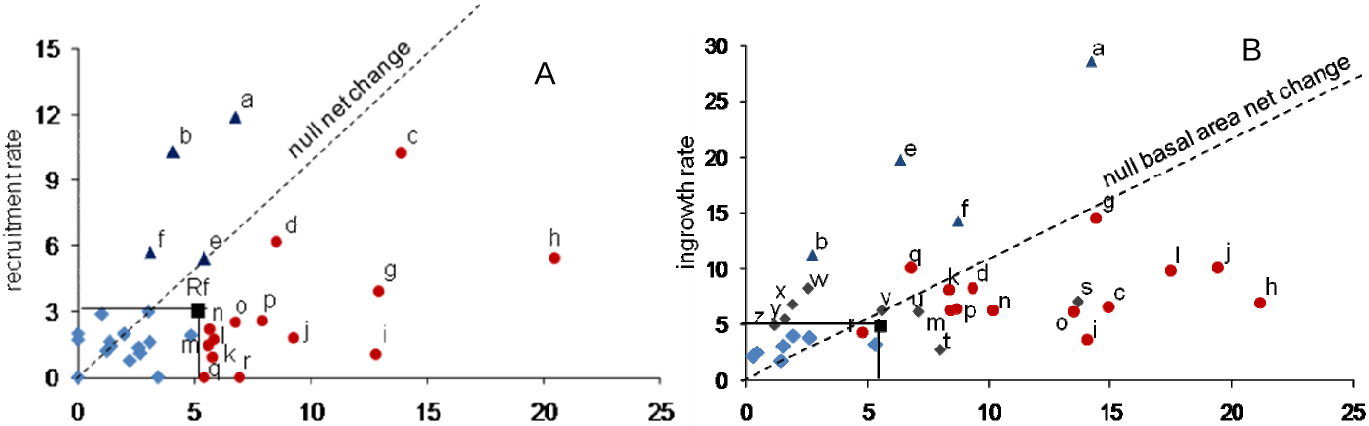

Figure 1.4 - Comparison of mortality/recruitment rates (A) and outgrowth/ingrowth rates (B) in species with 10 or more individuals in a riparian forest in southern Brazil. Square = community net change, triangles = species with positive individuals net changes, circles = species with negative individuals net changes, blue diamond = species with individual net changes lower than community and black diamond = species with individuals net changes lower than community but with area basal net change higher than community. Species Legend in Table 5.

Species with high individuals turnover (higher than 3.5 of TN, Table 1.5) are most typically from understory or forest edge (*Siparuna guianensis*, *Myrcia laruotteana*, *Byrsonima laxiflora*, *Xylopia aromatica*, *Bauhinia ungulata*, *Coccoloba mollis*, *Cousarea hydrangeaefolia*, *Erythroxylum daphnites*, *Matayba guianensis*, *Cordieria sessilis*, *Alibertia edulis* and *Casearia sylvestris*), except *Inga vera*, *I. laurina* and *Hirtella gracilipes*; three sub-canopy/canopy species which are related to moist environments (riparian and gallery forests). Otherwise, those species with low rates (lower than 3.0 of TN, Table 1.5) are mostly from canopy (*Apuleia leiocarpa*, *Terminalia glabrescens*, *Tabebuia roseo-alba*, *Ficus* sp1, *Protium heptaphyllum*, *Copaifera langsdorffii*, *Hymenaea courbaril*, *Platypodium elegans*, *Pouteria torta*, *Andira anthelmia*, *Salacia elliptica*, *Acacia polyphylla*) excepting *Cheiloclinium cognatum*, a typical understory species.

Most of the high turnover species (11 of 15) had a negative balance between mortality/recruitment and outgrowth/ingrowth, except *S. guianensis*, *B. laxiflora*, *E. daphnites* and *C. sessilis* (Table 1.5). However, some low turnover (7 of 13) species have a zero/positive balance between mortality/recruitment and/or outgrowth/ingrowth. From these low turnover species analyzed, four have both negative balances (*Tapirira guianensis*, *Ormosia arborea*, *Zanthoxyllum riedelianum* and *Myrcia splendens*) and one a zero/positive balance (*Unonopsis lindimanii*). Thus, most of the species (19, Table1.5) have a negative overall net rate (ONR) and seven show an ONR lower than five negative. Then, from 14 species with positive ONR (Table 1.5) only *B. laxiflora* exceeded five.

All these results indicated that few species (mostly from canopy) were subjected to only minor changes, and can be considered stable in this riparian forest even after flow reduction. Most species, however, had experienced a high death rate and/or loss of basal area, demonstrated the negative effects of lack of moisture. Moreover, the most severe negative effects of moisture reduction occurred to understory species and those associated with water resources. Figure 1.5 represents the species with major contributions to community changes. *H. gracilipes*, *I. vera*, *A. edulis* (water associated sub-canopy understory species), *I. laurina* (canopy water associated species), *C. hydrangeaefolia*, *C. mollis* and *P. torta* (understory-subcanopy species) were the more negatively affected species which strongly influences the community density and/or basal area reduction.

Only two species experienced major positive changes in this riparian forest: *S. guianensis*, within an increase in number of individuals and *A. leiocarpa* with an increase in basal area – Figure 1.5). In general, the cover value (that evaluates species density and dominance) of three species, all water associated, were the most negatively affected with strong mortality and basal area loss (*H. gracilipes*, *I. vera*, *A. edulis)*. With the high reduction in individuals and basal area of some species below canopy and a population increase (*S. guianensis*) in a short time period (four years), it is possible to summarizee that layers below the canopy (understory and sub-canopy) are going through a major restructuring of their features.

Table 1.5 - Species dynamic rates between T0 – T4 in descending order of Overall net change (ChN+ChAB)/2) to species with 10 or more individuals. M = mortality rate, O = outgrowth rate, R = recruitment rate, I = ingrowth rate, TN = individuals turnover, TBA = basal area turnover, ChN = individuals net chance, ChBA = basal area net change, ONC = overall net change; U = understory, I = intermediary, C = canopy (*Rodrigues et al 2010 classification) in a riparian forest in southern Brazil.

								T0-T4									
		Nº Individuals			**Basal Area (m^2)**			**Dynamics rates**				**Turnover rates**		**Net Changes**			**Layer***
Codes	**Species**	**T0**	**T2**	**T4**	**T0**	**T2**	**T4**	**M**	**O**	**R**	**I**	**TN**	**TBA**	**ChN**	**ChBA**	**ONC**	
b	*Byrsonima laxiflora*	13	12	17	0.21	0.25	0.26	4.09	2.73	10.31	11.24	7.20	6.98	6.94	5.45	6.19	U
e	*Erythroxylum daphnites*	10	10	10	0.03	0.03	0.04	5.43	6.39	5.43	19.83	5.43	13.11	0.00	7.62	3.81	U
a	*Siparuna guianensis*	78	91	98	0.34	0.40	0.37	6.74	14.23	11.91	28.54	9.33	21.38	5.87	1.62	3.74	I
x	*Apuleia leiocarpa*	25	27	27	0.99	1.13	1.20	1.02	1.90	2.90	6.79	1.96	4.34	1.94	5.04	3.49	C
y	*Ficus* sp1	12	13	13	2.31	2.23	2.71	0.00	1.61	1.98	5.47	0.99	3.54	2.02	4.04	3.03	C
w	*Unonopsis lindmanii*	26	27	26	0.34	0.41	0.43	3.02	2.56	3.02	8.24	3.02	5.40	0.00	5.51	2.76	U
f	*Cordieria sessilis*	17	20	19	0.09	0.09	0.09	3.08	8.72	5.74	14.33	4.41	11.52	2.82	1.04	1.93	U
z	*Tabebuia roseo-alba*	13	12	13	0.11	0.12	0.12	1.98	1.16	1.98	4.91	1.98	3.03	0.00	2.94	1.47	C
-	*Cheiloclinum cognatum*	14	14	15	0.29	0.30	0.31	0.00	2.62	1.71	3.87	0.86	3.24	1.74	1.13	1.43	U
-	*Protium heptaphyllum*	94	96	95	3.23	3.38	3.50	1.36	1.92	1.62	4.04	1.49	2.98	0.26	2.09	1.18	C
-	*Salacia elliptica*	12	12	12	0.62	0.65	0.67	0.00	0.28	0.00	2.18	0.00	1.23	0.00	1.95	0.97	I
-	*Hymenaea courbaril*	35	33	33	5.64	5.92	6.13	2.22	0.45	0.77	2.52	1.49	1.48	-1.46	2.11	0.32	C
-	*Andira anthelmia*	21	22	21	0.80	0.83	0.81	1.21	1.43	1.21	1.75	1.21	1.59	0.00	0.22	0.11	C
-	*Copaifera langsdorffii*	49	48	46	6.59	6.72	7.03	2.65	1.52	1.11	3.10	1.88	2.31	-1.57	1.61	0.02	C
v	*Platypodium elegans*	20	20	19	0.52	0.52	0.53	2.60	5.61	1.34	6.32	1.97	5.96	-1.27	0.63	-0.32	C
q	*Acacia polyphylla*	10	9	8	0.09	0.10	0.10	5.43	6.79	0.00	10.19	2.71	8.49	-5.43	3.79	-0.82	I
u	*Tapirira guianensis*	61	60	54	1.77	1.87	1.69	4.85	7.09	1.91	6.16	3.38	6.62	-3.00	-1.24	-2.12	I
d	*Matayba guianensis*	34	33	31	0.43	0.42	0.40	8.34	9.33	6.20	8.72	7.27	9.02	-2.28	-2.02	-2.15	I
-	*Terminalia glabrescens*	23	22	20	1.59	1.61	1.46	3.43	5.31	0.00	3.26	1.72	4.29	-3.43	-2.12	-2.78	C
k	*Myrcia splendens*	33	29	27	0.15	0.14	0.14	5.79	8.34	0.94	8.08	3.36	8.21	-4.89	-0.83	-2.86	U
t	*Pouteria torta*	18	18	18	1.18	1.20	0.92	1.42	8.01	1.42	2.71	1.42	5.36	0.00	-5.85	-2.92	C
m	*Coussarea hydrangeifolia*	63	56	53	0.53	0.50	0.47	5.61	8.42	1.45	6.34	3.53	7.38	-4.23	-2.63	-3.43	U

r	*Zanthoxyllum riedelianum*	12	10	9	0.30	0.31	0.30	6.94	4.80	0.00	4.33	3.47	4.56	-6.94	-0.50	-3.72	C
n	*Hirtella gracilipes*	174	171	151	4.41	3.78	3.68	5.63	10.20	2.23	6.30	3.93	8.25	-3.48	-4.45	-3.97	I
c	*Casearia sylvestris*	25	22	20	0.18	0.16	0.15	7.88	8.65	2.60	6.37	5.24	7.51	-5.43	-3.26	-4.34	U
s	*Ormosia arborea*	17	17	16	0.45	0.49	0.33	3.08	13.66	1.60	7.09	2.34	10.37	-1.50	-7.26	-4.38	C
g	*Bauhinia ungulata*	40	40	27	0.23	0.27	0.21	12.92	14.44	3.93	14.60	8.42	14.52	-9.36	-1.74	-5.55	U
l	*Inga laurina*	70	61	59	4.49	3.57	3.13	5.85	17.49	1.74	9.77	3.80	13.63	-4.18	-8.68	-6.43	C
o	*Alibertia edulis*	74	68	62	0.70	0.53	0.49	6.73	13.49	2.51	6.16	4.62	9.83	-4.33	-8.81	-6.57	U
c	*Xylopia aromatica*	20	19	17	0.37	0.27	0.24	13.88	14.94	10.31	6.97	12.10	10.96	-3.98	-10.29	-7.14	I
j	*Coccoloba mollis*	59	52	43	0.61	0.52	0.38	9.26	19.41	1.79	10.11	5.53	14.76	-7.60	-11.05	-9.33	I
i	*Inga vera*	38	29	23	1.00	0.86	0.63	12.77	14.03	1.11	3.60	6.94	8.81	-11.80	-10.90	-11.35	I
h	*Myrcia* aff. *laruotteana*	10	9	5	0.14	0.10	0.07	20.47	21.19	5.43	6.91	12.95	14.05	-15.91	-16.14	-16.02	U

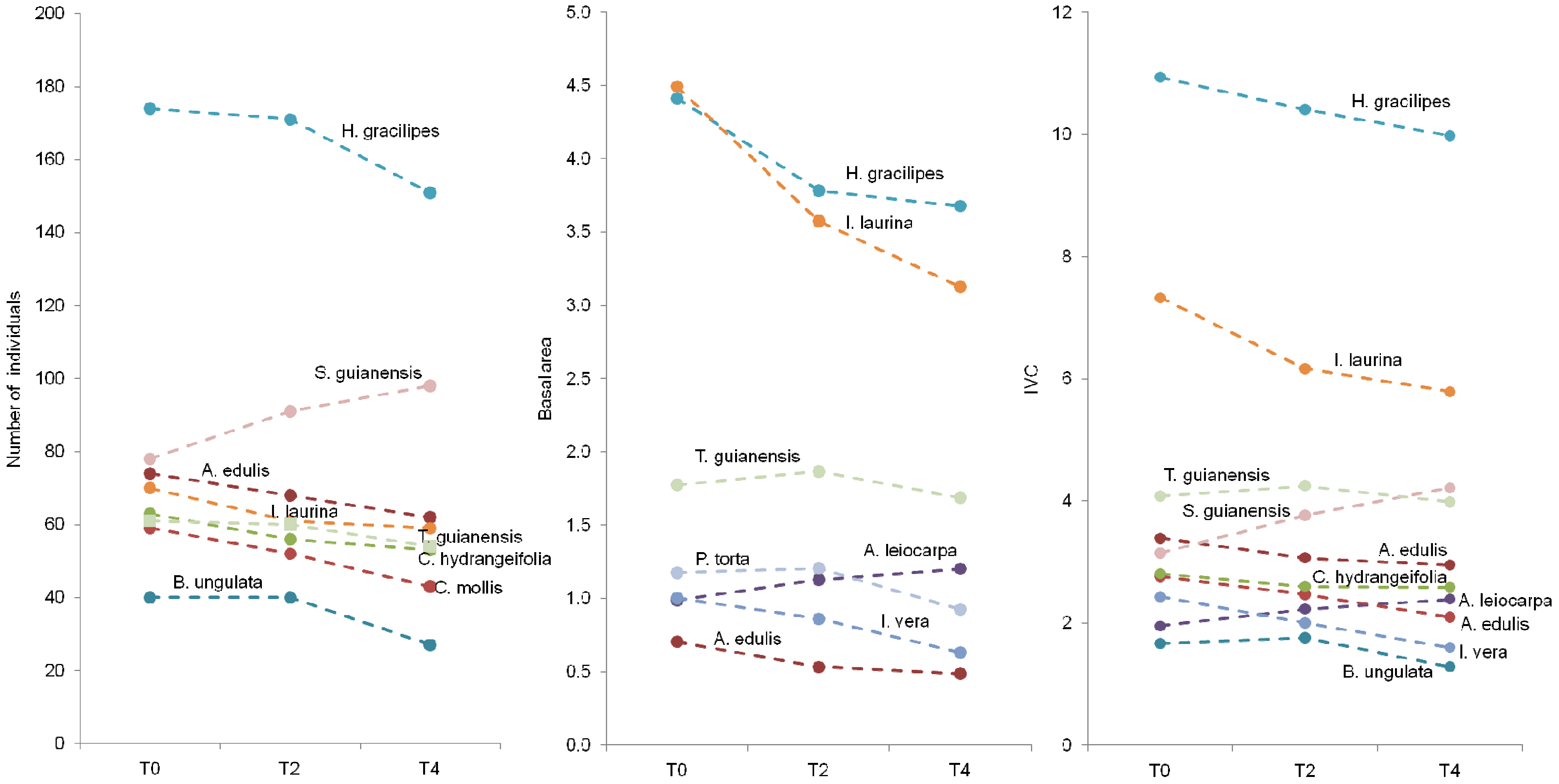

Figure 1.5 - Number of individuals, basal area and cover value to non-stable species in a Riparian Forest in southern Brazil, only those with 10 or more individuals (see figure 4A and 4B) and only those species that represents most changes in the tree community.

Discussion

Soil – Partly confirming our hypotheses the analysis successfully shows the spillway effects in soil moisture over dry season, mainly at 0 – 10cm, but not to other depths. Thus, the soil surface layer is more affected by water flow reduction. Studies in riparian forests demonstrate importance of groundwater and soil moisture storage to streamflow and vegetation (Fujieda et al. 1997; Munoz-Reinoso 2001); thus, at deep layers (20–50cm), the effect of spillway construction was not severe compared to soil in the surface. This soil moisture reduction was enough to cause structural modifications in this riparian forest. Many trees died after flow reduction, and obviously with greater light penetration onto soil surface provided greater evaporation and soil moisture loss. Trees also increase water infiltration by retaining soil moisture and also their absorption capacity (Joffre & Rambal 1988); therefore, tree mortality could contribute to soil moisture reduction.

Plant cover absence reduces water storage (Joffre & Rambal 1988, 1993), raises the temperature in warmest months (Breshears & Barnes 1999) and raises water evapotranspiration (Breshears et al. 1997); thereby, a long dry period due the water flow reduction will probably change water cycle on the riparian forest´s surface. Even though the tests have not shown a strong soil moisture reduction, in lower layers there is a clear trend of reduction especially in end of dry season at all depths. Despite soil surface being the most affected layer, at end of dry season very little water is stored in soil from rain (average rainfall in each month in region during dry season is just 36.5mm and in wet season is 236.5mm (Silva & Ribeiro 2004), and then, during these periods, deeper layers are dependent on moisture stemming from river.

Even soil moisture at end of dry season was reduced only one year after flow reduction, but not at all of three depths, probably due to rain before soil analysis on T3. If heavy rains have hot fallen at end of dry season, lack of moisture stemming from river can reach all soil depths. However, if first heavy rains have occurred (which actually happened for T3 measurement) it is hard to detect differences in soil moisture. The end of dry season is commonly driest period of the year and few rains could raise the soil moisture. Despite this, there is a clear soil moisture reduction tendency for all dry seasons and for all three depths of soil (but more strongly at 0-10cm) and in middle of dry season, before spillway construction was humid as rainy season, became as dry as the end of dry season.

Structure - Contrary to our hypothesis, in a period of four years after river water flow reduction, both richness and diversity did not decrease. Indeed, although two species have not been resampled, a further three were present in last remeasurement (T4). We have three explanations for this: 1) the time period of four years was not sufficient to provoke harsh changes in an arboreal community, hence longer periods could clarify the moisture reduction effects on riparian tree flora, 2) the intense rains during the rainy season were sufficient to keep soil moist enough for vast majority of species, and 3) and most probably, more abundant species are characteristic of high moist soils; therefore, these species still have many individuals to support their presence in community.

Explanations 1 and 3 are most acceptable. Riparian trees are known to be sensitive to reductions in stream flow or groundwater availability (Schume et al. 2004) and many species present in this riparian forest are water-dependent or at least, successful in riparian environments (*H. gracilipes, I. vera, A. edulis, I. laurina, C. hydrangeaefolia , T. guianensis, P. heptaphyllum*) (Felfili et al. 2001; Ferreira & Ribeiro 2001; Gould et al. 2006; Matos & Felfili 2010; Pinheiro & Ribeiro 2001; Schiavini et al. 2001; Silva Júnior et al. 2001; Veneklaas et al. 2005). The high mortality of these species were main factor in density reduction over four years, therefore in medium-long period with flow reduction and remoteness of riverbed, we believe there will be more severe changes in both richness and diversity. Water regime variations can strongly influence species composition, community structure and biological diversity (Munoz-Reinoso 2001; Stromberg 2001) and this riparian forest will suffer drought effects and soil moisture reduction for several years. Reduction in available water is considered responsible for the extinction of moist dependent plant species from rivers and streams (Lopes & Schiavini 2007).

Mortality and loss rates - The rapid mortality and outgrowth rates found in riparian forest confirming our hypotheses and exemplify effects of drought due to water flow reduction. These rates were higher than any other tropical forest, including long dynamics studies (Gomes et al. 2003; Korning & Balslev 1994; Manokaran & Kochummen 1987; Phillips et al. 2004; Phillips et al. 1994; Swaine et al. 1987), semideciduous dry forest (Appolinário et al. 2005; Machado & Oliveira-Filho 2010; Oliveira-Filho et al. 2007; Paiva et al. 2007; Silva & Araújo 2009), gallery forests (Felfili 1995; Lopes & Schiavini 2007), deciduous forest (Swaine et al. 1990; Werneck & Franceschinelli 2004) and even dynamic studies on severe drought periods originated by “El Niño” events (Condit et al. 1995). These results elucidated how water flow reduction could be dramatic to a riparian forest. Moreover, these values evidenced the prolonged drought effects in these forests because, before spillway construction, the river provided a superior supply of water to trees.

The length of dry periods and annual rainfall are important determinants of plant distribution (Barton & Teeri 1993; Fay et al. 2002; Gitlin et al. 2006), but less intense droughts may not increase mortality (Condit et al. 2004). In riparian forests the dry periods may be milder and lower due river moisture; however, absence of the river clearly makes dry period more intense with lower soil moisture. Drought effects, or at least soil moisture reduction, causes changes in leaf area index, leaf water status, and a decrease in carbon fixation and gas exchange (Leuzinger et al. 2005; Orwig & Abrams 1997), which lead to smaller photosynthetic, productivity and growth rates (Ciais et al. 2005; Kljun et al. 2006; Krishnan et al. 2006; Otieno et al. 2006), occasionally causing the death of trees (Allen & Breshears 1998; Breshears et al. 2005; Gitlin et al. 2006) or, at least, the falling of branches. Thus, the fitness of each individual may be reduced in environment and consequently the surving trees will become more vulnerable to pathogens, herbivory and wind. Therefore, is not surprising that high falling of branches and mortality providing strong decrement and basal area loss.

Turnover - Dead and mutilated trees open new spaces that are occupied by new recruits and growing trees (Everham & Brokaw 1996; Swaine et al. 1990). However, many species present high mortality and basal area loss (outgrowth) rates and they are not the same as those few species with a high recruitment and basal area gain (ingrowth) rates. Increase in importance of some dry resistant species over non dry resistance was noticeable in other works (Breshears et al. 2005; Condit et al. 1995) and promotes high species turnover. However, the intense mortality leads to negative net changes among the most abundance species and this was confirmed in community, in number of individuals and basal area. Most of high turnover species with negative overall net change and are understory species (except three sub-canopy/canopy water related species), unlike most low turnover species; which are mostly canopy species. However, we highlight species with a major contribution to community changes. All species with negative net change (to basal area or individuals) are water related species (*H. gracilipes*, *I. laurina*, *A. edulis*, *I. vera*), or are understory/sub-canopy species (*C. hydrangeaefolia*, *C. mollis* and *P. torta*), confirming the moist reduction impact on community.

Large-scale disturbances could cause drastic negative effects on tree diversity (Oliveira et al. 1997), change the structure and promote high turnover rates with sudden and dramatic changes in the abundance and spatial arrangement of dominant plants (Gitlin et al. 2006). In our riparian forest, the four years of water flow reduction was not sufficient to provoke a reduction in diversity or richness, but clearly influence structural changes in arboreal flora. More drought resistant species tend to reach low water potential more rapidly than less resistant species (Ladiges 1975) and their physiological aspects should not be severely affected by drought.

The consequences of reduced water availability may affect germination and growth (Stone & Bacon 1994), but we did not evaluate seeds or saplings, probably the most negatively affected forest compartment. Thus, we believe in a greater turnover in lower layers of forest which directly affects the future of community, nevertheless recruitment, was not apparently influenced and recruitment rates fit those found in other forests. Still the great recruitment of non-water dependent species, despite low recruitment of “moist dependent species”, confirm the river moisture reduction consequences on species changes in this riparian forest.

Large and small trees - Due to drought intensively and length effects on each tree, many individuals of drought sensitive species die or lose branches, which not only modifies the local flora as well as tree community structure. In our study, most dead trees have a diameter at breast height smaller than 20cm. The smaller trees were most important to trees density declines, while large trees mortality strongly influenced basal area reduction. It is common in tropical forests, for greatest mortality (even in % per size class) to be concentrated in the smaller diameter classes, especially in dry years (Chazdon et al. 2005; Clark & Clark 1996). The drought negatively affects all community; however it was more harmful to smaller classes, because the soil surface (0-10cm) suffered greatest soil moisture reduction.

Woody species differ with respect to depths from which they extract water (Breshears et al. 1997; Evans & Ehleringer 1994; Flanagan et al. 1992; Kiley & Schneider 2005). Large trees generally have deep roots and can access deeper soil water resources, which were less affected by droughts (Breshears & Barnes 1999; Otieno et al. 2005; Saha et al. 2008) and also become less river dependent. The 33 thickest trees, for example, had no mortality proving a non-river-water dependence. However, small trees have shallow-roots and cannot access deeper water, thus soil drying probably was the major mortality influence and provokes high mortality and loss rates to these sizes. Small trees have a greater lifetime risks of being hit by falling debris (Deng et al. 2008) such as branches and dead trees, then the mortality is also greater in seedlings than small trees (Kitajima & Augspurger 1989; Swaine & Hall 1988).

Water and light are linked to disturbances that affects the dynamics of smaller trees (Machado & Oliveira-Filho 2010); soon, was not surprising the forced drought caused by damming, negatively affects many understory species. Except for *Apuleia leiocarpa* (a canopy species), most species with high positive overall net change are from understory species (*Byrsonima laxiflora*, *Erythroxylum daphnites*, *Siparuna guianensis*, *Unonopsis lindmanii*, *Cordieria sessilis*). However, only *A. leiocarpa* and *S. guianensis* are well represented in this community. Still, these positive net changes for a few species are much smaller than negative effects in many others and all of these are generalist species.

We conclude that the negative effects of water flow reduction in riparian forests far outweigh any positive effects on a few generalist species and in some important ecosystems services, like carbon storage (represented by basal area reduction), soil protection (represented by high mortality), as well as biodiversity changes (with the disadvantage of many moisture dependent species and a high mortality of understory species), and environmental changes (with an increase of direct sunlight inside the forest). On large-scale, precipitation and temperature are the most important climatic factors controlling ecological processes (Liu et al. 2004). However, for riparian forest, the river water clearly has a great importance to ecological processes and many of these could be lost or at least reduced. Floods create heterogeneity within riparian zone, favoring coexistence of different species (Naiman & Decamps 1997) and with river detachment these patch were loss and environment´s heterogeneity was clearly reduced in some riparian sites.

With reduction in density and basal area, we can classify this riparian forest in a *degradation* phase (according Machado & Oliveira-Filho 2010). A hypothetical model of sylvigenesis, based on tree community dynamics. In this model a strong disturbance could lead a forest to lose its density and basal area; however, this forest still could recover and become a forest with an initial density and basal area, returning to a stage of stability (the forest maintains its resilience – capacity to recover from disturbances) (Gunderson 2000; Neubert & Caswell 1997; Pimm 1984).

In our studied riparian forest, and we believe to many other riparian forest which have reduced riverbeds by deviations and damming to irrigation or supplies to cities or dams, this “degradation phase” will not recover to a mature sucessional stage, due the moist reduction being a constant disturbance (thus, forest will lose its resilience). This constant disturbance will cause droughts for several years and change to this riparian forest (with many moist-dependent species) will lead to another physiognomy (with more generalist species), probably a semideciduous seasonal forest, a type of forest which is strongly influenced by seasonal climatic changes. This study strongly contributes to understanding of consequences of human damming in riparian forests and provides a valuable insight into what may be happening with riparian forests whose river water has been reduced (of course with distinct degrees of water reduction).

Appendix I. Kruskall wallis (K) and median test to soil moisture at three distinct depths (0-10 cm, 20-30 cm and 40-50 cm). Below trace are p values and aboce “Z” values; in a riparian forest affected by dam on southern Brazil.

Results for Kruskal Wallis to 0-10 cm (df =11, N = 120) = 89.30, p <0.001 and median test (Chi-square = 80.99, df = 11, p<0.001).

	T0 ED	T1 ED	T3 ED	T0 HR	T1 HR	T3 HR	T0 ER	T1 ER	T3 ER	T0 HD	T1 HD	T3 HD
T0 ED	-	1.552	0.677	3.222	4.006	2.479	2.171	2.551	3.630	1.847	0.904	1.163
T1 ED	1.000	-	0.875	4.774	5.558	4.032	3.724	4.103	5.182	3.400	0.648	0.348
T3 ED	1.000	1.000	-	3.899	4.683	3.157	2.849	3.228	4.307	2.525	0.227	0.503
T0 HR	0.084	0.000	0.006	-	0.784	0.742	1.050	0.671	0.408	1.374	4.126	4.298
T1 HR	0.004	0.000	0.000	1.000	-	1.527	1.834	1.455	0.376	2.159	4.910	5.062
T3 HR	0.869	0.004	0.105	1.000	1.000	-	0.308	0.071	1.151	0.632	3.384	3.576
T0 ER	1.000	0.013	0.290	1.000	1.000	1.000	-	0.379	1.458	0.324	3.076	3.276
T1 ER	0.710	0.003	0.082	1.000	1.000	1.000	1.000	-	1.079	0.703	3.455	3.645
T3 ER	0.019	0.000	0.001	1.000	1.000	1.000	1.000	1.000	-	1.783	4.534	4.696
T0 HD	1.000	0.045	0.764	1.000	1.000	1.000	1.000	1.000	1.000	-	2.752	2.961
T1 HD	1.000	1.000	1.000	0.002	0.000	0.047	0.139	0.036	0.000	2.752	-	0.283
T3 HD	1.000	1.000	1.000	0.001	0.000	0.023	0.069	0.018	0.000	0.203	1.000	-

Results for Kruskal Wallis to 20-30 cm (df =11, N = 120) = 79.96, p<0.001 and median test (Chi-square = 87.20, df = 11, p<0.001).

	T0 ED	T1 ED	T3 ED	T0 HR	T1 HR	T3 HR	T0 ER	T1 ER	T3 ER	T0 HD	T1 HD	T3 HD
T0 ED	-	1.446	0.787	3.330	3.394	2.973	2.102	1.032	2.777	0.418	0.530	1.472
T1 ED	1.000	-	0.659	4.776	4.840	4.419	3.548	2.478	4.223	1.029	0.916	0.026
T3 ED	1.000	1.000	-	4.117	4.182	3.761	2.889	1.819	3.564	0.370	0.257	0.685
T0 HR	0.057	0.000	0.003	-	0.064	0.357	1.228	2.298	0.553	3.748	3.860	4.802
T1 HR	0.045	0.000	0.002	1.000	-	0.421	1.292	2.362	0.617	3.812	3.924	4.866
T3 HR	0.195	0.001	0.011	1.000	1.000	-	0.871	1.941	0.196	3.391	3.503	4.445
T0 ER	1.000	0.026	0.255	1.000	1.000	1.000	-	1.070	0.675	2.520	2.632	3.574
T1 ER	1.000	0.872	1.000	1.000	1.000	1.000	1.000	-	1.745	1.450	1.562	2.504
T3 ER	0.362	0.002	0.024	1.000	1.000	1.000	1.000	1.000	-	3.195	3.307	4.249
T0 HD	1.000	1.000	1.000	0.012	0.009	0.046	0.775	1.000	0.092	-	0.112	1.054
T1 HD	1.000	1.000	1.000	0.007	0.006	0.030	0.560	1.000	0.062	1.000	-	0.942
T3 HD	1.000	1.000	1.000	0.000	0.000	0.001	0.023	0.811	0.001	1.000	1.000	-

Results for Kruskal Wallis to 40-50 cm (df =11, N = 120) = 65.20, p<0.001 and median test (Chi-square = 65.98, df = 11, p<0.001).

	T0 ED	T1 ED	T3 ED	T0 HR	T1 HR	T3 HR	T0 ER	T1 ER	T3 ER	T0 HD	T1 HD	T3 HD
T0 ED	-	1.720	1.000	2.671	2.918	2.472	1.462	0.697	2.256	0.897	0.726	1.308
T1 ED	1.000	-	0.720	4.390	4.638	4.191	3.182	2.417	3.976	0.823	0.993	0.411
T3 ED	1.000	1.000	-	3.671	3.918	3.471	2.462	1.697	3.256	0.103	0.273	0.309
T0 HR	0.499	0.001	0.016	-	0.247	0.199	1.209	1.973	0.415	3.568	3.397	3.979
T1 HR	0.232	0.000	0.006	1.000	-	0.447	1.456	2.221	0.662	3.815	3.645	4.227
T3 HR	0.888	0.002	0.034	1.000	1.000	-	1.009	1.774	0.215	3.368	3.198	3.780
T0 ER	1.000	0.097	0.912	1.000	1.000	1.000	-	0.765	0.794	2.359	2.189	2.771
T1 ER	1.000	1.000	1.000	1.000	1.000	1.000	1.000	-	1.559	1.594	1.424	2.006
T3 ER	1.000	0.005	0.075	1.000	1.000	1.000	1.000	1.000	-	3.153	2.983	3.564
T0 HD	1.000	1.000	1.000	0.024	0.009	0.050	1.000	1.000	0.107	-	0.170	0.411
T1 HD	1.000	1.000	1.000	0.045	0.018	0.091	1.000	1.000	0.189	1.000	-	0.582
T3 HD	1.000	1.000	1.000	0.005	0.002	0.010	0.369	1.000	0.024	1.000	1.000	-

CHAPTER 2

Dynamic groups and riparian forest heterogeneity: distinct responses after river water flow reduction.

Resumo: Grupos de dinâmica e heterogeneidade de uma floresta ciliar: distintas respostas após redução na vazão de água de um rio

Mudanças no regime hídrico de um rio, como a redução do fluxo de água devido à implantação de barragens, podem afetar comunidades ciliares adjascentes. Contudo, devido à alta diversidade destes ecossistemas, é mais viável avaliar as mudanças ocorridas nessas comunidades por meio do estudo dos grupos de dinâmica (ou grupos de resposta), pois, apesar de diferentes florestas ciliares apresentarem diferentes espécies, são esperados semelhantes grupos de resposta para o impacto em questão. Este estudo tem por objetivo avaliar os efeitos da redução do fluxo de água em diferentes setores de uma floresta ciliar, testando a hipótese de que existem diferentes grupos de espécies com distintas respostas ao impacto, e que os setores mais próximos ao antigo leito do rio são os mais negativamente afetados. Foram testadas as alterações na umidade do solo próximo e distante do antigo leito do rio para confirmar os reais efeitos da redução do fluxo de água na umidade do solo, durante três anos. Para verificar as modificações na estrutura arbórea de diferentes setores da comunidade estudada, foi realizada uma DCA entre as parcelas de amostragem, utilizando-se a densidade das espécies por parcela. Além disso, grupos de resposta dinâmica foram analisados a partir das taxas de dinâmica das espécies, calculadas com base em quatro anos de efeito da redução da vazão de água do rio, para espécies com mínimo de 10 indivíduos. Tais grupos foram formados por meio da elaboração de um dendrograma de similaridade, baseado nestas taxas. A DCA apresentou três setores distintos: próximo ao rio, longe do rio e parcelas próximas a um córrego presente na área. Os resultados demonstraram uma maior redução na umidade do solo nos setores próximo à beira do rio por isso o setor próximo ao leito do rio foi de fato o mais negativamente afetado, apresentando alta mortalidade e perda em área basal. O dendrograma utilizado para a formação dos grupos de resposta apresentou quatro grupos, dos quais dois foram os mais prejudicados pela redução na umidade do solo, formados por espécies de sub-bosque e/ou associadas a ambientes ripários. Além de serem os mais afetados em setores próximos ao leito do rio, essas espécies também foram negativamente afetadas nos demais setores, demonstrando que a redução na vazão de água atingiu toda a comunidade estudada. O único grupo com resposta claramente positiva possui espécies de sub-bosque, generalistas, que estão ocupando espaços vagos na comunidade. Apesar de existir um grupo de dossel estável na comunidade, a floresta ciliar deve continuar sofrendo alterações até possuir espécies com menor dependência da água advinda do rio, principalmente no sub-bosque, e se assemelhar, no futuro, a uma floresta estacional.

Palavras chave: barragens; grupos de resposta; represamento.

Key-word: dam; response groups; impoundment.

Introduction

Sixty percent of the largest rivers on earth are strongly or moderately fragmented by dams and diversions (Ravenga et al. 2000), and most of dams are used for hydropower production (Truffer et al. 2003). Reservoirs and dams are responsible for a wide variety of environmental problems, as sediment retention (Nilsson & Berggren 2000), fish migration interruption (Friedl & Wuest 2002), extinction of many amphibians, birds and fishes associated with water (Ravenga et al. 2000) and greenhouse emission (Fearnside 2002). Moreover, there are many important direct effects to flora such as fragmentation by water storage (Jansson et al. 2000; Nilsson & Berggren 2000), changes in species composition, structure and/or richness (Nilsson & Grelsson 1995; Nilsson et al. 1997; Chapter 1).

Modifications on plants are dramatic because as primary producers, they represent the basal component of most ecosystems (Loreau et al. 2001), therefore changes on vegetation component will lead to changes in many other taxa. However, some dams and diversous ecological problems are not well scientifically studied and are neglected. Effects of drought in tree riparian communities after river diversion or damming are understudied, with few exceptions (Horner et al. 2009; Chapter 1).

Changes in streamflow, due to man-made diversion or reservoirs projects, can affect riparian community and water-stressed environment may decrease vegetation cover (Smith et al. 1991), by death of many trees or stems and, consequently, reduction in basal area (Chapter 1); or, in extreme cases, destroy riparian ecosystem altogether (Smith et al. 1991). In general, many riparian species are sensitive to reductions in stream flow or groundwater availability (Schume et al. 2004; Chapter 1), but not necessarily all tree species are affected by the same way. In a seasonal environment, for example, tree riparian species strongly dependent of the water from river cohabit with others capable of withstanding longer droughts (Chapter 1). This species richness is commom in riparian environments largely due heterogeneity explained by occasional floods, variations in topography, groundwater soils and upland floristic influence (Naiman et al. 1993; Nilsson & Svedmark 2002; Rocha et al. 2005; Rodrigues & Nave 2000); therefore, stream flow reduction will alter directly water available to plants (Chapter 1). This scenario is certainly disadvantageous to a particular group of river-dependent species and favors others (Chapter 1). However, this effect should be more severe near to the shore, a more dependent environment of river water, and less intense farther from river. Thus, understand the link between distinct riparian patches with floristic composition can elucidate some disturbance consequences. Still, high richness in riparian forests makes difficult extrapolations and forecasts about the future of these systems, after a serious disturbance.

A way to assess the complexity of these communities is grouping species with similar responses to disturbance, competitive abilities or that have similar effects on ecosystem process (Kelly & Bowler 2002; Lavorel & Garnier 2002; Lyon & Sagers 2003). Thus, many important ecosystem functions features can be revealed (Swaine & Whitmore 1988) as key habitats, zones of richness and vegetation interactions, facilitates predictions about communities changes (Craine et al. 2001; Girão et al. 2007; Lavorel & Garnier 2002; Lyon & Sagers 2003). However, establishing these groups is problematic due to many obstacles to develop an effective classification (Naeem & Wright 2003)**.** To better groups formation is useful to employ specific parameters, as dynamics rates, to an objective community evaluation (Gourlet-Fleury et al. 2005).

The water flow reduction provoke distinct species responses in a riparian forest (Chapter 1), object of this study, and we believe that the dynamic rates (mortality, recruitment, ingrowth and outgrowth) will perform in cohesive “dynamic response groups” to this odd disturbance. The use of mortality, recruitment and ingrowth is already important and usual in dynamic groups formation (Gourlet-Fleury et al. 2005). However, in our case, the use of outgrowth, related with mortality, is important due many stems broke but the individual remained alive. Thereby, a good response group comprehension can help us to lead management efforts in other riparian areas with similar disturbance and even in restoration projects, favoring specific species groups.

Considering the lack of studies which evaluates, with empirical data, the effects of water flow reduction caused by river diversion or dams to tree communities and the importance of these kind of achievement to society (irrigation and hydroelectric power), this work aims to quantify the effects of drought caused by a dam in a heterogeneous riparian forest. This dam reduced flow discharge in the river and the riparian forest which was near the riverbed and now is 10 to 50 meters away the waterline. This is the first work in the world who analyzes dynamic response groups related with environment heterogeneity after a marked water flow reduction in a tree community. Thus, we seek to clarify the follow hypothesis: 1) soil water moisture reduction effect was more severe near the shore than in distant patches; 2) riparian forest has a marked floristic heterogeneity that influences the forest dynamic after water flow reduction, then patches previously near to the shore will have higher dynamics rates; 3) there are distinct dynamics groups in the forest and those groups suffer different effects influenciated by the distinct riparian patches.

Material and Methods

Study area - This study was conducted in a riparian forest located between coordinates 18°47´40”S, 48°08’57”W and 18°47´51”S, 48°08’43”W located in the Amador Aguiar Dam influence area. For dam construction part of the river was diverted by a 27m spillway, reducing the flow in 10km sector since December 2005. The river sector where the water flow was reduced is called “Reduced Outflow Stretch” (Trecho de Vazão Reduzida – TVR Figure a1). Spillway reduces the water flow by $359m^3.s^{-1}$ to $7m^3.s^{-1}$ of water (CCBE 2005). Thus the riparian forest near the riverbed in 2005 (Figure 1a) is now about 10–50 meters farther from the direct water influence (Figure 1b). The average altitude is 595m with low slope. The climate is of Aw type (Koppen 1948) with dry winter and rainy summer with an average annual temperature of 22°C and average precipitation around 1595mm (Santos & Assunção 2006).

Soil is eutrophic argisoil and cambisoil, being very heterogeneous in nutrients and physical components along riparian forest with average acidity (Rodrigues et al. 2010). The floristic heterogeneity is noted too, partially by surroundings matrix, with distinct landscapes and conservation levels, being common the presence of species typical of dry soils *(Myracrodruon urundeuva* Allemão), cerrados (*Miconia albicans* (Sw.) Triana) and sometimes pastures like *Urochloa* sp. nearby the riparian forest (Rodrigues et al. 2010).

Soil analysis – To compare flow reduction effects on different distances from the riverbed we made Friedman test amongst five meters distant and 15 meters distant to the shore at all different depths and all seasons (middle of rainy, end of rainy, middle of dry and end of dry season) in three times, one year before water flow reduction by spillway (T0), one year (T1) and three years (T3) after this impoundment. “Friedman” is a non-parametric test used to detect differences among repeated multiples measures. Finally, to compare the flow reduction effects between five meters distance (close to the shore) and 15 meters distance (distant to the shore) we made Wilcoxon test. These analysis were made in Statistica Program (StatSoft 2005).

Riparian heterogeneity – A tree inventory was carried out during the 2005 year after rain season on 110 plots of 10x10m in the riparian forest, at 0-10 m and 10-20 m of distance to the river. To verify floristic gradients, before spillway construction in the riparian forest we made a DCA (Detrended Correspondence Analysis) using the species density per plot. The heterogeneity in riparian forest probably influences community answers to disturbances. After DCA, we made some analysis comparing riparian sectors showed DCA. We perform Shannon diversity index followed by Hutcheson t test to infer diversity variations between each riparian sectors.

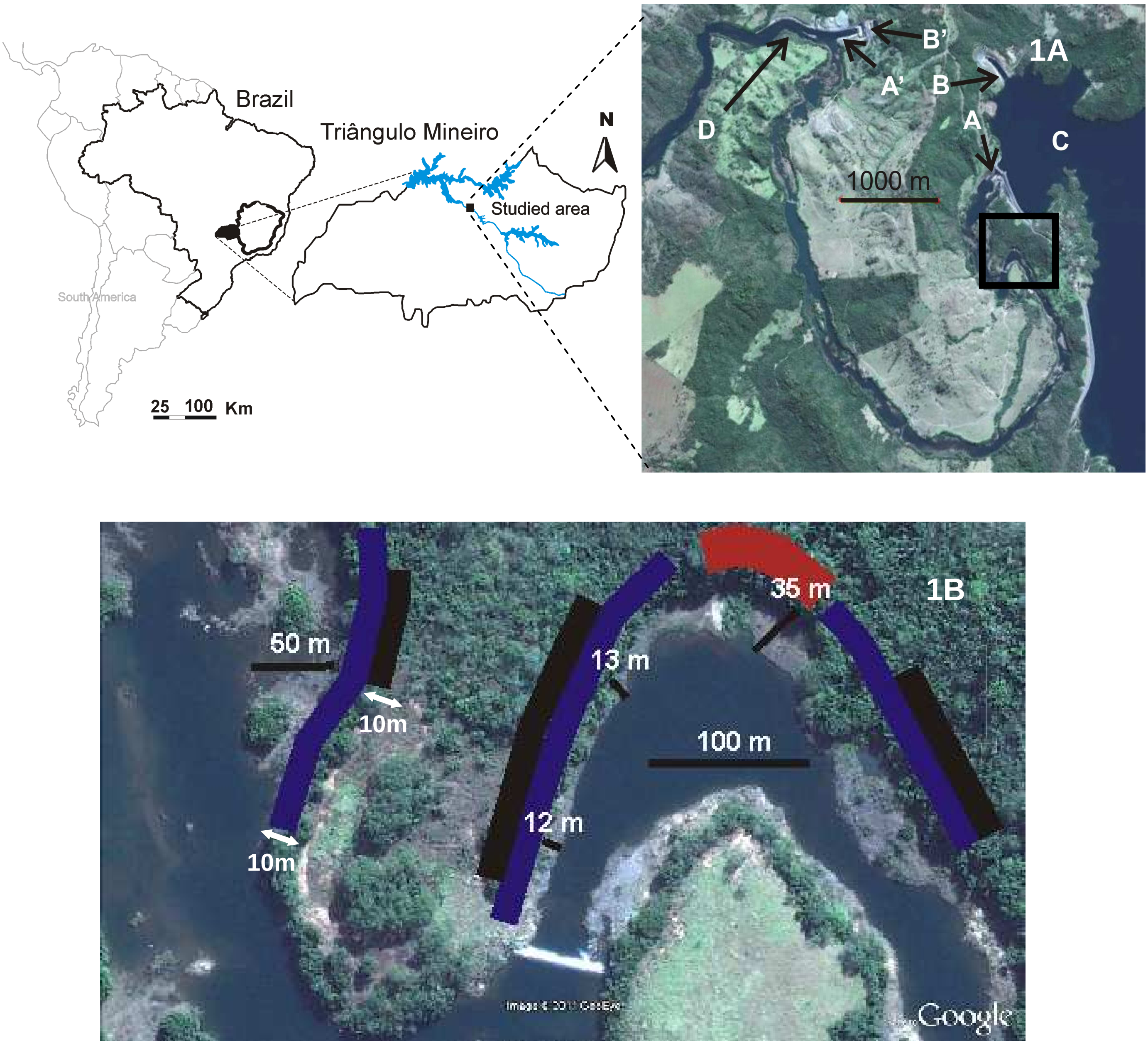

Figure 2.1 – Satellite image of a riparian forest on southern Brazil. “1A” Study area image with square showing plots locations. A = Spillway and the beginning of Reduced Outflow Stretch, A’ = end of Reduced Outflow Stretch, B = hydroeletric dam, B’ = end of hydroelectric dam, C = artificial lake created by dam, D = river patch returns to normal flow. The square ilustrates the study area. “1B” Riparian forest sectors: stream (red), riverside (blue) and distant to the shore (black) plots. Plots were allocated along the tracks.

To check data normality we perform Lilliefors test, but all the arboreal data did not presented normality. Then, we perform some Kruskall-Wallis with median test analysis (a non-parametric alternative test to ANOVA) to recruitment and mortality in T0-T4 period (where T0 is the measurements before flow reduction and T4 represents four years with low water flow), recruits and dead basal area in T0-T4 period, number of individuals in T0 and T4, and basal area in T0-T4 among plots groups found in DCA analysis. We did a Wilcoxon test between plot groups comparing plots individuals and basal area before and four years after damming and a Hutcheson t test to compare Shannon diversity index per plot group in T0 and T4. Then, to evaluate floristic differences between T0 and four years after dam construction, we perform a similarity comparison in these years based in species (T0 and T4), recruits and mortality with species number of individual values in each plot groups using Morisita-Horn index. All these analysis were made in FITOPAC program (Shepherd 2004) and Statistica program (StatSoft 2005).

Dynamics groups - To verify how the water flow reduction affected the species, we performed a cluster analysis using simple Euclidean distance and Ward method as clustering technique. We tried to detect statistically consistent “species dynamics groups” which suffered similar effects after water flow reduction. For this analysis we used mortality, recruitment, ingrowth and outgrowth species rates with at least five individuals in measurements and perform the cluster analysis (formulas are explained below (see Appendix 3 to species rates). To verify groups formation efficiency we perform a discriminant analysis made in Statistica Program (StatSoft 2005). The purpose of this analysis is test significant differences between groups (in this case, those resultant from cluster) and determine discriminant functions that allow reclassifying species wrongly classified in one of the groups obtained by the cluster. So this technique estimates the probability of correct classification (in this case, performed by the cluster) and reclassifies some species making more consistent groups (Gotelli and Ellison 2010). We still calculated all dynamics rates (mortality, recruitment, ingrowth, outgrowth, individuals turnover, basal area turnover, individuals net change, basal area net change and overall net change to each “species dynamics group”).

Annual mortality (m), recruitment (r), outgrowth (o) and ingrowth (i) rates were calculated in terms of annual exponensial rates (see formulas Sheil et al. 1995, Sheil et al. 2000 and Oliveira-Filho et al 2007). Outgrowth annual rates (o) refers to basal area of dead trees plus dead branches basal area of living trees (decrement) and ingrowth annual rates (i) refers to basal area of recruits plus growth in basal area of surviving trees (increment). To evaluate changes in forest we compute turnover rates to individuals and basal area groups through mortality-recruitment rates and outgrowth-ingrowth rates (Oliveira-Filho et al. 2007). Then we evaluated the net change (Korning & Balslev 1994) to individuals (ChN) and basal area (ChBA) and develop an overall net change based in average of ChN and ChBA rates (see all formulas on Chapter 1). All these analysis were done to each species groups.

Heterogeneity x Species dynamics groups - After found species dynamics groups and plot groups (sectors), we compare dynamic groups in each plot group (individuals and basal area per hectare) in T0 and T4. Finally to verify variations in four years of water flow reduction on each group, we made a Wilcoxon test (a paried student “t” alternative non parametric test) with each species groups in each plot group (using plots like samples) among T0-T4 period to number of individuals and basal area. To perform these analysis we use Statistica software (StatSoft 2005).

Results

Soil - When we compared results among close (five meters) and distant (15 meters) to the shore there was no significant differences among rainy months in all year periods analyzed and to all depths (Table 2.1). At 0 – 10 cm and 20 – 30 cm depths there are a clearly reduction of soil moisture in the middle and end of dry season near to the shore after flow reduction (Table 2.1). In 0 – 10 cm at middle of dry season soil moisture was reduced far from the shore too (Table 2.1), however less intense than near to the shore. Thus, flow reduction was more severe near to the shore than 15 m at distance, making soil drier in these locations. There are no significant differences distant to the shore in other drier periods. The lack of rain on dry season is strong and distant to the direct influence of the river, the soil had already little water available, and then the water flow reduction did not affect soil moisture at 15 m distance from the river. Soil moisture not varies at 40 – 50 cm soil deep in any distance and in any season; thereby flow reduction does not affect the soil layer below 40 cm.

The Wilcoxon test comproved spillway construction effected on soil moisture (Table 2.2). Before this building, at 0 – 10 cm and 20 – 30 cm in middle of rainy season and end of dry season, soil was moister close to the shore. However after flow reduction this environment became drier as far to the shore (Table 2.2). Thus the flow restriction affects both rainy and dry seasons up to 30 cm depth. A dramatic result (marginal significative but a important tendency) was noted in middle dry season because near the shore is becoming drier than far to the shore (Table 2.2). Therefore riverside probably was extremely dependent on the river moisture and changed greatly. At 40 – 50 cm the soil moisture near and far to river was similar before and after flow reduction which proves that the river water did not influence soil in deeper layers.

Table 2.1 - Friedman test before, one year and three year after dam construction to 0 – 10 cm, 20 – 30 cm and 40 – 50 cm deep near and far the shore in a riparian forest in southern Brazil. In bold, $p < 0.05$. In bold $p < 0.05$ and in italic $p < 0.10$.

	5m - Near		15m - Far	
	p	F	p	F
Season	0 - 10 cm			
Middle Rainy	0.819	0.4	0.165	3.6
End Rainy	*0.091*	4.8	0.247	2.8
Middle Dry	**0.022**	7.6	*0.074*	5.2
End Dry	**0.015**	8.4	0.165	3.6
	20 - 30 cm			
Middle Rainy	0.400	0.8	0.400	0.8
End Rainy	0.247	2.8	0.247	8.4
Middle Dry	**0.015**	2.8	0.692	0.7
End Dry	**0.015**	8.4	0.692	0.7
	40 - 50 cm			
MIddle Rainy	0.819	0.4	0.819	0.4
End Rainy	0.247	2.8	0.165	3.6
Middle Dry	0.819	0.4	0.247	2.8
End Dry	0.247	2.8	0.165	3.6

Table 2.2 - Wilcoxon test in three depths between near and distant to the shore for four season in a riparian forest in southern Brazil. MR = middle of rainy season, ER = end of rainy season, MD = middle of dry season, ED = end of dry season. In bold $p < 0.05$ and in italic $p < 0.10$. Median on Appendix 2.

		0 - 10 cm		20 - 30 cm		40 - 50 cm	
		p	Z	p	Z	p	Z
MR	T0	**0.043**	-2.023	**0.043**	-2.023	0.138	-1.483
	T1	0.893	0.135	0.345	-0.944	0.138	-1.483
	T3	0.500	-0.674	0.893	-0.135	0.225	-1.214
ER	T0	0.138	1.483	0.225	1.214	0.686	0.405
	T1	*0.080*	-1.753	0.500	-0.674	0.500	-0.674
	T3	0.345	-0.944	0.138	-1.483	0.225	-1.214
MD	T0	0.893	0.135	0.686	0.405	0.893	-0.135
	T1	*0.080*	1.753	0.893	-0.135	0.138	-1.483
	T3	*0.080*	1.753	0.138	1.483	0.345	0.944
ED	T0	**0.043**	-2.023	**0.043**	-2.023	**0.043**	-2.023
	T1	0.225	1.214	0.345	0.944	0.893	0.135
	T3	0.500	0.674	0.686	0.405	0.686	0.405

Floristic heterogeneity - The DCA, based in floristic composition and species density, show a separation between riverside plots, plots distant 10 - 20 m to the shore and a few plots near a stream (Figure 2.2) running along these plots. The eigenvalues were high explaining more than 77% of data variations.Some plots from riverside, distant to the shore and near the stream were similar in species, but most plots are floristically distinct then we choose to use the separation “a priori” in riverside plots, distant to the shore plots (10 – 20 m) and stream plots to perform the analysis (Figure 2.1b).

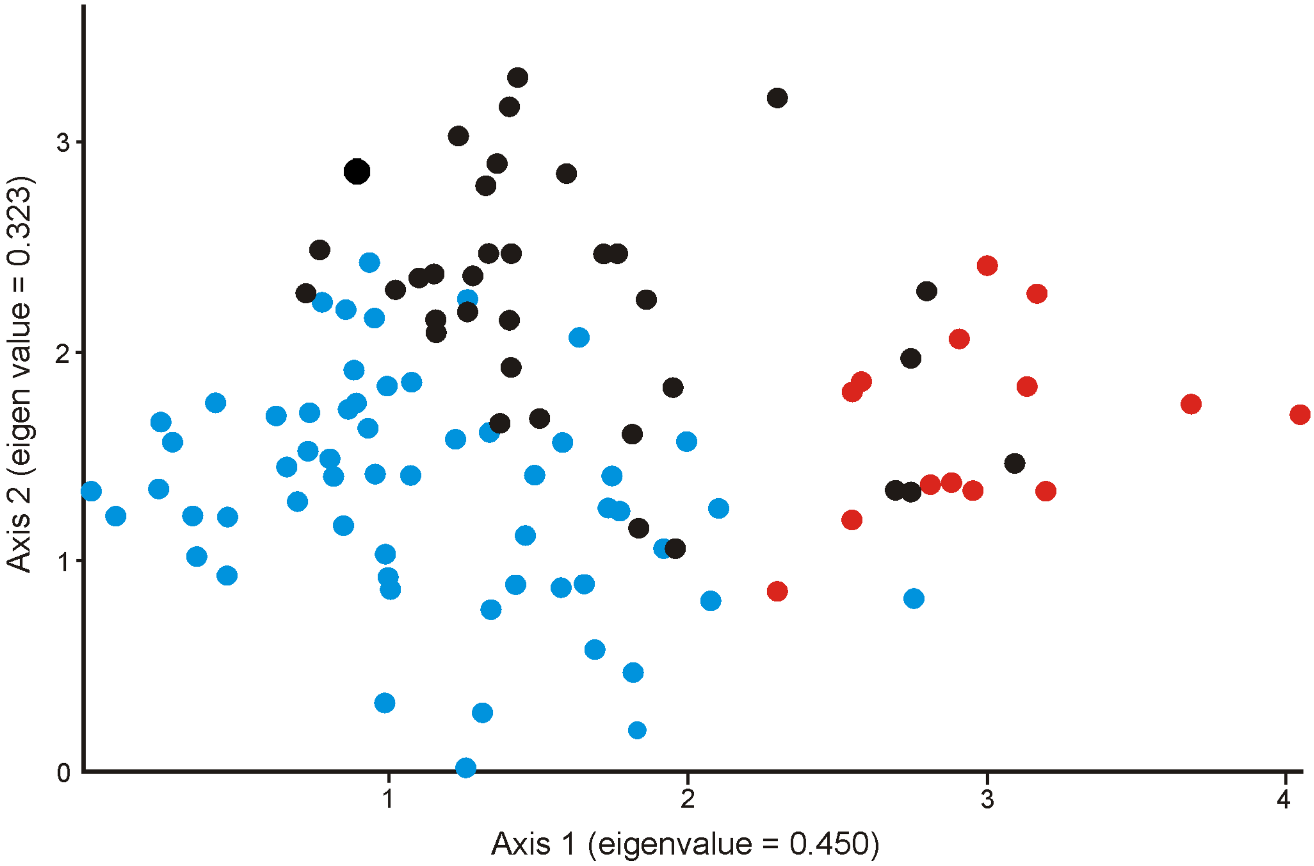

Figure 2.2 - DCA analysis based in species density per plot, among riparian plots in southern Brazil. Blue circles are plots near to the shore, black circles are plots distant to the shore and red circles are plots near a stream. Schematic representation is presented in Figure 1b.

Shannon diversity index did not vary in any riparian sectors (Hutcheson “t” test p > 0.1, df > 500 - Riverside, 3.335 in T0 and 3.378 in T4; Stream, 3.432 in T0 and -3.432 in T4; Distant, 3.491 in T0 and 3.429 in T4). However riverside, stream and distant to the shore sectors, demonstrated high structural variations in four years. Riverside is clearly more affected in these four years with low flow than stream and distant sectors. There were a great mortality (21.97% of individuals and 17.18% in basal area) and therefore a high reduction in individuals (11.52%) and basal area (8.73%) near the riverbed. In Stream and distant plots there was also a strong mortality (15.29% and 17.30%

in individuals), however with smaller depletion of basal area (8.18% and 5.35%). Recruitment did not follow the same pattern. Plots distant to the shore recruited more trees (14.02%) than riverside (11.81%) and stream (6.49%). sectors.

The dynamics rates confirm changes in forest structure; the riverside had the biggest mortality and outgrowth rates (Table 2.3) and distant the lowest ingrowth and outgrowh rates. These results reflect the large loss of stems and mortality of thick trees suffered by riverside plots (most of them were moisture dependent species and larger than 20 cm of diameter at breast height – Chapter 1). The same did not occur in stream and distant plots where most dead trees were thin. Thus turnover rates (Table 2.3) in riverside plot were higher than other two riparian forest sectors. Although all riparian forest is losing more trees than recruiting, but only in riverside this reduction was strong enough to produce a decrease in basal area (Table 2.3). In general only distant to the shore plots presented a positive overall net chance but in stream a small change was perceptible (Table 2.3).

Table 2.3 - Genenal changes and dynamics in riverside, stream and distant sectors four years after river reduction flow in a riparian forest in southern Brazil.

General data	Riverside	Stream	Distant
N° ind T0	842	170	393
N° ind T4	745	154	378
Mortality (N)	185	26	68
Recruits (N)	88	10	53
Basal area T0 (m^2)	28.75	6.48	11.22
Basal area T4 (m^2)	26.24	6.92	11.78
Dynamics			
Mortality (% ano^{-1})	6.01	4.06	4.64
Recruitment (% ano^{-1})	3.09	1.66	3.71
Ingrowth (% ano^{-1})	5.42	4.46	3.90
Outgrowth (% ano^{-1})	6.11	2.44	1.91
Individuals turnover (% ano^{-1})	4.55	2.86	4.17
Basal area turnover (% ano^{-1})	5.76	3.45	2.91
Individuals net change (% ano^{-1})	-5.94	-4.82	-1.93
Basal area net change (% ano^{-1})	-4.46	3.34	2.47
Overal net change (% ano^{-1})	-5.20	-0.74	0.27

The Kruskall-Wallis and median test were statistically significant only to basal area mortality in T0-T4 period and basal area in T0. These test showed a higher mortality in riverside than distant plots (Table 2.3 and 2.4). In same way, in T0, the riverside plots showed more basal area than distant plots (Table 2.3 and 2.4) but in T4 there were no significant differences. This occurred due to the strong loss in basal area in riverside plots (by mortality and stems loss) and basal area increase on 10 – 20 m distant to the shore plots. It is interesting to notice that the dead individuals near and far to the shore are more similar than those from the stream (Table 2.5) and the recruits in riverside and distant sector were more similar than stream sector. The new trees in riverside and distant to the shore were very similar in species (Table 2.5) promoting a rise in floristic similarity in these two sectors, nevertheless stream plots recruits was quite different and the stream sector became less similar than other two riparian sectors.

Table 2.4 - Kruskall Wallis test and median test (p\Z values) comparing riparian sectors before spillway construction (T0) and four years after water flow reduction (T4), in a riparian forest in southern Brazil. In bold, $p < 0.05$. BA = basal area.

Recruits	Riverside	Stream	Distant	BA Recruits	Riverside	Stream	Distant
Riverside	-	1.244	0.469	Riverside	-	0.819	0.732
Stream	0.640	-	1.465	Stream	1.000	-	1.251
Distant	1.000	0.429	-	Distant	1.000	0.632	-
Mortality	Riverside	Stream	Distant	BA Mortality	Riverside	Stream	Distant
Riverside	-	1.808	1.295	Riverside	-	1.784	2.601
Stream	0.212	-	0.790	Stream	0.223	-	0.117
Distant	0.587	1.000	-	Distant	**0.028**	1.000	-
N° Trees T0	Riverside	Stream	Distant	N° Trees T4	Riverside	Stream	Distant
Riverside	-	1.012	1.131	Riverside	-	0.851	0.144
Stream	0.934	-	0.168	Stream	1.000	-	0.882
Distant	0.775	1.000	-	Distant	1.000	1.000	-
BA T0	Riverside	Stream	Distant	BA T4	Riverside	Stream	Distant
Riverside	-	0.298	2.446	Riverside	-	0.165	1.036
Stream	1.000	-	1.382	Stream	1.000	-	0.854
Distant	**0.043**	0.501	-	Distant	0.901	1.000	-

Table 2.5 - Morisita-Horn similarity in riparian sectors (riverside, stream and distant) before water flow reduction (T0), four years after flow reduction (T4), dead trees and recruits, in a riparian forest in southern Brazil.

T0	Riverside	Stream	Distant	Dead	Riverside	Stream	Distant
Riverside	1			Riverside	1		
Stream	0.256	1		Stream	0.201	1	
Distant	0.547	0.375	1	Distant	0.487	0.319	1
T4	Riverside	Stream	Distant	Recruits	Riverside	Stream	Distant
Riverside	1			Riverside	1		
Stream	0.235	1		Stream	0.084	1	
Distant	0.559	0.296	1	Distant	0.624	0.022	1

When compare variation in each plot per riparian sector, only in riverside there were significant differences in number of individuals however to basal area only distant to the shore and in stream the test results were significant and marginally significant (Table 2.6). In fact the basal area decreasing (Table 2.3) in riverside but while some plots increases, others presents strong reduction and this affects the analysis. In stream and distant to the shore there were little but constant increase in basal area per plot (Table 2.6). Thus, distant and stream plots were more constant with no strongly variations unlike riverside plots were very variable.

Table 2.6 - Wilcoxon comparing changes during T0 – T4 in riverside, stream and distant sectors plots, in a riparian forest in southern Brazil. In bold $p < 0.05$ and in italic $p < 0.10$.

	N° Ind		AB	
	Z	p	Z	p
Riverside	-3.120	**0.002**	-0.445	0.656
Stream	-1.585	0.113	1.726	*0.084*
Distant	-0.727	0.467	2.225	**0.026**

Dynamics Groups - Cluster formed four groups with 18 (G1), 14 (G2), 11 (G3) and 6 (G4) species (Figure 2.3 - The species list are in Appendix III). However discriminant analysis reorganized two species from their groups, *Cordieria sessilis* left G4 and went to G1, and *Bauhinia ungulata* left G3 and went to G2 (see Figure 2.4). Discriminant analysis confirmed efficiency in group formation by cluster technique ($F_{12,111}$=19.29, p<0.001). To discriminant analysis three rates showed significant values and only to recruitment was marginally significant (Table 2.7). Still the total efficiency in groups formation was 95.6% and superior than 80 for all groups with only two species changing group (Table 2.8). The two first roots were significant with eingenvalue of 4.16 and 2.37 (p<0.001), then first two functions were high correlated to groups formation (90% of canonical correlation values – Table 2.9).

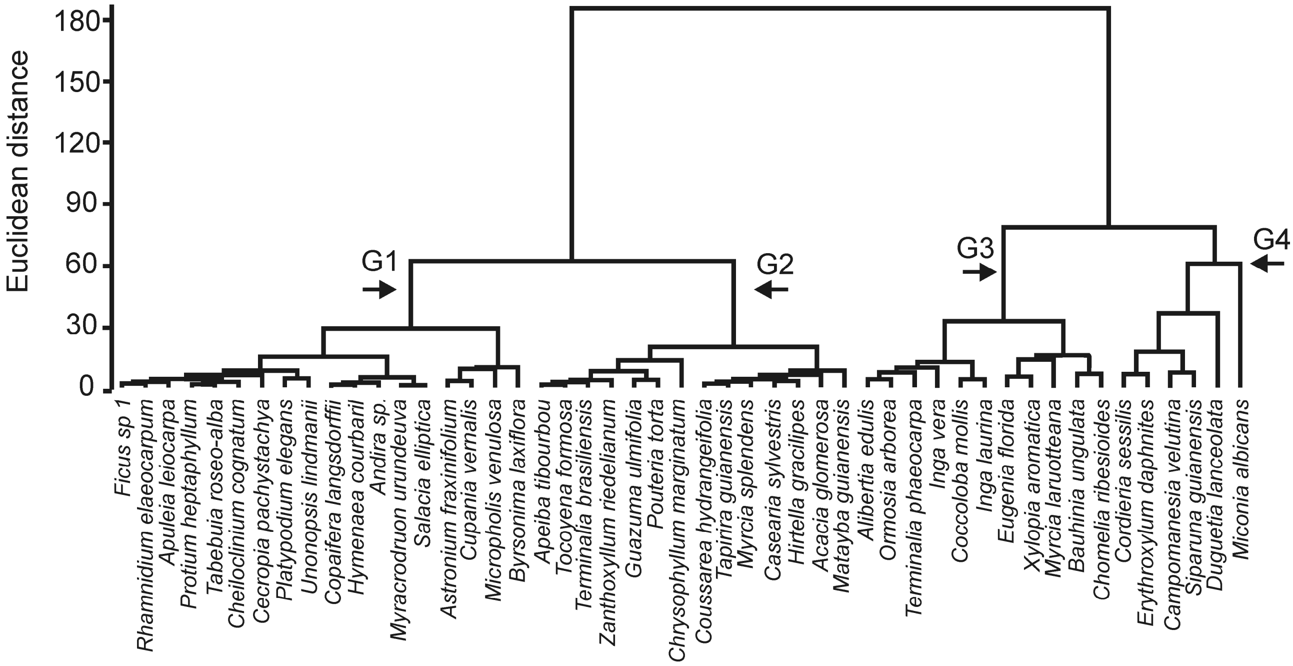

Figure 2.3 - Cluster analysis using Euclidian distance and ward method. Groups were formed based in species annual dynamic rates (mortality, recruitment, outgrowth and ingrowth), in a riparian in southern Brazil. G1 = group 1; G2 = group 2; G3 = group 3 and G4 = group 4. Dynamic rates used to groups formation and species phytossociological parameters were summarized on Appendix 3.

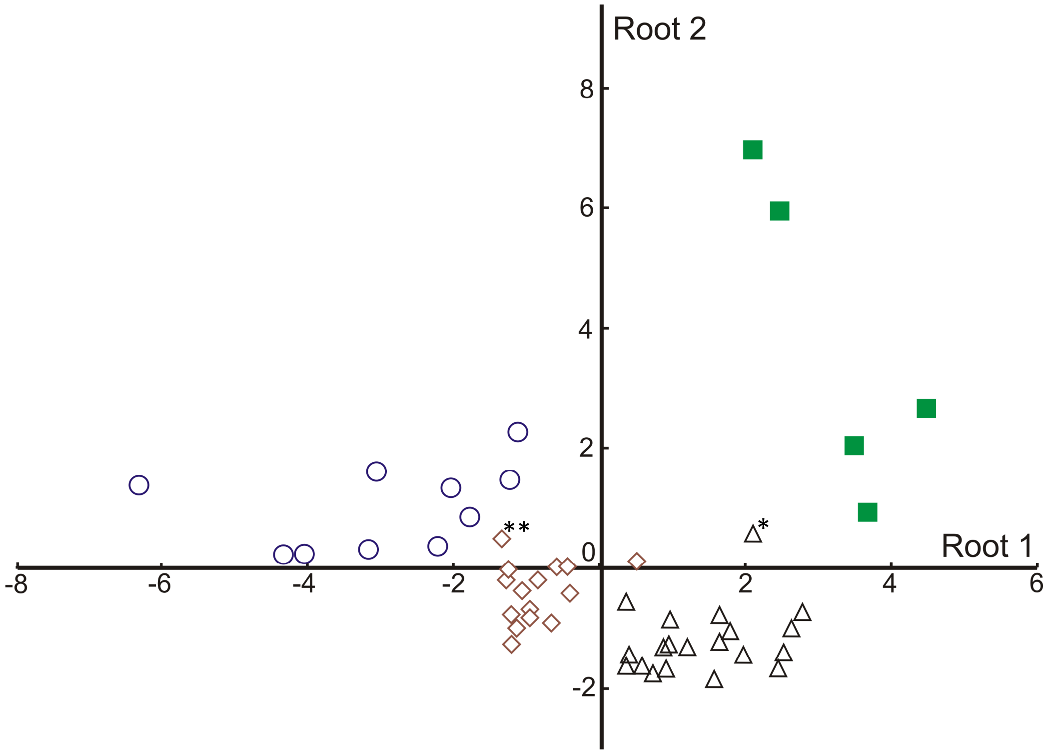

Figure 2.4 - Discriminant analysis based in species annual dynamic rates (mortality, recruitment, outgrowth and ingrowth) in a riparian forest in southern Brazil. Black triangle = group 1, Brown diamond = group 2, Blue circles = group 3; Green square = group 4; * = *Cordieria sessisis* (discriminant analysis changes this species from G4 to G1) and ** = *Bauhinia ungulate* (discriminant analysis changes this species from G3 to G2).

Table 2.7 - Discriminant function analysis summary to four rates in a riparian forest in southern Brazil. Wilks' Lambda: 0.05; F (12,111) = 19.289 p<0.001. In bold p < 0.05 and in italic p < 0.10.

N=49	*Wilks'*	*Partial*	*F-remove*	*p-level*
Mortality	0.077	0.665	7.038	**0.001**
OutGrowth	0.109	0.466	16.050	**0.001**
Recruitment	0.060	0.851	2.452	*0.076*
Ingrowth	0.154	0.331	28.326	**0.001**

Table 2.8 - Coherence between cluster groups and discriminant analysis groups in a riparian forest in southern Brazil.

		p=0.367	p=0.286	p=0.225	p=0.122
	Percent	G1	G2	G3	G4
G1	100.00	18	0	0	0
G2	100.00	0	14	0	0
G3	90.91	0	1	10	0
G4	83.33	1	0	0	5
Total	95.92	19	15	10	5

Table 2.9 - Discriminant function analysis summary to two first discriminant roots in a riparian forest in southern Brazil. In bold p < 0.05.

	Eigen-value	Canonical	Wilks'	Chi-Sqr.	df	p-level
1	4.15	0.90	0.050	130.86	12	**0.001**
2	2.37	0.84	0.263	58.70	6	**0.001**

Comparing their parameters, G2, G3 and G4 present high mortality and outgrowth rates; indeed G4 has the highest recruitment and ingrowth rates and owned a positive balance in these four years (Table 2.10). Further, G3 presented the highest mortality and outgrowth rates, and is the most affected by moist reduction (Table 2.10). G1 was the only group that showed low mortality/outgrowth rates. This group is formed mainly by canopy species with low or medium recruitment and ingrowth rates and was the less impacted group after the water flow reduction. This group can be considered in expansion in the forest.

Tabela 2.10 - Dinamics rates for "dynamics groups" in four years of water flow reduction in a riparian forest in southern Brazil. Ni = number of individuals, BA = basal area, Mort = mortality, Recr = recruitment, TN = individuals turnover, TAB = basal area turnover, ChN = individuals net change, ChBA = basal area net change and ONC = overall net change.

Group	Ni T0	Ni T4	BA T0	BA T4	Mort	Recr	Outg	Ingr	TN	TAB	ChN	ChBA	ONC
G1	392	404	22.69	24.82	1.22	5.73	1.89	5.53	3.48	3.71	0.76	2.27	1.51
G2	515	435	11.35	10.00	6.32	1.38	7.46	6.10	3.85	6.78	-4.13	-3.12	-3.62
G3	309	241	8.54	5.71	10.45	3.8	16.25	8.18	7.13	12.22	-6.02	-9.57	-7.80
G4	108	127	0.52	0.55	9.20	9.66	19.4	30.84	9.43	25.12	4.13	1.41	2.77

Group 1 – This group was formed by generalist canopy species with low mortality and outgrowth (Table 2.10) rates and there are no species related with moist environment. This dynamic group can be considered the most stable in this riparian forest and less affected by four years of water flow reduction (species list are in Appendix III).

Group 2 – This group was formed by many sub-canopy and understory species. Few species are related with moist environments (*Hirtella gracilipes, Coussarea hydrangeifolia and Chrysophyllum marginatum*). Mortality and outgrowth rates were superior to recruitment and ingrowth, and then this group presented a negative overall net value (species list are in Appendix III).

Group 3 – This group was formed by the most negative affected understory species and many species related with moist environments species (*Inga vera, Inga marginata, Eugenia florida, Alibertia edulis* – first three are sub-canopy/canopy species). The mortality and outgrowth rates were very superior to recruitment and ingrowth rates (Table 2.10). The negative net changes were superior to six, revealing the most negatively affected group by water flow reduction. Besides high mortality, many individuals lost stems increasing basal area loss.

Group 4 – This group was formed by only fast dynamic understory species. Despite of high mortality, the recruitment was higher (Table 2.10). This group presented a very high outgrowth and ingrowth rates, however the recruitment basal area and the older trees high growth compensate basal area lost by dead trees. Compared to other two groups with many understory species (G2 and G3), this is the only understory group which had a positive net change in the tree community.

Dynamic species groups x Riparian Forest Sectors - The number of individuals in each dynamic group varied in different riparian forest sectors. G1, a canopy group, did not vary greatly in any sector (Table 2.11), however the Wilcoxon test evidenced a raise in number of trees distant to the shore (Table 2.12). Two groups, G2 and G3, due to high mortality, decreased their participation in all sectors (Table 2.11) and these results were statistically confirmed (Table 2.12) though G3 did not vary greatly in stream sector. In both cases, the largest reduction in trees occurred in riverside (Table 2.12). G4 is the only group to did not vary over four years of reduced water flow (Table 2.12).

Tabela 2.11 - Density and Basal area to four dynamics groups found in cluster and discriminant analisys in three riparian sectors, in southern Brazil.

	G1		G2		G3		G4	
Density (ha^{-1})	T0	T4	T0	T4	T0	T4	T0	T4
Riverside	331	336	520	442	359	277	66	72
Stream	450	450	450	364	150	121	36	43
Distant	356	384	372	316	181	147	191	234
	G1		G2		G3		G4	
Basal area ($m^2 . ha^{-1}$)	T0	T4	T0	T4	T0	T4	T0	T4
Riverside	19.06	20.84	12.61	10.97	10.38	6.06	0.28	0.31
Stream	24.64	26.64	13.64	11.50	3.93	4.07	0.29	0.43
Distant	21.28	23.50	4.28	4.25	4.22	3.94	0.91	0.91

Tabela 2.12 - Wilcoxon test to number of individuals and basal área among dynamics groups in each riparian sectors in southern Brazil. In bold, $p < 0.05$.

	G1		G2		G3		G4	
Number of individuals	Z	p	Z	p	Z	p	Z	p
Riverside	0.837	0.403	-3.647	**0.000**	-4.240	**0.000**	0.522	0.601
Stream	0.000	1.000	-1.997	**0.046**	-1.633	0.102	1.000	0.317
Distant	2.324	**0.020**	-2.451	**0.014**	-2.392	**0.017**	1.391	0.164
	G1		G2		G3		G4	
Basal area	Z	p	Z	p	Z	p	Z	p
Riverside	4.380	**0.001**	-0.889	0.374	-3.327	**0.001**	1.265	0.206
Stream	3.170	**0.002**	-0.874	0.382	0.280	0.779	1.604	0.109
Distant	4.186	**0.001**	0.874	0.382	1.345	0.178	0.686	0.493

By comparing dynamics groups dominance in plot sectors (Tabela 2.11) there were significant increase in G1 dominance in all three plot groups but was more remarkable in riverside (Table 2.11 and 2.12). The opposite occurs with G3 which showed strong significant decrease in riverside. G2 dominance was more severe reduced more in riverside too (Table 2.11) but not statistically significant (Table 2.12). G4 variations, despite of increase in basal area were not significant. These results explain the wide variation seen in riverside. In riverside plots dominated by G1 basal area tends to increase while in plots dominated by G2 or G3 basal area tends to exhibit strong decrease.

Discussion

Water flow reduction decreases soil moisture in dry months until 30 cm deep and even in some rainy months, soil moisture was diminished, however more harsh near to the shore, confirming our hypothesis. More, drought induced by spillway reduces soil moisture in riverside making these sectors as arid as distant plots in both dry and wet months. A study in other riparian forest shows that major root biomass are concentrated until 30 cm (Kiley & Schneider 2005) hence many trees were affected by soil moist reduction. Impacts of drought could be complex due soil water reserves (Borchert 1994, 1998; Nepstad et al. 1994) and moisture near to the river was depleted by lack of water near the riverbank. Surface water and groundwater are important because strongly influence species composition, community structure and biological diversity (Ehleringer & Dawson 1992; Fujieda et al. 1997; Munoz-Reinoso 2001; Stromberg et al. 1996)**.** The reduce in soil water available can negatively affect the biomass conversion (Breshears & Barnes 1999) and consequently decreases riparian forest species performance and survival (Smith et al. 1991), increasing mortality**.** More severe disturbance makes community more unstable and quickly changing their structure (Machado & Oliveira-Filho 2010), therefore, in general, water flow reduction could be considered a intense disturbance in riverbed patches due to higher turnover of individuals and basal area but not to distant and stream sectors.

High mortality rates in riverbed is transforming this riparian sectors in more open areas with smaller density and basal area, therefore new patches are created and less dense species could become important (Chapter 1). Gaps created by dead of large trees, per example, favor establishment of pioneer and light demanding species (Oliveira et al. 1997; Vale et al. 2009), therefore, prolonged drought and their negatively effects on some species will lead to altered populations structures changing community composition (Smith et al. 1991). Underground water use to avoid drought are an ecophysiological traits in seasonal forest (Borchert 1998) but is not required in moist environments, therefore, many trees of moist adapted species in riverside should keep dying and open spaces to be occupied by species better adapted to prolonged drought as G4.

This group is constituted by species commonly found in seasonal environments. *Miconia albicans*, *Siparuna guianensis* and *Erythroxylum daphnites* are presents in savannas (Bridgewater et al. 2004; Ratter et al. 2003), and *Campomanesia velutina*, *Duguetia lanceolata* and *Siparura guianensis* are common in seasonal dry forest (Lopes et al 2011). These understory species have small heights and small species have a faster dynamics than canopy species due short lifespan (Lieberman et al. 1985; Manokaran & Kochummen 1987). However the G4 growth rates are too high and flow reduction should speed up their mortality and recruitment. This group has a positive net change in all riparian sectors and probably will gradually replacing some G2 and G3 understory moist species group. This process should be slow and take many years but will probably occur. These two groups were the most negative affected in four years, due low recruitment rates and high mortality and loss of stems.

Smaller trees are more prone to die than large trees after drought (Horner et al. 2009) and our work confirms this trend. Plants with different traits often differ in their responses to disturbance and/or stress as drought (Macgillivray et al. 1995) and smaller trees have less access to deeper ground water stocks than larger ones, and this is particularly critical in seasonal forests (Fortini et al. 2003). When riparian forest, before with plenty of water, became subject to a seasonal climate with severe dry months, only plots near to river statically become drier. However in all riparian sectors, riverside stream and distant, the density of G2 and G3 declines more than 15% after four years of spillway construction demonstrating flow reduction effects advanced to the distant sectors and even affected environment near the stream. Nevertheless basal area affected shows a very different pattern. G2 basal area reduced more than 13% in riverside and stream (not statistically) but remained stable in distant plots (reduction of 0.07%). Otherwise G3 had reduced their basal area in 41% in riverside but maintain stable in distant plots (less than 7% of basal area loss) and remained stable on stream plots. Comparing these two groups we could argue that flow reduction negatively affects G2 and G3 in all riparian sectors severely causing many small trees deaths (82% of died trees with diameter < 20 cm died belong these two groups). However in riverside more big trees died (> 20 cm of diameter), or at least, lose stems in G3 and loss of basal area were superior in riverside to G3. Dead and mutilated branches of big trees can hit smaller ones creating gaps that are occupied by recruits and growing trees (Deng et al. 2008; Swaine et al. 1990; Vale et al. 2009). We know that water and light would be inextricably linked with dynamics of smaller trees (Machado & Oliveira-Filho 2010) and gaps with low water availablility could lead to density and basal area reduction over the years. Despite of recruitment be superior in G1(canopy group) and G4 (understory group) and the ingrowth of G2 and G3 be high, their not compensate the community lost of individuals and basal area. Then the negative effects of water flow reduction in G2 and G3 was superior to any positive effects in G1 and G4 groups.

The more stable group (G1) consists in canopy species and few non-affected understory due to low mortality in all riparian forest sectors. Mortality of this group was concentrated in riverside (68% of total mortality of this group) but still much smaller than G2 and G3 in this sector (10% of total mortality in riverside). Canopy species usually presents deeper roots and these kinds of plants are the most able to obtain water in deeper soil layers (Breshears & Barnes 1999) even in drought months (Saha et al. 2008). We believe that low mortality of this canopy group is very important to forest moisture retention and will be a good group in recovery of areas with similar drought impacts by three reasons: 1) mortality itself is a disadvantage to water soil storage due to evapotranspiration increase (Breshears et al. 1997) outside tree cover (Joffre & Rambal 1993), thus plant composition influences soil moisture distribution (Breshears & Barnes 1999) and low mortality prevents moisture loss to the atmosphere; 2) this group has a good recruitment and ingrowth rates despite of riparian sector, then even saplings still survive after river water flow reduction and 3) canopy trees are critically important sources of fruits, flowers and shelters to animal populations due their reproductively dominance and strong influence in forest structure (McNeely 2002).

In general, riverbed sector become as drier as distant to the shore and this implies in water reduction distribution and consequently soil heterogeneity. Floristic gradients in forest are related with soil, slope, topography, and water available (Gartlan et al. 1986; Oliveira-Filho et al. 1998; Oliveira-Filho et al. 1994; van den Berg & Oliveira-Filho 1999), due to the diversity of microclimates generated enhance the diversity and productivity (Naiman & Decamps 1997). In this riparian forest, only water available was modified and severely reduced. Before flow regulation was expected that the river provides high soil moisture in many riparian forest patches. Occasional floods, per example, can create distinct regeneration niches that facilitate species coexistence (Naiman & Decamps 1997) and this kind of event became null**.** Therefore, water absence transformed moist patches making them similar to those outlying areas to the river, reducing moisture heterogeneity. The short term consequences were mortality of water dependent groups, especially from understory, and recruitment and high turnover of some understory generalist species. However the importance of these generalist species still low and cannot compensate the high loss in trees and basal area by other species. We predict, as a long term consequence, that the species of water dependent group will be replaced by generalist groups (especially in understory) and big canopy species will be maintained.

Considering that early life stages and herbs are likely to be particularly sensitive and vulnerable to extreme water available conditions (Fenner et al. 1985; Leuzinger et al. 2005; Stefan et al. 2007), we believe in more several changes to seedlings, saplings and non-arboreal flora than arboreal component in all riparian environments, but mainly to water dependent patches. Some indications of this tendency could be made by high recruitment and mortality of understory species (those with fast dynamic in the community), in riverside plots. More, the understory species recruited in riverside plots were common to the distant sectors, and the riverbed became floristically more similar to distant patches. We can conclude that the riparian patches nearest the river will be replaced, over the years, by less water dependent species and riparian moist forest will become in other physiognomy, less water dependent in dry season (probably a semideciduous dry forest, a kind of forest common in regions with marked seasonal climate and independent of direct river influence). Finally, If water flow reduction affected understory trees, will certainly cause a negative effects many herbs and scrubs changing the entire local flora.

Dynamics analysis based on groups and riparian sectors discrimination clarified spillway effects in riparian forest, confirming our hypothesis about the existence of species with distinct responses to dam impacts. First, a canopy group was not affected by water flow reduction in any riparian sector, including raise their basal area, then the spillway effects are positive to this group. Second, stream plots are a positive/stable place to three species groups (G1, G3 and G4), but many G3 trees died and even near the stream was negative affected by river flow reduction. Stream water certainly reduces drought effects, and not substantial drought could not cause severe plant mortality (Breshears et al. 2005; Gitlin et al. 2006; Gobron et al. 2005). However even though in general there were no major modifications, we demonstrated that at least two group of species suffering from the reduction of river water, even in plots close to the stream, then generalization should be cautions in any study. Third, G4 is a instable group with high turnover rates but also with major positive net values, therefore, this group should has the faster dynamic in community, and we predicts that this group will dominate the understory in the future. Fourth, our work focus in tree community and probably the more stable portion of entire vegetation, however in only four years of reduced water flow, the tree community was severely modified and this implies in more changes during next years.

Apendice II – Median values to soil moisture at three depths (0-10 cm, 20-30 cm and 40-50 cm) along seasons near (5m) and distant (15m) to the shore. T0 = before damming, T1 = one year after damming and T3 = three years after damming in a riparian forest affected by dam on southern Brazil.

		0 - 10 cm		20 - 30 cm		40 -50 cm	
		Near	Far	Near	Far	Near	Far
Half Rainy	T0	24.00	30.77	21.52	27.85	16.67	16.92
	T1	30.95	21.95	17.50	14.29	16.28	13.21
	T3	37.25	28.85	30.19	23.33	28.07	21.88
End Rainy	T0	32.08	29.55	32.00	22.64	28.00	21.43
	T1	30.43	32.73	32.08	27.12	32.76	21.74
	T3	28.77	24.68	24.59	25.76	29.23	21.62
Half Dry	T0	22.41	22.92	10.00	12.07	12.07	12.70
	T1	10.64	12.73	12.00	11.48	17.02	10.17
	T3	7.04	13.04	5.75	9.86	9.09	12.16
End Dry	T0	14.29	11.86	13.21	10.17	17.02	10.45
	T1	8.93	9.68	9.26	10.77	11.54	10.45
	T3	12.00	14.06	10.77	11.67	10.61	14.04

Appendix III – Tree species parameters to number of individuals and basal area, and dynamics rates to all species with at least five individuals in a riparian forest in southern Brazil. N0 = number of individuals in T0 (before water flow reduction), D = number of dead trees, R = number of recruits, N4 = number of individuals in T4 (four years after water flow reduction), BAT0 = basal area in T0, BAD = basal area of dead individuals, DcBA = decrement of basal area, IcBA = increment of basal area, BAR = basal area of recruits, BAT4 = basal area in T4, M = mortality rate, O = outgrowth rate, R = recruitment rate and I = Ingrowth rate.

Species	Number of individuals				Basal area (cm^2)						Dynamics rates used in analysis				Dynamic group
	N0	D	R	N4	BAT0	BAD	DcBA	IcBA	BAR	BAT4	M	O	R	I	
Acacia glomerosa	10	2	-	8	866.82	114.91	-	234.79	-	1005.80	5.43	6.79	-	10.19	G2
Alibertia edulis	74	18	6	62	7047.88	774.47	-1395.87	444.08	59.78	4874.39	6.73	13.49	2.51	6.16	G3
Cordieria sessilis	17	2	4	19	861.21	42.10	-143.80	208.37	114.37	897.77	3.08	8.72	5.74	14.33	G1
Andira sp.	21	1	1	21	8015.64	-	-67.64	332.79	31.83	8085.95	1.21	1.43	1.21	1.75	G1
Apeiba tibourbou	5	1	-	4	2136.83	-	-	195.74	-	1906.13	5.43	6.05	-	3.32	G2
Apuleia leiocarpa	25	1	3	27	9874.17	528.57	-	1946.12	72.22	12021.08	1.02	1.90	2.90	6.79	G1
Astronium fraxinifolium	5	-	2	7	841.73	-	-	94.76	21.66	998.00	-	-	8.07	5.47	G1
Bauhinia ungulata	40	17	4	27	2266.46	136.63	-10.21	568.48	91.22	2112.70	12.92	14.44	3.93	14.60	G2
Byrsonima laxiflora	13	2	6	17	1493.61	40.29	-	258.49	-	1846.47	4.09	2.73	10.31	11.24	G1
Campomanesia velutina	8	2	2	8	288.43	-	-16.91	45.14	17.90	305.38	6.94	11.47	6.94	24.12	G4
Casearia sylvestris	25	7	2	20	1750.08	350.76	-0.44	180.96	20.37	1532.78	7.88	8.65	2.60	6.37	G2
Cecropia pachystachya	7	-	-	7	553.60	-	-	85.10	-	782.70	-	-	-	8.29	G1
Cheiloclinum cognatum	14	-	1	15	2923.21	-	-83.20	151.93	-	3057.24	-	2.62	1.71	3.87	G1
Chomelia ribesioides	8	4	1	5	219.89	19.12	-	61.25	19.12	196.08	15.91	17.25	5.43	19.10	G3
Chrysophyllum marginatum	5	2	-	3	1291.48	17.90	-	151.97	-	1442.60	11.99	1.34	-	4.03	G2
Coccoloba mollis	59	19	3	43	6131.97	1760.03	-65.53	868.96	81.25	3838.07	9.26	19.41	1.79	10.11	G3
Copaifera langsdorffii	49	5	2	46	65942.73	2451.08	-384.72	4091.78	23.00	70285.98	2.65	1.52	1.11	3.10	G1
Coussarea hydrangeifolia	63	13	3	53	5280.79	774.66	-41.34	573.85	62.49	4746.45	5.61	8.42	1.45	6.34	G2
Cupania vernalis	4	-	1	5	1347.96	-	-	167.59	20.37	1729.95	-	0.15	5.43	6.54	G1
Duguetia lanceolata	5	-	-	5	811.01	-	-	727.48	-	962.73	-	31.07	-	33.96	G4
Erythroxylum daphnites	10	2	2	10	271.96	30.26	-	74.50	23.00	364.84	5.43	6.39	5.43	19.83	G4
Eugenia florida	8	3	2	7	1054.10	87.24	-	207.88	47.85	786.98	11.09	14.97	8.07	10.57	G3

Ficus sp1	12	-	1	13	23104.75	-	-1401.44	607.15	31.83	27072.43	-	1.61	1.98	5.47	G1
Guazuma ulmifolia	7	1	-	6	1240.17	362.57	-	136.77	-	1056.43	3.78	8.28	-	4.53	G2
Hirtella gracilipes	174	36	13	151	44130.26	5679.17	-4069.07	3452.00	210.36	36784.42	5.63	10.20	2.23	6.30	G2
Hymenaea courbaril	35	3	1	33	56417.67	143.74	-188.70	3100.45	20.37	61322.77	2.22	0.45	0.77	2.52	G1
Inga laurina	70	15	4	59	44943.54	8765.40	-1902.53	1459.33	25.78	31253.56	5.85	17.49	1.74	9.77	G3
Inga vera	38	16	1	23	10018.68	1288.26	-633.38	503.19	-	6313.22	12.77	14.03	1.11	3.60	G3
Matayba guianensis	34	10	7	31	4304.04	592.83	-6.60	490.02	87.06	3967.46	8.34	9.33	6.20	8.72	G2
Miconia albicans	7	5	4	6	345.39	244.08	-14.72	69.57	57.40	188.96	26.89	33.83	24.02	47.67	G4
Micropholis venulosa	7	-	3	10	4302.09	-	-45.16	346.94	66.21	4287.13	-	2.82	13.06	3.28	G1
Myracrodruon urundeuva	7	-	-	7	2288.32	-	-	126.39	-	2413.10	-	0.23	-	1.55	G1
Myrcia splendes	33	7	1	27	1492.45	210.92	-40.72	193.99	-	1443.57	5.79	8.34	0.94	8.08	G2
Myrcia laruotteana	10	6	1	5	1429.01	131.58	-355.63	58.93	20.37	706.77	20.47	21.19	5.43	6.91	G3
Ormosia arborea	17	2	1	16	4506.74	-	-	370.07	-	3333.07	3.08	13.66	1.60	7.09	G3
Platypodium elegans	20	2	1	19	5181.97	484.15	-450.59	914.24	20.37	5314.61	2.60	5.61	1.34	6.32	G1
Pouteria torta	18	1	1	18	11769.36	-	-171.67	450.07	-	9249.07	1.42	8.01	1.42	2.71	G2
Protium heptaphyllum	94	5	6	95	32266.54	64.02	-1207.19	2845.91	102.36	35044.13	1.36	1.92	1.62	4.04	G1
Rhaminidium elaeocarpum	8	-	1	9	769.01	-	-	98.72	23.00	919.10	-	0.13	2.90	5.21	G1
Salacia elliptica	12	-	-	12	6190.89	-	-15.60	309.34	-	6687.77	-	0.28	-	2.18	G1
Siparuna guianensis	78	19	39	98	3446.09	319.92	-227.93	1123.79	492.52	3674.38	6.74	14.23	11.91	28.54	G4
Tabebuia roseo-alba	13	1	1	13	1104.25	27.24	-	119.96	-	1240.01	1.98	1.16	1.98	4.91	G1
Tapirira guianensis	61	11	4	54	17736.76	367.27	-919.91	2210.48	96.93	16875.80	4.85	7.09	1.91	6.16	G2
Terminalia brasiliensis	23	3	-	20	15933.70	305.90	-353.18	874.83	-	14624.04	3.43	5.31	-	3.26	G2
Terminalia phaeocarpa	5	1	-	4	6364.23	-	-3178.73	140.29	-	3343.92	5.43	16.10	-	1.45	G3
Tocoyena formosa	5	1	-	4	259.04	25.78	-	10.09	-	235.73	5.43	4.87	-	2.60	G2
Unonopsis lindmanii	26	3	3	26	3439.15	59.07	-15.74	726.40	59.86	4262.84	3.02	2.56	3.02	8.24	G1
Xylopia aromatica	20	9	6	17	3722.25	1166.98	-166.87	321.93	100.90	2410.58	13.88	14.94	10.31	6.97	G3
Zanthoxyllum riedelianum	12	3	-	9	3042.38	121.06	-2.43	217.64	-	2981.98	6.94	4.80	-	4.33	G2

CHAPTER 3

Changes on seasonal forests community beneath dam impacts: a dynamic study

Resumo: Alterações na comunidade arbórea de florestas sazonais sob impactos de barragem: um estudo de dinâmica.

Mudanças locais causadas por represas podem ter drásticas conseqüências para o ecossistema, não somente alterando o regime hídrico como também modificando áreas as margens do lago gerado após seu enchimento. No entanto, não existem estudos temporais que avaliem empiricamente o aumento na umidade do solo após represamento e seus efeitos para florestas estacionais tropicais que passam a se situar ás margens do reservatório. Assim, avaliamos o efeito da aproximação da linha de água em três florestas tropicais, com demarcada estacionalidade climática, através de análises de umidade do solo e dinâmica da comunidade arbórea. Nossa hipótese é que haverá aumento na umidade do solo, com rápidas modificações nas comunidades, além das áreas mais afetadas serem aquelas mais próximas da margem imposta pelo reservatório. Diferente de outras barragens, a vazão destas estudadas é controlada por outras barragens a montante, não sofrendo flutuações sazonais. As parcelaforam alocadas a partir da margem da represa e acompanham a floresta perpendicularmente ao leito da represa, ao todo há 60 parcelas de 20 x 10 m. Análises de solo antes e depois do represamento foram realizadas em três profundidades (0-10cm, 20-30cm e 40-50cm) a 5m e 15m de distância da beira do reservatório. Foram realizadas amostras das comunidades arbóreas antes (T0), dois (T2) e quatro anos (T4) após o represamento, para árvores com diâmetro a altura do peito com mínimo de 4.77cm. Foram calculadas as taxas de dinâmica, o *turnover* e as mudanças líquidas da comunidade, bem como testes estatísticos para comprovar o aumento da umidade do solo e modificações ocorridas nas comunidades estudadas. As análises demonstraram aumento na umidade do solo nas três florestas, principalmente durante a estação seca e nos setores mais próximos ao reservatório. De fato, ocorreu aumento na área basal das comunidades devido ao rápido crescimento de diversos indivíduos (sobretudo os de grande porte), com altas taxas de dinâmica. Os maiores *turnovers* para indivíduos e área basal ocorreram nos dois primeiros anos após o represamento, demonstrando rápida troca de indivíduos após o inicio do distúrbio, próximo a beira do lago. Além disso, as mudanças líquidas neste setor demonstraram o rápido espaçamento das árvores na beira da represa, o conseqüente aumento na heterogeneidade espacial, com separação entre área afetada com árvores de maior porte (beira do rio, a 0-30m de distancia da margem) e área sem grandes alterações quanto a comunidade original (a 30–60m de distância da margem). Porém, em todas as florestas houve redução nas taxas de dinâmica nos anos seguintes (T2-T4), indicando que estas florestas tendem a estabilizar após o forte impacto inicial.

Palavras chave: mortalidade, recrutamento, incremento, *turnover*, mudanças líquidas

Key words: mortality, recruitment, ingrowth, turnover, net changes

Introduction

The knowledge about consequences of artificial reservoirs created by man can be seen into two different ways. Macro-scale consequences are related with fast landscape changes such as habitat fragmentation due to roads and highway, extensive flooding areas and, consequently, mortality of many biotic elements, downstream water flow reduction with biomass loss by death of trees and no longer carbon assimilation by photosynthesis. Then, just the act of building dams causes these problems and with many changes to landscape. These macro-scale consequences are easy to be noticed; nevertheless, local changes after dam construction are not. These local changes are micro-scale consequences, related to problems of difficult comprehension and deserve detailed studies.

There are several examples of micro-scale consequences after dam construction: generation of a large pulse of methane and carbon dioxide emissions (Duchemin et al. 1995; Fearnside 2001, 2002; Soumis et al. 2004), water biochemistry changes (Humborg et al. 1997), explosion of disease vectors such as mosquitoes (Fearnside 2005; Luz 1994; Patz et al. 2000), increase of illness incidence associated to stagnant water (Steinmann et al. 2006) and decrease fungal (Hu et al. 2010), herbs and shrubs diversity (Dynesius et al. 2004; Nilsson et al. 1997). All these local problems not change the landscape in a short time period, but their long-term effects can have drastic consequences. After conversion of a running-water (lotic) system to a stillwater (lentic) one, aquatic weeds cover the water enhancing methane flux to atmosphere (Fearnside 2002) and the carnivorous fish abundance increase, leading to a drastic reduction on fish diversity (Leite & Bittencourt 1991). To terrestrial environment, the biomass loss due the high mortality of tree species associated to water resources few years after water flow reduction on dam downstream, highly alters the forest understory (Chapter 1).

These changes in terrestrial environments are critical because plants represent primary producers and the basal component of most ecosystems (Loreau et al. 2001). However, most of studies focus on dam impacts to grass, herbs and shrubs (Mallik & Richardson 2009; Nilsson et al. 1991; Nilsson & Svedmark 2002) and are concentrated in cold environments with low diversity (Dynesius et al. 2004; Jansson et al. 2000; Nilsson et al. 1997), although most dams construction occur on high diversity tropical systems (Guo et al. 2007; Johansson & Nilsson 2002; Nilsson et al. 1997; Nilsson et al. 2005) dominated by trees. Comparisons between dammed and not dammed rivers are frequent too (Nilsson et al. 1997; Nilsson & Svedmark 2002), notwithstanding temporal studies which monitoring dam consequences over the years are nonexistent.

Monitoring studies which evaluates dynamic of mature forests (Condit et al. 1999; Lewis et al. 2004; Phillips et al. 2004; Sheil et al. 2000), or tree community changes related to natural or anthropogenic disturbances (Chazdon et al. 2005; Chazdon et al. 2007; Condit et al. 2004; Machado & Oliveira-Filho 2010; Oliveira et al. 1997) are widespread, hence still missing evaluate dam consequences to forest communities. Majority of world´s large rivers have a regulated flow (Nilsson et al. 2005) then dynamic studies should clarify dam impacts on many environments. Moreover most dams are built to generate electricity (Truffer et al. 2003), thereby are implanted on mountainous terrain (Nilsson & Berggren 2000) to increase efficiency of energy production (Truffer et al. 2003). Thus, we choose to evaluate upstream dam consequences to three dry forests on southern Brazil. These forests are associated to mountainous or at least steep terrain, and then are an excellent object of study to infer changes to other forests with similar impact. These dry forests have a marked dry season with lack of rains, then the water approach after dam construction means a total change on water relations to flora with uncertain consequences. Besides, dry forest are threatened environment (Espirito-Santo et al. 2009; Miles et al. 2006) enhancing the importance of evaluate dam impacts over these communities.

Considering that even small changes to water regime level induce changes on vegetation structure (Nilsson 1996), the new shorelines created by dam will enhance soil moisture and we predicted many changes on arboreal structure such as: great dynamic rates (because dam are severe disturbance and disturbed forests presents high dynamics rates), high ingrowth rates with basal area increase (because wet forests had more basal area than dry forests – Murphy & Lugo 1986) and the most impacted sites will be those closest to the river, where the water approach will enhance soil moisture.

Material and Methods

This study was carried out in three dry forest (18°47´40”S, 48°08’57”W, 18°40’31’’ S, 42°24’30’’ W and 18°39’13’’ S, 48°25’04 W; Figure 3.1) located in the Amador Aguiar Complex Dam (two dams located in Araguari River with 52m and 55m depth). All areas had slope terrains, however the deciduous forest inclinations were much more pronounced than semideciduous forest (in some plot the inclination was above 30°). The predominant soil types are dystrophic and eutrophic podzolic soil and dystrophic cambissoil with basalt outcrops with micaxist and biotite-gnaisse (Baccaro et al 2004; Baruqui & Mota 1983, Nishiyama 1989). The first dam (Amador Aguiar Dam I, from here AD1), finished flooding in 2005 and is at 624 meters of elevation (relative to sea level), with a flooded area of 18.66km^2 (CCBE 2007). The second dam (Amador Aguiar Dam II, from here AD2) ended flooding in 2006 and at 565 meters of elevation, with a flooded area of 45.11km^2 (CCBE 2006).

After damming three seasonal dry forests (two deciduous and one semideciduous forests), before distant at least 200m from any water source (see Figure 3.2A as representation), now has the riverbed on its edge since 2005 (AD1)/2006 (AD2). Different than other dams, water level is controlled by water flow of an upstream dam; thereby there are no water fluctuations and no floods occurrences in any period of the year. The climate is Aw (Koppen 1948) with a dry winter (april to september) and a rainy summer (october to march), with an average annual temperature of 22°C and average rainfall of around 1595 mm (Santos & Assunção 2006).

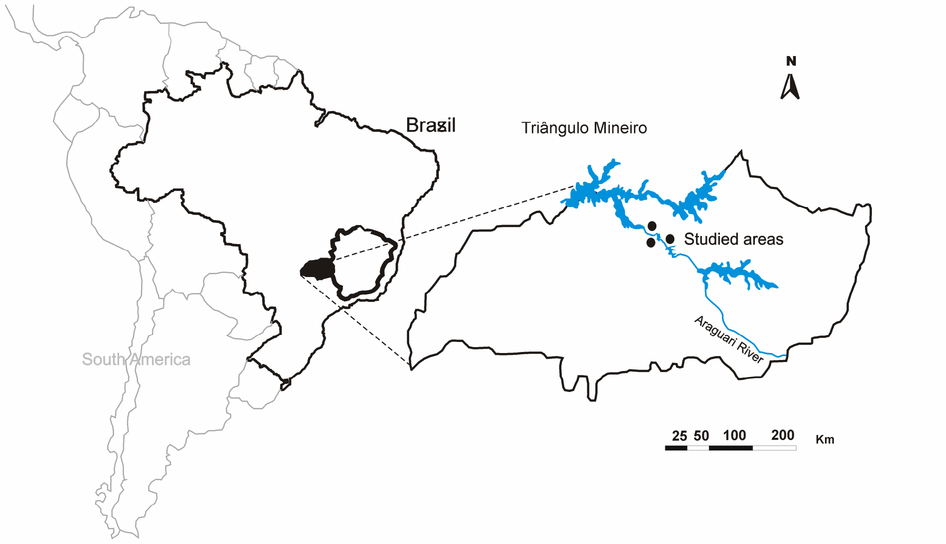

Figure 3.1 - Location of three seasonal dry forests which were affected by Amador Aguiar Complex Dam, on Triângulo Mineiro, southern Brazil. Before dam construction these forests were distant from water sources and now there are on the artificial lake margin.

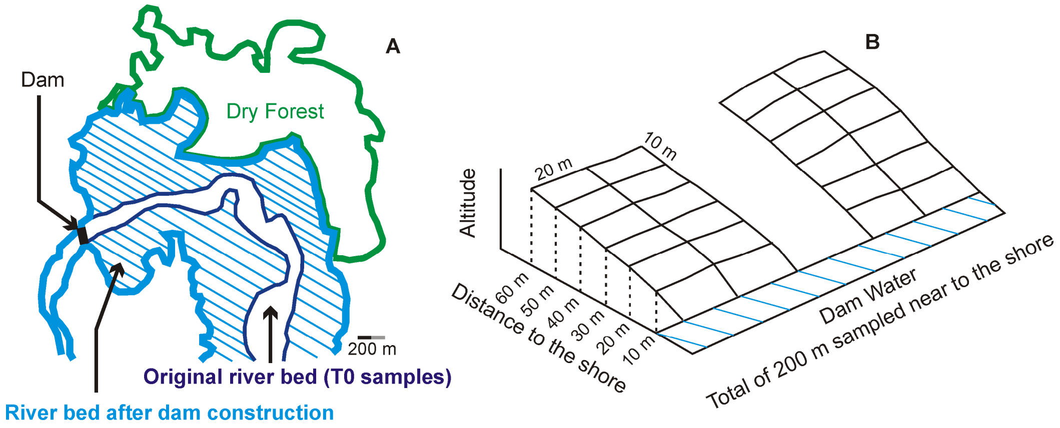

Figure 3.2 - “A” - Representation of upstream landscape changes and water proximity to dry forest after dam construction. “B” - Plots scheme used to tree community samples in three dry forests.

Soil - To each forest we made ten soil samples at three distinct depths: 0–10cm, 20–30cm and 40–50cm (total of 90 samples), five samples near the riverbed (five meters distant to water line) and five 15m distant to the artificial lake. This separation was important because we try to reveal how the water increase affects the soil moisture at different depths and the distance of dam influence in soil moisture. To calculate soil moisture variation we based on EMBRAPA methodology (EMBRAPA 1997). We repeated soil samples every three months to cover the middle and the end of rainy and dry seasons. We also repeated soil moisture collections in three distinct years: before spillway construction (T0 - original condition without artificial lake influence created by dams), after (T1) and during the third year after damming (T3).

We performed some soil moisture analyses for three periods (T0, T1 and T3). First, to check soil data normality we perform Lilliefors test, but the soil data did not presented normality. Then, to verify damming effects on the different distances, we made Friedman tests amongst 5m distant and 15m distant to shore separately. These tests were made in each soil depth to each season (middle of rainy, end of rainy, middle of dry and end of dry season), comparing the three years of measurement (T0, T1 and T3). “Friedman” is a non-parametric test used to detect differences among repeated multiples measures.

Finally, to compare damming effects on soil moisture between 5m (close to shore) and 15m of distance (distant to shore) we made Wilcoxon tests. We performed this test in each soil depth to each season, comparing soil moisture near x far to shore before damming (T0), one year after damming (T1) and three years after damming (T3). Thus, we realized 12 Wilcoxon tests in each soil depth (three tests every season), totalizing 36 tests in each forest (12 comparing near x far to shore before and 24 after damming) and 108 in total. All the analysis were made in Statistica Program (StatSoft 2005).

Plant Sampling - The first inventory (T0) was carried out before damming in 2005 (on AD1) and 2006 (on AD2). In each forest were marked 60 permanent plots of 20x10m, totalizing 1.2ha by area (total of 3.6ha sampled). A total of 10 plots (with 200m width) were installed at the site where river reached flood elevation after damming and remaining plots were installed perpendicular to river margin (Figure 2B). Thus, samples were distributed every 10m of perpendicular distance to the river (0-10m, 10-20m, 20-30m, 30-40m, 40-50m and 50-60m of distance – Figure 2B), with some samples linked. All trees with diameter at breast height (DBH) of at least 4.77cm were tagged with aluminum labels. The stem diameter was measured at 1.30m from the ground and in multiple stems all live tillers were measure at 1.30m too. The first inventory (T0) results are published in 2009

(Kilca et al. 2009; Siqueira et al. 2009). The second (T2) and third (T4) inventories were carried out in two and four years after damming. These samplings followed the same procedure of first inventory. New individuals that met inclusion criteria (recruits) were measured and identified, and mortality refers to standing dead trees, fallen trees or not found individuals. All these analyses were performed in Systat 10.2 program (Wilkinson 2002).

After test data normality to number of individuals and basal area in all forests to three measurement times with Liliefors test (data were normal), we compared the number of individuals and basal area of three plant inventories (T0, T2 and T4) in each forest with ANOVA test, followed by post-hoc Tukey test. Then, we test the number of dead trees and recruits in T0-T2 and T2-T4 periods with paired "t" test. The same procedure was done with basal area of death trees, recruits, increment and decrement. We still made paired "t" test between plots based on distances to river, comparing T0-T2, T2-T4 and T0-T4 periods apart. To this analysis, after some exploratory investigations, we combined both deciduous forest as a single forest (the damming effects were very similar to both forest) and separate plots in two distance groups too, samples near riverbed (0-30m of distance) and samples distant to riverbed (30-60m of distance). All these analyses were performed in Systat 10.2 program (Wilkinson 2002).

Dynamics rates - We based the community dynamics on mortality, recruitment, outgrowth and ingrowth rates. Annual mortality (m) and recruitment (r) were calculated in terms of annual rates (see formulas on Sheil et al. 1995; Sheil et al. 2000. Outgrowth annual rates refers to basal area of dead trees plus dead branches basal area of living trees (decrement) and ingrowth annual rates refers to basal area of recruits plus growth in basal area of surviving trees (increment). To evaluate changes in forest we computed turnover rates for individuals and basal area through mortality-recruitment rates and outgrowth-ingrowth rates (Oliveira-Filho et al. 2007). Then we evaluated the net change (Korning & Balslev 1994) to individuals and basal area and compute an overall net change (ONC, average between individuals and basal area net change – all detailed formulas on Chapter 1).

Results

Soil - Friedman tests near and far to lakeshore showed soil moisture increase due to approach of water line on the forest margin. There were significant increase in soil moisture after damming to all forests; however, damming effects in soil moisture were higher in end of dry season. To both deciduous forests soil moisture increased significantly to all depths ($p<0.05$, $F>7.6$) and marginally significant in semideciduous forest ($p<0.10$, $F=5.2$) both near (5m) and far (15m) to lakeshore. In middle of dry season the results were similar. In both deciduous forest there were a significant increase in soil moisture in all depths ($p<0.05$, $F>7.6$) and both near and distant to water line. The semideciduous forest was an exception, with no significant changes in soil moisture in middle of dry season distant to the shore; but near to water line, soil moisture increased marginally significant in 0-10cm and 20-30cm ($p<0.10$, $F>5.2$) and significant in 40-50cm ($p<0.05$, $F=6.4$). Damming consequences to soil moisture throughout the dry season were summarized on Figure 3.3A and 3.3B.

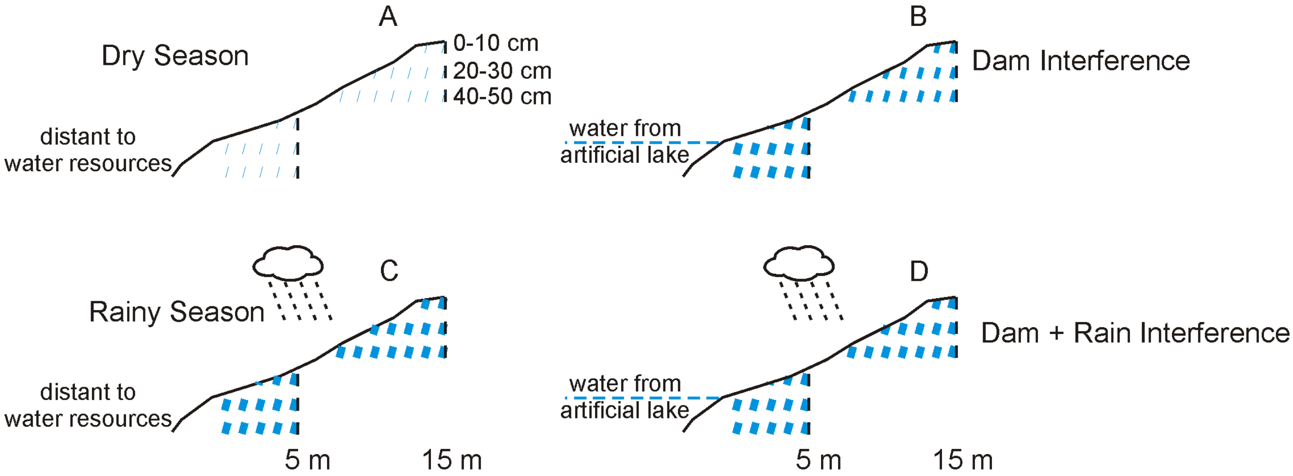

Figure 3.3 - Summary of soil moisture changes occurred due construction of dams. A and C represents soil moisture on dry forests before damming and B and D represents soil moisture after damming construction. A → B shows high damming influences to soil moisture on dry season; the soil moisture increase after dams construction breaking the strength of dry season. C → D show low damming influences to soil moisture on rainy season; the increase in soil moisture was less conclusive due rainfall variation in these periods. The continuous line represents soil surface, vertical black bars represents soil samples sites, blue bars represents soil moisture and their thickness illustrates soil moisture, then thicker bars represents more water available on soil; 5m and 15m represents the distance to the lakeshore after damming. After dam influence, soil moisture increase mainly on dry season and mainly near to the lakeshore. Detailed tests about soil moisture were summarized on Appendix IV, V, VI, VII, VIII and IX.

In rainy season (middle and end) results were more variable. In deciduous forest 2 (DF2) and semiciduous forest (SF) there was an increase in soil moisture to 0–10cm depth both near and distant to lakeshore ($p<0.05$, $F>7.6$) in middle of rainy season after damming. To deciduous forest 1 (DF1), near to shore presented a marginally significant increase only near to shore ($p<0.10$, $F=5.2$) in this soil depth. To 20–30cm deep near to shore, in DF2 and in SF forest there were an increase in soil moisture (significant to deciduous - $p<0.05$, $F=7.6$; and marginally significant to semideciduos - $p<0.1$, $F=5.2$) after damming. Far to shore, in 20–30cm deep, there was an increase in soil moisture only in deciduous forest 2. Finally in 40-50cm deep, far to shore, soil moisture becomes higher after damming in both deciduous forest (marginally significant to DF2 - $p<0.1$, $F=5.2$ and significant to DF1 – $p<0.05$, $F=6.4$). However, in DF1 the soil moisture increase occurs only three years after damming. In end of rainy season all data was inconclusive. Sometimes soil moisture were higher in T0, other times were higher after damming (occasionally in T1, in other cases in T3). These results probably happened due heavy rains in this period making any soil moisture sample more influenced by rainfall than by water from dam. Damming consequences were noted with certainty only near to the lakeshore throughout the rainy season (Figure 3.3C and 3.3D).

The comparison between the distances from the lakeshore demonstrated that, to all soil depths, the increase of soil moist was higher near than far to margin, especially in dry seasons (middle and end) after damming (T1 and T3). In 36 Wilcoxon tests after damming (T1 plus T3) between near x far to dry periods, in 21 soil were significant moister near to shore, and in others 15 present no differences. Just to contrast, before damming (T0) in 18 tests comparing near x far to shore, only in four the soil moisture was higher in soil samples in sites which become near to shore after damming. Then, the soil clearly becomes moister in samples near water line after damming.

To rain periods, before damming (T0), the 18 tests comparing near x far to shore showed no differences in soil moisture in these distances. However of the 36 comparison after damming (T1 plus T3) between near x far, in seven the soil near to shore was moister than far to shore. These results evidence that soil really become moister near than far to shore and raise in moisture was much more evident in dry periods (damming consequences to soil moisture were summarized on Figure 3.3). Damming consequences to soil moisture throughout the dry season were summarized on Figure 3.3A and 3.3B.

Structural changes in forests -There were no strong variations in number of individuals in three times to any forest (Table 3.1) with no significant differences (Table 3.2). Indeed mortality and recruitment of trees in T0-T2 and T2-T4 period were similar (Table 3.1), masking possible effects in communities. However, basal area presents a clear increase (Table 3.1) statistically confirmed to both deciduous forest in T0-T4 but not to semideciduous forest (Table 3.2). This enlargement occurred mainly due strong trees increment during the four years of damming (Table 3.1). Even to SF, basal area gain was high (11.59% in four years) not only due increment and mortality in T2-T4 but also to low mortality lost in T2-T4 (Table 3.1).

There was a contrast between first two years of damming and after the second year (T2-T4) among deciduous and semidecidous forests. In the first two years the mortality and recruitment (number of individuals and basal area) were similar in all forest while the increment in deciduous forest was higher than in semideciduous forest. However in T2-T4, mortality in semideciduous forest was much smaller than in deciduous while recruitment and increment were similar between the three forests (Table 3.1).

These results lead, in general, to high dynamics rates in all forests in T0-T4. However the dynamics rates were high in T0-T2 and much smaller in T2-T4, especially in both deciduous forests to all rates and for mortality and outgrowth in semideciduous forest. The ingrowth rates, for example, decreased from 11%.year^{-1} to less than 7.5%.year^{-1} in deciduous forest, and recruitment decreased from ~7% year^{-1} to 4.7%.year^{-1} in deciduous 1 and ~7% to ~2%.year^{-1} in deciduous (Table 3.1); otherwise, ingrowth and recruitment rates in semideciduous forest were just slightly higher in T0-T2 than T2-T4 period (Table 3.1).

Table 3.1 - Tree community parameters on three measurement times, before damming (T0), two (T2) and four years (T4) after damming and tree community dynamics between measurement times (T0 to T2, T2 to T4 and T0 to T4 periods). DF1 = Deciduous Forest 1, DF2 = Decidudous Forest 2 and SF = Semideciduous Forest.

	T0			T2			T4		
Parameters	DF1	DF2	SF	DF1	DF2	SF	DF1	DF2	SF
Number of individuals	1124	1626	1501	1136	1670	1489	1133	1649	1573
Mortality (N) (ha)	118.33	158.33	134.17	90.83	81.67	39.17	209.17	240.00	173.33
Recruitment (N) (ha)	130.00	185.00	125.00	86.67	62.50	108.33	216.67	247.50	233.33
Basal area	16.80	18.03	27.99	19.16	20.23	29.16	20.54	21.75	31.66
Mortality (m^2)	1.19	1.48	1.89	0.95	0.88	0.45	2.14	2.36	2.33
Recruitment (m^2)	0.37	0.48	1.50	0.44	0.17	0.50	0.81	0.65	1.99
Decrement (m^2)	-0.16	-0.25	-0.19	-0.29	-0.22	-0.18	-0.45	-0.47	-0.36
Increment (m^2)	2.95	3.23	2.72	1.93	2.18	2.21	4.88	5.41	4.92
	T0-T2			T2-T4			T0-T4		
Mortality rate (% ano^{-1})	6.53	6.02	5.52	4.92	2.98	1.59	6.12	4.76	3.66
Recruitment rate (% ano^{-1})	7.12	6.88	5.17	4.70	2.30	4.22	6.31	4.84	4.78
Outgrowth rate (% ano^{-1})	4.92	5.94	4.55	3.97	3.31	1.42	4.98	5.08	3.10
Ingrowth rate (% ano^{-1})	10.99	11.69	6.49	7.21	6.70	5.45	9.62	9.67	6.07
Turnover (N) (% ano^{-1})	6.82	6.45	5.34	4.81	2.64	2.91	6.22	4.80	4.22
Turnover (BA) (% ano^{-1})	7.96	8.82	5.52	5.59	5.00	3.44	7.30	7.38	4.58
Net change rate (N) (% ano^{-1})	0.53	1.34	-0.40	-0.13	-0.63	2.78	0.20	0.35	1.18
Net change rate (BA) (% ano^{-1})	6.79	5.92	2.07	3.54	3.71	4.20	5.15	4.81	3.13
Overal net change (% ano^{-1})	3.66	3.63	0.84	1.70	1.54	3.49	2.67	2.58	2.15

Table 3.2 - Analysis of variance (ANOVA) to number of individuals and basal area between three measurement times (T0, T2 and T4), to three Dry Forest on southern Brazil. In bold $p < 0.05$.

		Number of individuals		Basal area	
Forests	df	F	p	F	p
Deciduous forest 1	2	0.126	0.882	5.250	**0.006**
Deciduous forest 2	2	0.200	0.980	3.540	**0.031**
Semideciduous forest	2	1.199	0.304	1.482	0.230

To mortality and outgrowth, T0-T2 rates were higher than T2-T4 to all forest, however these rates declined sharply in semideciduous forest (mortality from 5.2 to 1.6% year^{-1} and outgrowth from 4.5 to 1.6% year^{-1} – Table 1). The contrast between T0-T2 and T2-T4 leads to high turnover rates (to individuals and basal area) in the first years of damming to all forests, but superior in both deciduous forest (Table 3.1).

Net change rates helped to differentiate damming effects in deciduous and semideciduous forest. In first two years to both deciduous forests individuals net change were low, opposed to the basal area where net increase was large (Table 3.1). However, semideciduous forest presented negative individuals net change and a low basal area net increase (Table 3.1), therefore the first two years of damming modified more deciduous forests than semideciduous forest, and was more positively too.

After two years of impoundment, increase in soil moisture presents distinct results. The subsequent period analyzed (T2-T4) showed low negative individuals net chance (very close to 0) in deciduous forest, but with a positive value to semideciduous forest (Table 3.1). This occurred due very low mortality in T2-T4 period in semideciduous forest because recruitment was similar at T0-T2 and T2-T4 periods (Table 1). The basal area net change to all forest were positively similar to all forest; however, unlike first two years, net change in semideciduous forest was superior than deciduous forests. Thus, overall net change (ONC) to deciduous forest in first two years were higher than semideciduous forest (Table 3.1) otherwise, in T2-T4 ONC in semideciduous was superior.

The paired t-test just confirmed that damming impacts were higher in T0-T2. Mortality was significantly higher in all forests (Table 3.3), but only to deciduous forest the recruitment was superior in T0-T2. To basal area, only increment was higher in the first two years to all forests and recruitment in basal area was superior in deciduous forests in this period (Table 3.3). An interesting result was related to basal area mortality in semideciduous forest, which was extremely smaller than two deciduous forest in T2-T4 (Table 3.1) and significant inferior to basal area of death trees in first two years of damming (Table 3.3).

The paired t-test between distances confirmed damming effects in all forest (Table 3.4). However, largest variations occurred near lakeshore (0-30m of distance). The number of recruits was higher in 0-30m to T2-T4 (both deciduous and semideciduous forest) but to T0-T4 period, only to deciduous forests (Table 3.4). This occurred due to major recruitment in 30-60m samples in semideciduous forest on T0-T2 period, meanwhile at this period, deciduous forests recruited more trees on 0-30m plots than 30-60m. To recruits basal area the effect was clearer, near the lakeshore recruitment were higher than 30-60m to T0-T2 period (Table 3.4). To deciduous forests, this happened in T2-T4 too reflecting in more recruits near the dam after four years of damming. However, to semideciduous forest, in T2-T4 period few trees were recruited near stream (0-30m), and more recruits were found distant to riverbed (30-60m), showing that damming effects was intense only in the first two years (Table 3.4).

Table 3.3 - Paired “t” test to number of individuals and basal area among T0-T2 and T2-T4 period to three Dry Forests on southern Brazil. In bold $p < 0.05$ and in italic $p < 0.10$. Positive values refers to higher values on T0-T2 and negative values refers to higher values on T2-T4 period.

		Number of individuals				Basal Area							
		Mortality		Recruitment		Mortality		Recruitment		Increment		Decrement	
Forests	df	t	p	t	p	t	p	t	p	t	p	t	p
DF1	59	1.985	*0.052*	3.278	**0.002**	0.811	0.421	3.351	**0.001**	3.299	**0.002**	1.833	*0.072*
DF2	59	4.623	**0.001**	7.894	**0.001**	1.489	0.142	7.347	**0.001**	3.774	**0.001**	-1.132	0.262
SF	59	4.998	**0.001**	1.096	0.277	3.184	**0.002**	1.434	0.157	3.413	**0.001**	-0.147	0.884

Table 3.4 - Paired “t” test among plots close to the lakeshore (0-30 m) and plots distant to margin (30-60 m) to number of individuals and basal area to all periods (T0-T2, T2-T4 and T0-T4) in three Dry Forest on southern Brazil. In bold p < 0.05 and in italic p < 0.10.

		Number of individuals											
Forests		Death Trees						Recruits					
		T0-T2		T2-T4		T0-T4		T0-T2		T2-T4		T0-T4	
		t	p	t	p	t	p	t	p	t	p	t	p
DF	df = 118	0.68	0.497	0.48	0.634	0.80	0.428	1.57	0.119	2.61	**0.010**	2.55	**0.012**
SF	df = 58	1.46	0.151	-0.12	0.904	1.27	0.211	-1.44	0.154	1.66	*0.100*	0.34	0.737

		Basal area											
		Death Trees						Recruits					
		T0-T2		T2-T4		T0-T4		T0-T2		T2-T4		T0-T4	
		t	p	t	p	t	p	t	p	t	p	t	p
DF.	df = 118	-0.93	0.356	1.04	0.303	-0.22	0.827	1.69	*0.095*	2.74	**0.007**	2.59	**0.011**
SF	df = 58	1.50	0.140	-0.90	0.375	1.20	0.236	-2.21	**0.031**	2.00	*0.051*	-0.45	0.658
		Increment						Decrement					
		T0-T2		T2-T4		T0-T4		T0-T2		T2-T4		T0-T4	
		t	p	t	p	t	p	t	p	t	p	t	p
DF.	df = 118	2.26	**0.026**	1.35	0.181	2.07	**0.040**	0.81	0.417	-1.53	0.128	0.08	0.934
SF	df = 58	1.18	0.243	0.99	0.326	1.24	0.220	1.77	*0.082*	0.18	0.860	1.30	0.198

Discussion

Soil moisture and vegetation - The soil moisture analysis successfully showed damming effects in soil moisture over dry season (middle and end of this season) to all depths (0-10cm, 20-30cm and 40-50cm) and to all three dry forests. More, soil moisture increases both near and until 15m of distance to lakeshore. These results infer changes on soil hydrology on these forests leading to many forest alterations, mainly because 80% root biomass in dry forests are found in first 50cm of soil depth (Raherison & Grouzis 2005), and same occur forests in general too (Kiley & Schneider 2005; Tufekcioglu et al. 1998). In seasonal environments, the soil water shortage is high due lower overall rainfall on winter (Stefan et al. 2007), thus was expected that changes in soil moisture up to 50cm of soil depth implies in several changes on structure of dry forests.

The approach of water line may have two consequences for a single tree; in a worst case the water overage creates anaerobic conditions which can lead to anoxia and cell death in roots, ceasing plant growth and consequently killing the tree (Blom & Voesenek 1996; Nilsson & Berggren 2000; Vartapetian & Jackson 1997); otherwise, soil moisture increase can broke the lack of water, making the environment less stressful, reducing the chance of root dissection and enhance the plant growth. Into the deciduous forests for example, soil are fertile (Kilca et al. 2009; Siqueira et al. 2009), but are deficient in water supply (Pennington et al. 2009) overdue water shortage in drought season. These droughts reduce photosynthesis efficiency (Goulden et al. 1996; Reichstein et al. 2002) and have adverse effect on growth (Suarez et al. 2004). Therefore, if this lack of water is suppressed, there are no restrictions to plant growth (Guilherme et al 2004), and an increase in their metabolically efficiency (like photosynthesis) should be expected, leading to fast tree growth.

Raise in Basal Area - In general, changes occurred mainly on basal area with strong increment rates and gain (ingrowth) superior of 9% year^{-1} in deciduous forest and 6% year^{-1} in semideciduous forest after four years of damming (correspond to a increase in basal area superior of 4.5m^2.ha^{-1} in all forests. These values were higher than major other dynamic studies in other tropical rain forests (Korning & Balslev 1994; Lewis et al. 2004; Manokaran & Kochummen 1987; Phillips et al. 2004; Rolim et al. 1999; Sheil et al. 2000), tropical seasonal forests (Carvalho & Felfili 2011; Marin et al. 2005; Swaine et al. 1990; Werneck & Franceschinelli 2004), riparian forest (Braga & Rezende 2007; Felfili 1995; Chapter 1) and even higher than forest which suffer with intense impacts (Chazdon et al. 2007; Condit et al. 2004; Machado & Oliveira-Filho 2010), mainly considering both deciduous forest.

These high rates in deciduous forest probable are due combination of two favorable abiotic factors: the soil moisture supply (by water approach) and good soil nutrition support (already occurring in these forests). Drought in deciduous forests is more pronounced due rocky terrain with steep slopes than semideciduous forests, then drought had more negative consequences to these vegetations. Is known that drought represents a major constraint on plant growth and productivity in most terrestrial plant communities (Churkina & Running 1998; Hinckley et al. 1979) due decline plants photosynthetic capacity (Martin et al. 1994), otherwise soil fertility increases net productivity (Malhi et al. 2004; Phillips et al. 2004). Therefore, the combination of water supply and high fertility leads to greater above-ground forest productivity in deciduous forest than semideciduous. The ingrowth rates in semideciduous forest was inferior because the soil is much less fertile than in deciduous forest (Kilca et al. 2009; Siqueira et al. 2009), then still exist a limiter to plants growth.

High turnovers in first years after damming - When a severe disturbance occurs is expected a reduction in basal area and density (Machado & Oliveira-Filho 2010) but, for the three studied forests, basal area not only increased, but also the density remained constant, due to offsetting of recruitment over mortality. Whilst many small trees died (93% of died trees were smaller than 20cm of diameter), a great number of trees were recruited (major recruit were between the minimum of 4.77cm until 10cm of diameter), therefore this change of trees provided high turnover, not only in basal area, but in density too. The high values of turnover (individuals and basal) was superior to the vast majority of other studies in forests (see papers compared in the previous paragraphs), exemplified the damming impact on dry forest and how these forests were disrupted.

This change in turnover probably occur due interaction of some factors. First, long winter season limits net primary productivity (Pregitzer & Euskirchen 2004; Stephenson & van Mantgem 2005) by water scarcity or cold (here, the scarcity of water is much important due the sites be located in tropical zone) and it means less change in basal and individuals; however, this limitation was broke by water approach enhancing plant growth. Second, richer soils had higher turnover rates compared to less fertile soils (Phillips et al. 2004; Stephenson & van Mantgem 2005), then a fertile soil favors survivor of more individuals if other environment factors are not limiter. Third, the proximity of water table reduces the seasonality effects on soil moisture and aseasonal environments favors the organisms that attack plants (fungi and nematodes for example) leading to higher death rates (Givnish 1999). More, death individuals open spaces to other trees leading to higher recruitment rates (Stephenson & van Mantgem 2005), consequently high turnover rates. Forth, in absence of a significant disturbance, little net change in living mass is expected (Stephenson & van Mantgem 2005). Others deciduous forest presents less than 4.5% year^{-1} of turnover rates (Carvalho & Felfili 2011; Marin et al. 2005; Swaine et al. 1990; Werneck & Franceschinelli 2004) due low mortality and basal area loss. However, in dry forests analyzed here not only the small trees died, but some large trees died too (7% of all died trees were bigger than 20cm of diameter). Even so these big trees represents 43% of total basal area lost, and fall of these individuals can hit and die many other small trees enhance gaps creation (Pearson et al. 2003); then, trees turnover in gaps tend to be faster than non-gap environment. Turnover rates are important because are positive correlated with net primary productivity (Runyon et al. 1994), and had important implications to carbon capture. More turnover will means more carbon in organic matter allocated in trees on these forests and more carbon exchange between trees and entire community.

This result however was tendentious. The organic carbon allocated in these forests not compensated the carbon lost by flooding of original riparian forest. An original riparian forest in these region possess about 40m^2 ha^{-1} (Rodrigues et al. 2010), and many were entire flooded. More, increase in basal area and turnover rates were extremely high in first two years after damming, but reduced in T2-T4 (which assess the period of two to four years of measurement). This occurs to all parameters analyzed for individuals (mortality and recruitment) and basal area (mortality, recruitment, increment and decrement - however, deciduous forests follow a pattern and semideciduous follow another pattern, we will discuss this contrast follow). T2-T4 dynamic rates were more closely compared to other dry forest whose mortality varies to 2.3 – 5.0% year^{-1} and recruitment varies to 1.5 – 4.5% year^{-1} (Appolinário et al. 2005; Machado & Oliveira-Filho 2010; Oliveira et al. 1997; Paiva et al. 2007; Silva & Araújo 2009; Swaine et al. 1990; Werneck & Franceschinelli 2004), but many rates still high, mainly ingrowth rates (highly influenced by increment in basal area, superior than 5.45% year^{-1}).

Then, we believe in a stabilization of the forest rates in next years, just like occur with a natural forest which suffer an antropogenic disturbance (Toniato & de Oliveira-Filho 2004). In these forests the first years after the disturbance, forest presents high dynamic rates (mainly high turnover rates); however, over the years rates tend to stabilize. The difference in dam impact was basal area gain over in few years after the impact. Despite of most work of forest dynamics be held every five years, the disturbance caused by dam was much more intense in first two years. Therefore we suggest that forests with similar impact monitor may be monitored at least each two years.

Forest heterogeneity and stabilization - The forests changes and stabilization not follow the same pattern to deciduous and semideciuous forest and even inside each forest. For deciduous forest, recruitment (in individuals and basal area) and increment was intense near to shore (0-30cm) but not semideciduous forest (with less recruit near to the shore in T0-T2 and more recruits at this distance in T2-T4, with no differences in four years of damming and no differences in increment between distances). Thus, we could found a regular arboreal response pattern to deciduous forest, where recruitment and increment of trees was positively affect after damming in patches near water table and strongly in first two year of impoundment.

The basal area could drive the succession (van Breugel et al. 2006) and high increase in basal area occurs in early successional stage (Harper et al. 2007) and/or after disturbance (Aide et al. 1995; van Breugel et al. 2006). The most impacted patch was near lakeshore, which expected a succession period; however, a non-common succession, because forest before damming was not in early succession stage. The water approach creates a new condition with water supplies, that enhance environment heterogeneity and divided the community in two main communities: a “riparian-deciduous forest” near to shore and water stressful distant to the margin (more similar to original deciduous forest). Riparian systems commonly presents high heterogeneity perpendicular to the watercourse (Ribeiro & Walter 2001; Rodrigues & Nave 2000); however, this occur naturally in environment.

Here, we had a rise of heterogeneity caused by a dam construction, a human impact. The water supply improves tree growth, increasing basal area in many trees that become ticker near to the shore. Without rains in dry season, low available subsoil moisture prejudice plant performance (Borchert 1994), declining their growth (Suarez et al. 2004). Of course those remote areas were impacted by the river dam however with less intensity. This addition in heterogeneity and great increase in trees increment near to shore was less intense in T2-T4 period too. The damming first impact clearly caused major effects in tree structure. Other works demonstrated that to other taxa, first years after dam disturbance were more pronounceable like rapid increase in aquatic vegetation (Nilsson 1983), fishes and invertebrates explosion (Leite & Bittencourt 1991; Luz 1994; Patz et al. 2000) but many changes tend to stabilize over the years (Lima et al. 2002), and with here the same occurred to trees, with a stabilization after first two years after impact. Great impacts tends to change greatly arboreal structure, but tend to stabilize over the years too (Aide et al. 1995; Chazdon et al. 2005). We could infer that the “riparian-deciduous forest” parameters tend to stabilize, even near to river. However, unlike other transitory disturbances (winds, intense dry periods, cold), damming is a permanent alteration and should transform landscape to another scenario, probably with major long-term consequences to environment.

Apendice IV – Friedman soil moisture test results comparing three sample periods (T0, T1 and T3) for each year season to each soil depth in deciduous forest 1. In bol $p < 0.05$ and in italic $p < 0.10$. Always soil moisture was larger before damming (T1 and T3 periods). In "1" soil moisture was bigger only in T3.

	Near		Far	
	p	F	p	F
Season	0 - 10 cm			
Half Rainy	*0.074*	5.2	1.000	0.0
End Rainy	0.165	3.6	**0.015**^{1}	8.4
Half Dry	**0.007**	10.0	**0.015**	8.4
End Dry	**0.041**	6.4	**0.007**	10.0
	20 - 30 cm			
Half Rainy	0.949	0.1	0.854	0.3
End Rainy	0.549	1.2	*0.074*	5.2
Half Dry	**0.022**	7.6	**0.022**	7.6
End Dry	**0.022**	7.6	**0.007**	10.0
	40 - 50 cm			
Half Rainy	0.819	0.4	**0.041**^{1}	6.4
End Rainy	0.247	2.8	**0.036**^{1}	6.6
Half Dry	**0.007**	10.0	**0.022**	7.6
End Dry	**0.022**	7.6	**0.015**	8.4

Apendice V – Friedman soil moisture test results comparing three sample periods (T0, T1 and T3) for each year season to each soil depth in deciduous forest 2. In bol $p < 0.05$ and in italic $p < 0.10$. Always soil moisture was bigger before damming (T1 and T3 periods). In “1” soil moisture was smaller on T1 and in “2” soil moisture was bigger in T3.

	Near		Far	
	p	F	p	F
Season	0 - 10 cm			
Half Rainy	**0.007**	10.0	**0.022**	7.6
End Rainy	0.549	1.2	0.022[1]	7.6
Half Dry	**0.016**	8.3	**0.015**	8.4
End Dry	**0.007**	10.0	**0.007**	10.0
	20 - 30 cm			
Half Rainy	**0.022**	7.6	0.091	4.8
End Rainy	0.022[2]	7.6	0.015[2]	8.4
Half Dry	**0.015**	8.4	**0.022**	7.6
End Dry	**0.015**	8.4	**0.008**	9.6
	40 - 50 cm			
Half Rainy	0.128	4.1	*0.074*	5.2
End Rainy	0.015[2]	8.4	0.015[2]	8.4
Half Dry	**0.022**	7.6	**0.015**	8.4
End Dry	**0.022**	7.6	**0.007**	10.0

Apendice VI – Friedman soil moisture test results comparing three sample periods (T0, T1 and T3) for each year season to each soil depth in semideciduous forest. In bol $p < 0.05$ and in italic $p < 0.10$. Always soil moisture was bigger before damming (T1 and T3 periods). In “1” moisture was bigger in T3 and “2” moisture was bigger in T1.

	Near		Far	
	p	F	p	F
Season	0 - 10 cm			
Half Rainy	**0.022**	7.6	**0.019**	7.9
End Rainy	0.549	1.2	0.165	3.6
Half Dry	0.074	5.2	0.165	3.6
End Dry	**0.022**	7.6	0.074	5.2
	20 - 30 cm			
Half Rainy	0.074	5.2	0.165	3.6
End Rainy	**0.022**^{1}	7.6	0.074^{1}	5.2
Half Dry	0.074	5.2	0.949	0.1
End Dry	**0.007**	10.0	**0.008**	9.6
	40 - 50 cm			
Half Rainy	0.128	4.1	0.247	2.8
End Rainy	**0.022**^{2}	7.6	0.015^{2}	8.4
Half Dry	**0.041**	6.4	0.247	2.8
End Dry	**0.015**	8.4	**0.007**	10.0

Apendice VII – Wilcoxon test comparing soil moisture samples near (5m) against far (15m) to the lakeshore at season year to each by soil depth in deciduous forest 1. In bold $p < 0.05$ and in italic $p < 0.10$. Soil moisture was always bigger near to the lakeshore.

		0 - 10 cm		20 - 30 cm		40 - 50 cm	
		p	Z	p	Z	p	Z
MR	T0	0.345	0.944	0.345	-0.944	0.893	0.135
	T1	0.500	-0.674	0.345	-0.944	0.500	-0.674
	T3	**0.043**	-2.043	0.500	-0.674	0.893	-0.135
ER	T0	0.345	0.944	0.686	0.405	0.500	0.674
	T1	0.500	-0.674	**0.043**	-2.043	*0.068*	-1.826
	T3	0.138	-1.483	0.500	-0.674	0.500	-0.674
MD	T0	0.686	-0.405	**0.043**	-2.043	0.686	-0.405
	T1	*0.080*	-1.753	**0.043**	-2.043	**0.043**	-2.023
	T3	0.500	-0.674	0.345	-0.944	0.500	0.674
ED	T0	**0.043**	-2.023	0.893	-0.135	**0.043**	-2.023
	T1	0.225	-1.214	**0.043**	-2.043	**0.043**	-2.023
	T3	**0.043**	-2.023	0.225	-1.214	0.225	-1.214

Apendice VIII – Wilcoxon test comparing soil moisture samples near (5m) against far (15m) to the lakeshore at season year to each by soil depth in deciduous forest 2. In bold $p < 0.05$ and in italic $p < 0.10$. Soil moisture was always bigger near to the lakeshore.

		0 - 10 cm		20 - 30 cm		40 - 50 cm	
		p	Z	p	Z	p	Z
MR	T0	0.345	0.944	0.225	-1.214	0.686	0.405
	T1	0.500	0.674	0.345	-0.944	*0.080*	-1.753
	T3	0.500	0.674	0.893	-0.135	**0.043**	-2.043
ER	T0	0.500	-0.674	0.893	0.135	0.686	0.405
	T1	0.500	-0.674	**0.043**	-2.043	**0.043**	-2.043
	T3	0.500	0.674	0.686	-0.405	0.500	0.674
MD	T0	0.686	-0.405	*0.080*	-1.753	0.500	-0.674
	T1	**0.043**	-2.023	**0.043**	-2.023	**0.043**	-2.043
	T3	0.225	-1.214	**0.043**	-2.023	0.138	-1.483
ED	T0	0.686	-0.405	0.345	-0.944	0.225	-1.214
	T1	**0.043**	-2.023	**0.043**	-2.023	**0.043**	-2.043
	T3	0.225	-1.214	0.500	-0.647	*0.080*	-1.713

Apendice IX – Wilcoxon test comparing soil moisture samples near (5m) against far (15m) to the lakeshore at season year to each by soil depth in semideciduous forest1. In bold $p < 0.05$ and in italic $p < 0.10$. Soil moisture was always bigger near to the lakeshore.

		0 - 10 cm		20 - 30 cm		40 - 50 cm	
		p	Z	p	Z	p	Z
MR	T0	0.345	0.944	0.225	-1.214	0.686	0.405
	T1	0.500	0.674	0.345	-0.944	0.080	-1.753
	T3	0.500	0.674	0.225	-1.214	0.500	-0.674
ER	T0	0.500	-0.674	0.893	0.135	0.686	0.405
	T1	0.500	-0.674	**0.043**	-2.043	**0.043**	-2.043
	T3	0.500	0.674	0.500	0.674	0.686	-0.405
MD	T0	0.686	-0.405	*0.080*	-1.753	0.500	-0.674
	T1	**0.043**	-2.043	**0.043**	-2.043	**0.043**	-2.043
	T3	0.686	-0.405	0.345	-0.944	0.144	-1.461
ED	T0	0.686	-0.405	0.345	-0.944	0.225	-1.214
	T1	**0.043**	-2.043	**0.043**	-2.043	**0.043**	-2.043
	T3	**0.043**	-2.043	0.500	-0.674	0.500	0.674

CHAPTER 4

Fast changes on tree composition and diversity: Dams consequences to seasonal dry forests

Resumo: Mudanças rápidas na composição e na diversidade: consequências das represas para as florestas secas estacionais.

Plantas típicas de ambientes estacionais que passam a se situar próximo as margens de reservatórios construídos pelo homem, tanto podem aumentar seu crescimento, como serem mortas após expostas às novas condições, sob solo alta saturação hídrica. Tais consequências devem causar mudanças na diversidade florística das comunidades afetadas. Não se sabe, também, se as espécies situadas nas margens das represas irão desempenhar o mesmo papel realizado por espécies típicas de beira de rio. Desta forma, nós avaliamos os efeitos da aproximação da linha de água após a construção de duas barragens, sobre espécies arbóreas associadas com florestas estacionais, através de um estudo de dinâmica das espécies, buscando compreender quais são as mais beneficiadas e as mais prejudicadas pelas novas condições ambientais impostas pela implantação de barragens artificiais, além de mudanças na diversidade. Nós estudamos três florestas estacionais tropicais que passaram a se localizar nas margens de uma represa artificial, ao longo de quatro anos, e comparamos com a estrutura e composição das áreas antes do represamento (T0). Todos os indivíduos com 4,77cm ou mais de diâmetro foram amostrados em 60 parcelas permanentes de 20 x 10 m em cada floresta. Taxas de dinâmica para cada espécie foram calculadas, além do *turnover* e mudanças líquidas, e comparamos a diversidade entre os períodos amostrados. As taxas de dinâmica das espécies foram, em geral, muito elevadas (acima de 10% ano^{-1}). Porém, enquanto algumas espécies especialistas de florestas estacionais apresentaram alta mortalidade (frequentemente superior a 5%ano^{-1}), muitas outras apresentaram acentuado crescimento (acima de 10% ano-1). Tais conseqüências implicaram em mudanças na estrutura das comunidades, com variação na importância das espécies principais. Houve aumento na diversidade, devido à entrada de novas espécies, algumas comuns em florestas ciliares. No geral, a comunidade parece estar mudando de estágio, devido à aproximação da linha de água, passando de uma floresta tipicamente estacional para outro, mais semelhante as florestas ciliares da região. No entanto, a diversidade é muito menor daquela apresentada por outras florestas ciliares naturais, e algumas espécies típicas de ambientes estacionais apresentaram taxas de recrutamento altas, indicando a manutenção das mesmas na comunidade, mesmo após vários anos. Concluímos que as modificações impostas pela represa (incremento e aumento na diversidade) não suprem os efeitos negativos da construção de barragens, nem mesmo tornaram esta comunidade semelhante a qualquer floresta ciliar natural.

Palavras chave: dinâmica, *turnover*, mudanças líquidas, crescimento, represamento

Key words: dynamic, turnover, net changes, impoundment

Introduction

Dams have historically facilitated human life, initially to farming, transport and domestic services, and are currently mainly built to energy generation (Baxter 1977). At least 45.000 dams above 15m high obstruct 60% of fresh water that flows to the oceans (Nilsson et al. 2005). Dams construction increased because hydropower were considered a clean and cheap alternative for energy production (Kaygusuz 2004) responsible to hold 16% of the worldwide electricity generation in 2005 (Evans et al. 2009). Despite the spread of this “clean and cheap” idea, several problems are actually know such as entire watershed modification (Nilsson & Berggren 2000), sediments retention (Manyari & de Carvalho 2007; Vorosmarty et al. 2003), biochemistry alterations (Humborg et al. 1997) and greenhouse gases emission (Fearnside 2002; St Louis et al. 2000).

Water-dwelling organisms (fish, amphibians, planktons, benthos and macrophytes) are directly affected, causing drastic changes in food webs (Brandao & Araujo 2008; Fearnside 2001; Leite & Bittencourt 1991; Moura Júnior et al. 2011; Wootton et al. 1996). The artificial lake created by impoundment interfere on terrestrial organisms too. Wildlife can moved to habits outside the flooding area, but sessile organism such as plants are killed drowned (Fearnside 2002; White 2007; Chapter 3). Plant anaerobic decomposition relieses organic matter and depleted the water oxygen (Barth et al. 2003; St Louis et al. 2000), also liberating carbon dioxide (CO_2) and methane (CH_4) to atmosphere (St Louis et al. 2000). However, not only organism who lived on direct flood influence area are impaired. The vegetation before distant from any water source became near the lakeshore margin created by dam, and long-term consequences are dificult to prevent because this new “riparian vegetation” is completely distinct than an original riparian environment in species and characteristics.

Riparian vegetation presents species adapted to water saturation and species adapted to low water patches, then are commomly represented by high diversity (Naiman & Decamps 1997). These environments are associated to many ecological services such as linkage aquatic and terrestrial habitats (Dynesius & Nilsson 1994), provide resources to fish (Jansson et al. 2005) and other dispersors as birds and mammals (Gundersen et al. 2010; Naiman & Decamps 1997), promotes refuge for these animals (Bagno & Marinho-Filho 2001; Marinho-Filho & Guimarães 2001; Palmer & Bennett 2006) making key role in diversity maintenance.

However, the vegetation which became near the new margins created by dams are located in hillsides (Truffer et al. 2003), without species associated to high hidric saturation, in other words with different species composition (Acker et al. 2003) and with distinc traits than a tipical riparian vegetation. Terrain with steep slopes facilitates waterflow and reduce water infiltration into the soil (Sidle et al. 2006), moreover, the hills have rocky soil that makes water retention even more difficult. Due these conditions, species of these environments presents adaptation to reduce water loss due hidric stressant habitat, such as leaves loss during dry season, fuits and seeds with low water contend (Murphy & Lugo 1986) and tend to have high density wood to prevent drought-induced embolism (Choat et al. 2003). Thus, it is not only difficult to prevent the waterline proximity consequences to these drought adapted species, also if the “new riparian vegetation” will supply ecological services performed by a typical riparian vegetation.

Many dams have already been built and will still be, therefore understand vegetation changes after damming is crucial to better conservation and future management actions. Then, we monitoring three seasonal forests that suffer a hydroelectric dam impact to answers follow questions: Which were the species best adapted to new conditions imposed by the dam? Which were negatively affected? New species will be established? Are possible local extinctions? Finally, will the “new-riparian vegetation” maintain the ecological services performed by typical riparian vegetation? Previous works showed communities changes after damming (Chapter 3) but not focus species changes, the focus of our study.

Material and Methods

This study was conducted in three dry forest (18°47´40”S, 48°08’57”W, 18°40’31’’ S, 42°24’30’’ W and 18°39’13’’ S, 48°25’04 W; Figure 4.1) located in the Amador Aguiar Complex Dam (two dams located in Araguari River with 52m and 55m depth). All areas had slope terrains, however the deciduous forest inclinations were much more pronounced than semideciduous forest (in some plot the inclination was above 30°). The predominant soil types are dystrophic and eutrophic podzolic soil and dystrophic cambissoil with basalt outcrops with micaxist and biotite-gnaisse (Baccaro et al 2004; Baruqui & Mota 1983, Nishiyama 1989). The first dam (Amador Aguiar Dam I, from here AD1) finished flooding in 2005 and at 624 meters of elevation (relative to sea level), with a flooded area with 18.66km^2 (CCBE 2007). The second dam (Amador Aguiar Dam II, from here AD2) ended flooding in 2006 and at 565 meters of elevation (relative to sea level), with a flooded area with 45.11km^2 (CCBE 2006).

After damming, three seasonal dry forests (two deciduous and one semideciduous forest), before distant at least 200m (see Figure 4.2 A as representation) from any water source, now has the riverbed on its edge since 2005 (AD1)/2006 (AD2). The dam water flux is constant and then the water flux not varies over seasons and over years. Some analysis made in three areas confirms the damming effects to moisture increase in soil at least to 15m away the artificial lake margin created by dam (Chapter 3). This impact clearly affects the entire community (Chapter 3), and the analysis to species responses to river dam were analyzed here. The climate is Aw (Koppen 1948) with a dry winter (april to september) and a rainy summer (october to march), with an average annual temperature of 22°C and average rainfall of around 1595 mm (Santos & Assunção 2006).

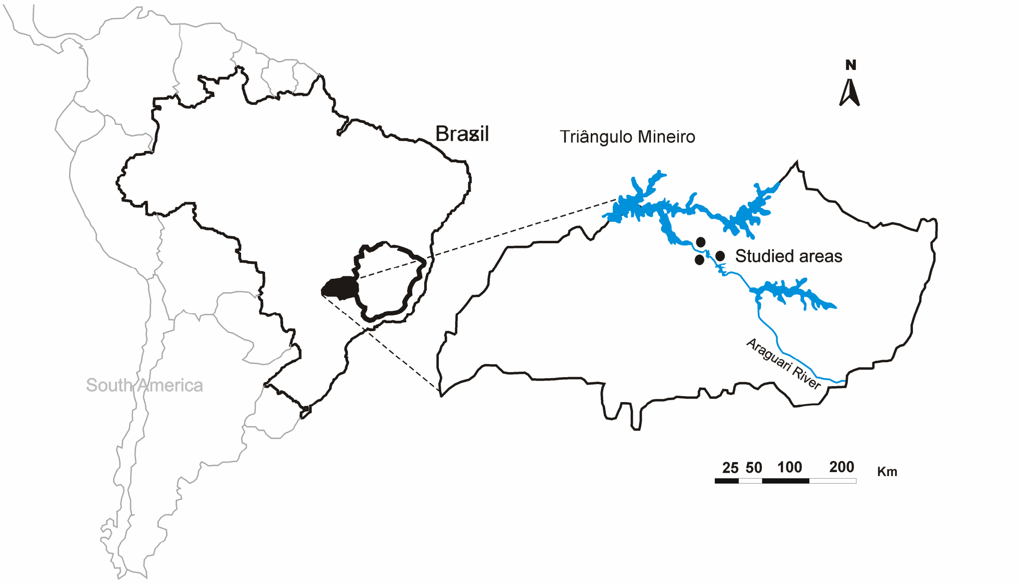

Figure 4.1 - Location of three Dry Forest which were affected by Amador Aguiar Complex Dam, on Triângulo Mineiro, southern Brazil. Before dam construction these forests were distant from water sources and now are on the artificial lake margin.

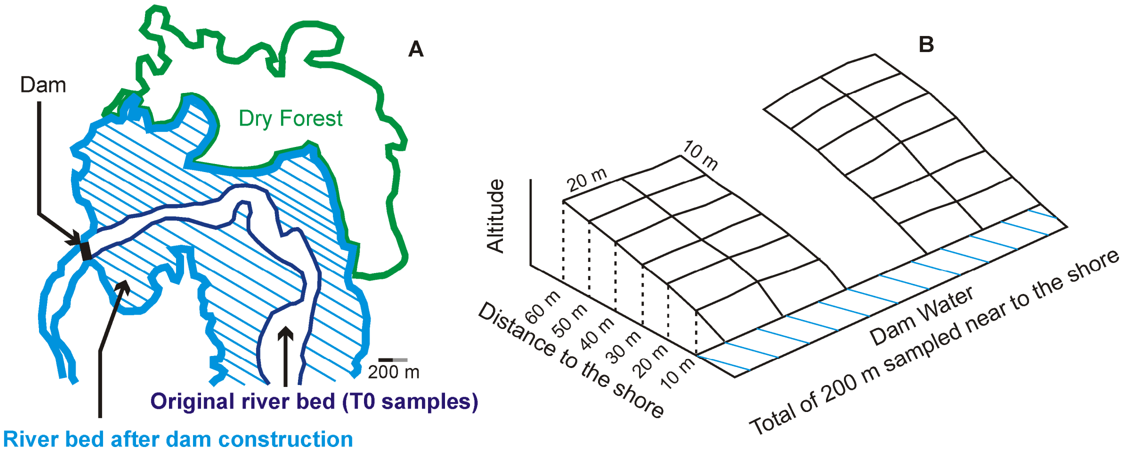

Figure 4.2 – “A” - Representation of upstream landscape changes and water proximity to Dry Forest after dam construction. “B” - Plots scheme used to tree community samples in three Dry Forests.

Plant Sampling -The first inventory (T0) was carried out in 2005 (AD1) and 2006 (AD2). In each forest were marked 60 permanent plots of 20 x 10m totalizing 1.2 ha by area (total of 3.6ha sampled). A total of 10 plots (with 200m width) were installed where the river reached the maximum flood elevation after damming and the remaining plots were installed perpendicular to the river margin (Figure 4.2B). Thus, the samples were distributed every 10 m of perpendicular distance to the river (0-10m, 10-20m, 20-30m, 30-40m, 40-50m and 50-60m of distance – Figure 4.2B) with some samples linked. All trees with diameter at breast height (DBH) of 4.77cm were tagged with aluminum labels. The diameter of the stem was measured at 1.30m from the ground and in multiple stems all live tillers were measure at 1.30m too. All the reproductive botanical material were inserted on Herbarium Uberlandense, and the species nomenclature and synonymies follow Missouri Botanical Garden web site (http://www.tropicos.org/).

The first inventory (T0) results are already published (Kilca et al. 2009; Siqueira et al. 2009). The second (T2) and third (T4) inventories were carried two and four years after damming. All inventories were realized at final of rain season (march-april) to standardize the sampling and to avoid dry season influence on the plant stem diameter by dehydration. These samplings followed the same procedure of the first inventory. The new individuals that met the inclusion criteria (recruits) were measured and identified. The mortality refers to standing dead trees, fallen trees or not found individuals.

Diversity - We calculated the Shannon-Weaver diversity index (Shannon 1948) to measure to changes in diversity over the three measurement periods (T0–T2–T4). We applied the Hutcheson t test (Hutcheson 1970) to compare the richness between T0-T2, T2-T4 and T0-T4 period in all forests.

Dynamic rates - Each species was availed regarding dynamic rates in T0-T2, T2-T4 and T0-T4 periods: mortality, recruitment, outgrowth and ingrowth rates (we focus species with at least 20 individuals, but all results are in appendix). The mortality (M) and recruitment (R) were calculated in terms of annual exponencial rates (formulas on Sheil et al. 1995; Sheil et al. 2000). The outgrowth annual rates (O) refers to basal area of dead trees plus dead branches basal area of living trees (decrement) and ingrowth annual rates (I) refers to basal area of recruits plus growth in basal area of surviving trees (increment). To evaluate changes in forest we computed turnover rates for individuals and basal area through mortality-recruitment rates and outgrowth-ingrowth rates (Oliveira-Filho et al. 2007). Then we evaluated the net change (Korning & Balslev 1994) to individuals and basal area and compute an Overall Net Change (ONC - the average between individuals and basal area net change).

To species with 20 or more individuals in any measurement period, we made dispersion graphic with mortality x recruitment rates and outgrowth x ingrowth rates in T0-T2 and T2-T4 periods, to compare distinct species response in the two first years of damming and the next two years.

Results

Floristic and dynamic - general changes in species

After four years of impoundment, both deciduous forests increase greatly their richness (Table 4.1). At deciduous forest 1 (DF1), the new species found in T2 were *Aspidosperma subincanum*, *Guapira areolata*, *Guarea guidonia*, *Luehea grandiflora*, *Siparuna guianensis*, *Trema micrantha* and *Xylopia aromatica,* and in the T4 were *Inga vera*, *Jacaranda caroba*, *Margaritaria nobilis*, *Myrsine umbellata*, *Trichilia elegans*, *Xylopia brasiliensis* and *Tocoyena formosa*. This last species was found in T0 with one tree but this tree died, however two recruits were sampled in T4. Another species, *Sterculia striata* were not sampled in T2 and T4.

At deciduous forest 2 (DF2), the new species found in T2 were *Cedrela fissilis*, *Eugenia florida*, *Genipa americana*, *Guarea guidonia*, *Luehea grandiflora*, *Nectandra cissiflora*, *Terminalia glabescens*, *Trichilia catigua*, *Trichilia elegans*, *Trichilia pallida* and *Zanthoxylum rhoifolium*. In T4 only two new species were found (*Ceiba speciosa* and *Matayba guianensis*). Otherwise, two species were not found, one in T2 (*Aegiphila sellowiana*) and another in T4 period (*Hymenaea courbaril*).

The semideciduous forest (SF), however, modified little in richness (Table 4.1). The new species sampled in T2 were *Albizia niopoides*, *Heteropteris birsonimifolia*, *Machaerium hirtum*, *Psidium rufum* and *Terminalia phaeocarpa*, and in T4 were *Hirtella gracilipes* and *Cecropia pachystachya*. Otherwise three species were not sampled in T2 period: *Dilodendron bipinnatum*, *Bauhinia rufa* and *Byrsonima laxiflora*.

Table 4.1 - Number of species and diversity Shannon index to three dry forest before (T0), two (T2) and four years (T4) after dam construction in southeastern Brazil. Letters refers to Hutcheson t test, same letters means same diversity, degree of freedom > 500, p < 0.05.

	Arboreal component					
	Number of species			Shannon Index		
	T0	T2	T4	T0	T2	T4
Deciduous 1	45	50	57	2.50a	2.57a	2.69b
Deciduous 2	57	67	68	2.66a	2.83b	2.89b
Semideciduous	89	91	93	3.43a	3.48ab	3.54b

When we consider the occurrence of new species according to the distance from the watertable, was notable the damming effect on richness rise. From 34 new species (accounting all forests) 28 were sampled near to the shore (0-30m of distance) and only six were not sampled in these patches, illustrating dammed influence to new species establishment. Then these sex species were sampled only distant to the shore (30 - 60m).

In all three dry forests the diversity index increase in the first two years and to second year for the fourth year of measurement too (Table 4.1). In DF1, the T2-T4 period presented more diversity changes than T0-T2 (significant, see Table 4.1), nevertheless in DF2, the greatest diversity increase occurred in T0-T2 period (significant, see Table 4.1). In SF the same difference in T0-T2 and T2-T4 were noted, but not significant in both periods. The most valuable results refer to the four-year effect of damming on forest. Comparing T0-T4 period to all dry forest investigated, the Shannon diversity index increase significantly (Table 4.1), confirming the soil moisture influence to richness and diversity raise.

Dynamic rates

The dynamic rates confirm the damming effects in tree community, especially in the first two years exemplified to those species with 20 or more individuals. In this period seven of 10 species showed, at least, a dynamic rate superior to 10% year^{-1} (Table 4.2), values considered extremely high. In DF2 the same were perceived, of 17 species (20 or more individuals) 15 had, at least, one dynamic rate superior than 10% year^{-1} (Table 4.3). However, the semideciduous species demonstrated be more stable regarding the species dynamic in T0-T2 period. Only 5 of 20 species with more than 20 trees had a dynamic rate superior than 10 % year^{-1} (Table 4.4).

These high dynamic rates in the first two years were did not follow the same pattern in subsequent years. To the same species analyzed in T0-T2 period, only one in DF1, four in DF2 and three in SF had a dynamic rate superior than 10% year^{-1}. This contrast between dynamic rates in T0-T2 and T2-T4 illustrate the damming effects on entire community. Many species presents mortality, recruitment, outgrowth and ingrowth rates superior than communities rates (Figure 4.3), thus the dam construction impact was a lot more intense in first two years. These effects were more severe for both deciduous forests (Figure 4.3) due more species present high rates than community (and the community's rates were very high – Table 4.2, 4.3, 4.4). If we analyzed the entire period (T0-T4) the results may not seem significant, because only five species in DF1, nine in DF2 and four in SF had a dynamic rate superior to 10% year^{-1}. This result mask the real and strong changes occurred in all forests, and mainly in the two deciduous forests, therefore the monitoring every two years was essential for understanding of damming effects (and consequently soil moisture increase) to seasonal dry forests.

Individuals Changes

Some changes occurred to more important species in individuals in four years of dam construction (Figure 4.4A, B, C). In DF1, *Myracrodruon urundeuva* was the most important species (individuals), however in T4, due to high mortality (superior to 8% year^{-1}), lost many trees in the community (Figure 4.4A). Same occurred with *Tabebuia roseoalba* and *Campomanesia velutina*, nevertheless with less intensity (Table 4.2). *Anadenanthera colubrina* and *Piptadenia gonoacantha* increase their number in these four years (Figure 4A) due new recruits which reached the inclusion criteria (DAP $\geq$ 5 cm, recruitment rates superior than 10% year^{-1} in T0-T4 period) and presented high recruitment rates. The mortlatity in *P. gonoacantha* was high too, but twice lower than recruitment rates (Table 4.2).

In DF2, again *Myracrodruon urundeuva*, lost many individuals (mortality rates of 4.5% year^{-1}) in T0-T4 period) but *Casearia rupestris*, *Guazuma ulmifolia* and *Aloysia virgata* were most negatively affected species, with high mortality rates (superior than 5% year^{-1}) in four years of monitoring (Table 4.3). Otherwise, *Anadenanthera colubrina*, *Inga sessilis* and *Lonchocarpus cultratus* were species with high increase in individuals (Figure 4.4B) due higher recruitment (rates superior than 7% year^{-1}, Table 4.3).

To semideciduous forest, less changes in more important species occurred (Figure 4.4C): *Casearia grandiflora*, *Cordiera sessilis* and *Terminalia glabescens* had mortality superior than 5% year^{-1} although *T. glabescens* had a great recruitment too (superior than 6% year^{-1}, Table 4.4) meanwhile these changes did not alter the relevance of those species (Figure 4.4C) unlike deciduous forest, whose many modifications were perceptible (Figure 4.4A, 4.4B and 4.4C).

* More information about dynamic of the species with less than 20 trees sampled on the three forests analysed were summarized on Appendix X, XI and XII.

* More information about turnover and net changes of the species with less than 20 trees sampled on the three forests analysed were summarized on Appendix XIII, XIV and XV.

Table 4.2 - Tree species parameters and dynamic rates to a Deciduous Forest (Deciduous Forest 1 – DF1) in southearn Brazil. T0 = before dam construction, T2 = two years after damming, T4 = four years after damming, M = mortality, R = recruitment, O = outgrowth, I = ingrowth. Only species with at least 20 individuals are shown.

Deciduous Forest 1	Number of Individuals			Basal Area			Dynamics Rates T0-T2				Dynamics Rates T2-T4				Dynamics Rates T0-T4			
Species	T0	T2	T4	T0	T2	T4	M	R	O	I	M	R	O	I	M	R	O	I
Acacia polyphylla	40	39	35	0.25	0.25	0.28	19.4	18.4	19.2	19.9	5.3	0.0	3.4	8.5	13.9	11.0	12.6	15.3
Anadenanthera colubrina	141	170	187	3.18	4.67	5.14	1.8	10.9	2.4	19.5	4.8	9.0	4.8	9.2	4.0	10.5	5.0	15.7
Bauhinia ungulata	28	30	29	0.09	0.12	0.13	3.6	6.9	3.5	18.3	5.1	3.5	4.8	6.4	4.8	5.6	5.4	13.7
Campomanesia vellutina	92	89	86	1.23	1.19	1.18	6.2	4.6	6.3	4.8	1.7	0.0	3.2	2.8	4.0	2.4	4.9	4.0
Casearia rupestris	24	21	19	0.18	0.17	0.18	13.4	7.4	11.4	8.6	4.9	0.0	5.5	8.9	9.6	4.2	9.0	9.3
Guazuma ulmifolia	19	22	23	0.17	0.25	0.29	0.0	7.1	2.6	19.0	4.7	6.7	2.9	9.4	2.7	7.3	3.5	15.0
Myracrodruon urundeuva	279	238	199	5.33	5.61	5.44	8.2	0.6	3.7	6.1	9.0	0.5	4.2	2.7	8.7	0.6	4.2	4.7
Piptadenia gonoacantha	71	96	99	0.80	1.02	1.05	11.2	23.6	16.9	26.5	7.0	8.4	14.2	15.3	11.8	18.8	22.8	27.9
Platypodium elegans	21	21	20	0.58	0.65	0.72	2.4	2.4	6.4	11.9	2.4	0.0	0.5	5.5	2.5	1.3	3.6	8.8
Tabebuia roseoalba	246	237	231	2.86	3.05	3.12	3.9	2.1	2.4	5.5	3.0	1.7	1.9	3.1	3.5	2.0	2.3	4.4
Entire Community	1124	1136	1133	16.80	19.16	20.54	6.5	7.1	4.9	11.0	4.9	4.7	4.0	6.6	6.1	6.3	5.0	9.2

Table 4.3 - Tree species parameters and dynamic rates to a Deciduous Forest (Deciduous Forest 2) in southearn Brazil. T0 = before dam construction, T2 = two years after damming, T4 = four years after damming, M = mortality, R = recruitment, O = outgrowth, I = ingrowth. Only species with at least 20 individuals are shown.

Deciduous Forest 2	Number of Individuals			Basal Area			Dynamics Rates T0-T2				Dynamics Rates T2-T4				Dynamics Rates T0-T4			
Species	T0	T2	T4	T0	T2	T4	M	R	O	I	M	R	O	I	M	R	O	I
Allophylus racemosus	13	22	22	0.05	0.08	0.09	3.9	26.1	3.1	23.8	2.3	2.3	3.3	7.4	4.1	15.9	4.5	17.1
Aloysia virgata	43	35	25	0.20	0.16	0.12	12.4	2.9	14.3	5.3	15.5	0.0	15.5	1.7	14.5	2.1	15.9	4.6
Anadenanthera colubrina	68	88	93	2.20	2.53	2.97	3.7	15.4	11.9	17.8	1.7	4.4	0.4	8.2	3.1	10.4	6.4	13.2
Aspidosperma parvifolium	20	23	24	0.34	0.19	0.38	0.0	6.7	0.0	11.1	2.2	4.3	5.2	32.3	1.3	5.7	1.5	23.0
Campomanesia velutina	27	31	32	0.13	0.16	0.17	1.9	8.4	3.3	12.0	0.0	1.6	1.9	5.5	0.9	5.1	2.9	9.1
Casearia gossypiosperma	21	32	37	0.08	0.13	0.16	0.0	19.0	0.1	21.9	0.0	7.0	0.4	8.9	0.0	13.2	0.4	15.8
Casearia rupestris	165	149	136	0.88	0.93	0.89	9.2	3.8	8.6	11.0	4.9	0.4	4.3	2.2	7.3	2.3	7.1	7.2
Celtis iguanaea	29	31	27	0.25	0.25	0.22	5.3	8.4	12.6	13.2	6.7	0.0	13.1	7.8	6.7	5.0	15.3	13.0
Coccoloba mollis	33	31	32	0.23	0.23	0.24	6.3	1.6	11.9	11.2	1.6	3.2	4.6	6.7	4.0	2.4	9.0	9.6
Dilodendron bipinnatum	46	45	46	0.35	0.46	0.54	2.2	1.1	3.9	16.2	0.0	1.1	0.4	7.8	1.1	1.1	2.2	12.2
Guazuma ulmifolia	478	430	414	4.75	5.08	5.54	7.6	2.1	5.7	8.7	2.6	0.6	2.7	6.7	5.2	1.4	4.5	8.0
Handroanthus vellozoi	20	18	17	0.16	0.16	0.17	5.1	0.0	3.7	6.6	2.8	0.0	1.0	3.2	4.0	0.0	2.4	5.0
Inga sessilis	170	202	200	1.14	1.45	1.53	4.5	11.8	6.9	17.1	3.0	2.5	4.0	6.5	4.2	7.7	6.4	12.8
Lonchocarpus cultratus	32	52	54	0.27	0.44	0.54	3.2	21.6	3.4	23.9	2.9	4.7	0.9	10.2	4.2	14.4	2.5	17.6
Machaerium brasiliensis	22	24	26	0.11	0.17	0.20	0.0	4.3	0.0	20.3	0.0	3.9	0.1	9.1	0.0	4.1	0.0	14.9
Myracrodruon urundeuva	225	208	197	3.75	4.10	4.34	5.5	1.0	4.3	8.6	3.4	0.5	2.1	4.6	4.5	0.8	3.4	6.8
Rhamnidium elaeocarpum	34	39	37	0.20	0.25	0.26	1.5	8.0	1.5	11.9	2.6	0.0	4.4	5.5	2.3	4.3	3.6	9.3
Entire Community	1626	1670	1649	18.03	20.23	21.75	6.0	6.9	5.9	11.7	3.0	2.3	3.3	6.7	4.8	4.8	5.1	9.7

Table 4.4 - Tree species parameters and dynamic rates to a Semideciduous Forest (SF) in southearn Brazil. T0 = before dam construction, T2 = two years after damming, T4 = four years after damming, M = mortality, R = recruitment, O = outgrowth, I = ingrowth. Only species with at least 20 individuals are shown.

Semideciduous Forest	Number of Individuals			Basal Area			Dynamics Rates T0-T2				Dynamics Rates T2-T4				Dynamics Rates T0-T4			
Species	T0	T2	T4	T0	T2	T4	M	R	O	I	M	R	O	I	M	R	O	I
Apuleia leiocarpa	26	29	33	0.82	0.93	1.01	0.0	5.3	0.0	6.3	0.0	6.3	0.1	3.7	0.0	5.8	0.1	5.0
Aspidosperma discolor	121	119	127	2.31	2.59	2.90	2.9	2.1	2.8	8.2	0.0	3.2	0.1	5.6	1.5	2.7	1.4	6.9
Astronium nelson-rosae	99	104	107	1.13	1.27	1.40	2.0	4.4	1.9	7.4	0.5	1.9	0.5	5.3	1.3	3.2	1.2	6.4
Callisthene major	55	56	56	4.21	4.45	4.51	2.8	3.6	0.8	3.5	2.7	2.7	2.0	2.7	2.8	3.3	1.5	3.2
Casearia gossypiosperma	19	23	24	0.22	0.26	0.28	0.0	9.1	0.3	9.4	0.0	2.1	0.0	3.8	0.0	5.7	0.2	6.6
Casearia grandiflora	136	115	119	0.99	0.89	0.91	11.7	4.4	12.3	7.7	3.1	4.3	4.3	5.4	7.6	4.5	8.7	6.9
Cheiloclinium cognatum	103	100	105	0.64	0.62	0.73	5.0	3.6	7.4	6.1	0.5	2.9	1.5	8.8	2.8	3.3	4.6	7.5
Copaifera langsdorffii	81	76	77	6.74	7.12	7.64	3.1	0.0	2.5	5.2	0.0	0.7	0.0	3.5	1.6	0.3	1.3	4.3
Cordiera sessilis	152	137	125	1.14	1.05	1.01	6.1	1.1	7.8	3.9	4.5	0.0	5.5	3.8	5.3	0.6	6.9	4.1
Diospyrus hispida	25	21	21	0.57	0.52	0.56	8.3	0.0	12.6	8.4	0.0	0.0	0.1	4.1	4.3	0.0	6.6	6.3
Duguetia lanceolata	54	56	60	0.46	0.50	0.55	4.7	6.5	5.5	9.4	0.9	4.3	1.8	6.9	2.9	5.4	3.9	8.3
Heisteria ovata	35	34	33	0.26	0.27	0.31	5.9	4.5	6.7	8.2	1.5	0.0	0.5	6.7	3.8	2.4	3.7	7.5
Luehea grandiflora	30	33	40	0.30	0.29	0.30	5.1	9.5	9.8	8.1	1.5	10.6	6.6	7.3	3.5	10.2	8.9	8.3
Mabea fistulifera	25	27	32	0.53	0.59	0.57	6.2	9.7	2.4	8.0	3.8	11.6	8.5	6.7	5.4	11.1	6.3	8.2
Micropholis venulosa	33	31	33	0.56	0.65	0.75	4.7	1.6	5.0	12.1	0.0	3.1	0.2	6.5	2.4	2.4	2.7	9.4
Myrcia floribunda	101	105	104	0.80	0.84	0.92	2.5	4.4	5.9	8.2	1.4	1.0	1.2	5.6	2.0	2.8	3.7	7.0
Protium heptaphyllum	37	37	39	0.53	0.56	0.60	5.6	5.6	2.1	5.4	1.4	3.9	1.4	4.6	3.6	4.8	1.8	5.1
Siparuna guianensis	16	16	24	0.06	0.05	0.14	33.9	33.9	30.9	24.5	3.2	20.9	7.7	9.7	21.7	29.3	22.9	10.1
Siphoneugena densiflora	19	22	23	0.19	0.21	0.23	0.0	7.1	0.5	4.6	2.3	4.4	0.8	5.3	1.3	5.9	0.7	5.0
Terminalia glabescens	73	71	77	0.72	0.79	0.92	8.6	7.3	8.1	11.9	1.4	5.3	1.1	8.5	5.2	6.4	4.8	10.4
Entire Community	1501	1489	1573	27.99	29.16	31.66	5.5	5.2	4.5	6.5	1.6	4.2	1.4	5.5	3.7	4.8	3.1	6.1

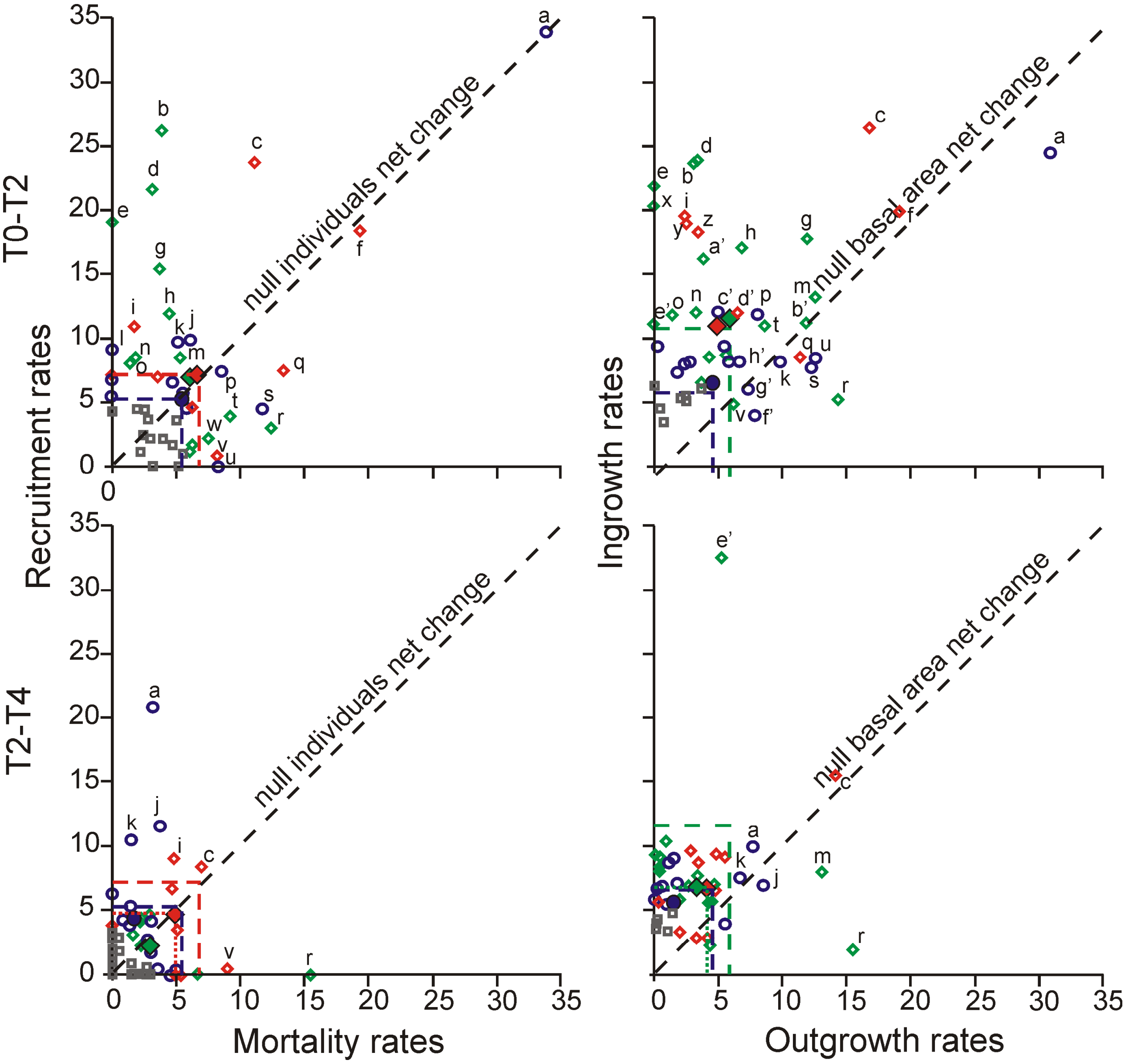

Figure 4.3 - Comparison of recruitment/mortality (A and C) and ingrowth/outgrowth rates (B and D) in species with 20 or more individuals in Dry Forests in southern Brazil. Blue circles = semideciduous forest species; green diamond = deciduous forest 1 species; red diamond = deciduous forest 2 species; green square are species with low dynamic rates even in T0-T2 period; close blue circles = entire semideciduous forest; close green diamond = entire deciduous forest 1; close red diamond = entire deciduous forest 2; dashed lines indicates the entire community rates on T0-T2 period. **a**=*Siparuna guianensis*; **b**=*Allophylus racemosus;* **c**=*Piptadenia gonoacantha*; **d**=*Lonchocarpus cultratus*; **e**=*Casearia gossypiosperma*; **f**=*Acacia polyphylla*; **g**=*Anadenanthera colubrina*; **h**=*Inga sessilis*; **i**=*Anadenanthera colubrina*; **j**=*Mabea fistulifera*; **k**=*Luehea grandiflora*; **l**=*Casearia gossypiosperma*; **m**=*Celtis iguanaea*; **n**=*Campomanesia velutina*; **o**=*Rhamnidium elaeocarpum*; **p**=*Terminalia glabescens;* **q**=*Casearia rupestris*; **r**=*Aloysia virgata*; **s**=*Casearia grandiflora*; **t**=*Casearia rupestris*; **u**=*Dyospirus hispida*; **v**=*Myracrodruon urundeuva*; **w**=*Guazuma ulmifolia*; **x**=*Machaerium brasiliensis*; **y**=*Guazuma ulmifolia*; **z**=*Bauhinia ungulata*; **a’**=*Dilodendron bipinnatum*; **b’**=*Coccoloba mollis*; **c’**=*Micropholis venulosa*; **d’**=*Platypodium elegans*; **e’**=*Aspidosperma olivaceum*; **f’**=*Cordiera sessilis*; **g’**=*Cheiloclinium cognatum*; **h’**=*Duguetia lanceolata.*

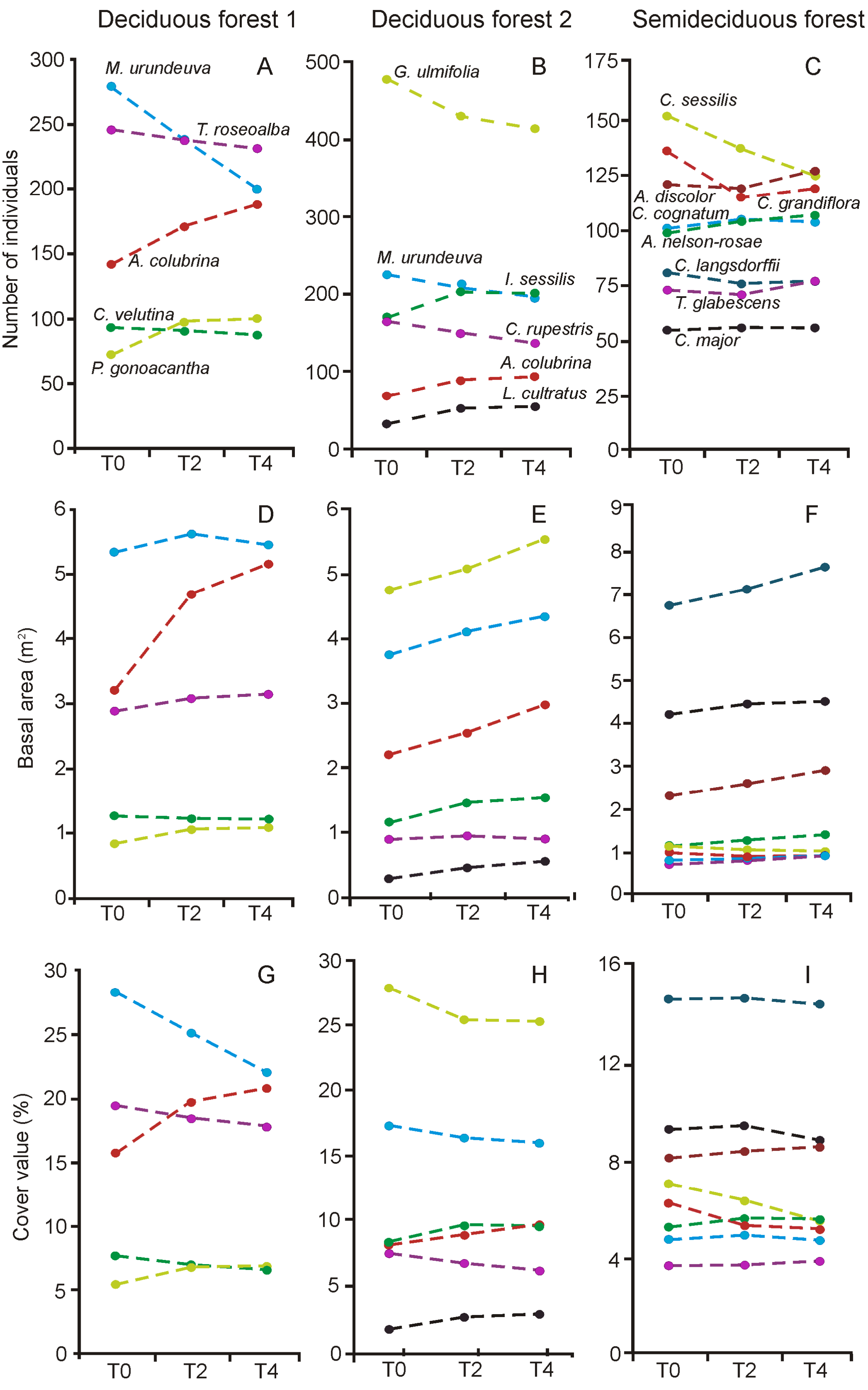

Figure 4.4 - Number of individuals, basal area and cover value to non-stable species in three Dry Forests in southern Brazil, only those with 20 or more individuals and only those species that represents most changes in the tree community. A, D and G represents deciduous forest changes 1; B, E and H represents deciduous forest changes 2 and C, F and I represents semideciduous forest changes.

Basal area changes

Although many species lost basal area due to high mortality, to both deciduous forests was the vast majority species showed an extremely high increment (gain in basal area Figure 4.4D and 4.4E), especially in first two years after impoundment (frequently bigger than 10% year^{-1} - Tables 4.2 and 4.3). In SF increments were high to many species too (Tables 4.5), but this growth in basal area has been less intense thereby, for all three forest the increment hide the high basal area loss (outgrowth) by mortality. Thus, for all seasonal forests, species outgrowth rates (mortality plus broken stem loss basal area) frequently were lower than ingrowth (increment plus recruitment gain basal area) with only three exceptions to DF1, two in DF2 and six in SF. Among these exception, only one species in DF2 (*Aloysia virgata*) and one in SF (*Siparuna guianensis*) had an outgrowth rates much larger than ingrowth, which means that species are seriously affected by the damming, although *S. guianensis* still had many recruitment rates.

Turnover and Net Changes

The high dynamic rates in first two years (Tables 4.5, 4.6 and 4.7) leads to major species turnover rates (individuals and basal area) in T0-T2 than T2-T4 period (Figure 4.3). Even to some important species (either to individuals or to basal area), turnover were high. Exchange of died individuals by new recruits determines dynamic speed to each species, therefore those species with high turnover represents unstable species. Despite of first two years impact in species, the T2-T4 period turnover was important because indicate species that are still very affected by damming. Thus, in all forests, *Aspidosperma olivaceum*, *Celtis iguanaea*, *Piptadenia gonoacantha* and *Siparuna guianensis* were the only ones with individuals or basal area turnover rates superior than 10% year^{-1}. These species had to be considered indicator to infer changes in communities which suffered similar impacts. If impact still affects the community, these species will present high turnover rates.

The individuals net changes varies greatly in analyzed species, nevertheless to basal area, major species had a positive value (Figura 4.3), and this exemplifies the strong trees increment after water approach. Both for individuals and basal area, most species follow the same dynamic path, those that had positive net change in first two years, had the same pattern in next period (T2-T4), however with lower values (especially for basal area – Tables 4.5, 4.6 and 4.7).

Finally, the overall net chance - ONC (average between individuals and basal area net change) showed that few species presents a negative value after four years of impoundment (Tables

4.5, 4.6 and 4.7), then few species clearly reduces their importance to each area (Figure 4.4G, H, I). This is the effect of the high tree increment and recruitment after damming, and demonstrated that all three communities (and many important species) responded similarly if we compared T0-T4 value. However, contrasting both measurement periods, SF had an ONC superior in T2-T4 than deciduous and the opposite occurred in T0-T2.

Table 4.5 - Tree species turnover and net change to a Deciduous Forest (Deciduous Forest 1 - DF1) by descending order of overall net change in T0-T4 period, in southeastern Brazil. Ind = individuals, BA = Basal area. Only species with at least 20 individuals are shown.

Deciduous Forest 1	Turnover (Ind)			Turnover (BA)			Net Change (Ind)			Net Change (BA)			Overall Net Change		
Species	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4
Anadenanthera colubrina	6.3	6.9	7.2	10.9	7.0	10.4	9.8	4.9	7.3	21.2	4.9	12.8	15.5	4.9	10.0
Guazuma ulmifolia	3.5	5.7	5.0	10.8	6.1	9.3	7.6	2.2	4.9	20.4	7.2	13.6	14.0	4.7	9.2
Piptadenia gonoacantha	17.4	7.7	15.3	21.7	14.8	25.4	16.3	1.6	8.7	13.1	1.4	7.1	14.7	1.5	7.9
Bauhinia ungulata	5.3	4.3	5.2	10.9	5.6	9.5	3.5	-1.7	0.9	18.2	1.7	9.7	10.8	0.0	5.3
Entire Community	**6.8**	**4.8**	**6.2**	**8.0**	**5.3**	**7.1**	**0.5**	**-0.1**	**0.2**	**6.8**	**3.5**	**5.1**	**3.7**	**1.7**	**2.7**
Platypodium elegans	2.4	1.2	1.9	9.1	3.0	6.2	0.0	-2.4	-1.2	6.2	5.3	5.7	3.1	1.4	2.3
Tabebuia roseoalba	3.0	2.4	2.8	4.0	2.5	3.4	-1.8	-1.3	-1.6	3.2	1.2	2.2	0.7	0.0	0.3
Acacia polyphylla	18.9	2.6	12.4	19.6	6.0	13.9	-1.3	-5.3	-3.3	0.9	5.6	3.2	-0.2	0.2	0.0
Campomanesia velutina	5.4	0.8	3.2	5.6	3.0	4.5	-1.6	-1.7	-1.7	-1.5	-0.4	-1.0	-1.6	-1.1	-1.3
Casearia rupestris	10.4	2.4	6.9	10.0	7.2	9.2	-6.5	-4.9	-5.7	-3.1	3.8	0.3	-4.8	-0.5	-2.7
Myracrodruon urundeuva	4.4	4.8	4.7	4.9	3.4	4.5	-7.6	-8.6	-8.1	2.6	-1.5	0.5	-2.5	-5.1	-3.8

Table 4.6 - Tree species turnover and net change to a Deciduous Forest (Deciduous Forest 2 – DF2) by descending order of overall net change in T0-T4 period, in southeastern Brazil. Ind = individuals, BA = Basal area. Only species with at least 20 individuals are shown.

Deciduous Forest 2	Turnover (Ind)			Turnover (BA)			Net Change (Ind)			Net Change (BA)			Overall Net Change		
Species	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4
Casearia gossypiosperma	9.5	3.5	6.6	11.0	4.7	8.1	23.4	7.5	15.2	27.9	9.4	18.3	25.7	8.4	16.7
Lonchocarpus cultratus	12.4	3.8	9.3	13.6	5.5	10.0	27.5	1.9	14.0	28.4	10.3	19.0	27.9	6.1	16.5
Allophylus racemosus	15.0	2.3	10.0	13.4	5.4	10.8	30.1	0.0	14.1	27.1	4.5	15.2	28.6	2.2	14.6
Machaerium brasiliensis	2.1	2.0	2.0	10.2	4.6	7.5	4.4	4.1	4.3	25.5	10.0	17.5	15.0	7.0	10.9
Anadenanthera colubrina	9.6	3.1	6.7	14.8	4.3	9.8	13.8	2.8	8.1	7.1	8.5	7.8	10.4	5.7	8.0
Inga sessilis	8.2	2.8	6.0	12.0	5.2	9.6	9.0	-0.5	4.1	12.6	2.7	7.6	10.8	1.1	5.9
Dilodendron bipinnatum	1.7	0.5	1.1	10.0	4.1	7.2	-1.1	1.1	0.0	14.7	8.1	11.3	6.8	4.6	5.7
Campomanesia velutina	5.1	0.8	3.0	7.7	3.7	6.0	7.2	1.6	4.3	9.9	3.8	6.8	8.5	2.7	5.6
Rhamnidium elaeocarpum	4.7	1.3	3.3	6.7	4.9	6.5	7.1	-2.6	2.1	11.8	1.1	6.3	9.5	-0.7	4.2
Aspidosperma parvifolium	3.4	3.2	3.5	5.6	18.8	12.2	7.2	2.2	4.7	-24.9	40.0	2.5	-8.9	21.1	3.6
Entire Community	**6.5**	**2.6**	**4.8**	**8.8**	**5.0**	**7.4**	**1.3**	**-0.6**	**0.4**	**5.9**	**3.7**	**4.8**	**3.6**	**1.5**	**2.6**
Myracrodruon urundeuva	3.2	2.0	2.6	6.4	3.4	5.1	-3.9	-2.7	-3.3	4.6	2.8	3.7	0.4	0.1	0.2
Guazuma ulmifolia	4.9	1.6	3.3	7.2	4.7	6.2	-5.2	-1.9	-3.5	3.4	4.4	3.9	-0.9	1.3	0.2
Coccoloba mollis	3.9	2.4	3.2	11.5	5.7	9.3	-3.1	1.6	-0.8	-0.2	2.2	1.0	-1.7	1.9	0.1
Handroanthus vellozoi	2.6	1.4	2.0	5.1	2.1	3.7	-5.1	-2.8	-4.0	3.0	2.3	2.7	-1.1	-0.3	-0.7
Celtis iguanae	6.9	3.3	5.8	12.9	10.4	14.2	3.4	-6.7	-1.8	0.7	-5.8	-2.6	2.0	-6.2	-2.2
Casearia rupestris	6.5	2.6	4.8	9.8	3.2	7.1	-5.6	-4.5	-5.1	2.6	-2.2	0.2	-1.5	-3.4	-2.4
Aloysia virgata	7.6	7.7	8.3	9.8	8.6	10.3	-9.8	-15.5	-12.7	-9.6	-14.1	-11.9	-9.7	-14.8	-12.3

Table 4.7 - Tree species turnover and net change to a Semideciduous Forest (SF) by descending order of overall net change in T0-T4 period, in southeastern Brazil. Ind = individuals, BA = Basal area. Only species with at least 20 individuals are shown.

Semideciduous Forest	Turnover (Ind)			Turnover (BA)			Net Change (Ind)			Net Change (BA)			Overall Net Change		
Species	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4
Siparuna guianensis	33.9	12.1	25.5	27.7	8.7	16.5	0.0	22.5	10.7	-8.5	62.6	22.0	-4.2	42.6	16.3
Casearia gossypiosperma	4.6	1.1	2.8	4.8	1.9	3.4	10.0	2.2	6.0	10.1	3.9	6.9	10.0	3.0	6.5
Apuleia leiocarpa	2.7	3.1	2.9	3.2	1.9	2.6	5.6	6.7	6.1	6.7	3.7	5.2	6.1	5.2	5.7
Siphoneugena densiflora	3.5	3.4	3.6	2.5	3.1	2.8	7.6	2.2	4.9	4.3	4.7	4.5	5.9	3.5	4.7
Mabea fistulifera	8.0	7.7	8.3	5.2	7.6	7.3	3.9	8.9	6.4	6.0	-1.9	2.0	5.0	3.5	4.2
Terminalia glabescens	7.9	3.4	5.8	10.0	4.8	7.6	-1.4	4.1	1.3	4.4	8.1	6.2	1.5	6.1	3.8
Duguettia lanceolata	5.6	2.6	4.2	7.5	4.3	6.1	1.8	3.5	2.7	4.3	5.5	4.9	3.0	4.5	3.8
Astronium nelson-rosae	3.2	1.2	2.2	4.6	2.9	3.8	2.5	1.4	2.0	5.9	5.1	5.5	4.2	3.3	3.7
Micropholis velunosa	3.1	1.5	2.4	8.6	3.3	6.0	-3.1	3.2	0.0	8.0	6.8	7.4	2.5	5.0	3.7
Aspidosperma discolor	2.5	1.6	2.1	5.5	2.9	4.2	-0.8	3.3	1.2	5.9	5.9	5.9	2.5	4.6	3.5
Luehea grandiflora	7.3	6.0	6.9	9.0	7.0	8.6	4.9	10.1	7.5	-1.9	0.7	-0.6	1.5	5.4	3.4
Protium heptaphyllum	5.6	2.6	4.2	3.7	3.0	3.4	0.0	2.7	1.3	3.4	3.4	3.4	1.7	3.1	2.4
Myrcia floribunda	3.4	1.2	2.4	7.0	3.4	5.3	2.0	-0.5	0.7	2.5	4.7	3.6	2.2	2.1	2.2
Entire Community	**5.3**	**2.9**	**4.2**	**5.5**	**3.4**	**4.6**	**-0.4**	**2.8**	**1.2**	**2.1**	**4.2**	**3.1**	**0.8**	**3.5**	**2.2**
Cheiloclinium cognatum	4.3	1.7	3.0	6.7	5.1	6.1	-1.5	2.5	0.5	-1.4	8.1	3.2	-1.4	5.3	1.9
Heisteria ovata	5.2	0.7	3.1	7.4	3.6	5.6	-1.4	-1.5	-1.5	1.6	6.6	4.1	0.1	2.6	1.3
Callisthene major	3.2	2.7	3.1	2.1	2.4	2.3	0.9	0.0	0.5	2.8	0.7	1.8	1.9	0.4	1.1
Copaifera langsdorffii	1.6	0.3	1.0	3.8	1.8	2.8	-3.1	0.7	-1.3	2.8	3.6	3.2	-0.2	2.1	1.0
Diospyrus hispida	4.2	0.0	2.1	10.5	2.1	6.4	-8.3	0.0	-4.3	-4.6	4.2	-0.3	-6.5	2.1	-2.3
Casearia grandiflora	8.1	3.7	6.1	10.0	4.8	7.8	-8.0	1.7	-3.3	-5.1	1.4	-1.9	-6.6	1.5	-2.6
Cordiera sessilis	3.6	2.2	3.0	5.9	4.6	5.5	-5.1	-4.5	-4.8	-4.0	-1.8	-2.9	-4.6	-3.1	-3.9

Discussion

Richness and diversity increase

A surprisingly finding of this study was quick changes in the richness and diversity of dry forest tree communities. Many studies in impacted forests demonstrated structural changes few years after a great disturbance like storms (Burslem et al. 1995; Laurance et al. 2006; Pascarella et al. 2004; Walker 1991), fragmentation due edge effects (Laurance et al. 2002; Laurance et al. 2006), timber removal (Guariguata et al. 2008) and severe dry periods (Chazdon et al. 2005; Chapter 3), nevertheless with forest recovery of structure and composition over the years (Chazdon et al. 2007). In general, only long time works showed tree species changes and their probable consequences to community (Laurance et al. 2006) because trees could be long lived and changes by disturbances should be slowly detected. These quick richness and diversity increase verified to all dry forests analyzed support the hypothesis of great changes caused by dam construction, even in tree community. The main factor was the increase in amount of water available, a barrier to growth in the dry season before the damming (Chapter 3) to many species. With water available on dry periods after impoundment (Chapter 3) there was no water restriction and more plant species could growth until reach the inclusion criteria.

Most of the new species recruited probably were already present in the community as small individuals or saplings with deficiency in growth due to water stress. Summarizing these new species sampled, at least 20 are water associated species, found in non-amazonian riparian forest (Rodrigues & Nave 2000) moist atlantic forest (Oliveira & Fontes 2000) or even distributed in wet environments of gallery and riparian forests (Oliveira-Filho & Ratter 2002; Silva Júnior et al. 2001) or flooded forests (Silva et al. 2007). Hence prolonged dry periods could act as a negative filter to these species in original dry condition, killing or at least difficult their establishment. The rise in soil moisture due dam construction (Chapter 3) broke the markedly moist seasonality presented on soil of these forests, favoring the establishment of water-associated species.

Is important note that the new condition created by damming is not transitory, then other tree species can be establish in these community over the years, and community will never come back to original stage. Germination is influenced by water (Breshears et al. 1998) and some species should meet good condition to stabilize and even fruits and seed dispersed from other areas should increase species richness too. The short period of monitoring and the inclusion criteria (only trees with five or more centimeters of diameter were sampled) difficult these affirmations of germination and dispersion influences on richness. However, a regeneration work in these areas show distinct seedling and saplings responses of the two most important species in these forests (*A. colubrina* and *M. urundeuva*) demonstrated that *M. urundeuva* had a more negative response than *A. colubrina* to soil water increase (Gusson et al. 2011), then the effects on germination were verified. More, other dam studies comparing free-flowing rivers with regulated rivers and shows some positive effects of damming to plant richness due dispersion (Jansson et al. 2000) and germination (Andersson et al. 2000). The rise in richness and diversity should be treated with caution. The increase in richness and diversity will never overcome the loss of species drowned by the damming. This increase in richness may be treated as one more impact of dams to the flora.

Studies in cold environments affected by dams found species changes (Jansson et al. 2000; Nilsson et al. 2002), but concluded that both richness and diversity were not the most sensitive indicator of effects of flow regulation (Dynesius et al. 2004). Our results, however, suggest high modification in both richness and diversity in only four years of impact. Probably the impacts on species pool were high, due high biodiversity in tropical environments, and several shifts in species should be expected in any overflow in tropics. This is a key problem because the most diverse tropical system are affected by dams (Nilsson et al. 2005) and all forests on tropics whose suffer similar overflow after damming tend to presents high species chances.

It is difficult to imagine how damming affects forest communities all of the world, but the changes presented here direct to a dramatic scenario with huge modifications. More, the damming influence on recruitment of water-associated species was strongest on patches near the river (0-30 m of distance to the shore), which was twice bigger of those sampled distant to shore (30-60 m). Thus damming effect to community and to some species in special were concentrated near the riverbed (Chapter 3), just the main area to conservation efforts due ecological services like soil protection against erosion and siltation (Guo et al. 2007; Hubble et al. 2010; Kiley & Schneider 2005), aquatic fauna and corridor to fauna movements (Gundersen et al. 2010), pathways to plant dispersion (Naiman & Decamps 1997; Nilsson & Berggren 2000). These areas which become near artificial lakeshore presented high impact and should be monitored for several years, then many implications for ecosystems been more understood.

Water restriction is a common event to seasonal environments, but is harsher in deciduous than semideciduous forests. The mountainous terrains with high slope and rocky soils in deciduous forests (Oliveira-Filho & Ratter 2002) facilitate the water flow in rain periods and difficult water infiltration (Baker et al. 2002). On semideciduous forest the water stress is less intense due more clayey soils and less slope terrain hence less new species were found.

Due to be a more water stressful environment, deciduous forests support less basal area and species (Scariot & Sevilha 2005) and had stronger deciduousness during dry season than semideciduous forests. With dam construction the watertable proximity to forest increase subsoil water reserves (Chapter 3) which is the ecophysiological basis for evergreen maintenance (Borchert 1998; Nepstad et al. 1994), therefore in deciduous forest, environment during dry season become milder (Chapter 3), facilitating growth of evergreen species (most new species sampled were evergreen). Evergreen species had more advantages when environment is not water deficient. Deciduous species had more photosynthetic capacity (Reich et al. 2003), but lost part of the carbon acquired due leaves fall, otherwise evergreen not loss much carbon during dry season then were photosynthetic active during dry season (Chabot & Hicks 1982). In general, evergreen species had depth roots with more secondary and lateral roots (Markesteijn et al. 2010), then is expensive maintain root biomass with less carbon gain during dry seasons (Wright & Vanschaik 1994).

However, with water supply all over the year, photosynthesis not had a limiter and evergreen plants could present high growth rates. Thus, in long-term we expected a conversion of physiognomies near riverbed, of an original deciduous forest to a more perennial environment (a semideciduous forest, but still with marked deciduousness due long lived deciduous trees). What about new deciduous species sampled? From all these “new-deciduous species” only three present intermediary-high density wood (superior than $0.65g.cm^{-3}$). Deciduous trees with lower wood density are more vulnerable to drought-induced embolism and cavitation (Choat et al. 2003; Choat et al. 2005), then intense dry periods tend to be more negative for low density wood deciduous plants (Markesteijn et al. 2010). With rise in soil moisture, the risk of low water problems to sap transportation is reduced; favors plant fitness and their survivor on the new conditions, then low density wood were favored.

However, enhance in diversity occur to all forest due two main changes in these communities. First and obviously were the new species found; richness interferes in diversity index (Smith & Wilson 1996). Second and less obviously is the number of individuals from each species. A community dominated by few species tends to present low diversity and this occurs in deciduous forest. In these forests two or three species tend to dominate the community, mainly *Myracrodruon urundeuva* and *Anadenanthera colubrina*. Water proximity kills many trees of some important species in deciduous forests analyzed (*Myracrodruon urundeuva*, *Tabebuia roseoalba*, *Guazuma ulmifolia* and *Casearia rupestris*), but the total number of individuals for all species remained similar after four years. This means that other species is rise their importance and, by consequence, the community diversity. The same effect occur to semideciduous forest too, by many trees died in most dense species (*Cordiera sessilis*, a understory high density wood species) after damming.

The “Riparian Effect”

The precipitation water regime and groundwater depths strongly influence species composition, community structure and biological diversity (Ehleringer & Dawson 1992; Munoz-Reinoso 2001; Naiman & Decamps 1997; Stromberg et al. 1996), and water-stressfull environment can raise the richness and diversity of trees after water availability changes (Xu et al. 2009). In global scale, moist forests have more biodiversity (Gaston 2000) in places without energy restriction like tropics (O'Brien et al. 2000). Considering that energy on these systems not varies on studied forests, the clear factor that enhances richness was change of a “common dry forest” to an “artificial riparian dry forest” due soil moisture increase (we call this event “Riparian Effect”).

Riparian forests are a transition zone between land and aquatic systems and support more plant richness than surrounding areas (Naiman & Decamps 1997; Nilsson & Berggren 2000) because presents flora associated to moisture and dryer patches (Chapter 1). The increase in richness and diversity, however, will not means a “total” conversion of these dry forests in a typical riparian forest due maintain of most species on the community, and few species lost. Riparian forest were richness systems (Rodrigues & Nave 2000), due high heterogeneity like floods (Lopes & Schiavini 2007), distinct water flow (Jansson et al. 2000), and great soil moisture variations (Rodrigues et al. 2010; Chapter 2) and, despite new species sampled, some characteristics of original forest become constant.

Some of these “heterogeneity creators” in a natural riparian forest did not occur in three forests analyzed here. First, floods did not occur due dam flow be controlled by an upstream dam, thus watertable not varies and soil moisture near stream should have few changes over next years. Flood frequency and variations of watertable depth alters increase habitat complexity (Naiman & Decamps 1997) creating conditions to distinct species growth (Lopes & Schiavini 2007). Second, new artificial lake created did not had a current water flow, then the sediments and seeds deposition from upstream plants will not occur. Flow regime influence species composition and distribution at small scales (Bendix & Hupp 2000; Hughes & Rood 2003), because many seeds are dispersed by hidrochory (Jansson et al. 2000) and soil deposition creates patches with distinct soil infiltration and nutrients (Rodrigues et al. 2010), increasing environment heterogeneity patches to distinct species establish. Therefore, free-flowing rivers become more species-richness than regulated ones after long periods (Dynesius et al. 2004; Nilsson et al. 1997).

Finally, and specifically to deciduous forest, major species are dispersed by wind, therefore not produce fleshy fruits to fauna consumption. However, typical riparian forests tend to had many zoochoryc dispersed species (Pinheiro & Ribeiro 2001; Rodrigues et al. 2010) and attract high animal diversity. Even though animal dispersed species increase their importance in the community (*Siphoneugena densiflora*, *Micropholis venulosa*, *Protium heptaphyllum*, *Duguetia lanceolata*, *Copaifera langsdorffii*, *Siparuna guianensis*, *Cheiloclinium cognatum*, *Cecropia pachystachya*, and others), this process should be very slow and this important function was virtually lost after original riparian forest destruction by dam construction.

This "Riparian Effect" happened in the three dry forest studied not only enhances richness and diversity but leads to high exchange in individuals, due high mortality and recruitment. The total number of individuals had few changes due balance in recruit and death trees; however, turnover to each species was high because a lot of species not only lost but recruits many trees.

Many tropical studies in no disturbed sites suggest few changes on species density and turnover (Chazdon et al. 2007; Lieberman et al. 1985; Swaine et al. 1987), suggesting stable environments with more recruitment and mortality from most dense species (Pinto & Hay 2005). However, an imbalance after damming occurred, some species presented high recruitment rates and low mortality, consequently a great positive net change, against species highly negatively affected, with mortality rates highter than recruitment. Even little soil moisture change may induce vegetation changes (Nilsson & Svedmark 2002) then water soil increase cause different responses on species. On one hand, water could kill roots by oxygen stress and consequently anoxia (Nilsson et al. 1997; Vartapetian & Jackson 1997; White 2007) and upland plants usually are intolerant to riparian environment (Johnson 1994; Nilsson & Berggren 2000), on the other hand can broke intense seasonal dry period enhance plant growth.

The scenario was an instable period with intense tree changes and several consequences to species few years after damming. Instable periods occur after strong perturbation and some conclusion were difficult to be made, but is clearly that most important species not response equally and those high negatively affected should not be used in management of areas with similar impacts. Many of these species with negative response to impoundment were inclusive used in "recuperation management" of many areas affected by damming (*Myracrodruon urundeuva*, *Platypodium elegans*, *Tabebuia roseoalba*, *Casearia rupestris* (Guimarães et al. 2009). Thus we concluded that these species are not good to be used in recuperation management if planted in areas with similar soil condition studied here (rocky terrain with high slopes), which means unnecessary money spent.

Increase in basal area and mortality rates

Despite of species changes, the most impressive was species increment (growth) and consequent gain in basal area. Even species with high mortality obtained positive basal area net change, mainly in the first two years after impoundment (most evident in the two deciduous forests, see Figure 4) due high trees increment. In secondary forests, biomass accumulation is more influenced by growth than density (Chazdon et al. 2007), soon are good predictors of possible species paths in these communities. In this way, survivor individuals that won the soil anoxia and not died presents high growth.

When analyzed the more important species, many of them show high ingrowths rates leading to thick woods on the communities (even many high mortality species presented high ingrowths rates, commonly higher than 5% year^{-1}), strongly influencing the forests basal area enlargement (Chapter 3). The great species basal area rise on deciduous forest occurred probably due broke the water period limitation (Chapter 3) and, because semidecidous species had a strong floristic similarity to moist Atlantic forest (Oliveira & Fontes 2000), thus trees growth should not be so limited in this milder environment. Many semideciduous species had not strong variations on basal area gain (see Figure 3), reinforcing the hypothesis that species on milder environments suffer less changes with raise in moisture changes than drought environments.

All high changes in individuals (high turnover due recruitment and mortality) and basal area gain leads to strongly modifications on tree communities. Species with few trees sampled could had high recruitment and/or mortality (case of *Siparuna guianensis* and *Cecropia pachystachya*). However, in few years these low important species not caused several changes on community thought their aggregate value could be significant over the years. Then, we focus on changes of most important species whose strongly affect entire community. Some of the most important species (with major cover value) reduced their importance due high mortality: *Myracrodruon urundeuva* and *Tabebuia roseoalba* in DF1, *Guazuma ulmifolia* and *M. urundeuva* in DF2 (all canopy species) and *Cordiera sessilis* and *Casearia grandiflora* (understory species) in SF.

Without disturbances, more abundant species tend to had more mortality (Felfili 1995), but more recruitment (Hubbell et al. 1990; Pinto & Hay 2005). However, these species recruitment rates were very low too, thereby the individuals negative net change occur due these two factors: high mortality and low recruitment. Otherwise, species with positive responses to damming in deciduous forests are canopy ones: *Anadenanthera colubrina* and *Piptadenia gonoacantha* in DF1 and *Inga sessilis*, *A. colubrina* and *Lonchocarpus cultratus* in DF2. Then, great structural changes on canopy in future should occur because dead and recruits are of distinct species from same forest layer. Great majority of these dead trees in deciduous forest were low thickness too (smaller than 20cm of diameter – Chapter 3) and the recruits had low height too, then these changes will be felt only in the future. However, less future changes tend to occur on semideciduous forest because the species with high density increase (*Aspidosperma discolor*, *Astronium nelson-rosae* and *Terminalia glabescens*) not present high density and are canopy species, different than those species with more mortality that were from understory. There was strongly rising of these canopy species too, than semideciduous forest will maintain similar to the original forest structure.

Times of records assessment

Not only was a range of answers as to forest type (species in semideciduous forest responded less intensely to the effects of the dam) as the responses were concentrated in the first two years after impoundment (see Figure 3). Most species suffer fewer changes in T2-T4 period in all three forests, demonstrating that damming impacts tend to stabilize few years after impoundment. Works with other taxa, like macrophytes and insects boom after damming (Fearnside 2005; Luz 1994; Moura Júnior et al. 2011; Patz et al. 2000), demonstrated that intense changes occur after dam construction, but tend to stabilize over years (Lima et al. 2002). Even abiotic changes, like carbon emission, were concentrated in first years after damming (Fearnside 2002).

Hence the analysis every two years was necessary and satisfactory in represent the scenario after damming, with high changes just after impact followed by a stabilization. The damming effect for many species would have been masked, if analyzed only T0-T4 period because rates in T2-T4 were three to most of ten times smaller to these species. More, we avoid error associated to tree hydration, when measurement occur in different seasons of the years (Phillips et al. 2004), because all measurements (T0, T2 and T4) occurred in the end of wet season (March-April), increasing the reliability of results. The interval of measurement interfere on dynamic rates too (Phillips et al. 2004), however the species rates differences between T0-T2 and T0-T4 were just too high too (frequently more than 5% year^{-1}) to infer that the two or four years cause so high interferences on rates. Finally, the rates could be influenced by species number of individuals and our criteria of analyzed just species with 20 or more individuals minimized this problem and many tendencies were general.

Conclusions

We could argue that a succession phase start just after damming, first due high mortality rates. This high mortality should be obviously accompanied by basal area lost, a “degradation phase” (Machado & Oliveira-Filho 2010). An intense mortality creates spaces to trees of other species stabilize (van Breugel et al. 2006), “replacement phase” occur. These two theoretical phases were concurrent, but the first one is totally independent from the second, however “replacement phase” depends from new open spaces created by mortality of trees of “degradation phase”. The differences of damming effects than other disturbance is the fact of recruitment be less dependent than mortality, because new conditions facilitates new trees establishment, independent of mortality. Hence mortality and recruitment presented same weight to community and a “degradation phase” did not really occur. What happened was a species change, because majority of dead tree were distinct than those recruited, then this “replacement phase” really occur. This replacement will favor moisture adapted species over the years.

A typical disturbance caused dead of big trees leading to basal area lost (Clark & Clark 1996), but basal area had incredible increase due soil moisture. In “building phases” a forest recovers from a disturbance (that cause basal area lost) with trees growths (Machado & Oliveira-Filho 2010). Damming disturbances studied in this seasonal forests act different. A rise in trees growths caused quick basal area gain not to rebuild the forest, but due new conditions, then we call this “growth phase”. This phase is distinct than “self-thinning” (Machado & Oliveira-Filho 2010) too by not lost density due fast recruitment.

We can infer that forest affected by dams presented a “replacement phase” and a “growth phase” with species change (prevail of water associated species) and basal area gain. The “growth phase” tends to stabilize first because many species rates in T2-T4 were lower than T0-T2. However, changes in more important species (with more mortality of few ones and more recruitment of others) and establishment of new species should be more complex and take years to stabilize. The original condition probably will had vestige of original condition due large trees had long longevity and persist in environments for a long period (Carey et al. 1994), but the forest will never return to their original state.

We conclude that forest is reorganize their structure, species establishment and total basal area supported and damming new condition will change these forests for several years, leading to a different kind of forest. These changes were most severe on deciduous forest too, because soil are dryer than semideciduous forest environments, than rise in soil moisture caused more changes. Continuation of monitoring of these areas is fundamental to understand dam effects on seasonal forests.

Appendix X - Tree species parameters and dynamic rates to a Deciduous Forest (Deciduous Forest 1 – DF1) in southearn Brazil. T0 = before dam construction, T2 = two years after damming, T4 = four years after damming, M = mortality, R = recruitment, O = outgrowth, I = ingrowth. Only species with less than 20 individuals are shown.

Deciduous 1	Number of individuals			Basal Area			Dinamics Rates T0-T2				Dinamics Rates T2-T4				Dinamics Rates T0-T4			
Species	T0	T2	T4	T0	T2	T4	M	R	O	I	M	R	O	I	M	R	O	I
Acrocomia aculeata	11	11	11	0.60	0.61	0.61	0.0	0.0	1.3	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.8
Aegiphyla sellowiana	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.6
Aloysia virgata	3	2	1	0.02	0.01	0.00	18.4	0.0	17.9	0.4	29.3	0.0	52.6	0.0	24.0	0.0	38.1	0.9
Allophyllus sericeus	5	6	4	0.04	0.05	0.05	0.0	8.7	0.4	12.1	18.4	0.0	4.9	3.2	12.0	6.9	3.5	8.5
Apeiba tibourbou	5	5	5	0.03	0.06	0.10	10.6	10.6	3.1	30.4	0.0	0.0	0.0	23.8	5.4	5.4	1.5	27.2
Aspidosperma parvifolium	2	2	2	0.01	0.01	0.01	0.0	0.0	0.0	5.1	0.0	0.0	1.9	0.0	0.0	0.0	1.1	2.7
Aspidosperma subincanum	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	14.3	-	100.0	-	100.0
Astrionium fraxinifolium	5	5	7	0.06	0.08	0.09	0.0	0.0	0.0	12.5	0.0	15.5	0.0	5.4	0.0	8.1	0.0	9.0
Casearia gossypiosperma	10	12	15	0.05	0.06	0.41	5.1	18.4	9.2	21.5	0.0	6.9	2.6	3.3	2.6	12.0	6.5	3.2
Casearia sylvestris	9	11	12	0.03	0.03	0.04	5.7	14.7	5.6	17.9	0.0	4.3	0.4	5.4	2.9	9.6	3.1	12.0
Cecropia pachystachya	6	5	9	0.11	0.08	0.12	18.4	10.6	28.9	18.3	0.0	25.5	0.0	17.8	9.6	18.4	15.7	18.0
Celtis iguanae	10	14	15	0.05	0.08	0.09	10.6	24.4	9.6	25.2	3.6	6.9	4.8	10.5	8.5	17.3	9.2	19.9
Chrysophyllum gonocarpum	1	1	1	0.01	0.01	0.01	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0
Chomelia pohliana	5	5	5	0.01	0.02	0.02	10.6	10.6	7.5	19.1	0.0	0.0	0.0	8.6	5.4	5.4	3.8	14.0
Chomelia ribesioides	5	6	4	0.02	0.02	0.02	0.0	8.7	8.6	8.3	18.4	0.0	24.9	5.1	12.0	6.9	20.3	10.3
Cordieria sessilis	1	2	2	0.00	0.01	0.01	0.0	29.3	0.0	40.8	0.0	0.0	0.0	2.8	0.0	15.9	0.0	24.1
Cordia sp.	7	6	6	0.07	0.06	0.06	7.4	0.0	8.0	1.4	0.0	0.0	3.3	0.3	3.8	0.0	5.7	0.9
Cupania vernalis	1	1	2	0.01	0.01	0.02	0.0	0.0	0.0	0.0	0.0	29.3	0.0	15.9	0.0	15.9	0.0	8.3
Dilodendron bipinatum	4	5	5	0.01	0.02	0.02	0.0	10.6	0.0	27.3	0.0	0.0	0.0	6.2	0.0	5.4	0.0	17.4
Enterolobium contortisiliquum	3	3	3	0.12	0.14	0.15	0.0	0.0	0.0	8.6	0.0	0.0	1.2	4.1	0.0	0.0	0.7	6.5
Eugenia florida	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	6.2	0.0	0.0	0.0	4.5	0.0	0.0	0.0	5.4
Guapira opposita	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	11.1	-	100.0	-	100.0

Guarea guidonia	0	1	4	0.00	0.00	0.01	-	100.0	-	100.0	0.0	50.0	0.0	59.8	-	100.0	-	100.0
Hymenaea courbaril	1	1	1	0.04	0.05	0.05	0.0	0.0	0.0	3.0	0.0	0.0	0.0	1.3	0.0	0.0	0.0	2.2
Inga vera	0	0	2	0.00	0.00	0.01	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Jacaranda caroba	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Lonchocarphus cultratus	4	2	4	0.06	0.02	0.02	-	0.0	43.6	14.4	29.3	50.0	30.2	24.9	29.3	29.3	44.4	28.9
Luehea grandiflora	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	8.6	-	100.0	-	100.0
Machaerium aculeatum	5	5	5	0.19	0.23	0.25	0.0	0.0	0.0	9.7	0.0	0.0	0.0	4.1	0.0	0.0	0.0	6.9
Machaerium hirtum	6	6	7	0.04	0.05	0.06	0.0	0.0	0.0	11.5	0.0	7.4	0.0	7.3	0.0	3.8	0.0	9.4
Maclura tinctoria	7	8	10	0.04	0.05	0.31	0.0	6.5	0.0	16.0	0.0	10.6	0.0	2.2	0.0	8.5	0.0	2.3
Margaritaria nobilis	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Matayba guianensis	9	10	16	0.05	0.06	0.09	18.4	22.5	9.9	16.3	0.0	20.9	0.2	16.4	9.6	21.7	5.2	16.4
Myrcia splendens	1	2	3	0.00	0.00	0.01	0.0	29.3	0.0	33.5	0.0	18.4	0.0	24.2	0.0	24.0	0.0	29.0
Myrsine guianensis	15	11	9	0.10	0.09	0.09	18.4	4.7	13.4	8.3	9.5	0.0	3.9	5.0	14.5	2.9	9.1	7.0
Myrsine umbelata	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Pouteria torta	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	10.0	0.0	0.0	0.0	18.4	0.0	0.0	0.0	14.3
Rhamnidium elaeocarpum	10	9	8	0.04	0.04	0.04	5.1	0.0	2.6	6.8	5.7	0.0	4.9	2.3	5.4	0.0	4.2	5.0
Siparuna guianensis	0	1	3	0.00	0.00	0.01	-	100.0	-	100.0	0.0	42.3	0.0	100.0	-	100.0	-	100.0
Sterculia striata	1	0	0	0.13	0.00	0.00	100.0	-	100.0	-	-	-	-	-	100.0	-	100.0	-
Handroanthus crysotrichus	2	1	1	0.01	0.00	0.00	29.3	0.0	25.7	9.1	0.0	0.0	4.5	0.0	15.9	0.0	16.2	5.1
Tapirira guianensis	1	3	4	0.00	0.01	0.01	0.0	42.3	0.0	35.9	0.0	13.4	0.0	20.3	0.0	29.3	0.0	28.5
Tocoyena formosa	1	0	2	0.00	0.00	0.01	100.0	-	100.0	-	-	100.0	-	100.0	100.0	100.0	100.0	100.0
Trema micrantha	0	1	2	0.00	0.00	0.01	-	100.0	-	100.0	0.0	29.3	0.0	48.6	-	100.0	-	100.0
Trichilia elegans	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Xylopia aromatica	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	17.4	-	100.0	-	100.0
Xylopia brasiliensis	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Zanthoxylum rhoifolium	3	2	2	0.01	0.01	0.01	18.4	0.0	26.1	11.3	0.0	0.0	3.5	3.8	9.6	0.0	16.0	8.1

Appendix XI - Tree species parameters and dynamic rates to a Deciduous Forest (Deciduous Forest 2 – DF2) in southearn Brazil. T0 = before dam construction, T2 = two years after damming, T4 = four years after damming, M = mortality, R = recruitment, O = outgrowth, I = ingrowth. Only species with less than 20 individuals are shown.

Deciduous 2	Number of individuals			Basal Area			Dinamics Rates T0-T2				Dinamics Rates T2-T4				Dinamics Rates T0-T4			
Species	T0	T2	T4	T0	T2	T4	M	R	O	I	M	R	O	I	M	R	O	I
Acacia polyphylla	12	8	9	0.21	0.19	0.21	18.4	0.0	15.2	9.1	6.5	11.8	4.0	10.4	12.6	6.1	10.2	10.1
Aegiphila sellowiana	1	0	0	0.01	0.00	0.00	100.0	-	100.0	-	-	-	-	-	100.0	-	100.0	-
Albizia niopoides	5	7	7	0.02	0.05	0.06	0.0	15.5	0.0	34.5	0.0	0.0	0.0	10.1	0.0	8.1	0.0	23.3
Apeiba tibourbou	5	5	5	0.16	0.20	0.21	0.0	0.0	0.0	11.1	0.0	0.0	1.1	3.3	0.0	0.0	0.7	7.4
Apuleia leiocarpa	1	1	1	0.00	0.01	0.02	0.0	0.0	0.0	39.0	0.0	0.0	0.0	12.8	0.0	0.0	0.0	27.1
Aspidosperma cuspa	1	1	1	0.01	0.02	0.02	0.0	0.0	0.0	20.5	0.0	0.0	0.0	7.2	0.0	0.0	0.0	14.1
Aspidosperma parviflorum	4	7	7	0.34	0.36	0.14	0.0	24.4	0.0	3.3	7.4	7.4	38.1	1.6	6.9	19.1	23.6	5.3
Astrionium fraxinifolium	15	17	19	0.18	0.25	0.29	0.0	6.1	0.0	14.5	0.0	5.4	0.0	7.0	0.0	5.7	0.0	10.8
Casearia sylvestris	9	10	10	0.10	0.12	0.12	5.7	10.6	2.6	10.7	0.0	0.0	1.6	4.3	2.9	5.4	2.3	7.7
Cecropia pachystachya	1	1	9	0.02	0.03	0.06	0.0	0.0	0.0	10.8	0.0	66.7	0.0	28.8	0.0	42.3	0.0	20.3
Cedrela fissilis	0	2	4	0.00	0.00	0.01	-	100.0	-	100.0	0.0	29.3	0.0	39.7	-	100.0	-	100.0
Ceiba speciosa	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Citrus sp.	1	1	1	0.01	0.01	0.01	0.0	0.0	0.4	0.0	0.0	0.0	5.5	0.0	0.0	0.0	3.0	0.0
Cordia alliodora	7	6	6	0.14	0.12	0.14	7.4	0.0	17.7	10.5	0.0	0.0	0.0	7.0	3.8	0.0	9.3	8.8
Cordia trichotoma	1	2	2	0.03	0.04	0.05	0.0	29.3	0.0	12.2	0.0	0.0	0.0	12.1	0.0	15.9	0.0	12.1
Cupania vernalis	1	3	4	0.00	0.01	0.02	0.0	42.3	0.0	43.7	0.0	13.4	0.0	21.7	0.0	29.3	0.0	33.6
Enterolobium contortisiliquum	15	14	13	0.29	0.42	0.48	6.9	3.6	1.4	17.9	3.6	0.0	0.2	6.3	5.4	2.0	0.9	12.4
Eriotheca candolleana	2	2	2	0.08	0.08	0.09	0.0	0.0	0.0	2.6	0.0	0.0	0.0	3.7	0.0	0.0	0.0	3.2
Erythroxylum deciduum	1	1	1	0.00	0.01	0.01	0.0	0.0	0.0	21.8	0.0	0.0	0.0	12.0	0.0	0.0	0.0	17.0
Eugenia florida	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	21.1	-	100.0	-	100.0
Ficus sp.	1	1	1	0.04	0.04	0.04	0.0	0.0	0.0	3.9	0.0	0.0	0.0	3.0	0.0	0.0	0.0	3.4
Genipa americana	0	1	1	0.00	0.00	0.01	-	100.0	-	100.0	0.0	0.0	0.0	14.3	-	100.0	-	100.0
Guarea guidonia	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	10.0	-	100.0	-	100.0
Hymenaea courbaril	1	1	0	0.02	0.04	0.00	0.0	0.0	0.0	23.9	100.0	-	100.0	-	100.0	-	-	-

Jacaranda caroba	2	2	2	0.01	0.01	0.01	0.0	0.0	2.2	5.8	0.0	0.0	0.0	4.3	0.0	0.0	1.1	5.1
Luehea divaricata	10	9	9	0.18	0.22	0.26	5.1	0.0	2.9	11.2	0.0	0.0	0.3	7.7	2.6	0.0	1.6	9.5
Luehea grandiflora	0	3	4	0.00	0.01	0.01	-	100.0	-	100.0	0.0	13.4	0.0	20.2	-	100.0	-	100.0
Machaerium hirtum	1	1	1	0.13	0.14	0.00	0.0	0.0	0.0	4.5	100.0	100.0	100.0	100.0	100.0	100.0	-	100.0
Machaerium stiptatum	12	18	18	0.08	0.11	0.11	0.0	18.4	4.7	17.5	0.0	0.0	2.3	2.7	0.0	9.6	4.0	10.9
Maclura tinctoria	6	6	6	0.05	0.06	0.07	0.0	0.0	0.0	9.8	0.0	0.0	0.0	3.1	0.0	0.0	0.0	6.5
Matayba guianensis	0	0	2	0.00	0.00	0.01	-	-	-		-	100.0	-	100.0	-	100.0	-	100.0
Metrodorea stipularis	1	1	1	0.01	0.01	0.01	0.0	0.0	0.0	20.3	0.0	0.0	0.0	7.5	0.0	0.0	0.0	14.1
Myrcia splendens	7	10	10	0.04	0.06	0.06	7.4	22.5	2.8	17.7	0.0	0.0	1.2	5.0	3.8	12.0	2.3	11.8
Nectandra cissiflora	0	2	2	0.00	0.01	0.01	-	100.0	-	100.0	0.0	0.0	0.0	19.3	-	100.0	-	100.0
Piptadenia gonoacantha	3	4	6	0.06	0.05	0.06	0.0	13.4	25.3	18.9	0.0	18.4	2.3	13.3	0.0	15.9	15.1	16.7
Pouteria gardnerii	4	4	7	0.03	0.04	0.04	0.0	0.0	0.3	6.8	0.0	24.4	0.7	9.9	0.0	13.1	0.6	8.4
Protium heptaphyllum	1	2	2	0.00	0.01	0.02	0.0	29.3	0.0	33.8	0.0	0.0	0.0	17.6	0.0	15.9	0.0	26.1
Psidium guajava	19	10	5	0.07	0.04	0.02	27.5	0.0	30.4	2.7	29.3	0.0	36.9	3.6	28.4	0.0	35.2	5.3
Psidium sartorianum	4	4	4	0.02	0.03	0.03	0.0	0.0	0.0	3.5	0.0	0.0	0.0	6.0	0.0	0.0	0.0	4.8
Sweetia fruticosa	1	2	2	0.00	0.01	0.01	0.0	29.3	0.0	46.9	0.0	0.0	0.0	9.9	0.0	15.9	0.0	30.8
Handroanthus crysotrichus	1	3	3	0.01	0.02	0.02	0.0	42.3	0.0	27.3	0.0	0.0	0.0	4.3	0.0	24.0	0.0	16.6
Handroanthus impetiginosus	3	4	4	0.03	0.03	0.03	0.0	13.4	0.0	11.1	0.0	0.0	0.0	2.3	0.0	6.9	0.0	6.8
Tabebuia roseoalba	2	2	2	0.00	0.01	0.01	29.3	29.3	22.3	45.9	0.0	0.0	0.0	11.5	15.9	15.9	11.9	30.8
Handroanthus serratifolius	9	10	11	0.03	0.05	0.04	0.0	5.1	0.4	14.5	0.0	4.7	5.7	3.9	0.0	4.9	4.2	10.4
Terminalia glabescens	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	29.2	-	100.0	-	100.0
Terminalia phaeocarpa	1	1	1	0.03	0.03	0.03	0.0	0.0	0.0	0.8	0.0	0.0	0.0	3.3	0.0	0.0	0.0	2.1
Trichilia catigua	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	18.4	-	100.0	-	100.0
Trichilia elegans	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	11.8	-	100.0	-	100.0
Trichilia pallida	0	2	3	0.00	0.00	0.01	-	100.0	-	100.0	0.0	18.4	0.0	34.5	-	100.0	-	100.0
Unonopsis lindmanii	2	3	4	0.01	0.01	0.02	0.0	18.4	0.0	31.4	0.0	13.4	0.0	15.2	0.0	15.9	0.0	23.8
Xylopia aromatica	2	4	5	0.01	0.03	0.04	0.0	29.3	0.0	40.8	0.0	10.6	0.0	13.3	0.0	20.5	0.0	28.4
Zanthoxyllum riedelianum	2	3	3	0.02	0.03	0.03	0.0	18.4	0.0	19.5	0.0	0.0	0.0	4.2	0.0	9.6	0.0	12.2
Zanthoxylum rhoifolium	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	0.0	-	100.0	-	100.0

Appendix XII - Tree species parameters and dynamic rates to a Semideciduou Forest (SF) in southearn Brazil. T0 = before dam construction, T2 = two years after damming, T4 = four years after damming, M = mortality, R = recruitment, O = outgrowth, I = ingrowth. Only species with less than 20 individuals are shown.

Semideciduous	Number of individuals			Basal Area			Dinamics Rates T0-T2				Dinamics Rates T2-T4				Dinamics Rates T0-T4			
Species	T0	T2	T4	T0	T2	T4	M	R	O	I	M	R	O	I	M	R	O	I
Acacia polyphylla	7	8	11	0.07	0.10	0.11	15.5	20.9	7.5	19.6	0.0	14.7	0.0	7.0	8.1	17.9	3.8	13.5
Agonandra brasiliensis	3	3	3	0.06	0.05	0.06	0.0	0.0	15.1	6.7	0.0	0.0	0.0	5.9	0.0	0.0	7.9	6.3
Albizia niopoides	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	10.5	-	100.0	-	100.0
Aspidosperma cylindrocarpum	4	4	3	0.03	0.03	0.03	0.0	0.0	1.1	1.7	13.4	0.0	3.2	1.7	6.9	0.0	2.2	1.7
Astrionium fraxinifolium	4	4	4	0.01	0.02	0.02	0.0	0.0	0.0	17.5	0.0	0.0	0.0	8.8	0.0	0.0	0.0	13.3
Bauhinia rufa	1	0	0	0.01	0.00	0.00	100.0	-	100.0	-	-	-	-	-	100.0	-	100.0	-
Bauhinia ungulata	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	6.5	0.0	0.0	0.0	8.0	0.0	0.0	0.0	7.3
Byrsonima laxiflora	1	0	0	0.01	0.00	0.00	100.0	-	100.0	-	-	-	-	-	100.0	-	100.0	-
Campomanesia vellutina	11	11	15	0.04	0.05	0.06	0.0	0.0	0.0	5.9	0.0	14.4	0.0	11.4	0.0	7.5	0.0	8.7
Cardiopetallum calophyllum	4	10	12	0.01	0.03	0.12	0.0	36.8	0.0	34.1	0.0	8.7	0.0	46.7	0.0	24.0	0.0	40.7
Cariniana estrellensis	2	2	2	0.01	0.01	0.02	0.0	0.0	0.0	2.7	0.0	0.0	0.0	5.4	0.0	0.0	0.0	4.1
Casearia sylvestris	3	2	2	0.01	0.01	1.20	18.4	0.0	9.4	6.6	0.0	0.0	-	5.0	9.6	0.0	100.0	2.6
Cecropia pachystachya	0	0	4	0.00	0.00	0.01	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Cedrela fissilis	2	2	2	0.03	0.06	0.07	0.0	0.0	0.0	24.1	0.0	0.0	0.0	6.3	0.0	0.0	0.0	15.6
Ceiba speciosa	1	1	1	0.02	0.03	0.04	0.0	0.0	0.0	5.1	0.0	0.0	0.0	14.5	0.0	0.0	0.0	9.9
Celtis iguanae	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2
Coccoloba mollis	3	2	2	0.01	0.01	0.01	18.4	0.0	25.7	0.0	0.0	0.0	2.9	2.0	9.6	0.0	15.1	1.0
Coussarea hydrangeaefolia	13	14	16	0.08	0.08	0.09	16.8	19.8	18.3	19.5	0.0	6.5	1.8	7.1	8.8	13.4	10.9	14.0
Coutarea hexandra	4	5	4	0.02	0.03	0.02	0.0	10.6	0.7	12.0	10.6	0.0	6.2	3.0	6.9	6.9	4.5	8.5
Cupania vernalis	3	3	6	0.07	0.07	0.08	0.0	0.0	0.0	1.8	0.0	29.3	0.8	7.1	0.0	15.9	0.4	4.5
Dilodendron bipinatum	1	0	0	0.00	0.00	0.00	100.0	-	100.0	-	-	-	-	-	100.0	-	100.0	-
Dipterix alata	4	4	4	0.11	0.12	0.14	0.0	0.0	0.0	5.9	0.0	0.0	0.0	7.2	0.0	0.0	0.0	6.6
Erioteca condolleana	5	5	5	0.05	0.06	0.06	0.0	0.0	0.3	2.3	0.0	0.0	0.0	2.2	0.0	0.0	0.1	2.2
Erythroxylum dahpnites	8	7	7	0.04	0.03	0.04	13.4	7.4	23.9	14.2	7.4	7.4	5.8	13.1	11.1	8.1	16.4	14.7

Eugenia florida	2	5	5	0.01	0.01	0.01	29.3	55.3	30.3	48.8	0.0	0.0	0.0	5.3	15.9	33.1	16.5	30.3
Eugenia ligustrina	1	1	7	0.01	0.01	0.02	0.0	0.0	0.0	3.5	0.0	62.2	0.0	33.8	0.0	38.5	0.0	20.1
Faramea cyanea	1	1	1	0.01	0.01	0.01	0.0	0.0	0.0	9.7	0.0	0.0	0.0	3.1	0.0	0.0	0.0	6.5
Ficus cf. enormis	1	1	1	0.03	0.00	0.00	100.0	100.0	100.0	100.0	0.0	0.0	0.0	8.1	100.0	100.0	100.0	100.0
Garcinia gardneriana	1	1	1	0.00	0.00	0.00	0.0	0.0	10.5	0.0	0.0	0.0	0.0	5.6	0.0	0.0	5.4	2.8
Guapira opposita	1	1	1	0.01	0.01	0.01	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.8	0.0
Guazuma ulmifolia	1	1	1	0.02	0.02	0.02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Guettarda viburnoides	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.5
Heteropterix birsonimifolia	0	5	4	0.00	0.05	0.02	-	100.0	-	100.0	10.6	0.0	38.9	3.7	-	100.0	-	100.0
Hirtella gracilipes	0	0	1	0.00	0.00	0.00	-	-	-	-	-	100.0	-	100.0	-	100.0	-	100.0
Hymenaea courbaril	7	7	7	0.86	0.94	1.03	0.0	0.0	0.3	4.5	0.0	0.0	0.0	4.4	0.0	0.0	0.1	4.5
Inga laurina	3	3	4	0.04	0.04	0.05	0.0	0.0	0.0	4.4	0.0	13.4	0.0	6.7	0.0	6.9	0.0	5.5
Ixora brevifolia	10	9	9	0.19	0.18	0.21	5.1	0.0	8.1	6.0	0.0	0.0	0.0	6.5	2.6	0.0	4.1	6.2
Jacaranda caroba	1	1	1	0.01	0.02	0.01	0.0	0.0	0.0	38.5	0.0	0.0	37.5	0.0	0.0	0.0	100.0	100.0
Luehea divaricata	7	7	6	0.03	0.04	0.03	0.0	0.0	0.4	14.0	7.4	0.0	5.0	2.8	3.8	0.0	3.7	9.5
Machaerium acutifolium	1	1	1	0.01	0.01	0.01	0.0	0.0	0.0	3.7	0.0	0.0	0.0	3.6	0.0	0.0	0.0	3.6
Machaerium brasiliensis	1	1	1	0.00	0.00	0.00	100.0	100.0	100.0	100.0	0.0	0.0	0.0	5.9	100.0	100.0	100.0	100.0
Machaerium hirtum	0	2	2	0.00	0.01	0.01	-	100.0	-	100.0	0.0	0.0	0.0	5.7	-	100.0	-	100.0
Machaerium nyctitans	1	1	1	0.00	0.01	0.01	0.0	0.0	0.0	31.5	0.0	0.0	0.0	5.3	0.0	0.0	0.0	19.4
Machaerium stiptatum	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	10.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.6
Machaerium villosum	12	9	7	0.58	0.39	0.38	18.4	5.7	21.4	3.9	11.8	0.0	5.5	3.9	15.9	3.8	14.0	4.2
Maclura tinctoria	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	11.1	0.0	0.0	0.0	10.0	0.0	0.0	0.0	10.6
Maprounea guianensis	1	1	1	0.00	0.00	0.01	0.0	0.0	0.0	40.0	0.0	0.0	0.0	7.4	0.0	0.0	0.0	25.5
Margaritaria nobilis	1	2	2	0.00	0.01	0.01	0.0	29.3	0.0	23.3	0.0	0.0	0.0	0.0	0.0	15.9	0.0	12.4
Matayba guianensis	4	3	3	0.01	0.01	0.02	13.4	0.0	8.2	14.4	0.0	0.0	0.0	11.3	6.9	0.0	4.2	12.8
Maytenus floribunda	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	6.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.2
Metrodorea stipularis	9	8	8	0.51	0.49	0.50	5.7	0.0	6.4	3.7	0.0	0.0	0.1	1.7	2.9	0.0	3.3	2.7
Myracrodruon urundeuva	3	3	3	0.02	0.02	0.02	0.0	0.0	0.0	8.8	0.0	0.0	0.0	6.7	0.0	0.0	0.0	7.7
Myrcia splendens	7	7	6	0.02	0.03	0.02	0.0	0.0	0.4	3.4	7.4	0.0	12.4	5.2	3.8	0.0	7.1	4.8
Myrcia tomentosa	6	6	7	0.04	0.05	0.06	0.0	0.0	0.0	7.9	0.0	7.4	0.0	5.4	0.0	3.8	0.0	6.7

Ocotea corymbosa	5	5	5	0.12	0.14	0.15	0.0	0.0	0.0	7.4	0.0	0.0	0.0	2.9	0.0	0.0	0.0	5.1
Piptadenia gonoacantha	1	1	10	0.00	0.00	0.04	0.0	0.0	0.0	18.0	0.0	68.4	0.0	63.2	0.0	43.8	0.0	45.1
Platipodium elegans	7	7	7	0.17	0.19	0.22	0.0	0.0	0.0	6.5	0.0	0.0	0.1	5.9	0.0	0.0	0.1	6.2
Pouteria gardnerii	0	4	5	0.00	0.01	0.01	-	100.0	-	100.0	0.0	10.6	0.7	13.1	-	100.0	-	100.0
Pouteria torta	2	2	3	0.01	0.01	0.02	0.0	0.0	0.0	4.9	0.0	18.4	0.0	19.5	0.0	9.6	0.0	12.5
Psidium rufum	0	1	1	0.00	0.00	0.00	-	100.0	-	100.0	0.0	0.0	0.0	5.6	-	100.0	-	100.0
Qualea dichotoma	2	2	2	0.08	0.08	0.08	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Qualea jundiahy	9	7	9	0.47	0.22	0.26	18.4	7.4	42.2	17.0	0.0	11.8	0.0	7.9	9.6	9.6	24.0	12.6
Rhamnidium elaeocarpum	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	7.7	0.0	0.0	0.0	2.5	0.0	0.0	0.0	5.1
Rudgea viburnoides	3	3	3	0.03	0.02	0.02	0.0	0.0	11.1	2.2	0.0	0.0	0.0	1.9	0.0	0.0	5.7	2.0
Schefflera morototonii	6	9	11	0.20	0.22	0.26	0.0	18.4	0.0	6.2	0.0	9.5	0.0	7.6	0.0	14.1	0.0	6.9
Simira sampaioana	4	1	1	0.02	0.01	0.01	50.0	0.0	48.4	30.3	0.0	0.0	0.0	5.7	29.3	0.0	28.2	18.9
Sorocea bonplandii	1	1	1	0.00	0.00	0.00	0.0	0.0	0.0	29.8	0.0	0.0	0.0	2.1	0.0	0.0	0.0	17.1
Sweetia fruticosa	12	15	15	0.20	0.22	0.23	0.0	10.6	0.0	4.9	0.0	0.0	0.5	1.7	0.0	5.4	0.3	3.3
Handroanthus impetiginosus	1	1	2	0.09	0.11	0.11	0.0	0.0	0.0	6.0	0.0	29.3	0.0	0.6	0.0	15.9	0.0	3.4
Handroanthus serratifolius	7	6	7	0.05	0.05	0.05	7.4	0.0	9.7	7.3	0.0	7.4	0.0	6.7	3.8	3.8	5.0	7.0
Terminalia phaeocarpa	0	2	3	0.00	0.00	0.01	-	100.0	-	100.0	0.0	18.4	0.0	26.1	-	100.0	-	100.0
Trichilia catigua	3	3	3	0.02	0.02	0.02	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6
Trichilia pallida	4	5	4	0.01	0.02	0.01	0.0	10.6	0.0	9.3	10.6	0.0	10.3	3.5	6.9	6.9	6.6	7.7
Vatairea macrocarpa	1	1	1	0.06	0.07	0.08	0.0	0.0	0.0	8.0	0.0	0.0	0.0	8.3	0.0	0.0	0.0	8.1
Virola sebifera	5	5	6	0.05	0.05	0.06	0.0	0.0	0.0	7.1	0.0	8.7	0.0	2.4	0.0	4.5	0.0	4.8
Xylopia aromatica	1	2	2	0.00	0.00	0.01	0.0	29.3	0.0	27.7	0.0	0.0	0.0	37.5	0.0	15.9	0.0	32.8

Appendix XIII - Tree species turnover and net change to a Deciduous Forest (Deciduous Forest 1 - DF1) by descending order of overall net change in T0-T4 period, in southeastern Brazil. Ind = individuals, BA = Basal area. Only species with less than 20 individuals are shown.

Deciduous Forest 1	Individuals Turnover			Basal Area Turnover			Individuals Net Change			Basal Area Net Change			Overall Net Change		
Species	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4
Acrocomia aculeata	0.0	0.0	0.0	1.4	0.0	0.7	0.0	0.0	0.0	0.3	0.0	0.1	0.1	0.0	0.1
Aegiphyla sellowiana	0.0	0.0	0.0	10.0	0.0	5.3	0.0	0.0	0.0	25.0	0.0	11.8	12.5	0.0	5.9
Aloysia virgata	9.2	14.6	12.0	9.1	26.3	19.5	-18.4	-29.3	-24.0	-17.5	-52.6	-37.5	-18.0	-41.0	-30.8
Allophyllus sericeus	4.4	9.2	9.5	6.2	4.1	6.0	9.5	-18.4	-5.4	13.3	-1.8	5.5	11.4	-10.1	0.0
Apeiba tibourbou	10.6	0.0	5.4	16.7	11.9	14.4	0.0	0.0	0.0	39.3	31.2	35.2	19.6	15.6	17.6
Aspidosperma olivaceum	0.0	0.0	0.0	2.5	0.9	1.9	0.0	0.0	0.0	5.4	-1.9	1.7	2.7	-0.9	0.8
Aspidosperma subincanum	50.0	0.0	50.0	50.0	7.1	50.0	-	0.0	-	-	16.7	-	-	8.3	-
Astrionium fraxinifolium	0.0	7.7	4.0	6.3	2.7	4.5	0.0	18.3	8.8	14.3	5.7	9.9	7.2	12.0	9.4
Casearia gossypiosperma	11.7	3.5	7.3	15.4	31.0	24.6	9.5	11.8	10.7	6.6	161.1	66.9	8.1	86.4	38.8
Casearia sylvestris	10.2	2.1	6.3	11.8	2.9	7.5	10.6	4.4	7.5	15.0	5.3	10.0	12.8	4.9	8.7
Cecropia pachystachia	14.5	12.7	14.0	23.6	8.9	16.8	-8.7	34.2	10.7	-13.0	21.7	2.9	-10.8	27.9	6.8
Celtis iguanae	17.5	5.3	12.9	17.4	7.6	14.5	18.3	3.5	10.7	20.8	6.5	13.4	19.6	5.0	12.0
Chrysophyllum gonocarpum	0.0	0.0	0.0	0.9	0.0	0.4	0.0	0.0	0.0	-1.8	0.0	-0.9	-0.9	0.0	-0.4
Chomelia pohliana	10.6	0.0	5.4	13.3	4.3	8.9	0.0	0.0	0.0	14.3	9.5	11.9	7.2	4.7	5.9
Chomelia ribesioides	4.4	9.2	9.5	8.4	15.0	15.3	9.5	-18.4	-5.4	-0.3	-20.8	-11.2	4.6	-19.6	-8.3
Chomelia sessilis	14.6	0.0	8.0	20.4	1.4	12.1	41.4	0.0	18.9	68.9	2.9	31.8	55.1	1.5	25.4
Cordia sp.	3.7	0.0	1.9	4.7	1.8	3.3	-7.4	0.0	-3.8	-6.8	-3.0	-4.9	-7.1	-1.5	-4.3
Cupania vernalis	0.0	14.6	8.0	0.0	8.0	4.2	0.0	41.4	18.9	0.0	18.9	9.1	0.0	30.2	14.0
Dilodendron bipinatum	5.3	0.0	2.7	13.6	3.1	8.7	11.8	0.0	5.7	37.5	6.6	21.1	24.7	3.3	13.4
Enterolobium contortisiliquum	0.0	0.0	0.0	4.3	2.6	3.6	0.0	0.0	0.0	9.4	3.1	6.2	4.7	1.5	3.1
Eugenia florida	0.0	0.0	0.0	3.1	2.3	2.7	0.0	0.0	0.0	6.6	4.8	5.7	3.3	2.4	2.8
Guapira areolata	50.0	0.0	50.0	50.0	5.6	50.0	-	0.0	-	-	12.5	-	-	6.3	-
Guarea guidonia	50.0	25.0	50.0	50.0	29.9	50.0	-	100.0	-	-	148.9	-	-	124.4	-
Hymenaea courbaril	0.0	0.0	0.0	1.5	0.6	1.1	0.0	0.0	0.0	3.1	1.3	2.2	1.6	0.7	1.1
Inga vera	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Jacaranda caroba	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Lonchocarphus cultratus	14.6	39.6	29.3	29.0	27.5	36.7	-29.3	41.4	0.0	-34.2	-7.1	-21.8	-31.7	17.2	-10.9
Luehea grandiflora	50.0	0.0	50.0	50.0	4.3	50.0	-	0.0	-	-	9.4	-	-	4.7	-
Machaerium aculeatum	0.0	0.0	0.0	4.8	2.1	3.5	0.0	0.0	0.0	10.7	4.3	7.4	5.3	2.1	3.7
Machaerium brasiliensis	0.0	3.7	1.9	5.8	3.7	4.7	0.0	8.0	3.9	13.0	7.9	10.4	6.5	8.0	7.2

Maclura tinctoria	3.2	5.3	4.3	8.0	1.1	1.2	6.9	11.8	9.3	19.0	151.5	73.0	13.0	81.7	41.2
Margaritaria nobilis	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Matayba guianensis	20.4	10.5	15.7	13.1	8.3	10.8	5.4	26.5	15.5	7.7	19.4	13.4	6.5	22.9	14.4
Myrcia splendens	14.6	9.2	12.0	16.7	12.1	14.5	41.4	22.5	31.6	50.4	31.9	40.8	45.9	27.2	36.2
Myrsine guianensis	11.5	4.8	8.7	10.8	4.5	8.1	-14.4	-9.5	-12.0	-5.6	1.2	-2.3	-10.0	-4.2	-7.1
Myrsine umbelata	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Pouteria torta	0.0	0.0	0.0	5.0	9.2	7.1	0.0	0.0	0.0	11.1	22.5	16.7	5.6	11.3	8.3
Rhamnidium elaeocarpum	2.6	2.9	2.7	4.7	3.6	4.6	-5.1	-5.7	-5.4	4.5	-2.7	0.8	-0.3	-4.2	-2.3
Siparuna guianensis	50.0	21.1	50.0	50.0	50.0	50.0	-	73.2	-	-	94.2	-	-	83.7	-
Sterculia striata	50.0	0.0	50.0	50.0	0.0	50.0	-100.0	-	-100.0	-100.0	-	-100.0	-100.0	-	-100.0
Handroanthus crysotrichus	14.6	0.0	8.0	17.4	2.3	10.7	-29.3	0.0	-15.9	-18.2	-4.5	-11.7	-23.8	-2.3	-13.8
Tapirira guianensis	21.1	6.7	14.6	17.9	10.2	14.3	73.2	15.5	41.4	55.9	25.5	39.9	64.6	20.5	40.7
Tocoyena formosa	50.0	50.0	100.0	50.0	50.0	100.0	-100.0	-	18.9	-100.0	-	39.3	-100.0	-	29.1
Trema micrantha	50.0	14.6	50.0	50.0	24.3	50.0	-	41.4	-	-	94.5	-	-	67.9	-
Trichilia elegans	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Xylopia aromatica	50.0	0.0	50.0	50.0	8.7	50.0	-	0.0	-	-	21.1	-	-	10.5	-
Xylopia brasiliensis	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Zanthoxylum rhoifolium	9.2	0.0	4.8	18.7	3.6	12.0	-18.4	0.0	-9.6	-16.6	0.3	-8.6	-17.5	0.1	-9.1

Appendix XIV - Tree species turnover and net change to a Deciduous Forest (Deciduous Forest 2 – DF2) by descending order of overall net change in T0-T4 period, in southeastern Brazil. Ind = individuals, BA = Basal area. Only species with less than 20 individuals are shown.

Deciduous Forest 2	Individuals Turnover			Basal Area Turnover			Individuals Net Change			Basal Area Net Change			Overall Net Change		
Species	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4
Acacia polyphylla	9.2	9.1	9.3	12.2	7.2	10.2	-18.4	6.1	-6.9	-6.8	7.0	-0.1	-12.6	6.6	-3.5
Aegiphila sellowiana	50.0	0.0	50.0	50.0	0.0	50.0	-100.0	-	-100.0	-100.0	-	-100.0	-100.0	-	-100.0
Albizia niopoides	7.7	0.0	4.0	17.2	5.1	11.6	18.3	0.0	8.8	52.6	11.3	30.3	35.5	5.6	19.5
Apeiba tibourbou	0.0	0.0	0.0	5.5	2.2	4.1	0.0	0.0	0.0	12.5	2.2	7.2	6.2	1.1	3.6
Apuleia leiocarpa	0.0	0.0	0.0	19.5	6.4	13.5	0.0	0.0	0.0	64.0	14.6	37.1	32.0	7.3	18.6
Aspidosperma cuspa	0.0	0.0	0.0	10.2	3.6	7.0	0.0	0.0	0.0	25.7	7.7	16.4	12.9	3.9	8.2
Aspidosperma subincanum	12.2	7.4	13.0	1.6	19.8	14.4	32.3	0.0	15.0	3.4	-37.1	-19.3	17.8	-18.5	-2.2
Astrionium fraxinifolium	3.0	2.7	2.9	7.2	3.5	5.4	6.5	5.7	6.1	16.9	7.5	12.1	11.7	6.6	9.1
Casearia sylvestris	8.1	0.0	4.2	6.6	2.9	5.0	5.4	0.0	2.7	9.0	2.8	5.8	7.2	1.4	4.3
Cecropia pachystachia	0.0	33.3	21.1	5.4	14.4	10.2	0.0	200.0	73.2	12.1	40.5	25.5	6.1	120.3	49.4
Cedrela fissilis	-	14.6	-	50.0	19.8	50.0	-	41.4	-	-	65.8	-	-	53.6	-
Ceiba speciosa	-	-	-	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Citrus sp.	0.0	0.0	0.0	0.2	2.7	1.5	0.0	0.0	0.0	-0.4	-5.5	-3.0	-0.2	-2.7	-1.5
Cordia alliodora	3.7	0.0	1.9	14.1	3.5	9.0	-7.4	0.0	-3.8	-8.0	7.5	-0.5	-7.7	3.8	-2.2
Cordia trichotoma	14.6	0.0	8.0	6.1	6.0	6.1	41.4	0.0	18.9	13.9	13.7	13.8	27.6	6.9	16.4
Cupania vernalis	21.1	6.7	14.6	21.8	10.9	16.8	73.2	15.5	41.4	77.5	27.8	50.6	75.3	21.6	46.0
Enterolobium contortisiliquum	5.3	1.8	3.7	9.7	3.3	6.6	-3.4	-3.6	-3.5	20.1	6.5	13.1	8.4	1.4	4.8
Eriotheca candolleana	0.0	0.0	0.0	1.3	1.8	1.6	0.0	0.0	0.0	2.7	3.8	3.3	1.4	1.9	1.6
Erythroxylum deciduum	0.0	0.0	0.0	10.9	6.0	8.5	0.0	0.0	0.0	27.9	13.6	20.5	13.9	6.8	10.3
Eugenia florida	-	0.0	-	50.0	10.5	50.0	-	0.0	-	-	26.7	-	-	13.3	-
Ficus sp.	0.0	0.0	0.0	2.0	1.5	1.7	0.0	0.0	0.0	4.1	3.0	3.6	2.1	1.5	1.8
Genipa americana	-	0.0	-	50.0	7.1	50.0	-	0.0	-	-	16.7	-	-	8.3	-
Guarea guidonia	-	0.0	-	50.0	5.0	50.0	-	0.0	-	-	11.1	-	-	5.6	-
Hymenaea courbaril	0.0	-	-	11.9	50.0	0.0	0.0	-100.0	-100.0	31.4	-100.0	-100.0	15.7	-100.0	-100.0
Jacaranda caroba	0.0	0.0	0.0	4.0	2.2	3.1	0.0	0.0	0.0	3.8	4.5	4.2	1.9	2.3	2.1
Luehea divaricata	2.6	0.0	1.3	7.0	4.0	5.6	-5.1	0.0	-2.6	9.4	8.0	8.7	2.1	4.0	3.1
Luehea grandiflora	-	6.7	-	50.0	10.1	50.0	-	15.5	-	-	25.3	-	-	20.4	-
Machaerium aculeatum	0.0	100.0	100.0	2.3	100.0	50.0	0.0	0.0	0.0	4.7	-88.0	-64.5	2.4	-44.0	-32.3
Machaerium stiptatum	9.2	0.0	4.8	11.1	2.5	7.5	22.5	0.0	10.7	15.4	0.5	7.7	19.0	0.2	9.2
Maclura tinctoria	0.0	0.0	0.0	4.9	1.5	3.2	0.0	0.0	0.0	10.8	3.2	6.9	5.4	1.6	3.5

Matayba guianensis	-	-	-	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Metrodorea pubecens	0.0	0.0	0.0	10.1	3.8	7.1	0.0	0.0	0.0	25.4	8.1	16.4	12.7	4.1	8.2
Myrcia splendens	15.0	0.0	7.9	10.3	3.1	7.1	19.5	0.0	9.3	18.1	4.0	10.8	18.8	2.0	10.1
Nectandra cissiflora	-	0.0	-	50.0	9.6	50.0	-	0.0	-	-	23.8	-	-	11.9	-
Piptadenia gonacantha	6.7	9.2	8.0	22.1	7.8	15.9	15.5	22.5	18.9	-7.9	12.7	1.9	3.8	17.6	10.4
Pouteria gardnerii	0.0	12.2	6.5	3.5	5.3	4.5	0.0	32.3	15.0	6.9	10.2	8.5	3.5	21.2	11.8
Protium heptaphyllum	14.6	0.0	8.0	16.9	8.8	13.1	41.4	0.0	18.9	50.9	21.4	35.3	46.2	10.7	27.1
Psidium guajava	13.7	14.6	14.2	16.5	20.3	20.2	-27.5	-29.3	-28.4	-28.4	-34.5	-31.6	-27.9	-31.9	-30.0
Psidium sartorianum	0.0	0.0	0.0	1.8	3.0	2.4	0.0	0.0	0.0	3.6	6.4	5.0	1.8	3.2	2.5
Sweetia fruticosa	14.6	0.0	8.0	23.5	4.9	15.4	41.4	0.0	18.9	88.3	11.0	44.6	64.9	5.5	31.7
Handroanthus crysotrichus	21.1	0.0	12.0	13.7	2.2	8.3	73.2	0.0	31.6	37.6	4.5	19.9	55.4	2.3	25.8
Handroanthus impetiginosus	6.7	0.0	3.5	5.6	1.2	3.4	15.5	0.0	7.5	12.5	2.4	7.3	14.0	1.2	7.4
Tabebuia roseoalba	29.3	0.0	15.9	34.1	5.8	21.3	0.0	0.0	0.0	43.6	13.0	27.4	21.8	6.5	13.7
Handroanthus serratifolius	2.6	2.3	2.4	7.5	4.8	7.3	5.4	4.9	5.1	16.6	-1.9	6.9	11.0	1.5	6.0
Terminalia glabescens	-	0.0	-	50.0	14.6	50.0	-	0.0	-	-	41.2	-	-	20.6	-
Terminalia phaeocarpa	0.0	0.0	0.0	0.4	1.6	1.0	0.0	0.0	0.0	0.9	3.4	2.1	0.4	1.7	1.1
Trichilia catigua	-	0.0	-	50.0	9.2	50.0	-	0.0	-	-	22.5	-	-	11.3	-
Trichilia elegans	-	0.0	-	50.0	5.9	50.0	-	0.0	-	-	13.3	-	-	6.7	-
Trichilia pallida	-	9.2	-	50.0	17.3	50.0	-	22.5	-	-	52.7	-	-	37.6	-
Unonopsis lindmanii	9.2	6.7	8.0	15.7	7.6	11.9	22.5	15.5	18.9	45.9	17.9	31.2	34.2	16.7	25.0
Xylopia aromatica	14.6	5.3	10.2	20.4	6.6	14.2	41.4	11.8	25.7	69.0	15.3	39.6	55.2	13.6	32.7
Zanthoxyllum riedelianum	9.2	0.0	4.8	9.8	2.1	6.1	22.5	0.0	10.7	24.2	4.3	13.8	23.4	2.2	12.3
Zanthoxylum rhoifolium	-	0.0	-	50.0	0.0	50.0	-	0.0	-	-	0.0	-	-	0.0	-

Appendix XV - Tree species turnover and net change to a Semideciduous Forest (SF) by descending order of overall net change in T0-T4 period, in southeastern Brazil. Ind = individuals, BA = Basal area. Only species with less than 20 individuals are shown.

Semideciduous Forest	Individuals Turnover			Basal Area Turnover			Individuals Net Change			Basal Area Net Change			Overall Net Change		
Species	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4	T0-T2	T2-T4	T0-T4
Acacia polyphylla	18.2	7.4	13.0	13.6	3.5	8.7	6.9	17.3	12.0	15.0	7.6	11.2	10.9	12.4	11.6
Agonandra brasiliensis	0.0	0.0	0.0	10.9	3.0	7.1	0.0	0.0	0.0	-9.0	6.3	-1.7	-4.5	3.2	-0.8
Albizia niopoides	50.0	0.0	50.0	50.0	5.3	50.0	-	0.0	-	-	11.8	-	-	5.9	-
Aspidosperma cylindrocarpum	0.0	6.7	3.5	1.4	2.4	2.0	0.0	-13.4	-6.9	0.6	-1.6	-0.5	0.3	-7.5	-3.7
Astrionium fraxinifolium	0.0	0.0	0.0	8.7	4.4	6.6	0.0	0.0	0.0	21.2	9.7	15.3	10.6	4.8	7.6
Bauhinia rufa	50.0	0.0	50.0	50.0	0.0	50.0	-100.0	-	-100.0	-100.0	-	-100.0	-100.0	-	-100.0
Bauhinia ungulata	0.0	0.0	0.0	3.3	4.0	3.6	0.0	0.0	0.0	7.0	8.7	7.8	3.5	4.3	3.9
Byrsonima laxiflora	50.0	0.0	50.0	50.0	0.0	50.0	-100.0	-	-100.0	-100.0	-	-100.0	-100.0	-	-100.0
Campomanesia vellutina	0.0	7.2	3.7	3.0	5.7	4.4	0.0	16.8	8.1	6.3	12.9	9.6	3.2	14.8	8.8
Cardiopetallum calophyllum	18.4	4.4	12.0	17.0	23.4	20.4	58.1	9.5	31.6	51.7	87.8	68.7	54.9	48.7	50.2
Cariniana estrellensis	0.0	0.0	0.0	1.3	2.7	2.0	0.0	0.0	0.0	2.7	5.7	4.2	1.4	2.9	2.1
Casearia sylvestris	9.2	0.0	4.8	8.0	2.5	51.3	-18.4	0.0	-9.6	-2.9	955.4	220.1	-10.6	477.7	105.2
Cecropia pachystachia	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Cedrela fissilis	0.0	0.0	0.0	12.0	3.1	7.8	0.0	0.0	0.0	31.7	6.7	18.5	15.8	3.3	9.3
Ceiba speciosa	0.0	0.0	0.0	2.5	7.2	5.0	0.0	0.0	0.0	5.4	16.9	11.0	2.7	8.5	5.5
Celtis iguanae	0.0	0.0	0.0	1.2	0.0	0.6	0.0	0.0	0.0	2.4	0.0	1.2	1.2	0.0	0.6
Coccoloba mollis	9.2	0.0	4.8	12.9	2.5	8.0	-18.4	0.0	-9.6	-25.7	-0.8	-14.2	-22.0	-0.4	-11.9
Coussarea hydrangeaefolia	18.3	3.2	11.1	18.9	4.5	12.5	3.8	6.9	5.3	1.5	5.7	3.6	2.6	6.3	4.4
Coutarea hexandra	5.3	5.3	6.9	6.3	4.6	6.5	11.8	-10.6	0.0	12.8	-3.3	4.5	12.3	-6.9	2.2
Cupania vernalis	0.0	14.6	8.0	0.9	3.9	2.4	0.0	41.4	18.9	1.8	6.8	4.3	0.9	24.1	11.6
Dilodendron bipinatum	50.0	0.0	50.0	50.0	0.0	50.0	-100.0	-	-100.0	-100.0	-	-100.0	-100.0	-	-100.0
Dipterix alata	0.0	0.0	0.0	3.0	3.6	3.3	0.0	0.0	0.0	6.3	7.7	7.0	3.1	3.9	3.5
Erioteca condolleana	0.0	0.0	0.0	1.3	1.1	1.2	0.0	0.0	0.0	2.1	2.2	2.1	1.0	1.1	1.1
Erythroxylum dahpnites	10.4	7.4	9.6	19.1	9.5	15.5	-6.5	0.0	-3.3	-11.3	8.4	-2.0	-8.9	4.2	-2.6
Eugenia florida	42.3	0.0	24.5	39.5	2.6	23.4	58.1	0.0	25.7	36.1	5.6	19.9	47.1	2.8	22.8
Eugenia ligustrina	0.0	31.1	19.3	1.8	16.9	10.0	0.0	164.6	62.7	3.6	51.0	25.1	1.8	107.8	43.9
Faramea cyanea	0.0	0.0	0.0	4.8	1.6	3.2	0.0	0.0	0.0	10.7	3.2	6.9	5.4	1.6	3.5
Ficus cf. enormis	100.0	0.0	100.0	100.0	4.1	100.0	0.0	0.0	0.0	-74.0	8.8	-46.8	-37.0	4.4	-23.4
Garcinia gardneriana	0.0	0.0	0.0	5.3	2.8	4.1	0.0	0.0	0.0	-10.5	5.9	-2.7	-5.3	2.9	-1.3
Guapira areolata	0.0	0.0	0.0	0.0	0.8	0.4	0.0	0.0	0.0	0.0	-1.5	-0.8	0.0	-0.8	-0.4

Guazulma ulmifolia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Guettarda verbenoides	0.0	0.0	0.0	6.3	0.0	3.2	0.0	0.0	0.0	14.3	0.0	6.9	7.1	0.0	3.5
Heteropterix birsonimifolia	50.0	5.3	50.0	50.0	21.3	50.0	-	-10.6	-	-	-36.5	-	-	-23.5	-
Hirtella gracilipes	0.0	50.0	50.0	0.0	50.0	50.0	-	-	-	-	-	-	-	-	-
Hymenaea courbaril	0.0	0.0	0.0	2.4	2.2	2.3	0.0	0.0	0.0	4.4	4.6	4.5	2.2	2.3	2.3
Inga laurina	0.0	6.7	3.5	2.2	3.3	2.8	0.0	15.5	7.5	4.6	7.2	5.9	2.3	11.3	6.7
Ixora warmingii	2.6	0.0	1.3	7.0	3.2	5.2	-5.1	0.0	-2.6	-2.2	6.9	2.3	-3.7	3.5	-0.2
Jacaranda caroba	0.0	0.0	0.0	19.2	18.8	100.0	0.0	0.0	0.0	62.5	-37.5	0.8	31.3	-18.8	0.4
Luehea divaricata	0.0	3.7	1.9	7.2	3.9	6.6	0.0	-7.4	-3.8	15.9	-2.2	6.4	7.9	-4.8	1.3
Machaerium acutifolium	0.0	0.0	0.0	1.9	1.8	1.8	0.0	0.0	0.0	3.8	3.7	3.8	1.9	1.9	1.9
Machaerium brasiliensis	100.0	0.0	100.0	100.0	2.9	100.0	0.0	0.0	0.0	-23.8	6.3	-10.0	-11.9	3.1	-5.0
Machaerium hirtum	50.0	0.0	50.0	50.0	2.9	50.0	-	0.0	-	-	6.1	-	-	3.0	-
Machaerium nyctitans	0.0	0.0	0.0	15.7	2.6	9.7	0.0	0.0	0.0	45.9	5.6	24.1	23.0	2.8	12.1
Machaerium stiptatum	0.0	0.0	0.0	5.4	0.0	2.8	0.0	0.0	0.0	12.1	0.0	5.9	6.1	0.0	2.9
Machaerium villosum	12.0	5.9	9.8	12.6	4.7	9.1	-13.4	-11.8	-12.6	-18.1	-1.7	-10.3	-15.8	-6.8	-11.5
Maclura tinctoria	0.0	0.0	0.0	5.6	5.0	5.3	0.0	0.0	0.0	12.5	11.1	11.8	6.3	5.6	5.9
Maprounea guianensis	0.0	0.0	0.0	20.0	3.7	12.7	0.0	0.0	0.0	66.7	8.0	34.2	33.3	4.0	17.1
Margaritaria nobilis	14.6	0.0	8.0	11.6	0.0	6.2	41.4	0.0	18.9	30.3	0.0	14.2	35.9	0.0	16.5
Matayba guianensis	6.7	0.0	3.5	11.3	5.7	8.5	-13.4	0.0	-6.9	7.2	12.7	9.9	-3.1	6.4	1.5
Maytenus floribunda	0.0	0.0	0.0	3.1	0.0	1.6	0.0	0.0	0.0	6.7	0.0	3.3	3.3	0.0	1.6
Metrodorea pubecens	2.9	0.0	1.5	5.0	0.9	3.0	-5.7	0.0	-2.9	-2.8	1.7	-0.6	-4.3	0.8	-1.7
Myracrodruon urundeuva	0.0	0.0	0.0	4.4	3.3	3.9	0.0	0.0	0.0	9.6	7.2	8.4	4.8	3.6	4.2
Myrcia splendens	0.0	3.7	1.9	1.9	8.8	5.9	0.0	-7.4	-3.8	3.0	-7.6	-2.4	1.5	-7.5	-3.1
Myrcia tomentosa	0.0	3.7	1.9	4.0	2.7	3.3	0.0	8.0	3.9	8.6	5.7	7.1	4.3	6.9	5.5
Ocotea corymbosa	0.0	0.0	0.0	3.7	1.4	2.6	0.0	0.0	0.0	8.0	3.0	5.4	4.0	1.5	2.7
Piptadenia gonacantha	0.0	34.2	21.9	9.0	31.6	22.5	0.0	216.2	77.8	22.0	171.7	82.0	11.0	194.0	79.9
Platipodium elegans	0.0	0.0	0.0	3.2	3.0	3.1	0.0	0.0	0.0	6.9	6.2	6.6	3.5	3.1	3.3
Pouteria gardnerii	50.0	5.3	50.0	50.0	6.9	50.0	-	11.8	-	-	14.4	-	-	13.1	-
Pouteria torta	0.0	9.2	4.8	2.5	9.7	6.3	0.0	22.5	10.7	5.2	24.2	14.3	2.6	23.3	12.5
Psidium rufum	50.0	0.0	50.0	50.0	2.8	50.0	-	0.0	-	-	5.9	-	-	2.9	-
Qualea dicotoma	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.0	0.0	2.0	2.0	0.0	1.0
Qualea jundiahy	12.9	5.9	9.6	29.6	4.0	18.3	-11.8	13.4	0.0	-31.8	8.6	-14.0	-21.8	11.0	-7.0
Rhamnidium elaeocarpum	0.0	0.0	0.0	3.8	1.3	2.6	0.0	0.0	0.0	8.3	2.6	5.4	4.2	1.3	2.7
Rudgea virbunoides	0.0	0.0	0.0	6.6	0.9	3.9	0.0	0.0	0.0	-9.1	1.9	-3.8	-4.6	1.0	-1.9
Schefflera morototonii	9.2	4.8	7.0	3.1	3.8	3.5	22.5	10.6	16.4	6.6	8.3	7.4	14.6	9.4	11.9

Simira viridifolia	25.0	0.0	14.6	39.4	2.9	23.6	-50.0	0.0	-29.3	-26.0	6.1	-11.4	-38.0	3.0	-20.4
Sorocea bonplandii	0.0	0.0	0.0	14.9	1.0	8.5	0.0	0.0	0.0	42.4	2.1	20.6	21.2	1.1	10.3
Sweetia fruticosa	5.3	0.0	2.7	2.4	1.1	1.8	11.8	0.0	5.7	5.1	1.2	3.2	8.5	0.6	4.4
Handroanthus impetiginosus	0.0	14.6	8.0	3.0	0.3	1.7	0.0	41.4	18.9	6.4	0.6	3.5	3.2	21.0	11.2
Handroanthus serratifolius	3.7	3.7	3.8	8.5	3.4	6.0	-7.4	8.0	0.0	-2.6	7.2	2.2	-5.0	7.6	1.1
Terminalia phaeocarpa	50.0	9.2	50.0	50.0	13.1	50.0	-	22.5	-	-	35.3	-	-	28.9	-
Trichilia catigua	0.0	0.0	0.0	1.6	0.0	0.8	0.0	0.0	0.0	3.3	0.0	1.7	1.7	0.0	0.8
Trichilia pallida	5.3	5.3	6.9	4.7	6.9	7.1	11.8	-10.6	0.0	10.3	-7.0	1.3	11.0	-8.8	0.6
Vatairea macrocarpa	0.0	0.0	0.0	4.0	4.1	4.1	0.0	0.0	0.0	8.7	9.0	8.9	4.3	4.5	4.4
Virola sebifera	0.0	4.4	2.2	3.5	1.2	2.4	0.0	9.5	4.7	7.6	2.5	5.0	3.8	6.0	4.8
Xylopia aromatica	14.6	0.0	8.0	13.9	18.7	16.4	41.4	0.0	18.9	38.3	59.9	48.7	39.9	29.9	33.8

CHAPTER 5

Dynamic species groups due dam impacts on forests: a contrast between seasonal environments

Resumo: Grupos de resposta dinâmica de espécies devido aos impactos de reservatórios nas florestas: um contraste entre ambientes sazonais

As plantas respondem a distúrbios de forma diferenciada: algumas aceleram o crescimento, outras cessam ou mesmo morrem, causando conseqüências ao ecossistema. Por exemplo, as plantas respondem à disponibilidade hídrica no ambiente,com diferentes graus de tolerância ao excesso de água em ambientes sazonais. O aumento da disponibilidade de água em florestas antes condicionadas a períodos de estiagem deve proporcionar diferentes respostas das espécies arbóreas, sobretudo desfavorecendo aquelas mais especialistas para ambientes secos e favorecendo as generalistas à saturação hídrica. Nós avaliamos as respostas de espécies arbóreas de florestas sazonais, semidecidual e decidual, sob influência do aumento da umidade do solo após represamento artificial de um rio, com a hipótese que as florestas deciduais devem possuir mais grupos de resposta, sobretudo com resposta negativa, por ser um ambiente sob maior déficit hídrico durante a estação seca, e possuir mais espécies especialistas desta condição. Buscamos detectar grupos de resposta consistentes, utilizando as taxas de dinâmica das espécies arbóreas próximas a beira de um lago artificial gerado por uma barragem. Para isso nós utilizamos as parcelas próximo à beira de represa (0-30m) para aferir o impacto causado pelo aumento de umidade nos dois primeiros anos após o represamento e como controle, utilizamos parcelas distantes da margem do lago (30-60m) no período de dois a quatro anos após o represamento, período em que poucas mudanças foram notadas. Para a formação dos grupos utilizamos a distância euclidiana entre as taxas de dinâmica das espécies, e como método de agrupamento o método de Ward. As análises demonstraram a formação de poucos grupos no controle e de vários grupos após o impacto, para ambas as florestas, demonstrando que as alterações ocorridas devido ao represamento foram fortes. Houve a formação de mais grupos de resposta nas florestas deciduais em relação à semidecidual. Os grupos de resposta negativa na floresta decidual apresentaram mais espécies, muitas especialistas de florestas secas. No entanto, enquanto este foi o grupo com maior densidade na floresta decidual, o mesmo teve baixa representatividade na floresta semidecídual, onde um grupo mais estável foi predominante, indicando maior resistência desta floresta aos impactos causados pela represa. Para ambas as florestas existem grupos positivamente afetados, demonstrando capacidade de resiliência a este distúrbio; porém, esta capacidade foi maior na floresta decidual, por possuir grupos com taxas altas de recrutamento e incremento. Por outro lado, a floresta semidecidual formou menos grupos, sendo mais resistente, ecologicamente, às mudanças, provavelmente por se tratar de uma fisionomia caracterizada por solo sem forte déficit hídrico na estação seca.

Palavras chave: floresta decidual, floresta semidecidual, grupos de resposta, reservatório
Kew words: deciduous forest, semideciduous forest, response groups, reservoir

Introduction

Plants respond to disturbances, some accelerate growth, others die or stop growing. These responses to external factors as light and soil moisture led scientists to classify species in plants response groups. For example, some species on forest environments are classified according light demanding as pioneers or shade-tolerant (Denslow et al. 1990; Pearcy 1999; Swaine & Whitmore 1988); other classification use establishment under different hidric resources, with tolerant and intolerant species to drought, and those tolerant to high water saturation (Kozlowski & Pallardy 2002). These response groups formation facilitates the understanding of a particular impact on a community, because interpret few groups is easier than understand about a wide range of species.

Recently, the use of annual dynamic rates, that take into account annual growth rate (increment), establishment of new recruits and mortality (linked with loss in basal area, the outgrowths rates) had been considered efficient of species response groups' formation (Chazdon et al. 2010; Gourlet-Fleury et al. 2005; Chapter 2). These dynamic rates are based on monitoring of permanent plots distributed around the world (Condit et al. 1999; Lewis et al. 2004; Phillips et al. 2004), and those studies have helped to evaluate forest communities beneath different disturbances, natural or anthropogenic.

Among many studies based disturbances on forests, those with direct relation to human impacts are very important because serious changes on landscape due human intervention are known such as species lost (Brooks et al. 2002) and several structural changes (Foster 1992). Understanding these changes could help us to better mitigate human impacts and even use knowledge in restoration projects. Among most serious human impacts, two are closely related with water avaiability, the most valuable abiotic resource to life organisms: 1) climate change and 2) modification of hydrological regime of watersheds through dams. Climate change is related to increased of carbon dioxide concentration in atmosphere (Smith et al. 2000; Weltzin et al. 2003). Such changes increase in global temperature and enhance rain frequency in some areas and prolonged drought to others (Walther et al. 2002), affecting entire communities. Shifts in precipitation have an greater influence on ecosystem dynamics, however the impacts of climate change on rainfall are a slow and gradual process (Weltzin et al. 2003), which makes difficult to evaluate its consequences on forests (the largest holders of biomass on Earth's surface - (Dixon 1994).

Otherwise, damming of rivers changes the entire watershed and form huge artificial lakes, which cause drastic and quickly increase in soil water availability (Chapter 3). This water enhances leads to rapid trees responses (Chapter 4). Then, these forest affected by dams are an excellent object of study to evaluate increase of water availability consequences to ecosystem.

Although result in several changes to the landscape (Nilsson & Berggren 2000), dams are distributed worldwide, mainly on high diversity biomes (Nilsson et al. 2005), notwithstanding few works address changes for tree species changes (see exceptions at Dynesius et al. 2004; Xu et al. 2009), and studies dealing with response groups are non-existent. Many of these dams are located in sloped or mountainous terrain (Truffer et al. 2003), which were far away from any water sources before damming (Chapter 3). Therefore, such lands have in common a high slope and low water availability in the dry season. In this scenario often occur forests and, in South America, these patches are associated to Dry Forests.

Previous studies shows rise in basal area after impoundment (Chapter 3), but species had different answers to soil moisture increase (Chapter 4). As the number of trees per hectare remained constant, we hypothesized that species specialist of these environments (high tolerance to water deficit on dry seasons) were the most negatively affected by dam, which is offset by increase of individuals from generalist species. Spatial and temporal availability of water has ecological and evolutionary implications (Schwinning & Ehleringer 2001); then, increase on water availability will probably complicate drought specialist species establishment.

These dry forests can be subdivided in two types, called deciduous and semideciduous forest (Oliveira-Filho & Ratter 2002). Both are seasonal, but in first one drought is more severe due soil being shallower and retain the least amount of water in the dry. Thereby, we hypothesized that in deciduous forest will have more response groups, because the soil moisture increase was more drastic and change the entire systems water relations. Nevertheless, semideciduous forests are a milder environment with species less associated to drought avoidance or tolerance, and then is expect less severe responses less species groups.

Material and Methods

Study Area - This study was conducted in three dry forest (18°47´40”S, 48°08’57”W, 18°40’31’’ S, 42°24’30’’ W and 18°39’13’’ S, 48°25’04 W; Figure 5.1) located in the Amador Aguiar Complex Dam (two dams located in Araguari River with 52m and 55m depth). The first dam (Amador Aguiar Dam I, from here AD1) finished flooding in 2005 and is at 624 meters of elevation (relative to sea level), with a flooded area with 18.66km^2 (CCBE 2007). The second dam

(Amador Aguiar Dam II, from here AD2) ended flooding in 2006 and at 565 meters of elevation (relative to sea level), with a flooded area with 45.11km^2 (CCBE 2006).

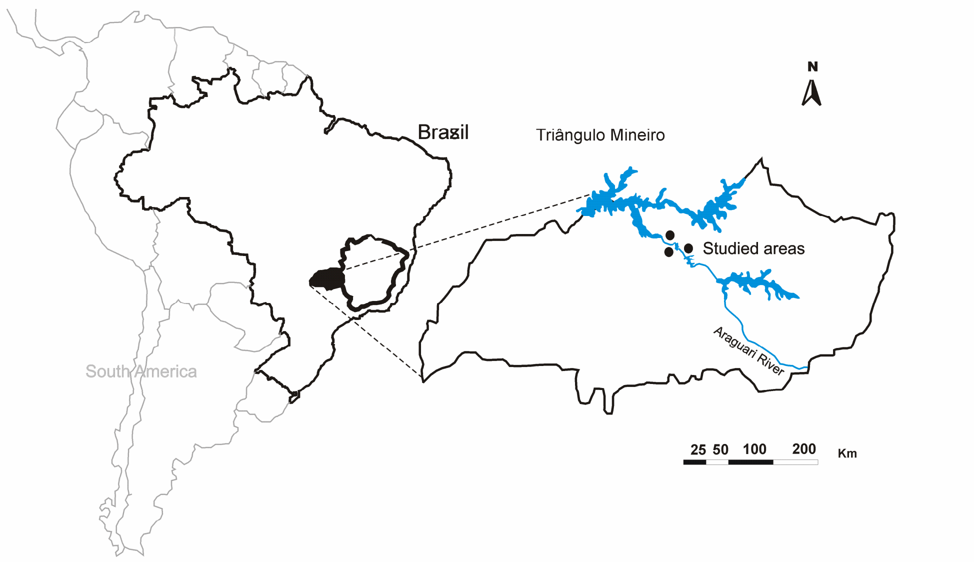

Figure 5.1 - Location of three Dry Forest affected by Amador Aguiar Complex Dam, in Triângulo Mineiro, southern Brazil. Before dam construction these forests were distant from water sources and now are on the artificial lake margin.

After damming, three seasonal dry forests (two deciduous and one semideciduous forest), before distant at least 200 m (see Figure 5.2A as representation) from any water source, now has the riverbed on its edge since 2005 (AD1)/2006 (AD2). Damming increased the amount of water in soil, mainly on dry season (Chapter 3) and several impacts on species in first two years were severe (Chapter 3 and Chapter 4), mainly in the first thirty meters of proximity to the shore. Then, we chose to evaluate in more detail the changes occurring in forest patches closer to the artificial lake created by dam (plots within 30m away from river – 0-30m, Figure 5.2A) and contrast to patches distant to the shore (plots 30 m to 60 m distant to the shore – 30-60m, Figure 5.2B) were dam effects were lower.

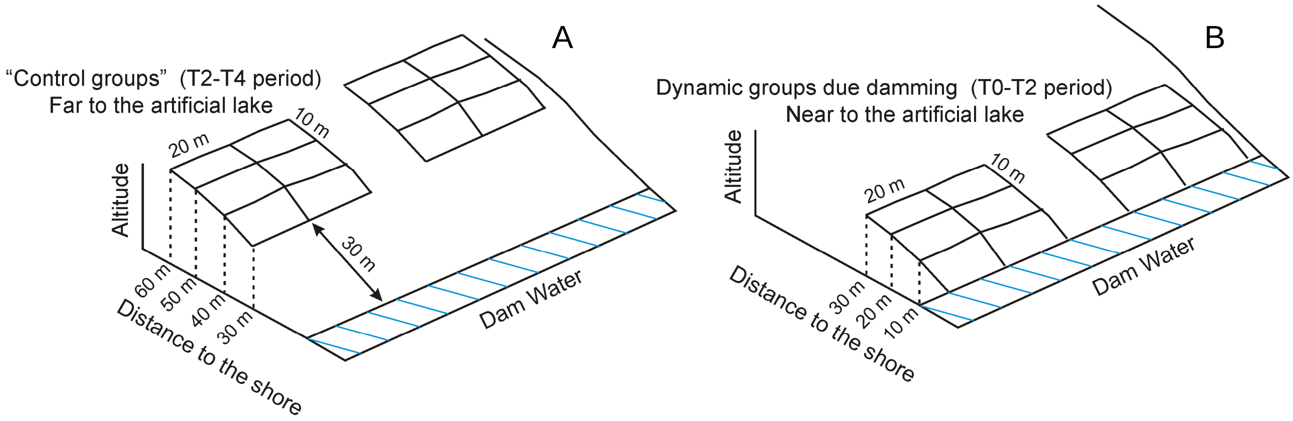

Figure 5.2 – "A" - Representation plots used to dynamic groups generation. Control sites were distant to dam effects, the T2-T4 was chosen by represent a normal deciduous forest without dam influence. considered and water proximity to Dry Forest after dam construction. "B" - Plots scheme used to arboreal community samples in three Dry Forests.

Previous works (Chapter 3 and Chapter 4) showed lower community modifications to these plots located far to river shore in the period after the first two years of damming (T2-T4) and we decided to use these plots as a control to damming modifications. We also grouped the two deciduous forests to analysis because their responses to increased soil moisture were very similar (Chapter 3). Thus, we evaluated dam impacts in deciduous and semideciduous forests using trees sampling carried out in three measurement times (T0, T2, T4) on these forests and species annual dynamic rates (mortality, recruitment, outgrowth and ingrowth rates – more details on Chapter 4) from each species. Climate in all forests is of Aw (Koppen 1948), with a dry winter and rainy summer with an average annual temperature of 22°C and average rainfall around 1595 mm (Santos & Assunção 2006).

Dynamics groups - To verify how the water increase near to forest (0-30m) affected species in first two years of damming (T0-T2), we performed a cluster analysis, using simple Euclidean distance and Ward method as clustering technique in deciduous and semideciduous forests apart. We tried to detect statistically consistent "species dynamics groups" which suffered similar effects after damming. For this analysis we used mortality, recruitment, outgrowth and ingrowth species rates with at least 10 trees sampled. The same proceeding was done in plots distant to the artificial lake (30-60m, Figure 5.2B) in T2-T4 period, to confirm the dynamic groups found near to shore occurred due damming impact (some species with few trees sampled, four or less, were removed due high interference on dynamic analysis).

To verify groups formation efficiency we perform a discriminant analysis. The purpose of this analysis was to test significant differences between groups (in this case, those groups resultant from cluster) and determines discriminant functions that allow reclassifying species wrongly classified. So, this technique estimates probability of correct classification and reclassifies some species making more consistent groups (Gotelli and Ellison 2010).

We still calculated all dynamics rates (mortality, recruitment, ingrowth, outgrowth, individuals turnover, basal area turnover, individuals net change, basal area net change and overall net change) to each “species dynamics group” to the control groups and to dam affected groups. The same procedure was done to dam affected groups in T2-T4 period to verify if dam impacts in each group were similar in these periods.

Results

Groups in deciduous forest

Dynamics groups distant to dam effects (control) – The cluster formed basically two groups (Figure 5.3A), a low turnover (19 species) and high turnover (five species) group. The second one could be divided in a stable (four species) and a negative group, with just one species at 40 of Euclidian distance. The discriminant analysis confirmed cluster efficiency (Table 5.1) and not reorganized any species, confirming coherence on group formation in cluster technique ($F_{4,19}$=15.56, p<0.001). However, only recruitment and ingrowth were significant to groups formation (Table 5.1, *Aloysia virgata* were not included because groups with one species cannot be included in discriminant analysis). The first root was significant (Table 5.2) with correlation of 87% to groups formation.

Dynamics groups near to the dam (first two years after damming) – The cluster formed basically two groups too, with low turnover (19 species) and with high turnover (nine species - Figure 5.3B), with distinct answers to dam impacts. Both groups could be divided in other two groups, based on each group dynamic rates, following same procedure used on control analysis. Thus, were evidenced a positive (11 species), a negative (eight species), a very positive (six species) and a unstable group (three species). Discriminant analysis confirmed cluster efficiency and not reorganized any species (Figure 5.4A), confirming coherence on group formation in cluster technique ($F_{12,55}$=11.16, p<0.001). However, mortality rates were not significant to groups formations (Table 5.1). The two first roots were significant, with eingenvalue of 5.73 and 2.19

(Table 5.2, p<0.001), with correlation of 92% and 83% to groups formation (see canonical values – Table 5.2).

Groups in semideciduous forest

Groups distant to dam effects (control) – Cluster formed two groups (one with 12 and other with four species) but with small distinct rates between both (Figure 5.3C). Discriminant analysis confirmed existence of two groups ($F_{4,11}$=12,39, p<0.001), but the only variable that separated groups were recruitment (Table 5.3). However, differences on the dynamic rates to these groups were very low, furthermore following same procedure adopted on deciduous forest, no groups were found at 40 of Euclidian distance, thus we consider existence of only one stable group (Figure 5.3C).

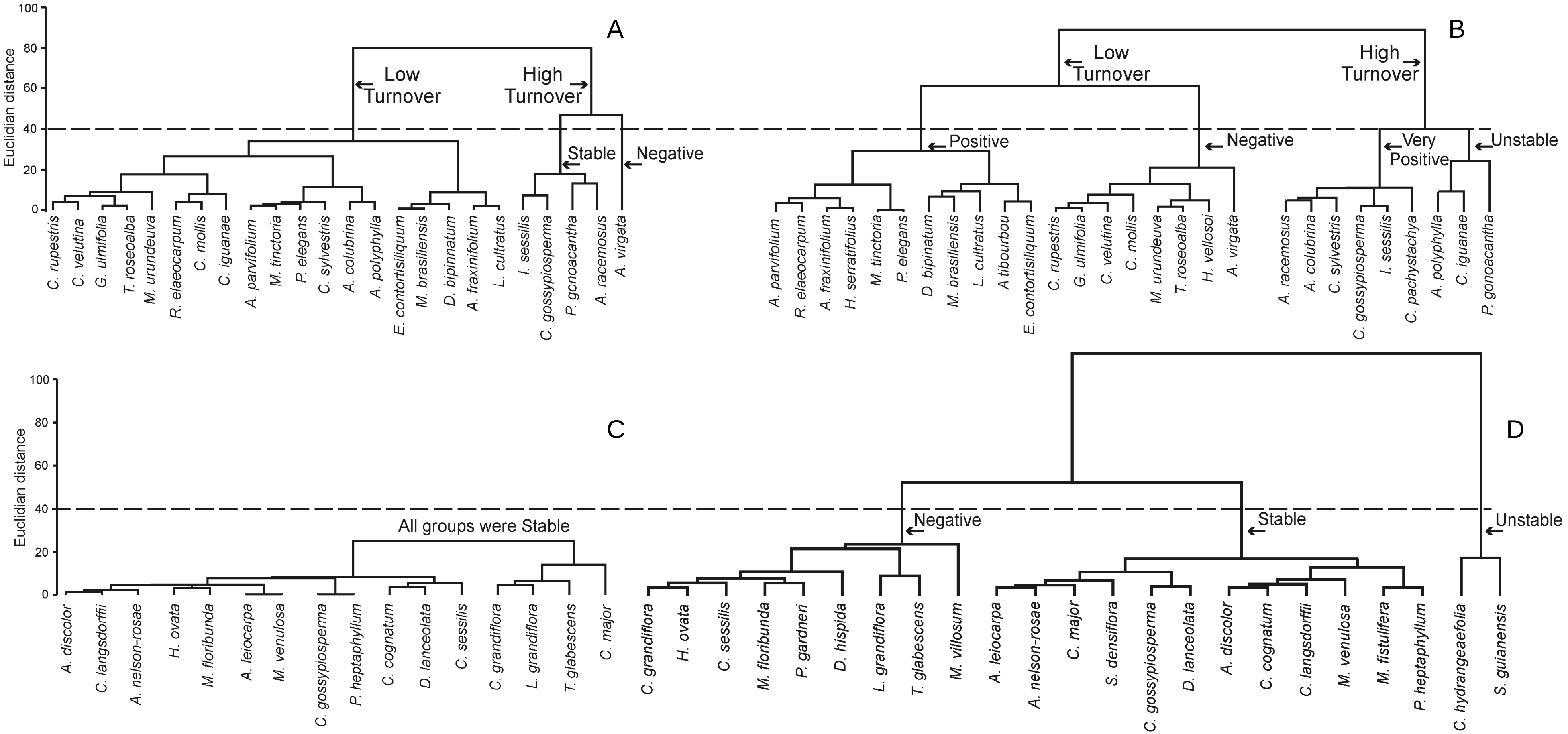

Figure 5.3 - Cluster analysis using Euclidian distance and ward method. Groups were formed based in annual dynamic rates (mortality, recruitment, outgrowth and ingrowth), in dry forests in southern Brazil. A = deciduous forest control dynamic groups, B = deciduous forest dam affected dynamic groups, C = semideciduous forest control dynamic groups, D = semideciduous forest dynamic groups.

Groups near to dam (first two years after damming) - Cluster formed only three groups (Figure 5.4D), with distinct answers to dam impacts. Based on each group dynamic rates, we evidenced three groups: a stable (nine species), a negative (12 species) and an unstable (two species). Discriminant analysis confirmed cluster efficiency and not reorganized any species (Figure 5.4B), confirming coherence on group formation in cluster technique ($F_{8,34}$=20.55, p<0.001). Unlike the deciduous forest, only ingrowth rates were not significant to groups formations (Table 3). The two first roots were significant with eingenvalue of 8.14 and 2.72 (Table 5.4, p<0.001). Then, first two functions were high correlated to groups formation (94% and 85% of correlation, see canonical values -Table 5.4).

Table 5.1 - Discriminant function analysis summary to four groups in deciduous forests in southern Brazil. In bold, p < 0.05, and in italic p < 0.10

	Wilks'	Partial	F-remove	p-level
Control				
Mortality	0.234	0.997	0.05	0.816
Recruitment	0.600	0.391	29.63	**0.001**
OutGrowth	0.236	0.992	0.15	0.412
Ingrowth	0.254	0.922	1.61	**0.010**
Dam Affected				
Mortality	0.041	0.947	0.38	0.764
Recruitment	0.122	0.321	14.76	**0.001**
OutGrowth	0.076	0.519	6.48	**0.002**
Ingrowth	0.054	0.731	2.57	*0.087*

Table 5.2 - Discriminant function analysis summary to two first discriminant roots in thre dry forests in southeasternBrazil. In bold, p < 0.05.

	Eigen-value	Canonical R	Wilks'	Chi-Sqr.	df	p-level
Control						
Function 1	3.277	0.87	0.23	29.06	4	**0.001**
Dam Affected						
Function 1	5.73	0.92	0.04	74.43	12	**0.001**
Function 2	2.19	0.83	0.26	30.59	6	**0.001**

Table 5.3 - Discriminant Function Analysis Summary to four groups in semideciduous forests in southern Brazil. . In bold, p < 0.05.

	Wilks'	Partial	F-remove	p-level
Control				
Mortality	0.183	0.990	0.112	0.744
Recruitment	0.462	0.393	17.001	**0.002**
OutGrowth	0.201	0.902	1.198	0.297
Ingrowth	0.208	0.874	1.590	0.233
Dam Affected				
Mortality	0.042	0.693	3.761	**0.044**
Recruitment	0.091	0.668	4.218	**0.032**
OutGrowth	0.043	0.320	18.029	**0.001**
Ingrowth	0.032	0.899	0.954	0.404

Table 5.4 - Discriminant function analysis summary to two first discriminant roots in a semideciduous forest in southern Brazil. In bold, $p < 0.05$.

	Eigen-value	Canonical	Wilks'	Chi-Sqr.	df	p-level
Control						
Function 1	4.508	0.90	0.181	20.47	4	**0.001**
Dam Affected						
Function 1	8.14	0.94	0.029	65.26	8	**0.001**
Function 2	2.72	0.85	0.268	24.33	3	**0.001**

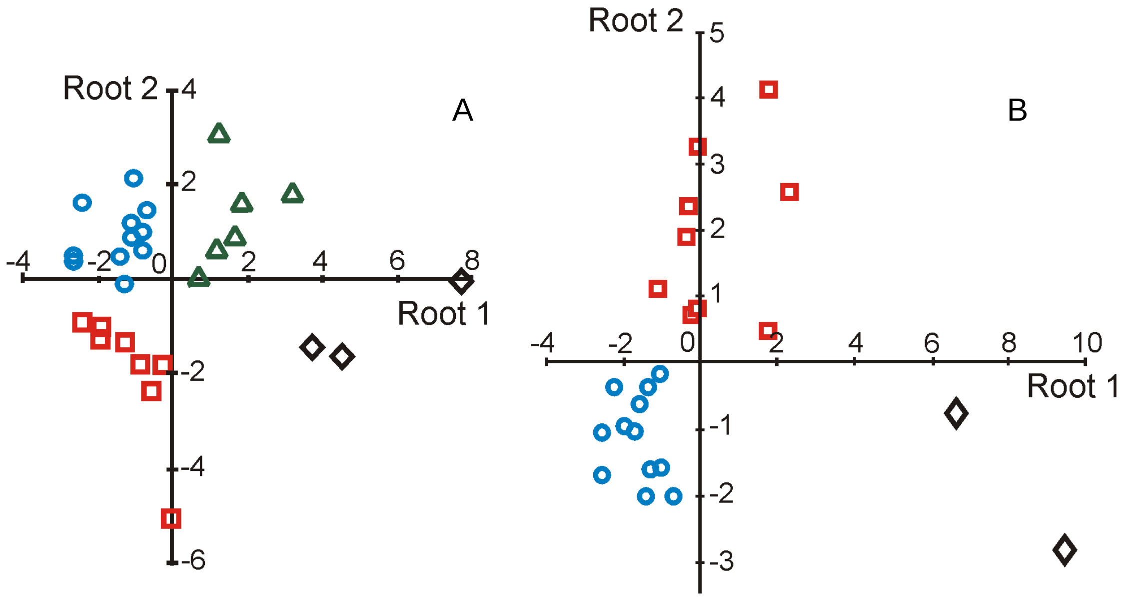

Figure 5.4 - Discriminant analysis based in species annual dynamic rates (mortality, recruitment, outgrowth and ingrowth) affected by dam in dry forests in southern Brazil. A = deciduous dynamic groups, red square = negative group, blue circles = positive group, green triangle = very positive group and black diamond = unstable group. B = semideciduous dynamic groups, red square = negative group, blue circles = stable group and black diamond = unstable group.

Dynamic groups in deciduous forest

Low dynamic x high dynamic groups - Analysis showed division between a low turnover and a high turnover groups in both control and dam affected sites (Figure 5.3A,B). The difference was number of species in each group. High turnover groups presented four more species in sites near to dam compared to control. Then, water approach after damming accelerated the dynamic to this set of species. The dynamic rates to high turnover groups reach values exceeding 10% year^{-1}, mainly to basal area, but dam affected sites the dynamic rates to individuals were very high too (Table 5.5). Moreover on sites close to dam, dynamic rates were much larger than control sites.

Groups with high turnover presented high increment and recruitment rates; however, the very positive group had low mortality and outgrowth rates (Table 5.5). Then, water line approach did not impaired trees of this group. Otherwise, the unstable group had a high mortality, compensated by a very high recruitment (Table 5.5). This group was distinct than the high turnover stable group, found on control analysis, because without dam influence, net change to stable group was close to zero, and with dam influence, net change exceed 15% year^{-1}.

The low turnover group without dam influence was a little distinct than same group direct affected by dam. Without dam direct influence, this group presented very low rates and can be considered very stable; however, after dam construction, there were clear separation between a group with high recruitment and other with high mortality (Positive and Negative group respectively, Table 5.5, Figure 5.3B). Nevertheless, these rates are still lower compared to high turnover groups.

Control x dam affected sites - One group was considered negatively affected in both analysis; however, to control site, only one species composed this group against eight species in dam affected sites. The low turnover stable group and high turnover stable groups (Figure 5.3A), change a lot with dam influence. With high dynamic rates, species that were previously stable presented positive responses to dam and formed other three groups (high turnover very positive and unstable groups, both with high recruitment rates and positive, and the low turnover positive group - Table 5.5). Then, dam impacted entire community, even the more stable groups.

The net change summarizes impacts of dam (Table 5.5). One group was negatively affected (negative) and lost many trees and the other three (very positive, positive and unstable) increasing their importance in community due positive net rates (Table 5.5). Despite of individuals changes, ingrowths were high to all groups in dam affected sites. Then, those trees which survive to impacts caused by water line approach increase their growth rates, therefore all groups presented positive basal area net change.

Deciduous first two years after dam impacts x next years -When comparing groups rates at T0-T2 with T2-T4 period in sites close to the dam, all dynamic rates decreased (Table 5.5). This leads to lower turnover rates to four groups and demonstrated that major dam impacts occurred in first two years after impoundment. However, tendencies maintain the same (Figure 5.5 A,B,C): three groups continue to enhance their importance to community, with recruitment bigger than mortality and ingrowth bigger than outgrowth rates (positive, very positive and unstable groups) and one group continues to decrease their importance, mainly due very low recruitment rates (Figure 5.5C).

Dynamic groups in semideciduous forest

Control stable group x dam affected groups - The analysis did not showed clear dynamic groups on semideciduous control (Figure 5.3C), otherwise three groups were found on sites directly affected by damming (Figure 5.3D). The single semideciduous group on control sites had low dynamic rates, such as turnover rates and net change (Table 5.5). One group on sites affected by damming remained stable, but other two groups not: a group with negative responses to impact (high mortality and outgrowth rates) and an unstable group, with high mortality and outgrowt rates compensated by high with recruitment and ingrowth rates (Table 5.5).

Table 5.5 - Dynamics rates for "dynamic groups" for deciduous and semideciduous forests in southern Brazil. Control refers to forest sites not affected by dam, T0-T2 refers to forest sites in the first two years after dam construction and T2-T4 refers to the next two years after damming. M = mortality, R = recruitment, O = outgrowth, I = Ingrowth, Ind = individuals, BA = basal area, ONC = overall net change, * = group with just a species.

		Rates Average				Turnover		Net Changes		
	Groups	M	R	O	I	Ind	BA	Ind	BA	ONC
Deciduous Control	Low turnover	3.89	1.83	2.70	5.82	2.86	4.26	-2.10	2.56	0.23
	Negative*	18.35	0.00	26.52	1.05	9.18	13.78	-18.35	-25.74	-22.04
	Stable	4.37	4.37	9.90	10.21	4.37	10.06	0.00	0.76	0.38
Deciduous T0-T2	Positive	2.16	7.68	1.38	11.82	4.92	6.06	5.97	11.89	8.90
	Negative	7.15	2.97	4.65	8.13	5.06	6.39	-4.31	3.67	-0.32
	Very Positive	3.60	12.29	3.93	18.60	7.95	11.26	9.92	18.00	13.96
	Unstable	9.42	23.02	15.49	26.50	16.22	21.00	17.67	14.97	16.32
Deciduous T2-T4	Positive	2.17	4.42	1.44	12.14	3.29	6.81	2.35	12.03	7.19
	Negative	3.84	0.84	2.92	4.76	2.34	3.75	-3.08	3.80	-0.58
	Very Positive	3.28	6.53	4.19	11.94	4.91	7.82	3.48	10.94	6.14
	Unstable	5.72	8.23	10.35	14.01	6.98	11.75	2.74	7.318	3.50
Semideciduous Control	Stable	1.59	2.97	1.71	3.99	2.28	2.85	1.42	2.44	1.93
Semideciduous T0-T2	Stable	4.08	3.78	2.56	6.29	3.93	4.42	-0.31	3.96	1.83
	Negative	7.68	3.55	11.99	6.47	5.62	9.23	-4.15	-5.91	-5.03
	Unstable	26.15	30.72	21.15	23.06	28.43	22.11	6.60	2.49	4.54
Semideciduous T2-T4	Stable	0.72	3.57	0.66	4.49	2.14	2.57	2.95	4.01	3.48
	Negative	2.58	2.31	3.45	7.61	2.45	5.53	-0.13	2.44	1.15
	Unstable	2.02	14.72	3.26	9.46	8.37	6.36	14.89	6.85	10.87

Semideciduous control site had low rates and seems to be stable. Then, the formation of three dynamic groups (two of them with rates superior than 10% year^{-1}) illustrated dam influence on community (Table 5.5). Even species on "unstable group" without dam influence presented low changes (*Siparuna guianensis* and *Cousarea hydrangeaefolia*, low density species in control sites), had only one recruits and no dead trees. Then, the high rates to these species confirm existence of this group, due dam effects.

The net change summarizes the impacts of dam (Table 5.5). One group was negatively affected (negative) and lost many trees and others two (stable and unstable) increasing their importance in community (Figure 5.5). Despite of individuals changes, ingrowth were high to all groups in dam affected sites. Those trees which survive to impacts caused by water line approach increase their growth rates, nevertheless basal area lost by negative group was not compensated by the ingrowth rates. Then, only this group lost basal area in dam affected sites, considering both forests.

First two years after dam impacts x next years - When comparing groups rates at T0-T2 with T2-T4 period in sites close to the dam, practically all dynamic rates decreased (Table 5.5) just as deciduous forest. This leads to lower turnover rates to three groups and demonstrated that major dam impacts occurred in first two years after impoundment. However, all groups becomes less negatively affected in T2-T4 period and showed lower mortality and outgrowth rates than T0-T2. Thus, the negative group becomes stable, and the stable and unstable groups becomes more positive net changes. Nevertheless, only unstable group enhance consistently their importance on community (Figure 5.5F) increasing number of individuals (Figure 5.5D) and basal area (Figure 5.5E).

Deciduous x semideciduous: dynamics groups, general aspects and contrasts

Comparison between deciduous and semideciduous forests showed formation of more dynamic species groups in deciduous than semideciduous forest, although both forest had some similar (not equal) groups. Unstable group occurred in both forest due damming influence; however, to deciduous forest the larger net change were found in T0-T2 period due very high recruitment and ingrowth rates, and to semideciduous forest the larger net change were found in T2-T4 period due lower mortality and outgrowth rates.

Both forests had a negative group too. However, on deciduous owned net changes close to zero in both periods of measurement, meanwhile in semideciduous only in first two years of impoundment and their rates stabilized in T2-T4 measurement (close to zero too). Both forests were very distinct in positive responses. While deciduous forest presented two groups clearly positive affected in first two years after damming, the semideciduous forest did not had a clear positive group and could considered less positive affected by damming.

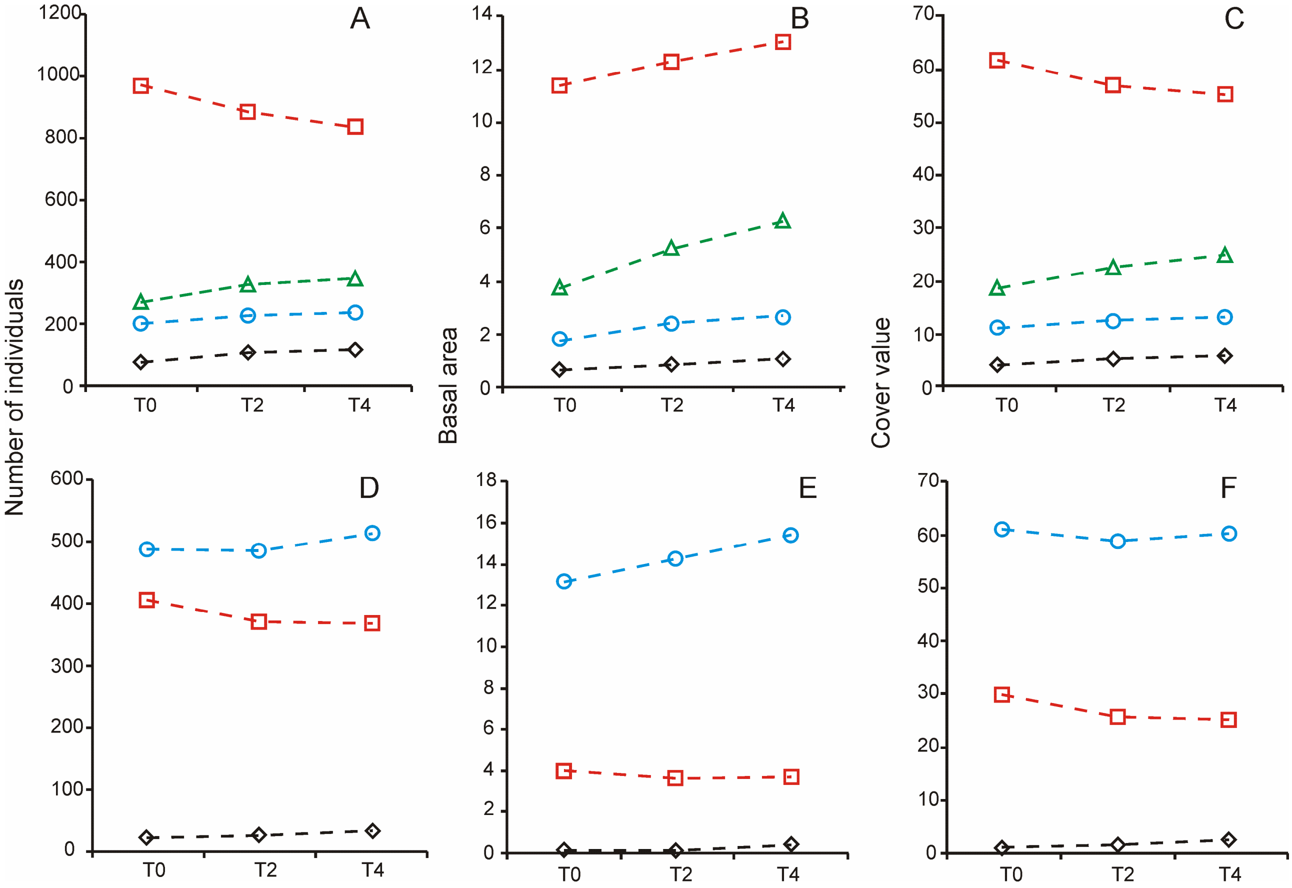

Figure 5.5 - Number of individuals, basal area and cover value to dynamic functional groups directly affected by dam in Deciduous Forest (A, B, C) and in Semideciduous Forest (D, E, F) in southern Brazil. A, B and C = deciduous dynamic groups, red square = negative group, blue circles = positive group, green triangle = very positive group and black diamond = unstable group. D, E and F = semideciduous dynamic groups, red square = negative group, blue circles = stable group and black diamond = unstable group.

Discussion

Negative group

Two dynamic groups found in semideciduous due dam impacts forest occur in deciduous forest too (negative and unstable groups). Unlike entire community, which maintains the number of trees constant (Chapter 3), many species had negative net change in first two years after dam. Low recruitment rates infer problems of group maintenance at new conditions. Mortality was concentrated in “young trees” (Chapter 4); then, many of probable “new recruits” must had died. Early life stages are likely to be sensitive and vulnerable to extreme water available conditions (Fenner et al. 1985; Leuzinger et al. 2005; Stefan et al. 2007), and even more to an environment wont to have an intense dry period. This tendency was reinforced by low recruitment rates on T2-T4 period to negative group in both physiognomies (extremely low on deciduous forest and much smaller than community recruitment rates in semideciduous forest).

In several surveys to these sites we diagnostically several mortality of young trees and saplings of *Myracrodruon urundeuva* (Gusson et al. 2011) and *Guazuma ulmifolia*, two important species in this group to deciduous forest. Besides these species, other species considered specialist of dry forests and well distributed in those environments (Linares-Palomino et al. 2011) had high mortality rates: *Acacia polyphylla*, *Celtis iguanae*, *Tabebuia roseoalba* and *Aloysia virgata* (first two in unstable group and the last in negative group).

Water deficit is an ecological filter which favors plants adapted to low water saturation and species specialists of these environments tend to decline their importance after dam impacts. Growth an survivor are associated to water conductance and even few changes in soil moisture could had a negative consequence to plant growth like roots anoxia (Nilsson et al. 1997; White 2007), mainly to young trees. Based in this group we could infer in several changes to lower layers of the community for many years. Indeed the recruitment in T2-T4 period were zero two these species and both could be extinct on sites near river.

Consequences to future of these forests are unknown, but we believe in a reduction on importance of species of this group over the years. Even in semideciduous forest (which this group maintain their number of trees equivalent) recruitment were lower than other groups, suggesting a reduction in density relative importance. Other works, for example, showed damming impacts on small trees and saplings (Xu et al. 2009) and many new species were found in four years after impoundment (Chapter 4), then major species changes will still occur.

Unstable Group

The presence of negatively affected groups was counterbalanced by recruitment of new species (Chapter 4), high recruitment rates of a positive and a very positive group in deciduous forest and an unstable group in both forests. In fact, this unstable group presents low abundant species that limit major conclusion. However, there were clear tendencies of density rise due rapid decrease on mortality rates after first two years on semideciduous forest, and we conclude that these species will enhance their importance on community in the next years.

In deciduous forest, however, this group is stabilizing with mortality rates near to recruitment ones. Despite of this stabilization, many other changes should occur in next years and this “unstable group” should be monitored and more studied to verify their raise in importance to community. Contrary to low dense unstable groups, the “positive” and the “very positive” groups contributes a lot with new recruits and their high recruitment leads to several changes to future of deciduous community due these high rates.

Positive Group

We could argue that positive groups (positive and very positive) and unstable group represents forest a kind of an “asleep resilience”. Resilience is the community capacity of recover after a disturbance (Walker 1992), but this concept is too wide. We showed that entire community can response positively to a determinate disturbance, but few species can answer very positively to the perturbation. If perturbation event not occurs (and here disturbance was anthropogenic) these groups no need to use this resilience capacity. Then, we consider that this group is “asleep” to this kind of perturbation. However, if disturbance occurs, resilience of group “wake up” and start their positive response to perturbation. This phenotypic plasticity could change the community characteristic after a disturbance and buffer the effects of soil changes (Burgess et al. 1998).

Phenotypic diversity increases the probability of occurred species that are more positively affected by the perturbation (Loreau 2000). In our case, very positively affected group was composed by only six species (only one is specialist of dry forests in deciduous forest *Anadenanthera collubrina* (Linares-Palomino et al. 2011), but five are evergreen and not specialist of dry environments (*Allophyllus racemosus*, *Casearia sylvestris*, *Casearia gossypiosperma*, *Inga sessilis* and *Cecropia pachystachia*) and many new species sampled after four years of damming were evergreen too. Then, this trait is being selected after disturbances, two of them are water associated species (*I. sessilis* and *C. pachystachia*). *C. pachystachia* presented only four trees in first sample and 16 four years after damming.

These changes can be no perceptible just few years after disturbance due long lived trees persistence on environment and because some disturbance (like damming approach) change the environment, but not destroy the community “climax” like hurricanes. Hurricanes and deforestation force the community to return to early sucessional stages (Burslem et al. 2000; Pascarella et al.

2004), and rapid increase of basal area has been noted in early succession forests (Denslow & Guzman 2000; Saldarriaga et al. 1988; Steininger 2000), with a linkage with light availability. In damming, the unstable groups (very positive and unstable) can lead to an unstable period which can change the forest "climax".

This new forest "climax" clearly will present high basal area because all groups showed fast growth. Water limitation and several droughts reduce photosynthetic capacity and plant growth (Goulden et al. 1996; Reichstein et al. 2002). Then, river approach by damming and, consequently, soil moisture increase (Chapter 3) broke this limitation. Mature moist forests tend to presents a high basal area (Murphy & Lugo 1986), and we believe that these forest still enhance their basal area over the years but is difficult to prevent when will stabilize.

Stable and positive groups: implications to conservation

This instability clear changes the community. Only on semideciduous forests we could found clearly a stable group in first two years after damming and, however, in control analysis 14 species in deciduous forests and all semideciduous species were stable. We could conclude that, despite of changes after damming, there still a resistance to perturbation, exemplified by stable group persistence on semideciduous forest. This ecological resistance promotes not only taxonomic maintenance, but maintenance of forest characteristics. Even without having a stable group on deciduous forest, the "positive group" is represented mainly by species canopy, deciduous and light demanding (typical traits of tree deciduous forest community). Then, not only stable groups could indicate community resistance, but also the increase in importance of a set of species with typical traits too.

Considerations and implications for management

Responses on deciduous forest to dam impacts were larger than on semideciduous forest, both for community level (Chapter 3) and species level (Chapter 4). The "dynamic response groups" analysis confirms this tendency. Not only deciduous forest had distinct groups of responses, but these groups showed the unstable period after damming. The groups rates frequently were higher than 10% year^{-1} in deciduous forest, even in T2-T4 period. Meanwhile, the rates on semideciduous were lower than deciduous groups rates, both to T0-T4 and to T2-T4 period. Despite of dry forests been subdivided into seasonal deciduous and semideciduous forests (Oliveira-Filho & Ratter 2002) and both be structurally similar, dam effects showed distinct responses in these forests.

Distinct forest answers to disturbance implies in high distinct management and restoration projects. The resistance (displacement from its equilibrium value following a perturbation, (Loreau 2000) to disturbance on semideciduous forest was high because this forest are a less stressfull environments. In the other hand, the major changes in deciduous forests indicates less resistance to this disturbance, otherwise great adaptability to the new situation, then high complementarity.

Species groups with distinct answers to an environment represent distinct ecological niches (Walker et al. 1999). This ecological niche to water availability are different than functional groups on these areas (Vale et al. 2010), and represents the forest capacity to maintain their attributes for several years even during a permanent disturbance. For example, *Piptadenia gonoacantha*, *Myracrodruon urundeuva* and *Platypodium elegans* presents some traits in common: are canopy trees, light demanding, deciduous and anemochoric species. However, these species presents distinct responses to water soil increase (See Figure 3B). Meanwhile *M. urundeuva* lost many individuals, *P. gonoacantha* recruited many trees and *P. elegans* remaining constant on environment, with high increment. Thereby, these characteristics will persist on community.

Complementarity is not sufficient to maintain the importance of these traits on forest community. There are no species with all above traits on positive group and many species were present on negative group. Then, the importance of these traits on deciduous forest will clearly to reduce. Otherwise, some zoochoric and perennial species had highly positive response (*Allophyllus racemosus*, *Casearia sylvestris*, *Inga sessilis*, *Cecropia pachystachia*). We can conclude that damming changes not only species, but functional groups too. In few years these changes not lead to several changes on community, but as damming condition is permanent, these changes will still occur (rates of recruitment of these species is still high, superior than 10% year^{-1}) and functions will change too.

We predict functional groups changes for next years, and the deciduous forests will become more evergreen and zoochoric, with functional traits more similar to semideciduous forests. The impact on deciduous forest clearly affects patches until 60m of distance to lake shore (see Positive group on “Control” analysis), favouring evergreen zoocoric species too (*Inga sessilis* and *Allophyllus racemosus*). Then, damming effects are very invasive and functional changes on deciduos tend to be more extensive than in semideciduous forest.

More, the most important group in individuals and basal area on semideciduous forest was the “stable group”, while in deciduous the most important group was the “negatively affected”. This scenario guide for much several future changes on deciduous forest for several decades. In natural forests, disturbance regime is driven primarily by tree-falls (Clark & Clark 1996), and high community mortality rates open space to new trees. In many cases high mortality rates are associated with high recruitment rates (Gourlet-Fleury et al. 2005), and is expected high turnover to pioneers species and low rates to non-pioneers (Kohler et al. 2000).

Nevertheless, many adversely affected species on deciduous forest (negative group) are long lived non-pioneer important canopy species (*M. urundeuva*, *T. roseoalba*, *G. ulmifolia*). This imbalance makes sudden soil moisture increase a singular disturbance to dry forest, by affect species independently from light demanding. Indeed, we conclude that deciduous forests are most vulnerable to environmental changes.

Stressful environment changes may leads to a total reformulation of communities, and this is problematic to deciduous forests. These threatened forest physiognomy (Espirito-Santo et al. 2009; Miles et al. 2006) occurs frequently in mountains regions, the more effective places to hydropower production (Truffer et al. 2003). Then, many others deciduous forest areas should had similar responses. In this way, analysis based on “dynamic response groups” was satisfactory because, in world scale, the floristic diversity is too high (Pennington et al. 2009), but the condition of low water forests on dry season occur largely on Earth. More, this singular impact is not a simple transitory disturbance which can be healed, but a permanent disturbance whose leads to a reorganization of these forest. We cannot forecast the stabilization of these forests, therefore we need to monitory this kind of forest by many other years to increase the strong of data and provide more explanations.

Considerações finais

Solo

Confirmando nossas hipóteses as mudanças ocorridas no meio ambiente devido a construção das barragens para geração de energia hidrelétrica Amador Aguiar I e Amador Aguiar II foram notáveis, sobretudo durante os períodos mais secos. Na floresta ciliar houve clara redução na umidade do solo, enquanto que nas florestas estacionais houve aumento da umidade, sobretudo na região mais próximo da margem e nas estações secas. Devido a tal comprovação, podemos inferir que a paisagem das florestas presentes em trechos com vazão de água reduzida e áquela que passa a se situar ás margens dos reservatórios tende a sofrer modificações ao longo do tempo, devido a mudanças fortes no regime hídrico. Para ambientes sob clima estacional em especial, estas variações alteram a relação solo-planta, pois sem água em abundância, o clima passa a ter papel mais importante para as florestas ciliares nestes setores com rio escasso. Já para florestas secas, a chegada do rio tem efeito oposto, amenizando a influência do clima seco, principalmente para as árvores localizadas ás margens da represa.

Modificações gerais nas comunidades

As consequências da alteração na disponibilidade de água no solo tiveram, como hipotetizamos, reflexo na comunidade arbórea. Em todas as comunidades altas taxas de mortalidade e perda em área basal ocorreram. As parcelas da floresta ciliar também puderam ser compartimentada em três setores floristicamente associados: beira do rio, distante do rio e próximo a um córrego. Foram notados maiores modificações justamente na beira do rio com alta mortalidade e recrutamento de espécies semelhantes ao setor distante do rio. Logo a florestaciliar não somente deve suportar menor área basal como também ser florística e estruturalmente menos complexa, indicando redução na heterogeneidade espacial.

Por outro lado, nas florestas estacionais, o crescimento acelerado de muitas árvores de várias espécies proporcionou um aumento na área basal geral da comunidade, enquanto que o recrutamento contrabalanceou a perda de indivíduos mortos após o represamento. As mudanças também foram mais severas nas florestas decíduas em comparação com a floresta semidecídua, provavelmente pelo fato das florestas decíduas terem originalmente uma condição déficit hídrico mais severo, e o aumento na umidade do solo significou uma alteração mais drástica a este ambiente, confirmando nossas hipóteses sobre as possíveis mudanças nestas comunidades. Por isso as taxas de dinâmica encontradas foram maiores do que as taxas já encontradas para demais florestas comparadas pelo mundo, mesmo àquelas com fortes impactos associados.

Espécies

Para as espécies, a redução na importância daquelas especialistas foi verificada para todas as comunidades. A perda de indivíduos de espécies características de ambientes ribeirinhos na floresta ciliar e a mortalidade de espécies consideradas especialistas de florestas secas param nas florestas estacionais ilustram o quão forte são os efeitos deste tipo de perturbação para ambientes naturais, mesmo em um curto período de tempo. Para a floresta ciliar, no sub-bosque estão sendo favorecidas espécies generalistas e comuns em florestas estacionais, indicando mudança florística e estrutural no futuro desta comunidade.

Nas florestas estacionais as mudanças foram mais perceptiveis com estabelecimento de espécies comuns em ambientes ribeirinhos, próximo á beira do lago artificial gerado pela barragem, aumentando a diversidade local da comunidade arbórea apenas quatro anos após o início no impacto, confirmando nossas expectativas. Nós chamamos a troca de espécies típicas de florestas estacionais por áquelas mais adaptadas á água de “Efeito Ciliar”, cuja floresta estacional passa a desempenhar o papel de floresta ciliar. Este efeito aumentou a heterogeneidade da comunidade, basicamente a separando em parcelas onde as modificações foram mais intensas (0-30 m) e parcelas onde as modificações foram menos perceptiveis (30-60 m).

Não podemos, porém ser equivocados e considerar os impactos gerais causados pela construção de barragens, seja para qualquer finalidade, como um evento benéfico para as comunidades naturais. A perda de hábitat e de espécies típicas das florestas que foram alagadas, ou seja, destruídas pelo enchimento das barragens é muito superior a qualquer aumento em espécies que possa ter ocorrido, ou que ainda pode ocorrer nos próximos anos nestas florestas.

Mudanças fisionômicas

Tantas alterações ocorridas nas florestas avaliadas, observadas neste curto espaço de tempo, demonstrou uma tendência de mudanças na própria fitofisionomia. Na floresta ciliar, por exemplo, as reduções na importância de grupos de espécies dependentes de solos com alta saturação hídrica e drásticas modificações no sub-bosque reduziram a área basal total desta comunidade. Sem conseguior repor a biomassa perdida a floresta ciliar se demonstrou incapaz de se recuperar deste distúrbio, logo não é totalmente resiliente á redução de umidade do solo, por isso consideramos esta floresta como em “fase de degradação”, cujo mais modificações devem ocorrer. Como a redução na vazão de água é um dano permanente, grupos de espécies mais generalistas e tolerantes a menores teores de umidade do solo devem continuar a ocupar os espaços vagos, deixados por indivíduos de espécies especialistas ás antigas condições. Isto significa uma mudança fisionômica, de uma floresta úmida associada a um recurso hídrico constante para uma floresta menos úmida mais associada com a típica estacionalidade da região, com menor capacidade de reter biomassa.

Já nas florestas deciduais, onde as modificações foram mais drásticas, o estabelecimento de várias espécies tolerantes á solos com alta saturação hídrica, foi relacionada a determinadas características favoráveis a ambientes não estacionais. Espécies típicas das florestas deciduais perdem as folhas na estação seca devido a baixa disponibilidade hídrica e tendem a possuir madeira pesada, evitando o embolismo e morte do indivíduos sob forte déficit hídrico. Contudo, a grande maioria das novas espécies ocorrentes nestas florestas após o represamento são perenefolias e/ou possuidoras de madeira mais leve. Assim, a nova condição está favorecendo não somente espécies, como também determinadas características que são mais próprias a ambientes sem marcada estação seca.

Relevância do estudo para a ciência, meio social e economia

Este estudo buscou preencher uma lacuna nos estudos de efeitos das construções de barragens sobre comunidades arbóreas para ambientes de clima tropical para demonstrar o quão sério é este tipo de perturbação para tais comunidades. As mudanças notadas foram drásticas e instigam para a continuidade do acompanhamento destas áreas e mais estudos para outras diferentes comunidades sob semelhantes condições.

Para o meio social, a criação de barragens gera diversos problemas como proliferação de vetores de doenças e aumento de incidência de doenças relacionadas com a água. Por outro lado a manutenção de comunidades vegetais á beira das represas representam proteção das águas dos rios, sobretudo contra assoreamento, assegurando a utilidade do empreendimento, no caso à geração de energia para a população. Devido à importância destas comunidades para o bom funcionamento da usina, programas de manutenção e re-vegetação destas florestas são realizados, assim como o Programa de Salvamento do Germoplasma e Coleta de Sementes realizadas pelo Consórcio Capim Brancas e Energia, responsável pela implantação da usina. Demonstramos que muitas das espécies propostas para os programas apresentaram alta mortalidade e baixo recrutamento nas áreas estudadas, logo não são espécies apropriadas para re-vegetação de áreas que passam a ficar ás margens da represa. Em contrapartida, encontramos espécies com respostas positivas ás mudanças, que podem servir de base para próximos programas de re-vegetação semelhantes.

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