The Political Ecology and Ecosystem Services in Yerba Maté (*Ilex paraguariensis*) Agroforestry of the South America Atlantic Forest

by

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Abstract

Agroforestry offers a land-use management methodology that may provide solutions to environmental degradation in the tropics. However, practitioners of agroforestry are faced with the dilemma of clearing more forest cover in order to increase crop size and sustain their income. The aim of this study is to understand the agroforester's dilemma and to measure the value of the agroforestry ecosystem stewardship in yerba maté (*Ilex paraguariensis* A. St. Hil.) agroforestry parcels of the South American Atlantic forest eco-region. Biodiversity, carbon sequestration and vegetation cover were measured to be considerably higher in yerba maté (*Ilex paraguariensis* A. St. Hil.) agroforestry plantations than in neighboring monoculture crops. Agroforestry vegetation cover values were measured to have between 65-89% cover while monocultures had roughly 25% cover. Agroforestry stored carbon values ranged between 154.7-172.7 Mg C ha^{-1}, compared to monoculture plantation values of 81.3 Mg C ha^{-1}. Finally, as measured using the Shannon index, values of species richness ranged from 2.7-3.5 in agroforestry parcels and between 0.9-1.3 in monocultures, and values of evenness ranged between 0.6 and 0.8 in agroforestry parcels, and 0.2 in monocultures. These findings illustrate that yerba maté agroforestry can potentially contribute as a regional climate change mitigation strategy. Valuating and monetizing ecosystem services and engaging smaller farmers with worldwide ecosystem marketplaces offer the potential to expand the dialogue around payments for the valuable ecosystem services that agroforesters are providing. An analysis of market prices available within the ecosystem marketplace for total ecosystem services being conserved on agroforestry parcels amounted to a range in value between \$16 – \$160 ha^{-1} yr^{-1}. To address environmental degradation in the Atlantic Forest region, in South America, governments should motivate environmental conservation to support a shift towards sustainable yerba maté production which supports livelihoods of small-scale farmers, economic justice and environmental sustainability.

Table of Contents

List of Tables

List of Figures

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Chapter I. Introduction and Research Outline

Research Focus

This study developed out of an effort to identify market-based mechanisms that promote large-scale environmental conservation. A process of inquiry began in a collaboration with an American tea company which purposely sources their products in small-scale agroforestry plantations of the Atlantic Forest eco-region of South America. This is taking place when large-scale agriculture in the tropics is developing to meet the demands for higher quantity and reduced costs. To engage in any form of agriculture, farmers tend to define their land-use practises around the dominant political ecology, which currently in the Atlantic Forest eco-region, insists on a larger agricultural output for a lower price. Within these economics of land-use, consideration of environmental consequences is secondary at best. Within this economic paradigm, humans do not consider the cost of environmental impacts resulting from our labour. Though the environment is utilized and relied upon, the environment rarely has a voice nor carries any intrinsic value that is interpretable to global markets (Turner 2003; 2008). In this lack of economic consideration of the value of environment and the services it provides, unsustainable development of the environment is inevitable.

This interdisciplinary thesis explores a process of evaluation and valuation of some environmental conservation properties of agroforestry, specifically in the cultivation of yerba maté (*Ilex paraguariensis* A. St. Hil.) The environmental elements that are analyzed in this thesis were chosen based on their marketability, i.e. the ecosystem services of the agroforestry parcels most likely to be reflected in world markets as having monetary value based on previous transactions and current markets prices (Goldman *et al*. 2008). Further, to elaborate on the lived-experience, and the motives of the agroforester, an analysis of their political ecology, and its recent historical roots, was used to illustrate their growing autonomy amidst the challenge of large-scale agricultural forces.

Theoretical Lens

This thesis uses various methodologies to analyze and interpret information central to the research focus with the intention of developing a practical perspective on the economic and social value of agroforestry in tropical regions.

Land-use methodologies are quantitatively analyzed with the purpose of extending human evaluations of economic worth to the natural environment with the intention of better aligning the elements of wealth with the realities of environmental sustainability. To achieve this goal, marketable ecosystem services within defined land parcels are quantified and related to current market transactions in order to illustrate their potential worth in an economic system that recognizes ecosystem service values. The qualitative component of this paper draws insights from the researcher's experiences with the local inhabitants in order to showcase the political ecology of the agroforester.

Significance and Overview of Outcomes

According to Costanza and Folke, "valuation ultimately refers to the contribution of an item to meeting a specific goal" (1997). In the agroforestry context, the goal is to create sustainable livelihoods in harmony with the natural environment. For agroforesters, the preservation of ecosystem services is therefore pivotal to their goal of sustainability. One aim of this research is to develop discourse around supporting practices of agroforestry with a methodology of valuation. Therefore, this thesis will help to identify a basis upon which to discuss the creation of payments for the stewarding of ecosystem services to agroforestry workers in the tropics. In doing so, momentum will generate towards popularizing payment for ecosysterm service programmes. Such program rewarding stewarding practices of rainforest remnants and restoration agriculture.

Organization of Thesis

Programmes allowing for payment for ecosystem services have strong potential for growth in the near future (Adgar 1995; Costanza *et al.* 1998; Acharya 2000; Assessment 2002; Baranzini *et al.* 2003; Kumar 2005; Boyd and Nanzhaf 2007; Carroll *et al.* 2008). These programmes are either building a discussion around payment for ecosystem services, or are already accessing various sources of financing within a market collectively termed “the ecosystem marketplace.” The carbon market is included within this marketplace however, smaller financial mechanisms based on the carbon market’s credit and offset programmes, which serve to preserve biodiversity and watershed intactness, are culminating and becoming significant sources of funding (Brinkman and Hebda 2008). In order for any project to gain candidacy for funding from the ecosystem marketplace, measuring the product one aims to conserve is a necessary first step. Therefore, evaluating and monetizing marketable ecosystem services on agroforested land parcels in South America was identified as a practical process to create a payment for ecosystem services initiative.

Chapter one includes the research outline and introduction to this thesis. Chapter two provides a context for the research region, describing the enduring ecological problems and the basic drivers of environment degradation and destruction. It also offers a description of agroforestry, and outlines the potential for conserved ecosystem services in agroforested landscapes. Further, a discussion around the ecosystem considerations in the environment is developed which presents the reader with an understanding of why this research is potentially valuable to the agroforester. Chapter three presents the political ecology of the agroforester in the Atlantic Forest eco-region by suggesting that the development of the yerba maté agricultural sector was manipulated by institutional policy promoting large-scale productivity that has marginalized agroforestry practises. Chapter four presents regional profiles and the descriptions of the tools used to measure and quantify ecosystem services in the field sites. Chapter four also contains an abstract, an introduction, and a methods section behind the quantitative assessment of ecosystem services in three agroforestry sites throughout the Atlantic Forest eco-region. Chapter five goes on to present the results from the qualitative assessment of ecosystem services in agroforestry sites, while Chapter six introduces the application of values to the measurements of ecosystem services. Chapter seven discusses the ecosystem service quantification findings and their environmental implications. Finally, Chapter eight provides a summary of the thesis and concludes with key findings.

Chapter II. Context

This Chapter contextualizes the thesis by describing the environmental circumstances of the Atlantic Forest eco-region resulting from current land-use methodologies. A discussion of alternative land-use methodologies follows, namely agroforestry, which in the context of yerba maté, is a viable option for biodiversity conservation without sacrificing economic development. Various ecosystem services offered by agroforested lands are described, and a roadmap towards monetization of ecosystem services in agroforested land is sketched.

Mata Atlântica

The Atlantic Forest or *Mata Atlântica* is being lost to relentless expansion of the agricultural frontier, urban expansion and land speculation (Galindo Leal and de Câmara, 2003). As of 2005, 2% of the remaining biome is protected, leaving the remainging 5% which is still covered in native forest, exposed to clearing from the potential expansion of the conventional agricultural frontier, e.g. slash and burn deforestation, a quick transition to monocultures and cattle grazing. Overall, 93% of the original forest in the Atlantic eco-region is already converted to agricultural lands or cleared due to urban expansion (de Lima Palidon and Guapyassu 2005).

Though the forest has been diminished, the Atlantic Forest eco-region blankets the Atlantic coast of Brazil, from Rio Grande do Norde covering South to Rio Grande do Sul. The Atlantic Forest cover continues inland along rugged coastal topography just skimming the border of Uruguay, and it reaches Eastern Paraguay, blanketing the northern province of Misiones, Argentina. Some off-coast islands, including the archipelago of Fernando de Noronha, are also included in this South American Atlantic forest ecoregion (Cartes 2003) (Figure 1).

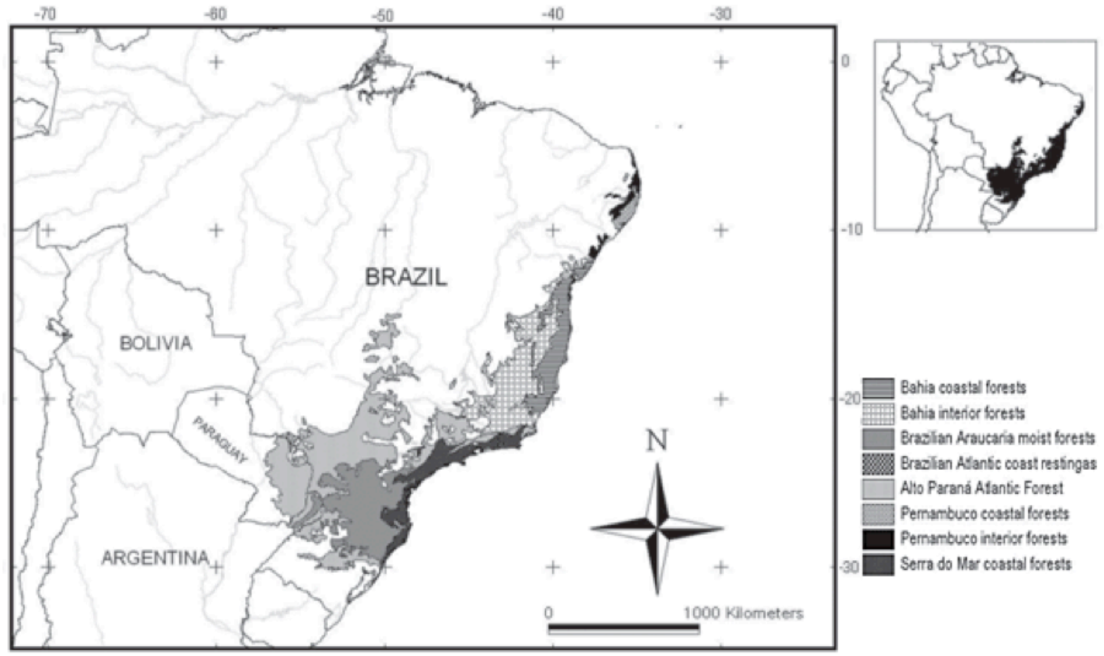

Figure 1. Atlantic Forest eco-regions.

Within this area, the coastal Atlantic rainforest covers a 50-100 km portion of coastline while the interior Atlantic Forest covers the southern foothills in the Serra do Mar into Southern Brazil, Paraguay and Argentina. The forest eco-region extends up to 600 km inland with elevation ranging between sea level and 2,000 meters. The vegetation cover across this range can be broken into three altitude types: the lowland forest of the coastal plain; montane forest; and the high-altitude grassland or campo rupestre (Cartes 2003).

Within the ecosystems of the Atlantic Forest there are over 20,000 plant species; 40 % of the species are endemic to their particular niches and are very dependent on their community structure and local climate (Jacobsen 2003). There are up to 30 critically endangered species throughout the region including six bird species, which are restricted to small patches of forest in northeastern Brazil, and three species of lion tamarins (Pardini *et al.* 2005). Further, 950 bird species make use of these regions in both a resident and migratory capacity, including the red-billed curassow, the Brazilian merganser, and numerous threatened parrot species (Bodrati *et al.* 2005).

The interior Atlantic Forest eco-region once covered large portions of eastern Paraguay, northeastern Argentina and southeastern Brazil, but today only 7.81 % of the original forest cover remains, and the remnants are highly fragmented (Galindo and Leal, 2003; Freitas et al 2005). Deforestation has been most severe in Brazil (Galindo and Leal, 2003) and continues to be driven by large-scale agricultural, urban and industrial development (Cartes 2003) (Figure 2). In 2004, Brazil developed a National Biodiversity Policy legislation to protect and preserve biodiversity (Fearnside 2003). For example, the Brazilian Government's Medida Provisória MP2.166-67 (a presidential decree pending approval into law) requires that forest clearing leave 20% (originally 50%) of the forest intact (Ferraz *et al.* 2003). Though a positive step, this legislation does little to address the conservation of biodiversity corridors. In Paraguay, 13 percent of the original forest coverage exists, but in an effort to grow the country’s economic viability Paraguay has opened its borders to the highest rate of intensive agriculture development in South America in recent decades and currently its remaining patches of forest are highly fragmented (Pardini *et al.* 2005). As can be clearly viewed with satellite imagery, Argentina has the largest area of continuous forest, accounting for roughly 50.9 percent of the original area and covering much of the province of Misiones with varying degrees of degradation (Galindo and de Câmara 2003; Lawson 2009).

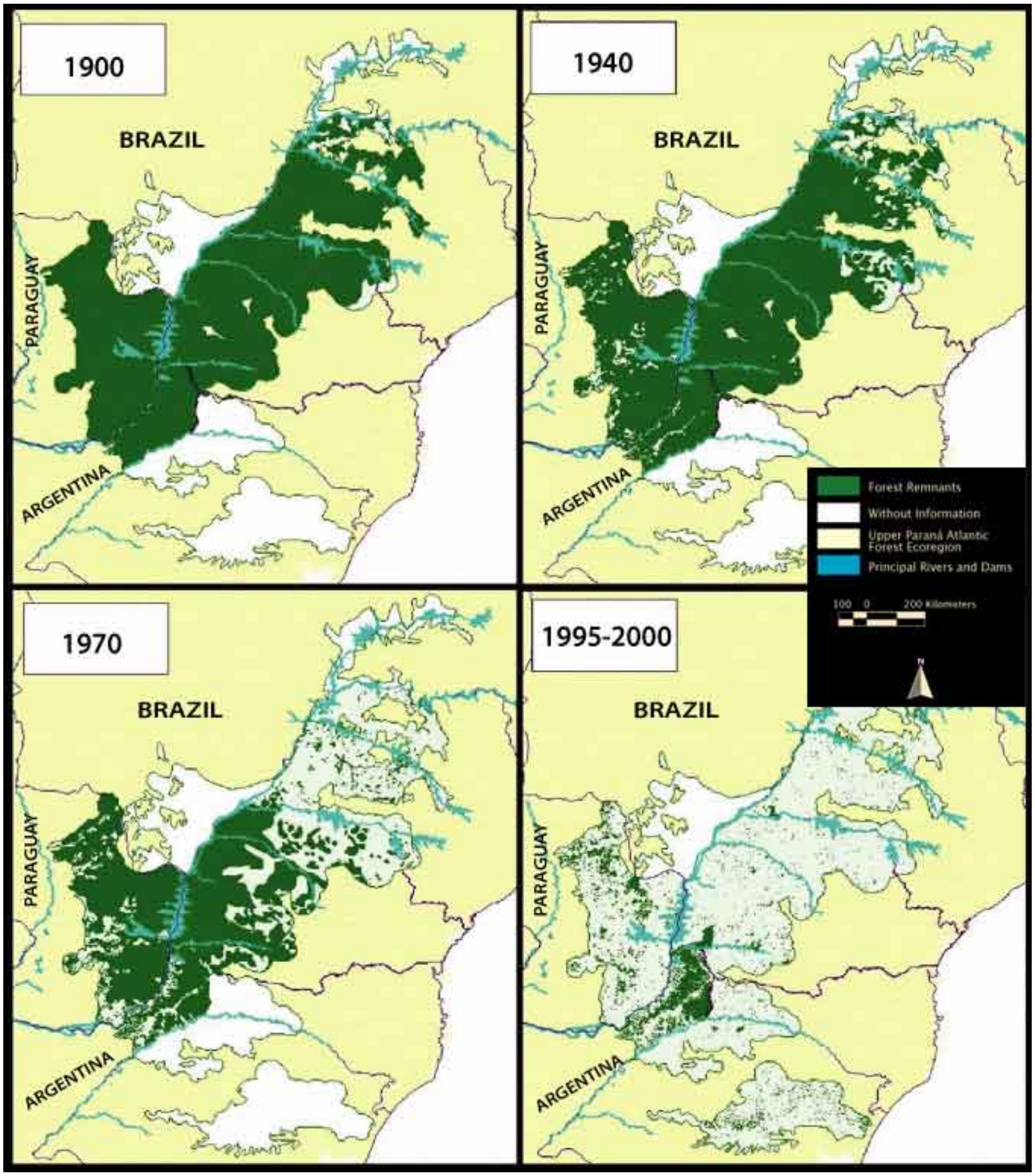

Figure 2. Deforestation of Interior Atlantic Forest of South America since 1900.

Losses of habitat, biodiversity and ecosystem functioning due to rainforest destruction and agricultural intensification are prime concerns for science and society alike. The growth of metropolitan and urban areas also dramatically impacts this eco-region as sprawl and other expansion initiatives interact rather indiscriminately with landscapes and ecosystems. Losing biodiversity threatens the stability and continuity of ecosystems as well as their provision of goods and services to humans (Kremen 1994; Scroth 2004; Raffaelli and Schmid 2006).

An Economic Case for Conservation Based Land-Use Alternatives

Forests have both ecological and economic importance (Adger *et al.* 1995; Pearce 2001). However, their sustainable use is vital in maintaining their ecological value. The South American Atlantic Forest eco-region directly and indirectly supports the livelihoods of 100 million people, and within it are some of the most socioeconomically complex areas of the world (Galindo and de Câmara 2003; Smith 2007). In areas where people are often poor, land is relatively cheap and agricultural methods are often intensive. Large-scale agriculture therefore threatens the future of smallholder organic agriculture and also the economic growth prospects of nations. Some 61 - 91% of the land in the Altantic Forest eco-region has experienced low to severe land degradation as a result of large-scale monoculture agricultural methodologies (Cartes 2003).

The degradation of land can be measured in lower soil quality, the loss of biodiversity and the land's reduced ability to fix carbon (de Lima and Guapyassu 2005; Vieira *et al.* 2008). Erosion and the conversion of forests to agricultural land has had an adverse effect on soil organic carbon which includes a decline in soil structure, soil compaction, reduction in activity and diversity of soil fauna, and nutrient depletion (Hamilton *et al.* 1983; El-Hassanin *et al.* 1993; Mutuo *et al.* 2005). Recent evidence demonstrates that deforestation not only influences the soil biological pools and fluxes, but also can modify the association of biological properties of the soils (Nourbakhsh, 2007). Biodiversity loss results when species-rich woodlands are converted to relatively species-poor farmlands and plantations. The negative effect of deforestation on species richness and overall biodiversity in this region of the world has been demonstrated by numerous scientific studies. The area is now recognized as one of the world's 25 recognized biodiversity hotspots, which are under extreme threats (Cartes 2003). The current figures for threatened species could be much higher since the full extent of the region's species diversity is still unknown. The amount of greenhouse gas emission due to conversion of the forest to agricultural land is high, accounting for between 20-40 % of man-made greenhouse gas emissions. According to the Intergovernmental Panel on Climate Change report (Watson *et al.* 2000), land-use changes (primarily deforestation) have been releasing 1.6 - 1.7 Pg[1] of carbon annually, which is about a third of the emissions from fossil fuels and cement production (Desanker *et al.* 1997). Other natural and anthropogenic processes, such as wild-land burning, clearance of land for cultivation, slash-and-burn agriculture and the cultivation of wetlands, also contribute unquantified amounts of trace gases to the atmosphere, as well as altering the nature of the land cover and hydrological processes (Desanker *et al.* 1997).

Agroforestry Alternatives

Because of the mounting negative consequences of large-scale agriculture, strong support and application of alternatives to land-use are of vital importance in regions where land degradation is occurring. Agroforestry is broadly defined as the union of forestry and agricultural practices to promote optimum results of land-use for all parties concerned, including the environment (Garrity 2004).

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[1] Pg: Pentagram, 10^{15} g

As an agricultural alternative to monocultures, agroforestry provides environmental and economic incentives (Garrity 2004). In many cases, adopting such practices would serve to improve the functioning of the ecosystem services which are under threat (Sell *et al.* 2007). Such services include an increase in carbon storage (Schroeder 1994; Pandey 2002; Albrecht and Kandji 2003; Montagnini and Nair 2004; Nair *et al.* 2009), reduced edge effects (Langton 1990; Scroth *et al.* 2004), reduced agricultural pathogen outbreaks (Sperber 2004), decreased fragmentation (Cullen *et al.* 2001), increased habitat potential (Cullen *et al.* 2001, Klein *et al.* 2002), increased soil conservation and nutrient cycling (Pattanayak, 1996; Scroth *et al.* 2002; Udawatta 2008), and increased watershed conservation (White 1989; Garrity 1989). All ecosystems show nonlinear responses to land-use intensification as noted by Steffan-Dewenter (2007) alternative management options that promote various expressions of native ecosystems, can limit ecological losses and while supporting less intensive agriculture to satisfy economic gains.

Conventionally, the objective of agroforestry research has been to identify those circumstances (biophysical, socio-economic and policy) in which mixing agricultural crops with growing trees would give benefits to farmers (Cannell 1996). As indicated by Nair (1997), larger-spatial-scale issues, such as carbon sequestration, water quality and biodiversity conservation, have been neglected because of the emphasis on field- and farm-scale studies. Fortunately, for a variety of economic and social reasons, agroforestry systems which combine native forests and perennial cash crops are becoming increasingly common. The increasing damage that conventional agricultural methods are having on local and regional productions of ecosystem services position agroforestry as a significantly less destructive alternative.

Applying agroforestry as a land-use management system in order to counter land and forest degradation and the loss of biodiversity, is viable for many crops throughout the tropics (Oke and Odebiyi 2007). In many circumstances, implementation of agroforestry can meet the conflicting goals of agricultural production and environmental stewardship. Many of the current endeavours in agroforestry development have focused on increasing crop yields and crop diversity to meet the needs for human subsistence (Sileshi and Mafongoya 2006). This pressing objective has tended to create management aimed at only maximizing the primary concern of 'soil fertility improvement'. Very few attempts have been made to review and synthesize other knowledge on the functions, processes and capabilities of agroforestry practices in promoting ecosystem services. This has led to minimal appreciation of the environmental benefits of agroforestry, and hence less attention being paid to accelerating its adoption in policy making processes (Sileshi *et al.* 2007).

In a few areas of the Atlantic Forest eco-region, agroforestry systems are being adopted to meet the needs of small-scale farmers (de Lima Palidon and Guapyassu 2005). *Ilex paraguariensis* (South American holly, or yerba maté) is one of the more common perennial species to be intercropped with other tree species in an agroforestry system (Nozzi *et al.* 2000). In the history of Atlantic Forest agriculture, the association of native trees such as *Araucaria angustifolia* and perennial cash crops such as yerba maté is an older practice. yerba maté is a native sub-canopy tree whose leaves are used as a traditional tea with widespread consumption in Argentina, Chile, Paraguay, Uruguay and Brazil. The tree is managed as a shrub, pruning it to heights of no taller than 3 meters. The leaves are harvested every two years in agroforested plantations beginning in the fourth year after planting, with an average annual yield of about 3,000 kg/ha of fresh green leaves. The trees maintain the same productivity for 30 years or more. yerba maté is marketed as a fresh product and is generally sold to processing plants that dry, grind and pack the yerba maté for local, national or international markets (Rao 2009). Presently, in order to increase productivity, maté is mainly grown in open plantations, over 90% of the market being supplied by sun grown maté (Nozzi *et al.* 2000). Monocultures are harvested annually, with an annual average yield of 3,000 kg/ha of fresh green leaves (Lawson 2009). However, this species occurs naturally as an understory tree in the native forest, so it is suitable for agroforestry associations (Nozzi *et al.* 2000). Currently there are measurable benefits to producing organic shade grown yerba maté. Medicinal properties inherent in yerba maté show higher concentrations when grown in the shade. Furthermore, the shade grown bird friendly certification is becoming quite popular in international markets (Bodrati *et al.* 2005; Heck and Mejia 2007).

This research aims to investigate yerba maté agroforestry designs for sustainable use in deforested areas of the Atlantic Forest eco-region by exploring a methodology of assessing ecosystem services on three field sites in the Atlantic Forest eco-region of South America, and by showcasing the current functioning of ecosystem services in field sites. Finally, a comparative economic valuation is applied to provide rough values to each field site respective of the measurements of ecosystem services provided by this study.

Conceptualizing Ecosystem Services

Earth's biosphere is a finite and chaotic system that relies on equilibrium. Such equilibria continually replenish the capacity of an ecosystem in order to support life on the planet. The slow decay of organic material on the forest floor provides some of the nutrient base to be cycled back through the roots of the trees, which in turn provides the infrastructure for diverse forest communities. This cyclical process of decay and growth supports various functions of local ecosystems which themselves lead to various other equilibria, each supporting dynamic webs that share the common theme of dependence and homeostasis. Not surprisingly, ecosystems are changed by humans; often human decisions are made to manipulate, disassemble or destroy ecosystems without knowing or considering the effects of losing those ecosystems. At the base of this dynamic, an effort has been made to interpret ecosystems in a way that world markets, and the human philosophies and mindsets which construct and fortify them, can include the consideration of ecosystems within economic transactions. In an anthropocentric fashion, the conceptually simple term "ecosystems services" has arisen in order to categorize the services that ecosystems can provide to humans and other life in their intact state. Such services include oxygen replenishing, ground water filtration, carbon sequestration and soil accumulation, all services being packaged up in order to have quantifiable units and noticeable costs if lost. These processes or products are just a few of the necessary and irreplaceable tasks that intact ecosystems accomplish; ultimately one can regard ecosystem services as the work ecosystems accomplish which sustains and improves the life of all living things, especially and particularly humans (Daily 1997).

Because human domination of the biosphere is rapidly altering the composition, structure, and function of ecosystems (Vitousek *et al.* 1997), often eroding their capacity to provide services critical to human survival (Palmer *et al.* 2004), the Millennium Ecosystem Assessment has classified ecosystem services into four categories: provisioning services, regulating services, supporting services, and cultural services (ME 2003). Provisioning services provide goods such as food, fuel, medicine and timber. Regulating services include climate and flood control. Supporting services include pollination, population control, soil formation, and other basic ecological properties upon which biodiversity and other ecosystem functions or services depend. Cultural services provide humans with recreational, spiritual, and aesthetic values (Kremen and Ostfeld 2005).

For the purposes of this report, agroforestry land-use parcels are described to contain, in the most conservative estimate: (1) provisioning services such as food, clean water, sources of energy and fodder; (2) regulatory services including microclimate modification, erosion control, mitigation of desertification, carbon sequestration and pest control; and (3) supporting services namely, soil fertility improvement, biodiversity conservation and pollination in the local region. The services provided by agroforestry in contrast to larger-scale monoculture are much more robust as it would appear that agroforstry is managed to promote ecosystem services, which is not the case in conventional monoculture practises. Agroforestry is being implemented specifically to capture the ultimate functioning of every possible ecosystem service as a product secondary to the cash crop (Sileshi *et al*. 2007).

Integrating Ecosystems with Economics

The green revolution and the effect that mechanization has had on the rate of agricultural development over the past century can be seen in the hillsides, traced in the atmosphere, and in the oceans and soils. The effect of *Homo sapiens* on this planet has been sufficient to be featured in the geological record; geologists are now terming this period of geological history the Anthropocene (Crutzen 2006). The overuse of our resources has degraded our environment and up until now the value of the ecosystem services provided by the environment have not been established. Interpreting ecosystem services for commercial markets or quantifying the outputs of ecosystems in terms comparable to economic services and manufactured capital could help to establish stronger foundations for conservation and sustainable development. Particularly because interpreting ecology by way of the ecosystems services it provides is essentially in itself a discourse intelligible the business world (Boyd *et al.* 2001; Boyd and Wainger 2003).

The effects that humans have on their environment, and the likelihood that such effects will continue to degrade our environment, will slowly corrupt the ecological life support systems of the world to the point that worldwide economies of the earth will most probably grind to a halt (Pereira *et al.* 2005). It seems clear that our effects on the functioning of ecosystem services must be considered in all future transactions (Constanza *et al.* 1998; Curtis 2004; Christie *et al.* 2006).

Pioneered by conservation sciences, ecosystem service approaches to land-use assessments are being championed as a new strategy for conservation, under the hypothesis that they will broaden and deepen support for biodiversity protection (Daily 1999). Whereas traditional conservation approaches focus on setting aside land by purchasing property rights, ecosystem service approaches aim to engage a much wider range of places, people, policies, and financial resources in promotion of ecosystem service conservation activities (Daily *et al.* 2000). This is particularly important given the projected intensification of human impacts, and resultant rapid growth in population size, and insatiable human consumption rates (Vitousek *et al.* 1997).

Interestingly, it has been found that conservation projects which showcase ecosystem services attract, on average, four times as much funding through greater corporate sponsorship and use of a wider variety of finance tools than biodiversity projects (Brinkman and Hebda 2009). Furthermore, ecosystem services projects are also more likely to encompass working landscapes and the people in them. It has also been established that ecosystem services projects not only expand opportunities for conservation, but they are no projects less likely other than biodiversity projects to include or create protected areas. Moreover, they do not draw down limited financial resources for conservation but rather engage a more diverse set of funders, such as watershed conservation funds, from local municipalities and carbon offsetting funds (Bingham *et al.* 1995; Acharya 2000; Chan *et al.* 2006; Corbera *et al.* 2006; Carroll *et al.* 2008).

Ecosystem Service Valuation

The process of measuring and valuing carbon in ecosystems and valuing ecosystem services, and then integrating the valuations into the business of offset trading, is complex and evolving rapidly. The process requires technical expertise in many fields: physical and biological sciences, economic and social systems, policies and legislation (Cowling 2008; Engel *et al.* 2008). Furthermore, this technical expertise has to be applied on a range of geographic scales. The specific tools and frameworks for measuring carbon, CO_2 emissions and ecosystem components and services, are developing quickly and are yet to be standardized (Chee 2004).

Motivated mostly by risk of financial loss, businesses are seeking ecosystem valuation support so as to develop strategies to manage business risks and opportunities arising from their company's dependence and impact on ecosystems (Gatto and Leo 2000; Farber *et al.* 2006). By quantifying ecosystem relationships and expressing them in monetary terms, valuation could provide a series of measures that can, in principle, be integrated with conventional financial measures and linked directly to a company's bottom line (Robbins 2004).

The application of ecosystem valuation techniques to business concerns is, however, still at an embryonic stage. An important question, therefore, arises as to whether and how the firlf, as currently practiced, lends itself to use by the corporate sector (Baranzini *et al.* 2003). As yet there is little guidance available on this topic and mounting publications which insist that environmental economists tread cautiously for fear of undervaluing features of the environment that are priceless and overlooking the unsustainable features of capitalist economic systems (Daily 2000; Ludwig 2000; Chee 2004; Robertson 2004; McCauley 2006).

Agroforestry's Contribution To Ecosystem Services

When comparing land-use methodologies, agroforestry, more so than conventional agricultural methods, can provide significant, measurable benefits to the biosphere, potentially providing an economic argument for agricultural planning for ecosystem services and conservation (Nozzi *et al.* 2000; Kremen *et al.* 2002; Tscharntke *et al.*

2005). The discussion below is structured using other Millennium Ecosystem Assessment classifications of ecosystem services specifically described within the agroforestry context (ME 2003).

Services Categories

Provisioning Services

Provisioning services are the products obtained from ecosystems, including genetic resources, food, energy, fibre and fresh water.

Regulating Services

Regulating services are the benefits obtained from processes, including the regulation of climate, control of floods and control of some human diseases.

Microclimate Modification

The various elements of the herbaceous and shrub layer, the canopy and the emergent layer of many tropical and subtropical forests can contribute to microclimate stability with shade and windbreak. Trees specifically influence to many environmental characteristics including not just the availability of light to species growing beneath the canopy, but also air temperature, humidity, soil temperature, soil moisture content, wind and air movements throughout the region, and pest and disease complexes (Sileshi *et al.* 2007). In combination, these dynamics contribute to the resilience of the forest community itself, and also to a wide array of crops in the region. There is increasing evidence demonstrating that the enrichment of natural shade agroforestry trees, particularly with planted leguminous trees is a promising management option to increase nitrogen cycling, improve yields of crops and to keep intact high functional biodiversity (Altieri 1999; Rice and Greenberg 2000).

Indigenous communities and farmers have traditionally cultivated crops and managed the shade rich environment under the canopy of native forests (Bennett 2002). The tree litter and canopy have been documented to influence the microclimate of the forest and regionin terms of improving rainfall infiltration, soil structure and microfauna, reducing evapo-transpiration and temperature extremes, and increasing relative humidity (Saka *et al.* 1994). In agroforestry crops such as maté and coffee are grown under a canopy of shade trees that may be remnants of the original forest or may have been deliberately planted. A typical example of this is the agroforestry system of the small-scale farmers in the 'Araucaria' forests of Turvo, Parana (Cardoso-Leite *et al*. 2010). These agroforestry systems maintain not only a high biodiversity, they are an old and very sustainable way of land-use that meets several different demands. However, introduction of soyabean that is sun-tolerant, and low prices for the yerba maté as a result of the large-scale expansion of this crop, endanger the Turvo small-scale farmer.

Erosion Control and Soil Conservation

Conversion of forest to cropland has led to soil erosion, continuous loss of nutrients and degradation of 90% of the eco-region's land (McNeil 1986; Gotari 2007). Further, severity of floods, landslides and desertification are indicators that water regulating ecosystem services are stressed (Guo *et al.* 2000). According to a recent analysis, with each 10% decrease in natural forest areas in some tropical regions flood frequency increased by up to 28% (Bradshaw *et al.* 2007). On average, the annual net nutrient depletion in conventional monoculture crops can exceed 30 kg of nitrogen and 20 kg of potassium per ha (Stoorvogel and Smaling, 1990). One of the main conceptual foundations of tropical agroforestry is that trees control soil erosion and improve the soil beneath them. Researchers have developed various agroforestry practices including contour planting, contour hedges and woodlots for soil and water conservation (Sanchez 1995). For example, *Leucaena* (commonly known as Leadtrees) contour hedges have effectively controlled soil erosion on steep slopes in Malawi (Banda *et al.* 1994).

Mitigating Desertification

Between 100 to 200 million people are directly threatened by the impacts of desertification worldwide (Pattanayak and Kramer 2001). Desertification is seen as a primary eventuality to global climate change in many areas already experiencing extreme annual hydrological fluctuations (Sileshi and Mafongoya 2006). As a result of desertification, persistent reductions in the capacity of ecosystems to provide services such as food, water, and other necessities, are leading to a major decline in the wellbeing of people living in drylands. There is also mounting evidence that desertification leads to adverse impacts on adjacent non-drylands, which may include downstream flooding, impairment of global carbon sequestration capacity, and climate change (Sileshi *et al.* 2007).

Agroforestry can play a role in arid and semi-arid areas in combating desertification (Watson 2000). The thematic programme network (TPN) in Asia, Africa and Latin America is pushing agroforestry to become one of their main activities, as established in a framework on the United Nations Convention to Combat Desertification (UNCCD) implementation (Sileshi 2007). Further, Senegal has implemented two successive phases of the International Fund for Agricultural Development (IFAD) initiated agroforestry projects to combat desertification with an aim to improve soil fertility, access to water and regeneration of tree cover.

Carbon Sequestration

Dramatic change to land-use will affect the ability of that land to fix and store carbon. Below-ground carbon stocks are equally as precarious as the forest canopy (Walker and Desanker, 2004). The conversion of forest to crop, or the clearing of woodland for pasture depletes terrestrial carbon stock by reducing the vegetation carbon and soil organic carbon pool. Further, a reduction in the capacity to sequester is most notable in the loss of canopy. Agroforestry arrangements increase soil organic matter and continually store large amounts of carbon in their woody biomass (Houghton *et al.* 1993). For smallholder agroforestry systems in the tropics, potential carbon sequestration rates range from 1.5 to 3.5 Mg[2] C ha^{-1} y^{-1} (Montagnini and Nair, 2004).

A study by Albrecht and Kandji indicates that if agroforestry systems are implemented on a global scale, 1100 – 2200 Tg[3] C could be removed from the atmosphere over the next 50 years (Albrecht and Kandji, 2003). This is about 1/5 of 10.2 Pg[4]; the amount required to be removed from the atmosphere in order to reduce carbon concentration at ground level from 387 ppm to 350 ppm thus reducing ocean acidification.

Three mitigating effects of agroforestry have been identified relating to carbon sequestration. The first is the direct uptake of CO_2 in trees and soil through accumulation in live tree biomass (3-60 Mg ha^{-1}), woody mass (1-100 Mg ha^{-1}) and soil organic matter (10-50 Mg ha^{-1} and the overall protection of existing forests (up to 1000 t ha^{-1}) (Kursten

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[2] Mg: Megagrams, 10^6 g
[3] Tg: Teragra, 10^{12} g
[4] Pg: Pentagram, 10^{15} g

and Burschel 1993). Secondly, additives within an agroforestry system are not required in the same quantities as they are in conventional monocultures. A calculated reduction of about 5-360 Mg ha^{-1} of greenhouse gas emissions are offset through energy substitutions such as less mechanization, up to 100 Mg ha^{-1} through material substitutions and 1-5 Mg ha^{-1} through fertilizer reductions (Sileshi *et al.* 2007). Thirdly, agroforestry can enhance carbon sequestration by decreasing pressure on natural forests, which are a terrestrial carbon sink. This process is measured most notably in the various studies espousing the edge effects of agroforestry plantations compared to monoculture plantations of the same size and crop (McNeely 2004).

About 1600 Tg of carbon are left unfixed per year as a result of an annual deforestation rate of 17 million ha y^{-1}. Palm *et al.* suggested that one hectare of agroforestry could potentially save five hectares from deforestation and that agroforestry systems could be established on up to twoo million hectares annually. As a result, a significant portion of carbon emissions caused by deforestation, could be reduced (Palm *et al.* 1999). Agroforestry is being positioned as a viable option for enhancing terrestrial carbon sinks (Pandey 2002; Garrity 2004), which is being backed by a growing scientific consensus. Further, recent analyses conducted in Australia (Wise and Cacho 2005) and in Peru (Antle *et al*. 2007) have shown that agroforestry systems are profitable at certain levels of carbon prices (Montagnini and Nair 2004).

Control of Crop Pests

Land-use methodologies termed “conventional” due to their wide spread activity indirectly promote a reduction of biodiversity because of the expansion of continuous monoculture crops and minimal rotation. These practices have a tendency to deplete the soil, thus diminishing resilience towards pests, resulting in various pest problems (Geist 1999). The shortening of fallow periods may increase the intensity of serious pests (Sileshi *et al.* 2006). Numerous agroforestry studies show that the spread of pests, such as termites in maize, can be drastically reduced with the use of agroforestry (Sileshi *et al.* 2005). The structural complexity and plant diversity that result in a section of agroforested land have various implications on pest population dynamics.

Simply increasing diversity will not necessarily increase the stability of all agro-ecosystems though, in general, diversity is closely related to stability because structural heterogeneity and genetic diversity regulate pest populations (Sileshi *et al.* 2007).

Supporting Services

Services provided by an ecosystem which are necessary for the production of all other ecosystem services can be defined as "supporting" ecosystem services. Examples of these include biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat.

Biomass Production and Soil Fertility Improvement

Agroforestry systems fix nitrogen and produce large amounts of biomass that improve soil quality. The repeated application of tree biomass to the soil increases soil organic matter that leads to important increases in soil water retention capacity thus providing a good environment for soil microbes and plant nutrients during its decomposition. These services cannot be offered under conventional crop monocultures (Udawatta *et al.* 2008). Further benefits of agroforestry systems include enhanced availability of nutrients resulting from production and decomposition of tree biomass (Akinnifesi *et al.* 2006;

Mafongoya *et al.* 2006; Doward *et al.* 2006); uptake and utilization of nutrients from deeper layers of soils by deep rooted trees (Doward *et al.* 2008); increased activity of soil biota (Banda 1994); and improvement in water dynamics (Doward *et al.* 2008). A recent synthesis shows that these improvements in soil quality in turn result in improved agricultural productivity and increased yields of staple crops (Sileshi *et al.* 2007).

Biodiversity Conservation

Many agroforestry systems are found in places that otherwise would be covered by natural forests; the natural forests have often been replaced by agroforestry systems. Human settlements always have profound influence on forests. Thus “natural” defined as “without human influence” is a hypothetical construct, though one that has assumed mythological value among many conservationists. Biodiversity is a forest value that does not itself carry a market price. It is the foundation, however, upon which productive systems depend. The relationship between agroforestry and the wild biodiversity contained in more natural forests is a complicated one, depending on the composition and nature of the agroforestry system itself and the way it is managed. Complex agroecosystems are obviously more supportive of biodiversity than monoculture systems. Shade coffee plantations are more conducive to integrating a native canopy than open canopy monocultured coffee plantations. Further, agricultural systems using native plants tend to be more biologically diverse (McNeely 2004). Non-native plants, especially potentially invasive species, threaten biodiversity and need to be avoided. The relationship between forests, agroforestry and wild biodiversity can be made most productive through applying adaptive management approaches that incorporate ongoing research and monitoring in order to feed information back into the management system.

Maintaining diversity in approaches to management of agroforestry systems will provide humanity with the widest range of options for adapting to changing conditions (McNeely 2004). The accelerated extinction of species that often accompany agricultural development may disrupt vital ecosystem processes and services (Sileshi *et al.* 2007). Even reduction in species abundance and richness are likely to have far-reaching consequences, affecting the general stability of the ecosystem by affecting populations of agricultural pests and increasing the spread of disease (Scroth *et al.* 2000).

Pollination

Among other impacts, large-scale agricultural practices can negatively impact honeybees and native insects that provide pollination services (Priess *et al.* 2007; Kremen *et al.* 2005). There have been declines in pollination services with agricultural intensification resulting in significant reductions in both diversity and total abundance of pollinators. Restoring pollination services in areas of greatest agricultural intensity would require both reducing insecticide use and restoring native vegetation to provide nesting habitat and food sources for bees when they are not pollinating crops. These habitat features are naturally provided by agroforestry plantations (Priess *et al.* 2007).

Chapter III. The Political Ecology of yerba maté (*Ilex paraguariensis*) Agroforestry

Introduction

The fieldwork behind this thesis took me on a journey through the South Atlantic Forest to remote locations, meeting with agroforestry farmers and their families, interacting with community members and ultimately developing the perspective that small-scale agroforesters are important environmental stewards of the planet. My observations lead me to conclude that the agroforester experience is an exception to the common farming paradigm where native ecosystems and the services those ecosystems provide are maximally displaced.

The process of gathering the quantitative data for this thesis in many ways mandated that I illustrate the underlying political and social motives behind the environmental stewarding activities of the agroforester. My exposure to the social, political and economic variables which coerce farmers into unsustainable land practices revealed to me that agroforesters tend to be independent individuals representing a distinct movement, which is in opposition to large-scale agriculture. This portion of the thesis describes the sustainable worldview of some of the agroforesters in the Atlantic Forest eco-region. Their common livelihood is one that provides a basis for food production while sustaining the environment. It is a view that society is more than ever beginning to value at a time when the conventional means of food production are increasingly recognized as coupled with significant environmental degradation (Daily 1997).

Here I attempt to capture agroforestry initiatives within the framework of increased autonomy and well-being of farmers, in juxtaposition to their conventional farming counterparts by discussing the history of agricultural ‘development’ in South America during the last 50 years. My ‘measurements’ are qualitatively analyzed for themes that constitute a subversive political ecology in the agroforester, competing with the dominant infrastructure of large-scale agriculture. The dominant agriculture has been driven by pervasive forms of neo-liberalized capitalism which fundamentally defines the political ecology of most farmers nowadays in this region. I will discuss the origins and expansion of the monoculture and ‘export-oriented’ large-scale agriculture in South America, and then position the agroforester as an oppositional force to the cultural assimilation imposed by dominant, economically driven, monoculture practices.

Political Ecology and the Apolitical Thinking of Environmental Degradation

The environmental issues which plague communities and society are the result of decisions constructed beneath the weight of various influences (eg. economic, social, environmental and political). These influences are generally insensitive to ecosystem costs and they fail to consider outward environmental effects. Political ecology attempts to understand decisions resulting in environmental degradation by identifying and analyzing the motives behind decisions and also uncovering the structures of power and coercion that create the circumstances that necessitate such decisions (Adger *et al.* 2001).

Paul Robbins (2006) illustrates a few structures of power and coercion relevant to the political ecologist approach when he describes how often the motives to deforest and expand monoculture crops are influenced by commodity markets that set the prices. Fiscal pressure to adopt unsustainable but ‘efficient’ agricultural practices have great influence on the farmers’ decision to abandon more sustainable land-use methodologies. Further, the coercion of individuals whose choices lead to a degradation of their immediate environment comes from largely removed institutions that operate along linked axes of money, influence and control. “These institutions are part of an established system of power and influence that...are tractable to challenge and…can be improved”. (Robbins 2006 p 32). A political ecology approach makes a diligent effort to identify and uncover broad and far-reaching systems of power and driving forces rather than holding accountable proximate and local forces. This is the difference “between viewing ecological systems as power-laden rather than politically inert; and between taking an explicitly normative approach rather than one that claims the objectivity of disinterest” (Robbins 2006 p 33). Therefore, a political ecologist will positions participants who directly degrade environments as the failed actors in the stewardship of the environment, but also as complicit representatives of a flourishing or decaying ecological ideal which is subject to outside forces of control and coercion that threatens livelihoods and demands integration (Budds 2004). The political ecology approach is essential to understanding the institutionally systemic issues that are the prime movers for a chain reaction ultimately leading to environmental degradation. In exploring the political ecology of yerba maté farmers, it is possible to identify the economic policies that require change if environmental problems are to be mitigated.

Interrogating (A)political Ecology Assumptions in the yerba maté Farmer

In discussing the political ecology of agroforesters, it seems essential to describe what may be considered as an apolitical analysis of their circumstance. In its most basic description, an apolitical ecology fails to capture and define the broader forces that contribute to the decision-making responsible for environmental degradation. The oversight in all apolitical ecologies often appeals to simple causality concerning the harm of nature. Robbins writes that “the ability to explain beliefs and practices [apolitically] is diminished by several typical assumptions that we tend to make about human beings and their environmental behaviours" (Robbins 2007 p 23). Those behaviours are considered to be free from coercion, suggestion, power, and exploitation. Further, an apolitical ecology would in general stop short of considering the 'bigger picture' and confidently defend environmental issues as being the result of choices free from coercion. For example, "people choose to recycle or not, to commute to work or not, to use responsibly produced products or not. Similarly in this way of thinking, companies choose to make their products safer, choose methods of advertising, and choose the prices and characteristics of their products. Indeed, choice is such a fundamental assumption about human behaviour that it is hard to imagine any other way of thinking about what we do" (Robbins 2006 p 29). Applying this assumption to the analysis of the conventional yerba maté farmer would have us insisting simply that their choices in land-use management are free of coercion. However, in the case of the yerba maté farmer, and indeed at most levels of production and consumption in complex markets, that which is considered 'free choice' is actually confined by limited options. These options confer survival in a society that demands of its subjects integration within the dominant economic system. The autonomy expressed by an individual is only in the actions of production, exchange and consumption of the commodities and services which existing power structures permit to be produced, exchanged and consumed. True autonomy is much more accommodating. As Low and Gleeson write,

> "Autonomy in human being entails allowing for choice [which] cannot be imposed, or it would not be a choice. The 'expanded self' cannot in any way be legislated. All that can be done is gradually to discover a form of society in which the free choice of self becomes possible, such that the choice is not limited to the restricted, and narrowed self-picture which can alone justify the existing capitalist system" (Low and Gleeson 1998 p 200).

In the Atlantic Forest, as will be shown, the political ecology of maté farming has become unnurturing of autonomy in small-scale farming communities. In the case of maté farmers, an apolitical ecology would lack a description of the forces that produce decisions to clear forested land rather than adopt or maintain agroforestry methodologies. As Robbins (2006) notes, "clearly people who participate in intensive farming have mixed desires, and indeed feel more ambivalence about large-scale agriculture than those who do not" (p 23). Without an analysis of the political and economic structures that influence the decisions of yerba maté farmers, plotting a course for sustainable growth of the industry is unlikely. It is my intent to identify those structures which coerce farmers to abandon sustainable land-use methodologies, so as to facilitate an approach in affecting change in policy which incentivizes environmental restoration, promotes sustainable livelihoods and diminishes environmental harm.

The Atlantic Forest and the Expansion of Large-scale yerba maté Agriculture

Prior to the end of the 19th century yerba maté was not an intensive agricultural product; rather individuals would harvest from naturally growing maté trees from the wild. Since then however, land area dedicated to cultivation of monocultured maté has increased from only only a few dozen ha to 152 000 ha in the northeastern part of Argentina (Misiones and Corrientes) (Rau 2009). This is equal to approximately 280 000 tonnes of maté per year, making Argentina the largest maté producer and Brazil and Paraguay are the 2nd and 3rd largest producers, respectively. Worldwide, 290 000 ha of area harvested with a production of 874 678 tonnes of maté were reported in 2002. The overall value of maté production around the world is estimated in U.S. $1 billion in 2004 (Schnepf *et al.*

2001; Smith 2007; FAOSTAT 2009). Today over 75% of all yerba maté farmers in these three countries are small-scale farmers with less than 10 hectares of land dedicated to their crop (INYM 2009). However, about 50 producers with between 200 and 1 000 hectares of land are under cultivation are currently supplying 16% of the maté being marketed (INYM 2009). As reported by Lawson (2009), the top ten leading yerba maté brand companies purchase and control 80% of the production in Argentina. The recent development of larger-scale yerba maté agriculture in the Atlantic Forest eco-region has positioned a relatively small number of businessmen who now dominate this resource. The consequence has been the marginalization of small-scale farmers.

Michael Dove (1993a) illustrates the trend that follows when individuals live in a forest and develop a resource; they eventually initiate an economic boom with that particular resource. According to Dove, because of the rising popularity and demand for the resource, external political and economic interests are attracted to the resource and assume control of it. Dove observes that the new dynamic serves to marginalize the people that live in the forest under the grip of individuals with much more political and economical clout. “The problem is not that the forest peoples are poor, but that they are politically weak (and the problem is not that the forest is environmentally fragile, but that it is politically marginal)” (Dove 1993a p 21). This view is applicable to the development of yerba maté cultivation. As noted by Lawson (2009), [yerba maté] was originally harvested by the Guaraní Indians from naturally occurring yerba maté groves (2009). The resource then began to be controlled through power structures of colonialism. This resulted in expansion of the production into plantations in the understory of the forest. The plantation agriculture brought with it land consolidation and mechanization. Yerba maté markets then became heavily controlled by the leading yerba maté companies in Argentina and Brazil (Lawson 2009). The resulting political economy has been that small-scale yerba maté farmers profit minimally after the cost of production (Netting 1993). The small-scale farmer is increasingly marginalized, perpetuating the trend of their migration, often with their families, to the city. The alterative appears to be their sequestration as labourers into other forms of agriculture.

The Effects of the Green Revolution on the yerba maté Industry

The implementation of green revolution technological innovations was promoted within the yerba maté industry in Argentina and Brazil to increase yields after World War II. Changes in government agricultural and economic policies lead to changes in the local management of agricultural systems. As a result environmental degradation and income inequalities increased (Peluso 1996). In Argentina, policies evolved to promote high-yielding commercial production techniques which reflected a comprehensive green revolution paradigm shift in agricultural production. Dove and Kammen (1997) describe this shift as a movement from moral ecology [such as swidden cultivation systems which is temporarily clearing an area for cultivation (Dove 1983)] to an immoral ecology, promoting a paradigm of unsustainable resource use.

> “The real significance of the green revolution is conceptual, not technological, that its real failing is ideological, and that its long-term viability is put in doubt by the immorality of its ecology” (Dove and Kammen 1997 pp 91-92).

The technology that came with the green revolution was manufactured without considering the whole farm system. The mechanization of farm practices is therefore leading to environmental degradation. The change in the way that farmers engaged with their land and resources shifted from emphasis on the sustainability of a farm system to a logic of ‘extraction from a farm system’ (Lawson 2009). Thus, farmers become dependent on technology and nonrenewable inputs such as fertilizers, herbicides, and fuel for tractors.

The Effects of the Green Revolution on Landscapes

Large-scale monoculture imposes a cultural perception of needing to keep the fields “clean” and “pretty” (Lawson 2009). This exported perception results in yerba maté fields suffering from erosion, nutrient loss and run off. The problem is that agricultural customs developed in North America are being misapplied in the ecological and social context of the South.

> “Problems sometimes arise because this technology does not suit tropical conditions. Compaction of the soil by the use of heavy machinery, deep plowing causing moisture loss and exposure of humus to the sun and to leaching by heavy rainfall,…line planting and low foliage protection are some of the problems that are caused by this technology” (Blaikie 1985 p 143).

Monoculture systems are widely adopted as the most productive, and most efficient means of meeting higher density demands. In these monoculture systems, plant diversity, shade and soil integrity are lost and along with them those plants that might harbour the natural enemies of pests. Further, the effects of drought are amplified and erosion becomes a constant problem in monoculture systems (Gortari and Oviedo 2001). Because of the shift in ecology of plantations, pesticide, herbicide and fertilizer use increases in order to combat pests that become pervasive in monoculture plantations and to maintain nitrogen levels (Gortari and Oviedo 2001). With government subsidies and agronomic research promoting higher-yields, monocultures and agricultural additives, the management techniques of small farmers change to incorporate the paradigms of the green revolution (Key and Runsten 1999). With this paradigm shift, the ecology of the yerba maté agricultural system changes as well, leading ultimately to degradation of the Atlantic Forest landscape.

The Effects of Neo-liberalization on the yerba maté Industry

To meet the terms dictated by financing organizations such as the World Bank and the International Monetary Fund, the governments of Brazil and Argentina were required to institute policies which in effect deregulated the agricultural sector (Rosin 2004). Regulated land distribution in the yerba maté sector was eliminated allowing farmers to expand their quotas. Thus, a combination of powerful economic and political interests eliminated regulation in this industry. The movement to neo-liberal policies in the early 1990s was symptomatic of the era as South American governments shifted away from reliance on commodity regulations to self- regulating market mechanisms:

> "Between World War II and the 1990s, governments had implemented state intervention, regulation, protectionism, state-owned enterprises, and collective production systems in well-intentioned attempts to implement economies that served national interests and to foster greater internal coherence by incorporating different sectors through protection policies and special-interest programs. But these efforts were critiqued because they fostered systems that were inefficient because they were not competitive. Instead neo- liberalism placed its faith in the engines of growth that individual initiative acting in free and self-regulating markets could bring about. Neo-liberal governments deregulated markets, eliminated subsidies, dismantled development programs, and privatized state enterprises" (Mayer 2002 pp 313-314).

The new paradigm supported an economic environment characterized by market competition and increased productivity in a system in which producers were expected to respond to market signals through a process of self-regulation. In classical economics, economic units are designated as either producers or consumers (Mayer 2002). However, farming households do not fit solely into one or the other of these categories (Mayer 2002). As Lawson (2009) puts it, the small farmer is unable to partake in an economy of scale and is therefore seen as inefficient by those producers who are deemed efficient, highly productive and more successful within the neo-liberal paradigm. The neo-liberals declare the peasant and small-scale household as uneconomic and as disappearing as players in the economic sector. According to the ideology of modernization, as Mayer characterizes this view, “Peasants are technologically backward and doomed by the forces of modernization and industrialization” (Mayer 2002 p 314). Policy shifts under neo-liberalism revert to these tendencies. As a result, the small-scale farmers are left in a ‘redundant’ and precarious position, this being overtly the result of the expansion of the neo-liberal market structure, forcing producers to respond to competition or to withdraw completely from the market.

> “Competition, quality, and efficiency encompass specific meanings as concepts of competitive markets. These meanings are formalized at the theoretical level as means to legitimize economic policies that rely on the assumed uniform rationality of economic actors in markets that objectively reward ‘correct’ response to market signals” (Rosin 2004 p 346).

In trying to meet the newly sanctified demands of the market, farmers are forced to increase their yields, decrease their input or unit costs and adopt agricultural techniques that often force them to abandon their agroecosystem’s sustainability. After the implementation of neo-liberal policies within the yerba maté sector, the land area devoted to maté expanded, leading to an oversupply of yerba mate, and maté began to flood the market. Unfortunately, the increase in supply was not countered with an expansion in demand for yerba maté (Gortari 1998). Without a increasing demand, the increasing production, and resultant overstocks of maté were unsellable. This resulted in a crisis within the yerba maté market and a plummeting of prices. As yerba maté prices fell, the production became less attractive for small-scale farmers, especially for their children to continue with maté agriculture. Farmers encouraged their children to earn an education and to find work through other opportunities. As the production of yerba maté does not provide an economic incentive for youth to stay on their farm, there has been an exodus of the youth from the farms to the city (Pasinato 2003). “What happens is that the young people all go to the city to look for work alternatives. The field becomes a place that does not provide a future for the youth” (Lawson 2009). Without a livelihood to carry on with on the farm and no future to be seen, the youth move on, thus severing the cultural legacy between generations. This creates a change in the social structure of farm families as they begin to disperse all throughout the towns and provinces (Lawson 2009). In lacking governmental support for the small-scale farmers in the form of market regulation, the social and ecological effects of neo-liberal strategies have been both the marginalization of the livelihoods and legacy of the small-scale farmer and degradation and the landscape of the Interior Atlantic Forest ecosystem.

A Positive Shift in yerba maté Culture

The increasing marginalization of small-scale farmers has resulted in some resistance. Small and medium-sized yerba maté farmers initiated a series of social movements in order to revitalize their pursuit of sustainable livelihoods (Yamada and Ghoiz 2002; Halberg 2006). In the 1990’s, small-scale farmers began to unite in response to the intense competition within the market: Rosin (2004) writes that otherwise competing economic agents within a region (much like yerba maté processors) often act cooperatively in response to external competition. In the last decade, many yerba maté farmers have organized into cooperatives, often being led by individuals initiating social movements. Within these movements, an appeal to solidarity, equality and integrity has benefited those cooperative members in the face of strong institutional structures and economic power within the yerba maté market. As cooperatives began to find their voice within the political ecology, they expanded their ethic to work in solidarity with other social movements in order to create an alternative market that would overcome competition with large yerba maté firms to support the small producers (Lawson 2006). Cooperatives have also worked to diversify the crops of rural farmers who are included in their membership (Jarosz 2000). This strategy recognized that dependence on one crop makes small farmers economically vulnerable to fluctuating market conditions. By diversifying the agricultural systems, the cooperatives have been instituting a land-use methodology that existed prior to the influence of the green revolution and the following neo-liberalization of economic and agricultural policy. The hope is that yerba maté farmers might be able to persist through changing commodity prices by economically relying on more than one crop, and be able to maintain their families and their farms. Because of the alternative political economy offered by cooperatives, small and medium sized farmers are weathering the neo-liberal storm and its long-term effects.

Restoring the Landscape

Farmers have been empowered to adapt their management techniques to their environment and to the competitive marketplace on behalf of the cooperative movement. Their social perceptions and knowledge of agricultural management systems has been expanded as a result of the feedback, workshops and community building that various cooperatives provide and encourage.

These partnerships and connections are being spread under the idea that “the ability and willingness of local producers to adopt alternate means of production and to realize efficiency gains differ widely depending on local perceptions of the potential and viability of cultivating the agroecosystem" (Rosin 2004 p 82).

Small-scale farmers have identified that intercropping their yerba maté trees with other native trees and cash crops within an agroforestry system is a better solution for diversifying their income. Furthermore, small-scale farmers are finding that agroforestry better reflects their preferences in their relationship with the land; they are assuming their preferred identities within the political economy. In Argentina, Brazil and Paraguay, the development of small and mediums-scale farms diversifying their agricultural systems through intercropping of both exotic and native trees species is a direct consequence of the cooperative movements arising in opposition to the effects of neo-liberalism. Farmers who are mostly members of cooperatives are beginning to explore the implementation of yerba maté agroforestry systems with native trees.

The 13 agroforestry farms that I visited in Paraguay, Brazil and Argentina had similar characteristics of planting their yerba maté under the shade of a native tree canopy. Farmers had thinned down the underbrush and partially the sub-canopy in order to plant and increase the concentration of yerba maté and other medicinal plants. As Lawson discovered, if the yerba maté receives more than 20% shade, yields will begin to decrease (2009). Therefore, farmers must control the amount of shade and its effects when incorporating native trees into their yerba maté agroforestry system. It would appear that for many farmers there has been a shift in the way they see the yerba maté production and as a result, farmers have started planting native trees within their yerba maté fields. This paradigm shift is ongoing, but noticeable in the agroforestry communities I visited.

Cooperatives have been successful in helping to establish niche markets for shade-grown and organic yerba maté which in turn is supporting a movement toward more sustainable agroforestry production (Rosset 2000). As Lawson found this transition entails the strengthening of the ecological relationship between farmer and forest and reconceptualization of the yerba maté agricultural system (2009).

The Future for the Maté Industry as a Conservation Strategy for the Atlantic Forest

Sustainable yerba maté agriculture was distorted by a change in the political ecology of the Atlantic Forest and it has historically marginalized small-scale yerba maté farmers. The isolation from the market that farmers experienced forced them to implement agricultural techniques and technologies that resulted in the degradation of this region, soil and landscapes. Promoted by state subsidies, agricultural extension agents and policy change, the green revolution paradigm of agricultural production had farmers adopt the use of costly technologies such as tractors and herbicides. These new land-use techniques were misapplied to the region and industry and had serious environmental and even health consequences in the communities they were forced upon. Still responding to the structural crisis of oversupply which has come as a result of neo-liberal economic policies that were instituted in the early 90's, smaller farmers who continued farming diminished their investments in maintaining regimes and conservation strategies in order to cope with the lower and constantly fluctuating market prices of mate. Though there has been some on-farm conservation by the state in Argentina, Brazil and Paraguay, farmers are mainly taking it upon themselves to implement more sustainable management techniques. Primarily, agroforestry with native canopies is being touted as the ideal agricultural system that emphasizes the cultivation of cover crops and the implementation of a diversified agroforestry system (Lawson 2009).

The future of small-scale yerba maté farmers seems to partly lie in their ability to work together, and in doing so cultivate and preserve an environment which reflects their ideals, their customs and their respect for nature. Surely, the adoption by the state of a coherent agrarian policy would serve to institutionally support the small-scale yerba maté farmer as an agroforester and aid in moving the rest of the industry towards implementing the restoration of landscapes. The state plays a significant role in affecting the agricultural systems and management techniques of small-scale farmers (Rosin 2004). To begin a shift towards environmentally sustainable yerba maté production that supports the livelihoods of the farmers, the state must foster policies that create economic and social conditions that foment agricultural sustainability:

> "It is unrealistic to claim productive, ecological management without making it compatible with 'social ecology'. The transition to socially and environmentally sustainable agricultural models is inextricably linked to the development of a state policy that is aimed at: improving the quality of life of producers; ensuring the future commercialization of the product through the development of new markets; guaranteeing the right of consumers to safe and healthy food; reducing unemployment and rural and urban marginalization; and preserving the environment from contamination (Gortari and Oviedo 2001 p 312).

For a selective few yerba maté farmers the organic market is providing incentives for the adoption of agricultural techniques that do not use herbicides and pesticides. However, there is an expense in certifying products as organic. The creation of an organic market necessitates that the farmer not only implements sustainable agroforestry but above all, restricts the use of herbicides and pesticides. This requires an even deeper paradigm shift in the way that farmers conceptualize their agroforestry systems.

Due to the current political economy being the cause of marginalization of small-scale farmers and degradation of the Atlantic Forest eco-region, an institutional change in state policies both economic and environmental is needed. New policies must support small-scale farmers by ensuring fair access to markets and create programmes which financially support the environmental stewarding that the farmer is willing do. To create this paradigm shift, the state should widen its programs to work with farmers to implement sustainable agroforestry techniques and reward those techniques already being adopted (Lawson 2009). Included in this process would be payment for ecosystem services programs, offsetting the environmental harm induced by large-scale agriculturalists and supporting sustainable agroforestry practices.

Chapter IV Exploring Ecosystem Services in Relation to yerba maté (*Ilex paraguariensis)* Agroforestry.

Introduction

Economics of Ecosystem Services

Interpreting local ecology through the ecosystem services that nature offers is becoming a common quantitative tool for conservationists interested in carbon or biodiversity offsetting initiatives (Costanza *et al.* 1998; Altieri 1999; Turner *et al.* 2000; Boyd and Wainger 2003; Costanza *et al.* 2004; Robertson 2004; Pereira *et al.* 2005; Chan *et al.* 2006; Reid 2006; Boyd 2007; Engel 2008). An estimated four-times more conservation funding is available to projects incorporating ecosystem services than those focusing simply on the preservation of biodiversity alone (Gatto and Leo 2000; Corbera *et al.* 2006; Christensen and Lawyers 2007; Brinkman and Hebda 2009). This rising popularity of ecosystem services terminology requires the development of standardized and effective methodologies of quantifying those ecosystem services most likely to have a market (Boyd and Banzaf 2007). Such ecosystem services include carbon, watershed maintenance and biodiversity credits.

As world markets begin to feel the impact of environmental degradation, a valuation system for the ecosystem services that have historically been ignored in economics will be a critical first step to improving the relationship our economic system has with the environment. Apprehension and opposition certainly exist in the conservation community regarding the application of dollar amounts to nature (see Daily, 2000; McCauley 2006) and many economists argue that such an endeavour is impossibly confounding (Bingham *et al.* 1995; Engel 2008). Nonetheless as we transition towards recognizing intrinsic value of intact ecosystems, the contemporary capitalist model necessitates that we articulate our drive to conserve in the accepted monetary language. Despite the fact that as ecosystem service valuations are opposed by some of the most credible conservationists and environmentalists, it can be argued that the transition of world economies towards a 'greener' future begins with interpreting ecology using the language of finance and cost. Such an application will ultimately shift the focus of value towards an ecosystem-based understanding that integrates the cost of human impact inherent in all operations into the operational design (see Salzman *et al.* 2001; Baranzini 2003; Salzman *et al.* 2003).

Pilot projects are already being launched around the world with the aim of quantifying ecosystem service functions (Reid 2006). For example, there are currently dozens of carbon forest projects that aim to tap into the large and rapidly growing carbon markets of the world (Brinkman and Hebda 2008; Farber *et al.* 2006). Further development of the ecosystem marketplace can be seen in the expansion of biodiversity measurements and the application of credits to encourage offsetting initiatives (Acharya 2000; Lupi 2002; Balvanera 2006; Nunes 2001). Investments into watershed maintenance rather than water treatment facilities are also examples of the more practical solutions that some localities have adopted (Pattanayak 2001). Such practices add to the momentum towards conceptualizing an ecosystem marketplace. Arguably any possible product of an ecosystem that can provide benefits to humans as a result of its intact existence represents a possible market value in the ecosystem marketplace. There is a strong demand for standardization of the methodologies for measuring such products (Boyd and Banzhaf 2007). Identifying the monetary worth is another task in itself, but as the markets are already suggesting, payments for ecosystem services are arising despite the aggressive opposition of some economists.

Biodiversity Indices, Carbon and Vegetation Coverage

The three most marketable ecosystem services in a land parcel were identified by the amount of money currently navigating the market in promotion of those services. The carbon market sets the stage for forestry conservation projects that aim to finance their creation from registering the measurable carbon storing that their conservation perpetuates (Kroeger and Casey 2007). Further biodiversity conservation projects are now being sourced in the ever-growing carbon market (Carrol *et al.* 2008). Finally, the financing of watershed conservation projects is ultimately growing in popularity. Watershed conservation projects are often proving to be cheaper than the construction and maintenance of water treatment centers to maintain cleanliness in an over-developed watershed (Stanton *et al.* 2009). These market-driven mechanisms are extremely valuable sources of conservation financing and show great potential for helping environmentally destructive industries in financing their transition towards environmentally more sensitive land-use methodologies and supporting initiatives such as agroforestry in their conservation efforts with payments for ecosystem services.

To accomplish these goals, baseline measurements of ecosystem service provisioning in conventional monoculture crops compared to agroforestry parcels of the same crop is a necessary primer for establishing quantified differences (Kremen and Ostefeld 2005). Integrating biophysical features of an cosystem service with valuation data is then important for creating a discussion around the ideal valuation that these services should receive (Turner *et al.* 2000; Villa *et al.* 2002; Troy and Wilson, 2005). This Chapter will give agroforesters a means of illustrating their stewarding practices in a language that is relevant to markets. Using three case studies involving yerba maté, I will illustrate both the rigorous provisions of three ecosystem services in each land parcel and the value as indicated by current market prices.

Biodiversity

Biodiversity, as measured by species evenness and richness, itself represents an ecosystem service in that it directly supports features necessary to the resilience of an agricultural landscape such as pollination, biological control and biomass accumulation (Grime 1998; Cardinale *et al.* 2007; Goldman *et al.* 2008). Because agricultural land-use methodologies range in the effects they have on terrestrial areas, choosing and managing the least invasive land-use methodology can be critical for successful conservation efforts (Tscharntke *et al.* 2005). Traditionally, agricultural land-use and biodiversity conservation have been viewed as incompatible. Ecologists and conservationists often focus on pristine or threatened and disturbed habitats to save the last remnants of wild nature. Only recently has there been an increasing recognition that such a conservation focus is of limited value (Elmqvist *et al.* 2003; Schroth *et al.* 2004) and that the importance of population exchanges among areas of different disturbance regimes and among early and late successional habitats needs to be acknowledged (Cullen *et al.* 2001). Intensified land-use in agriculture and forestry is irrefutably the direct cause of global change and biodiversity loss, but low-intensity land-use systems may provide environmental beneficial elements to large-scale conservation programs. Agricultural lands have received less attention than forests with respect to measuring biodiversity value or conservation success. Indicators of biodiversity within agricultural land are still partially developed. The European Environmental Agency and various other EU research programmes are currently attempting to develop agricultural sustainability indicators, including some on-farm biodiversity, with the aim of implementation in due course (Duelli and Obrist 2003).

Compared with forestry, there is generally less agreement about how the links between biodiversity and agriculture might be enriched, but much of the emphasis (where it occurs at all) goes towards measuring the detrimental impacts of agriculture on surrounding habitats (for instance through soil erosion, pollution run-off and edge effects) rather than looking at the biodiversity within specific systems (Duelli and Obrist 2009). Dudley argues that this is an important shortcoming because, while the most intensive farming systems support little biodiversity, as shown through a serious decline in species previously associated with farmland (2005), many traditional agricultural systems, rangelands and various forms of ecologically based agriculture can support a proportion of wild species. Residual natural and semi-natural vegetation and structures on farmlands, if not overly affected by inappropriate management, can provide important reservoirs of biodiversity and corridors and stepping stones between natural habitats, including protected areas. Low-intensity farming systems can mimic some of the attributes of more natural systems and can be associated with greater species diversity than other land-uses competing with natural ecosystems (Dudley *et al.* 2005)

In this study, biodiversity measures were the most rigorous field assessments, because in some cases they required counts of flora species in entire parcels of land. This information was used to determine the overall flora biodiversity conserved on the properties relative to bordering sections of land. Data was then interpreted using the

Shannon index the most commonly used tool for estimations of richness and evenness (van Jaarsveld 1998).

Watershed Maintenance and Remote Sensing

Soil erosion is one of the most important problems that human beings face. Many nations pay attention to soil erosion factors which influence water conservation (Blaikie 1985). Many new methods and techniques of monitoring soil erosion are being applied to understand soil erosions. Geospatial Information Systems (GIS) and remote sensing offer powerful measuring capabilities using vegetation cover analysis as an indicator for soil erosion potential (Running *et al.* 1995; Roy and Wilson 2006). Across the planet, vegetation cover change is one of the main indications of ecological and environmental change. As cover is reduced, various services that intact soil and landscapes provide are affected, including the infrastructure that prevents erosion and nutrient loss (El-Hassanin *et al.* 1993). For the purpose of this study, vegetation cover analysis was interpreted as an indicator for watershed maintenance. Vegetation cover is defined as the percentage of cover in a defined area, which, using remote sensing, is simply the shade of the pixel which corresponds to an establish percent cover value. Recent satellite imagery of field sites will be analyzed to estimate the extent of vegetation cover.

Regional Profiles

All locations being showcased here are representative for the primary regions of productivity and concentrated development in the yerba maté industry in South America. Figure 3 shows field site locations relative to each other. In the Southern Atlantic Forest the remnants of forests are fewer and the distances between native forested areas increases every year. The demand for greater agriculture output has very rapidly transitioned areas to intensive large-scale agriculture in the last 30 years. Very few small farmers are maintaining sustainable land-use methodologies, and they are marginalized by the large-scale agricultural pressures which construct a coercive political ecology complicit in environmental degradation.

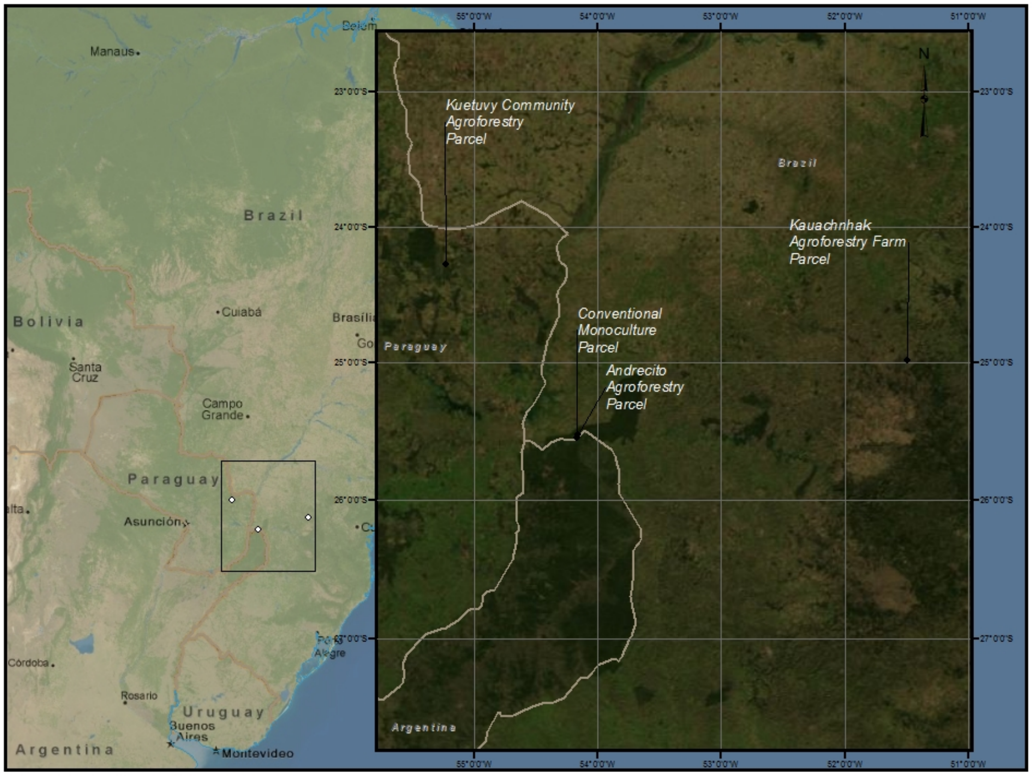

Figure 3. BING map's aerial imagery of field site locations in Brazil, Argentina and Paraguay.

Case Study #1: Turvo, Parana, Brazil

The municipality of Turvo is located in the central region of the state of Parana. The region itself is blanketed with some of the largest *Araucaria angustifolia* reserves in Brazil. The population of the region is approximately 16,000 inhabitants (Mikhailova and Mulbeir 2008). The region has an average altitude of 1040 m and its climate is considered to be Oceanic subtropical.

I initiate these field studies were initiated in early February, 2010. The small town of Turvo is located between Curitiba, the capital of the state of Parana, and Guarapuava. Intermittently paved streets stained red with the iron rich soil of the region lead into the town center. A smokey horizon and aroma of burning vegetation is dominant around dusk in this community. There are mules with carts carrying young farmers between fields, large Volkswagen 2-tonne trucks carrying propane tanks and green harvests back and forth between processing facilities and farms growing soybean and mate. The main street of the city is nestled in a depression and buildings stud the hillside that inclines up into a horizon of *Araucaria* trees. A 30-minute drive outside the city centre is the Kauchnhaki farm (Figure 4).

Across cattle farms and sprawling soyabean plantations, a red stained dirt road leads into a densely forested region. Small homesteads and vegetable gardens begin to appear through the tree line. The occasional tobacco crops being grown in full sun exposure were noticeable because they result in open-lit, green fields that stand out against the darker forest backdrop. Farmers dry their tobacco on-site and in most cases ship it to Cuba for cigar production. Between these open-sun crops the land is for the most part forested, and small homes stand behind the native trees. Each forested land parcel shares a common theme. They are noticeably thinned in their understory with the concentration in planted species slightly diminished as a result of the planting and cultivation of *Ilex paraguariensis*, or yerba maté. The region combats the encroaching and expanding industry of soyabean (*Glycine max* (L.) Merr) with a well-established yerba maté artisanal, or small-scale farming infrastructure (Rodigheri 1997). The Kauchnhaki farm is called Saudades, which in Portuguese means “longing.” The farm is owned by Christina and Reginal who live there with their three children in their late teens. On the farm there are 50 chickens, 6 cows, 6 pigs, 4 dogs and 2 horses. Of their 16 hectares only two hectares lack full canopy coverage. In this open area a vegetable garden is cultivated and there is area set aside for grazing land. The remaining area, which has canopy coverage, has yerba maté and other medicinal plants cultivated in the understory. They grow Cow's foot (*Bauhinia forticata* Link), globe artichoke (*Cynara scolymus* L), Rosemary, (*Rosmarinus officinalis* L), lemongrass (*Cymbopogan citratus* (DC.), Staph, 1906), Lemon balm or Melissa (*Melissa officinalis* L), oregano (*Origanum majorana* L), Carqueja (*Baccharis genistelloides* (Lam), Pers.), and most notably, and with highest density sometimes reaching 700 stems per hectare, yerba maté (*Ilex paraguariensis* A. St. Hil.).

According to the Institute of Brazilian Geography and Statistics (IBGE), the state of Parana is responsible for 70% of the national production of yerba maté. International companies will pay 50% more than the average national market price to import yerba maté. Yerba maté is typically imported in 50 kg bags and sold in 0.5 – 2.5 kg bags for around \$12 and \$53 dollars respectively (Rodigheri *et al.* 2005).

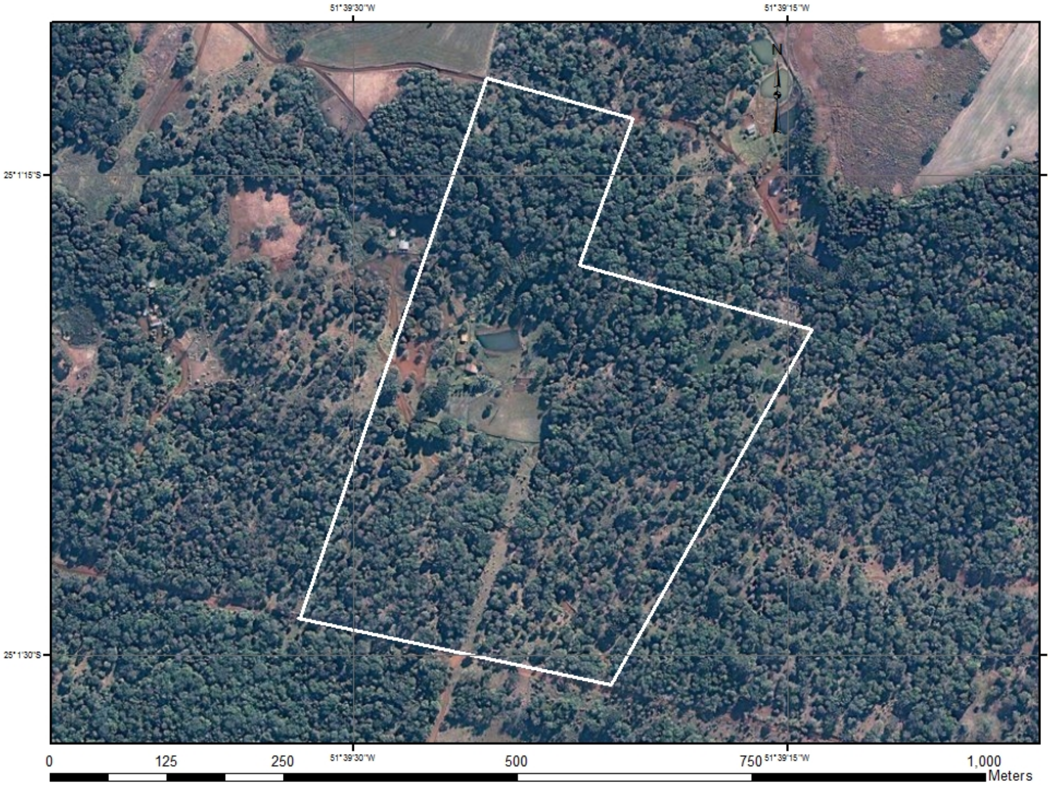

Figure 4. BING map's aerial imagery of Kauchnhaki medicinal plant and yerbe maté agroforested farm. Municipality: Turvo, Microregion: Guarapuava, State: Parana, Country: Brazil. Latitude: 25°1'22.3404"S; Longitude: 51°39'23.7918"W, Size: 15.9 Hectares. Date: April 2010.

Comandante Andrecito, Misiones, Argentina

Comandante Andrecito is a small community east of Puerto Iguacu on the Brazilian Argentine border in the Province of Misiones. As seen from aerial photos, the province boasts some of the most concentrated forest cover in all of Argentina, and certainly in most of the Atlantic Forest eco-region. The regional population is approximately 14,000 inhabitants according to the National Institute of Statistics and Census (INDEC) in 2001. Andrecito, the colloquial name for the city, has a main street that runs alongside a river, with shops and buildings that sit atop the riverbank; their entrances accessing the street. Directly surrounding the community is cleared land for cattle. The town streets jet into red roadways which lead in all directions out of the city into the surrounding well-forested land, common in this part of Argentina. Our destination is a small section of land north of Andrecito bordering the Iguaco River. The site is an 84 hectare parcel, formerly a farm where yerba maté was cultivated in a monoculture, but which was abandoned 6 years ago, allowing the prevailing native species to re-establish themselves within the farm's borders (Figure 4). It is owned by Guayaki, a company with the specific policy of restoring lands to productive agroforestry. Guayaki is replanting the fields with native trees grown from seeds that were recovered from the surrounding forest to intercrop with the overgrown yerba maté. They are restoring the native forest and integrating its biodiversity into their agriculture in what is a pioneering effort in this region. They term their efforts 'market-driven restoration' and market their product in North America with this strategy as a selling feature. Figure 6 is an aerial photo of a nearby conventional yerbe maté monoculture where I preformed various measurements in order to provide comparisons values.

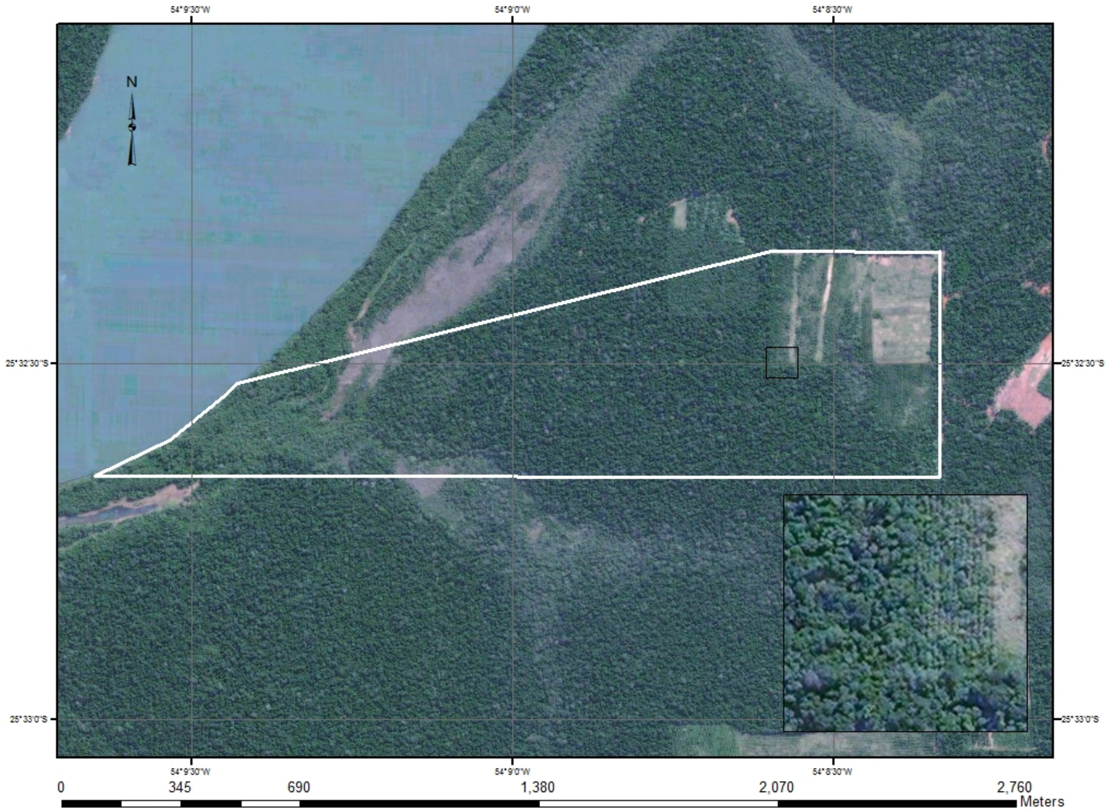

Figure 5. BING maps aerial imagery of Andrecito property, Location: Comandante Andrecito, Province: Misiones, Country: Argentina Latitude: 54°08'43.95"S; Longitude: 25°37'28.81"W. Size: 84 Hectares. Date: April 2010

Figure 6. BING maps aerial imagery of Conventional yerba maté Farm, Municipality: Comandante Andrecito, Province: Misiones, Country: Argentina. Latitude: 54°07'17.91"W; Longitude: 25°31'17.75"S, Size: 112.5 Hectares. Date: April 2010.

Finca #470, Mbaracayú, Ygatimi, Canindeyu, Paraguay.

The low hills of eastern Paraguay's Mbaracayú region are covered by some of the last remaining extensive subtropical forests in South America. The region is defined by the watershed and drainage basin of the Jejuí River. A diverse biome of temperate and tropical zones transitions in the western escarpment of Brazil's Paraná Plateau. In addition to its flora, the region is important for the diversity of its fauna, including unstudied species and endangered mammals, such as the giant armadillo (*Priodontes maximus*) (Hill and Padwe 2000).

Ranching, commercial agriculture, and national colonization schemes promote deforestation to accommodate the government's desire for foreign exchange and the rural dwellers need for land. In the last 20 years, over a million hectares of forest have been cleared for intensive agriculture, roads have been constructed in even the most isolated regions, and cotton (*Gossypium hirsutum* L.) and soybeans (*Glycine max* (L.) Merr.) have become the new mainstays of Paraguay's expanding export economy (Yahnke and Fox 1998).

Responses to environmental concerns in Paraguay are being met with conservation programmes aimed at protecting biosphere reserves (Schwartzman *et al.* 2000; Jacobson 2003). A main strategy within many governmental programmes is to "demonstrate sites of harmonious, longstanding relationships between man and the natural environments" (Batisse 1985 p 21). As a result, conservation strategies of biosphere reserves are expanding their sights in order to understand the rights and traditions of indigenous groups that are affected by environmental destruction. Various studies indicate that the indigenous people manage the so-called "natural" diversity of flora and fauna (Jacobsen 2003). Efforts are therefore being made to incorporate indigenous design and management methodologies into the biosphere reserves. Just to the south of the Mbaracayú Bioreserve is a 4 400 hectare reserve called Finca #470. It was purchased by the Paraguayan government and is home to approximately 200 Ache Indians who were relocated to this area in the early 1980's (Hill and Hurtado 1996). The community is called Kuetuvy and the people belong to the Ache Indian tribe, a traditional hunter-gatherer group. The Ache community manages the property as an indigenous reserve. Large areas of the forest support sustainable hunting, the collection of edible fruits and insects, growth of medicinal plants, and further, the enrichment of the forest with valuable native tree species such as yerba maté and minimal impact forestry with wood destined for internal consumption to construct houses, schools, clinics etc. A 16 hectare parcel has been established North of the community to house an agroforestry parcel (Figure 7). The parcel has been sectioned into four 4-hectare quadrants over a 5-year period, one quadrant being integrated into agroforestry every year over the 5-year period.

Despite many legal battles in a process towards transferring the land titles of Finca #470 to the Indigenous Ache community, no agreement has been reached, leaving the Ache residents unable to enact full autonomy concerning their management decision. Because of land ownership ambiguity, the Ache residents have fought battles against illegal loggers, speculators and so-called “landless peasants,” without legislative confirmation of their responsibilities towards the land and stewardship of the land.

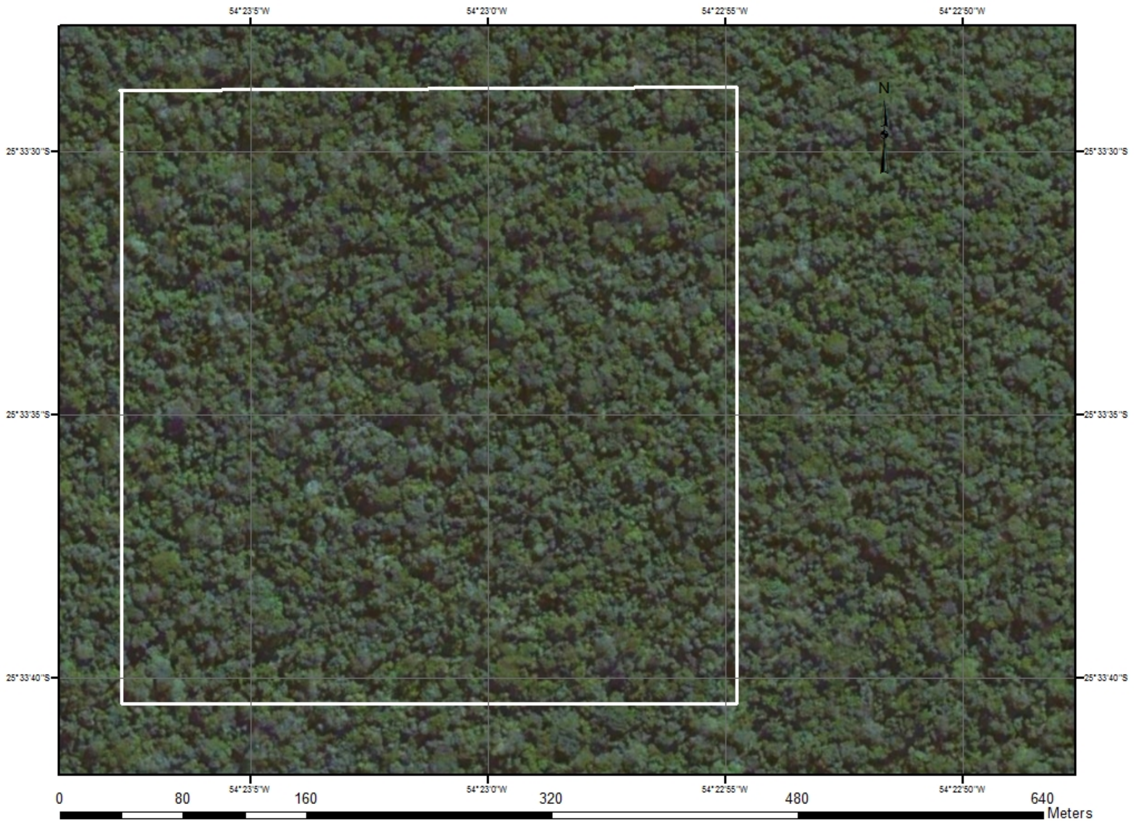

Figure 7. BING maps aerial imagery of Kuetuvy Aché Yerba maté agroforested parcel. Location: Finca #470 Region: Mbaracayú District: Ygatimi Department: Canindeyu Country: Paraguay Latitude: 24°16'26.62"S; Longitude: 55°25'02.97"W, Size: 16.1 Hectares. Date: April 2010

Methods

Field measurements and remote sensing were used to estimate ecosystem service functioning in three agroforestry parcels of the yerba maté industry in the Atlantic Forest eco-region of South America. The sites were visited between February and April of 2010. All three sites were in different stages of ideal agroforestry characteristics. While visiting each site, the land usage perimeter was defined using GPS tracking. Any specific field assessment was recorded by hand and tracked using Garmin's Oregon 300 handheld GPS. This information was later transformed into ArcGIS, a GIS spatial analysis software for Remote Sensing measurements of vegetation coverage. Maps were then made of each field site from satellite imagery. Evaluations in agroforestry yerba maté parcels were compared to baseline measurements in nearby conventional yerba maté plantations and local primary forested land. Each agroforestry plantation was geographically and environmentally unique. In Comandante Andrecito, Argentina, the agroforestry parcel was partially bordered by a sprawling monoculture plantation of yerba maté, while the Iguacu river shared part of the parcel border. The agroforestry parcels in Brazil and Argentina were entirely bordered by either forested or bordering agroforested land. Ecologically, all three sites were visually distinct and they were hundreds of kilometers apart from each other making their flora and fauna community structures unique (Press 2010). In the temperate forests around Turvo, the landscapes were articulated with the distinctive *Araucaria* (*Araucaria* Juss.) tree, this tree predominantly growing up through the canopy and shadowing other species. While in Argentina, which is more tropical in climate, palm (*Arecaceae* Schultz. Sch.) and acacia (*Acacia* Mill.) were most notable.

Biodiversity

Species diversity can be used to characterize the health of ecosystems. Several different indices have been developed to express diversity. There are two primary aspects of diversity when using the Shannon's index as an interpretive measure: species richness and species evenness. Species richness is a measure of the number of species per sample, while species evenness indicates the number of individuals per species in a sample. The evenness of a particular species increases as species become more evenly distributed such that the maximum evenness is attained when all the species are equally abundant (Dejong 1975; Jarvis 2008).

Several indices displaying species diversity are used throughout the literature that is available on biological diversity and ecological monitoring. Shannon’s index is the most frequently used index of species diversity and is a derivative of Shannon’s Information Theory of Communication; the uncertainty is measured by the Shannon Function “H.” Shannon’s Function H is the measure corresponding to the entropy concept defined by:

$$H = -\sum_{i=1}^{n} pi \ln pi$$

(Spielburg and Fedor, 2003)

In this index, *pi* will be estimated by *ni*/N, where *ni* is the number of individuals of one species and n is the total number of individuals in the community. Logarithms to the base, 2, e and 10 can be used in the equation. H is a measure of uncertainty, for example, if an individual is picked at random from an infinite population, H is a measure of how uncertain one is that the individual picked will be of a particular species. H is therefore thought to be an intuitive measure of species diversity or richness since diversity uncertainty will increase as species diversity in a population increases (DeJong, 1975). The value indicating richness reaches its maximum at 3.5, illustrating high species richness. Measures of evenness from this equation follow from dividing H by the same logarithms of the number of species in populations H_{max}, (following from the above noted equation, H_{max} is calculated as ln(N)), giving a measure of relative population sizes of all the species in a group. This value reaches its maximum at 1, illustrating high species evenness.

Flora counts (counts of the number of species their population size) in the field were used to measure species evenness and richness as a further indicator of biodiversity. These indices were calculated from data acquired using two methodologies in the field.

Parcel Wide Flora Counts

In the Mbaracuya Region, agroforested land sections were separated into meter wide rows using nylon line suspended between meter long stakes placed in the ground. Over the course of five days, the 16 hectare agroforestry plot was quarterd into four 4 hectare sections. In between 60-100 rows were designated in each section using nylon rope draped on wooden stake. Species were then identified and tallied within each row so to diminish replicatiin. This method ultimately proved to be very time consuming. To measure the accuracy of this method, forestry surveys adapted from Canadian forestry standards were adopted and the resulting measures compared to species richness and evenness measures from the larger biodiversity counts.

Random Forestry Surveys Within Land Parcels and Conventional Monocultures Nearby

A 2.5 meter long nylon cord was used to identify plot survey areas. A stake was placed in the ground and the 2.5 meter cord was tied to the stake. Plants species were then counted and classified in a circular motion walking around the stake. All flora individuals within the circular area were included in the survey. Following the more robust method of parcel wide counts of species and their populations, forestry surveys were administered per hectare.. Therefore there were 16 plots established throughout a 16 hectare parcel. Sites where plots were established were selected randomly using a grid system provided by gps guidance. Due to their speed and ease, forestry surveys were primarily used at the majority of agroforestry locations. Values interpreted by the Shannon index were averaged for the entire parcel.

Edge Effect Surveys

The same method of plot surveying species number and population size using a 2.5 meter cord to create a plot was used to analyze edge-effects of agroforestry parcels and conventional monocultured crops on the surrounding forested land. Six surveys were conducted at each distance of 20, 60 and 100 meters into the surrounding forested land.

Carbon

Field samples to tesr carbon concentration were collected by Gabriela Canto Pires Santos *et al.* of the Federal University of Parana in collaboration with Guayaki Sustainable Forest Products. The following methodology was dictated in a stepwise fashion and confirmed by members of the collecting team. The results were interpreted to reflect stored carbon and sequestration rate. Data was collected from the same field sites over a two-year period. It was possible to compare the changes in fixed carbon within the system, thus providing a rough rate of sequestration.

Sampling Sites

The data was gathered from samples taken from the agroforestry locations in 2007 and 2009. The sites within the agroforestry parcel were chosen in consideration of spatial variability so as to ensure that the sites chosen reflected the average vegetation cover in the land parcel.

Inventory Design

A field inventory design and collection protocol was established after Brown and Roussopoulos (1974), MacDicken *et al.* (1997), Brown *et al.* (2000), and Jaramillo *et al.* (2003). The aim was to measure above-ground biomass (in the canopy and herbaceous layer), litter, and soil, which were all in the same sampling plot.

Three quadrants were established in each sampling site, the largest measuring 20 x 25 meters. Within the larger quadrant sharing one of the vertices, a smaller sampling section of 1 x 1 meter was established, as was a section measuring 0.25 x 0.25 meters. A slope compensation was applied where necessary (Brown and Delaney, 2000). Furthermore, three squares of 50 cm x 50 cm were established at three corners in each larger quadrant.

Carbon Content in Above-ground Biomass

To estimate the above-ground biomass of trees in the full 500 m^2 quadrant, the diameters of trees within each quadrant were measured at 1.3 m above-ground (AB) while the diameter breast height (DBH) was measured for all trees taller than 1.30 m; while trees less than 1.30 m height were included in the herbaceous carbon assessments. The AB was estimated using the following equation based on a study by Viera *et al.* (2008), estimating AB in submontane forests of the Atlantic Forest eco-region:

$$AB = 21.297 - 6.953\,(DBH) + 0.740(DBH)^2 \qquad (1)$$

Tiepolo *et al.* (2002)

where AB is the above-ground biomass and DBH the diameter at breast height at 1.3 m above ground.

In 1 x 1 meter quadrants, herbaceous biomass was measured by cutting all the vegetation, including shrubs, vines and small herbaceous plants, within the m^2 quadrant, excluding large trees composing the canopy. The material were dried in the field over a drying rack, bagged and weighed using a handheld scale. All biomass values were then converted to carbon using a factor of 0.5 Mg C/Mg DM (Watson *et al.* 2000).

Carbon Content in Roots

The expansion factors proposed by Cairns *et al.* (1997) were used to estimate root biomass from above-ground biomass calculated at each site (Table 1)

Table 1. Root/Shoot ratio for different forest types cover and land-use cover.

Forest cover/land-use cover	Root/shoot ratio	Source
Deciduous	0.25	Cairns *et al.* (1997)
Coniferous	0.26	Cairns *et al.* (1997)
Herbaceous	0.26	Ordóñez *et al.* (2007)
Agriculture	0.10	Ordóñez *et al.* (2007)

Carbon Content in Litter

Carbon content of the litter layer was estimated from gathering the litter within the 0.25 x 0.25 m quadrants. Samples were dried individually for 48h at 105°C in an oven, ground and weighed. The carbon concentrations of the resulting powders were measured in a carbon analyzer (CA; Fisons Instruments, Beverly, MA 01915). The carbon content in litter was estimated by multiplying its mass by the carbon concentration in dry matter and converted to Mg C ha-1.

Carbon Content in Soil

Within each square from three of the corners of the larger quadrant, litter samples were taken at three depths. 0-20 cm, 20-40 cm and 40-60 cm. A total of 9 soil samples were collected per site and used to estimate the carbon content in soil per unit volume. Fresh soil samples were individually dried and weighed for 48 hours at 808°C. Samples were then re-weighed to give an estimate of bulk density. The dried samples of soil were sieved in a 2 mm mesh to remove coarse sands and to break up soil aggregates. The samples were then ground, and the total organic carbon content was measured using a dry combustion at 950°C in a carbon analyzer (CA; Fisons Instruments, Beverly, MA 01915). These concentrations, combined with bulk density, were used to estimate the amount of carbon per unit area.

Total Carbon Stocks

The total carbon stocks per unit area were estimated as (Sathaye *et al.* 2001):

$$Ct = Cv + Cl + Cs \text{ (Mg C ha}^{-1}\text{)} \qquad (2)$$

where Ct: total carbon content, Cv: carbon content in vegetation, which is the sum of the above-ground biomass and roots, Cl: carbon content in litter and Cs: carbon content in soil. The total carbon within the land parcel was calculated by adding the carbon content of each reservoir (vegetation, litter and soil), and then multiplying this figure by the area of the agroforestry parcel. Estimates for sequestration rates were calculated by determining the difference in carbon pool amounts over the two-year period for each land parcel.

Remote Sensing of Vegetation Cover

Satellite imagery was acquired from BING maps, a privately owned company, which provided land imagery current up to August 2010. GPS data was collected using a Garmin Oregon handheld GPS device in a WGS 1984 Wed Mercator Auxilliary Sphere format and uploading to BING map aerial layer files using Arcgis 9.3.1.Images of areas where land parcels were located were converted to .tiff files so as to create images capturing the red, green and blue wavelengths of each aerial photo separately. Parcel locations were then outlined using polygon features, and unsupervised classification performed on each .tiff file to separate pixels into 6 classes (Class 1: Unknown due to shadow; Class 2: 100% cover; Class 3: 90-100% cover; Class 4: 80-90% cover; Class 5: 50-80% cover; Class 6: 0-50% cover). ERDIS image analysis software was then used to summarize percent vegetation cover of each particular Class.

Chapter V. Results

Biodiversity

Comparing Sampling Methods

Richness is said to be optimum with a value of 3.5 unitsand evenness with a value of 1 (DeJong 1975). Figure 8 is a graphical comparison account of species richness and evenness as measured by the Shannon index using two different sampling methods. Richness measured from parcel-wide population counts and 16 sample plots was calculated to be 3.4 and 3.2 respectively while evenness was 0.8 and 0.78.

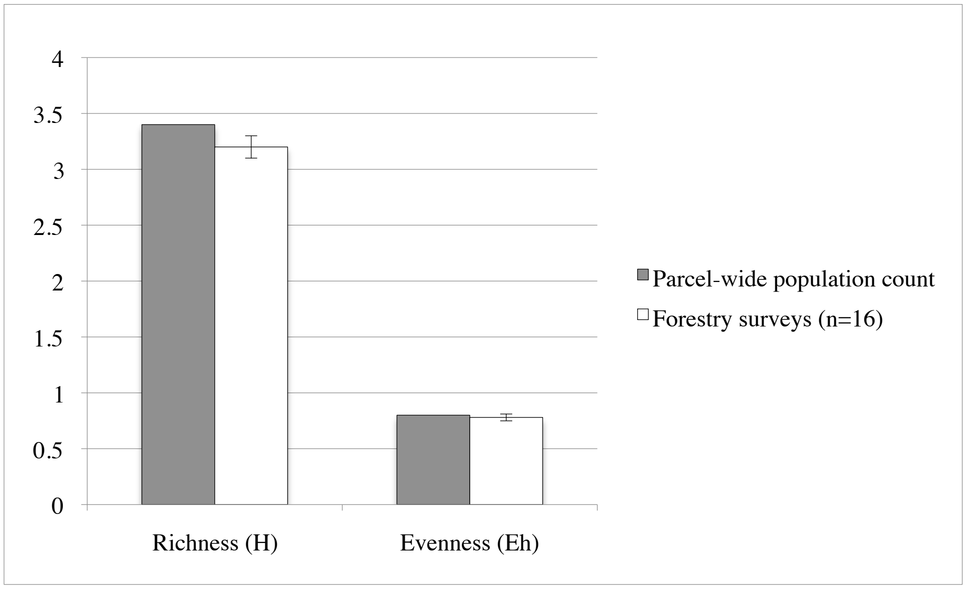

Figure 8. Bar chart comparing the biodiversity of two data samples acquired from using methodologies, in the flora community of a 16 hectare agroforestry parcel in Finca #470, Kuetuvy community, Paraguay. Biodiversity measured using the Shannon index.

Edge Effects

Edge effects in the bordering forests to a 16 hectare agroforestry parcel on Finca #470 and an expansive soyabean monoculture nearby as measured using the Shannon index's measure of species richness and evenness are displayed in Figures 9 and 10. Edge effects, as noted by disturbance, are distinct up to 20 meters into bordering forests of monoculture plantations, however, species evenness and richness in the flora community are not impacted to the same degree. The same trends were observed when comparing species richness and evenness between agroforestry parcels in Turvo and Andrecito. Disturbance was noticeable as indicated by decreased richness and evenness past 20 meters into the surrounding forest off yerba maté monocultures, but relatively no disturbance in the flora community was measured within the bordering forest off these agroforestry parcels. Figures 9-14 display these comparisons.

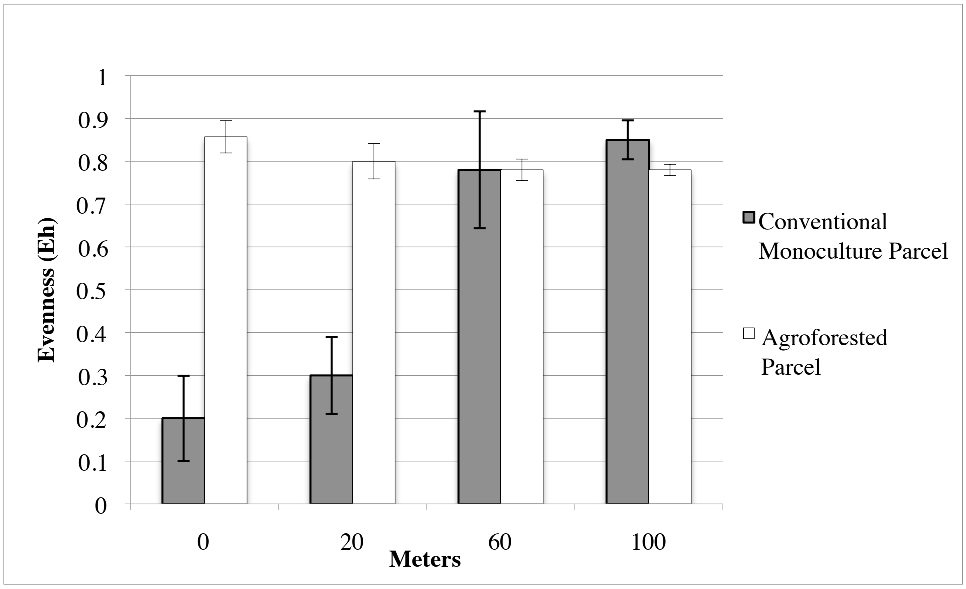

Figure 9. Bar chart comparing species evenness (Eh) as measured by the Shannon index in the flora community of the bordering forest to a conventional monoculture of 16 hectares and an agroforestry parcel of 16 hectares in the Finca #470, Kuetuvy community, Paraguay. n=6

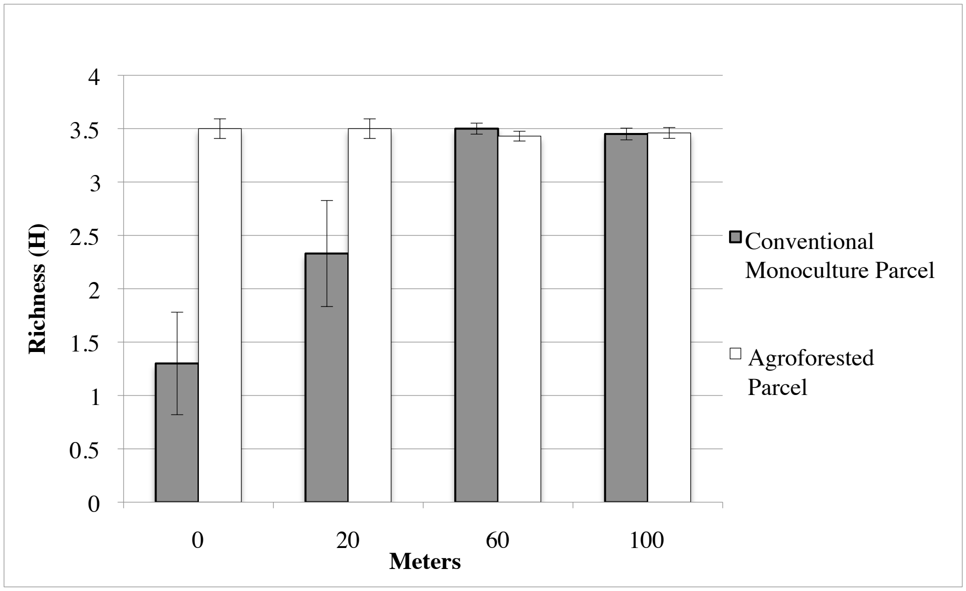

Figure 10. Bar chart comparing species richness (H) as measured by the Shannon index in the flora community of the bordering forest to a conventional monoculture of 16 hectares and an agroforestry parcel of 16 hectares in the Finca #470, Kuetuvy community, Paraguay. n=6.

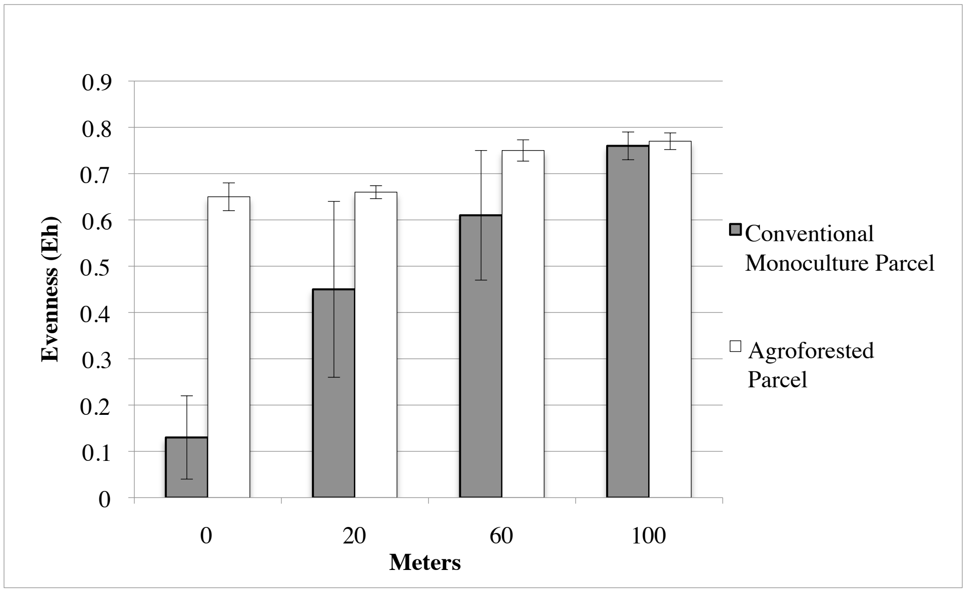

Figure 11. Bar chart comparing species Evenness (Eh) as measured by the Shannon index in the flora community of the bordering forest to a conventional monoculture of 16 hectares and an agroforestry parcel of 16 hectares in the Turvo, Parana, Brazil. n=6.

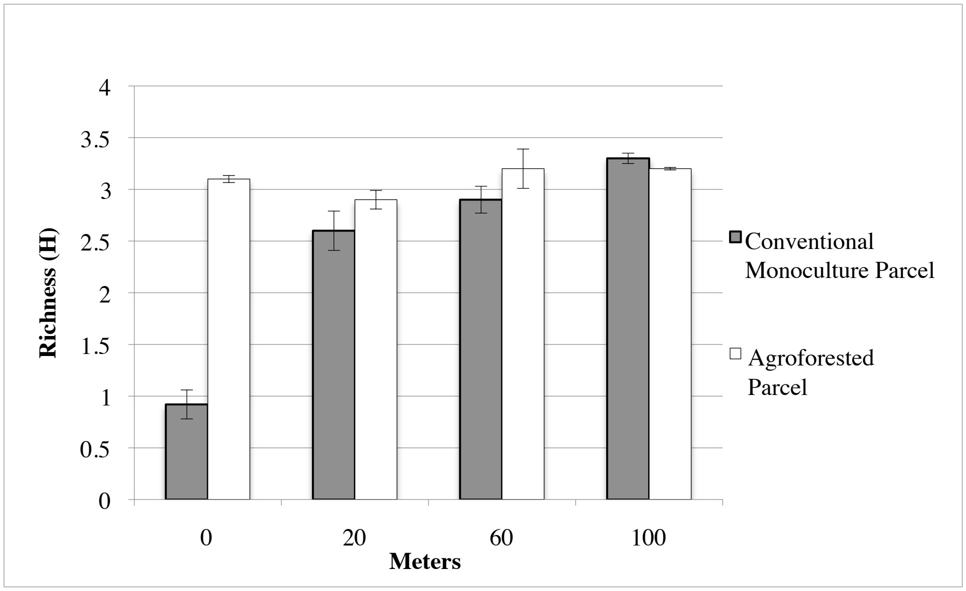

Figure 12. Bar chart comparing species Richness (H) as measured by the Shannon index in the flora community of the bordering forest to a conventional monoculture of 16 Hectares and an agroforestry parcel of 16 hectares in the Turvo, Parana, Brazil. n=6.

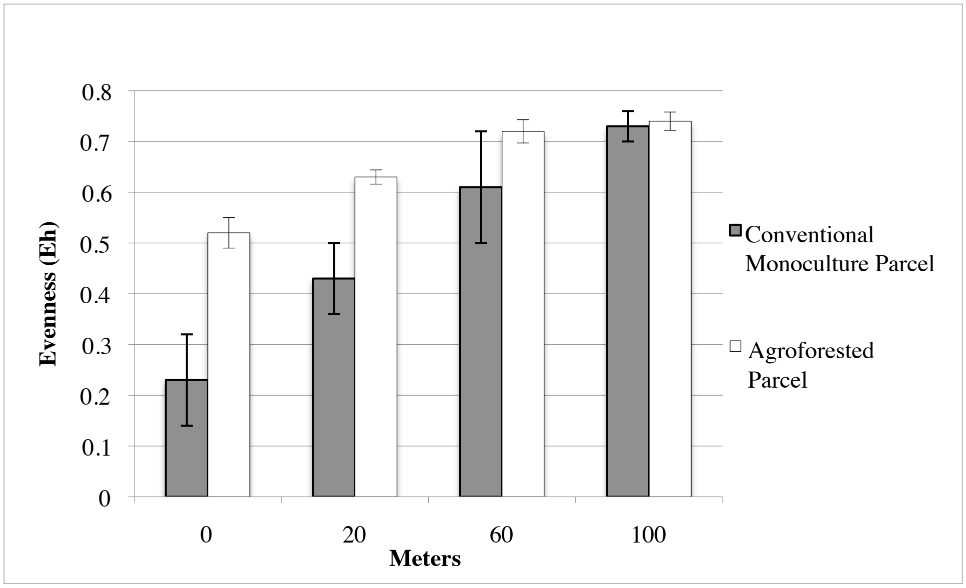

Figure 13. Bar chart comparing species evenness (Eh) as measured by the Shannon index in the flora community of the bordering forest to a conventional monoculture of 112.5 hectares and an agroforestry parcel of 28 hectares in the Comandante Andrecito, Misiones, Argentina. n=6.

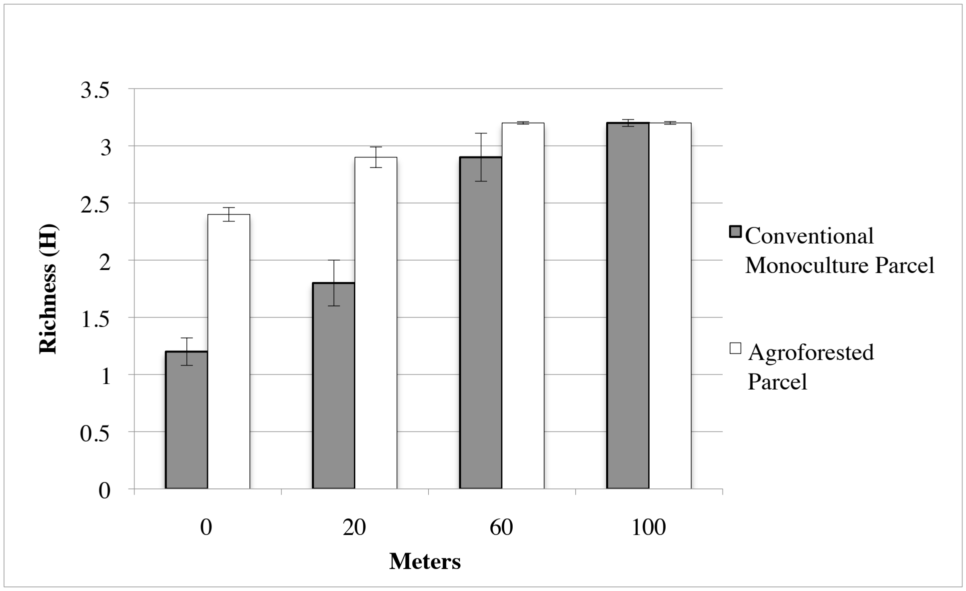

Figure 14. Bar chart comparing species Richness (H) as measured by the Shannon index in the flora community of the bordering forest to a conventional monoculture of 112.5 hectares and an agroforestry parcel of 28 hectares in the Comandante Andrecito, Misiones, Argentina. n=6.

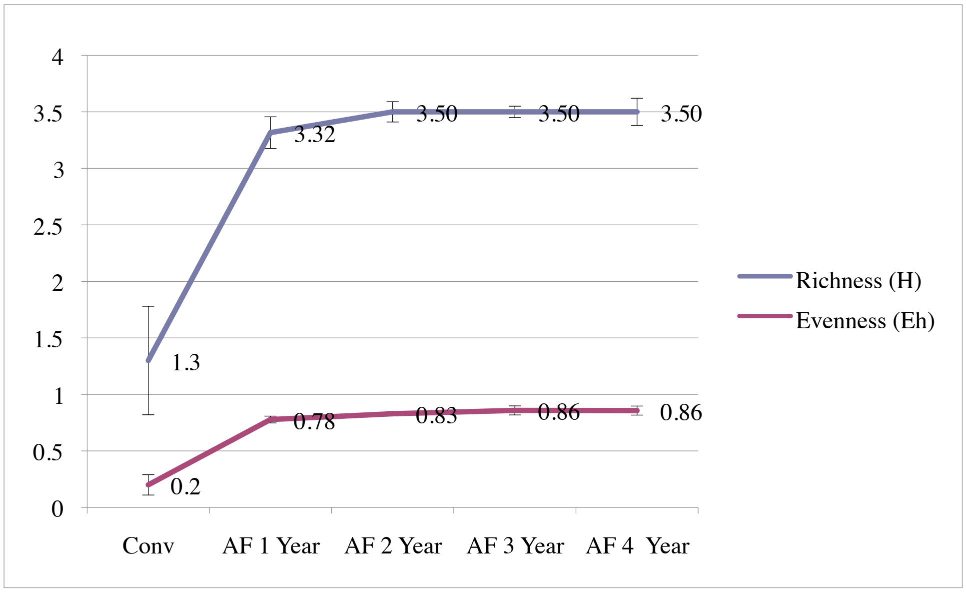

Figure 15. Line graph comparing species richness (H) and evenness (Eh) as measured by the Shannon index in a conventional parcel and agroforestry parcels up to 4 years old each of 4 hectares. In the Mbaracayu Region, Paraguay. n=4.

Biodiversity Measures within Land Parcels

Figure 15 displays the change in flora species richness and evenness between a conventional monoculture and agroforestry parcels 1-4 years of age. A conventional expansive soya monoculture shows diminished biodiversity in term of its measured species evenness and richness of 0.2 and 1.3 respectively, relative to agroforestry parcels in the same region. Forest floors converted to agroforestry plantations have lower measures of evenness and richness after one year compared to more mature agroforestry sites. Older agroforestry plantations have increased measures. Figure 16 illustrates the difference in richness and evenness between the agroforestry parcel in Turvo and a nearby yerba maté monoculture. Figure 17 shows the same distinctive trends in an Andrecito clearing. However, the agroforestry parcel in Andrecitio was characterized as having the higher disturbance than other agroforestry crops and therefore lower values of species richness and evenness were noted in comparison to other agroforestry parcels. Nevertheless, Andrecito was measured to be much higher in biodiversity than a nearby yerba maté conventional monoculture.

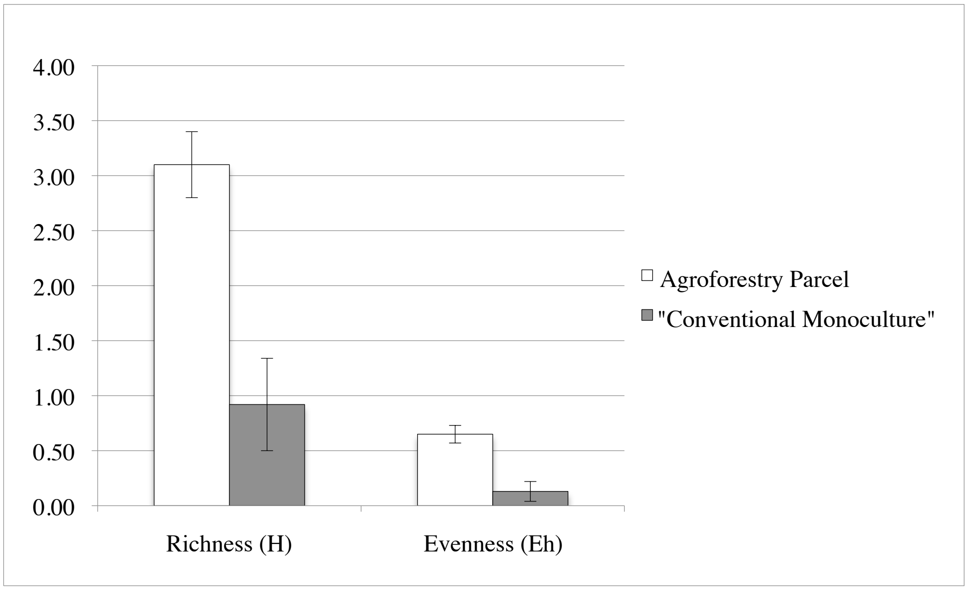

Figure 16. Figure comparing the biodiversity as measured with species richness (H) and evenness (Eh) of the Shannon index in the flora community of a 28 hectare agroforestry parcel Turvo, Parana, Brazil.

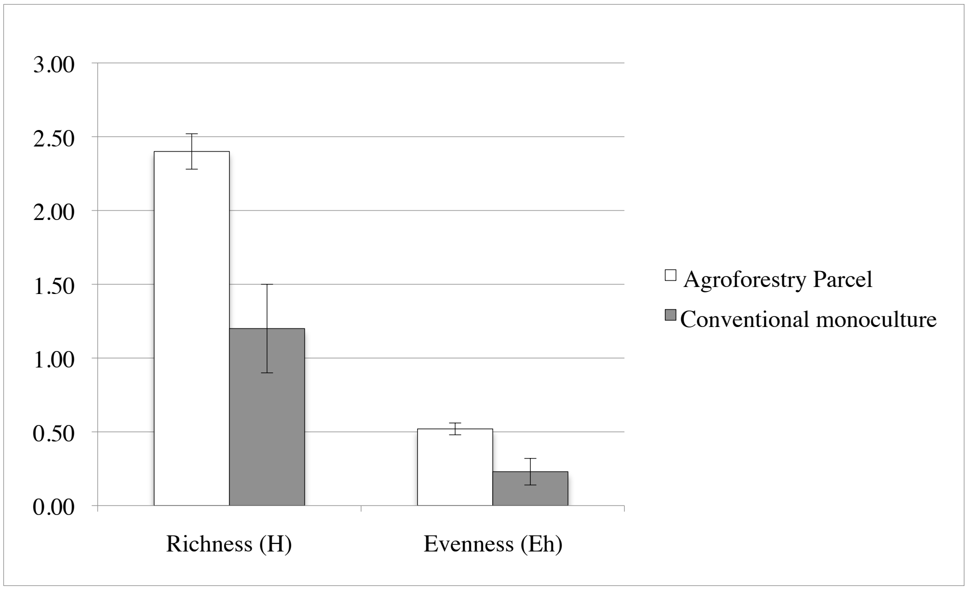

Figure 17. Figure comparing the biodiversity as measured with species richness (H) and evenness (Eh) of the Shannon index in the flora community of a 28 hectare agroforestry parcel in Comandante Andrecito, Misiones, Argentina.

Carbon Storage

Carbon in Trees

Quantities sequestered in above-ground tree carbon stocks vary between sites as shown in Table 2. Each field site was ecologically distinct from the previous, because microclimate, elevation and species mixtures varied between sites as did disturbance levels. The Andrecito agroforestry parcel in Argentina (Figure 4) was small, comprising connected fragments of forest, according to the classification system used by Pardini *et al.* (2005). This particular section of land was undergoing an early stage of forest regeneration, because much of the land parcel had formerly been a cleared area with a monoculture crop of yerba maté. The parcel was left untended (overgrown) for up to 12 years in some parts of the parcel, and native tree seedlings were intercropped with crop plants throughout the land parcel. Carbon concentration within the trees, or canopy, was calculated to be 17 Mg C ha^{-1}. On the other hand, the Kauchnaki family farm in Turvo, Brazil, resembled more a large fragmented forest with secondary to mature growth with a canopy biomass of 31.1 Mg C ha^{-1}. Finally, Finca #470 in Paraguay resembled a secondary continuous forest; having once been logged in the mid-century, there was very limited mature growth. However, in Finca #470, the forest canopy was the densest of the agroforestry plantations visited, with a canopy biomass of 38.8 Mg C ha^{-1}.

Table 2. Total carbon content (Mg C ha-1) in vegetation in three agroforestry parcels of Atlantic Forest eco-region. (Above-ground biomass: canopy, herbaceous and litter layer and below-ground biomass: roots)

Location	Above-ground biomass (Mg C ha^{-1})			Below-ground biomass	Total carbon
	Canopy	Herbaceous	Litter	Roots	Vegetation
Turvo, Brazil					
Agroforestry	31.1	14.3	3.2	8.1	66
Comandante Andrecito, Argentina					
Agroforestry	17.0	7.8	3.4	4.25	32.45
Conventional Monoculture	-	1.1	0.6	-	2.7
Finca #470, Paraguay					
Agroforestry	38.8	9.2	4.7	9.7	53.1

Carbon in Herbaceous and Litter Layers

The parcel in Andrecito, Argentina boasted the highest density of herbaceous material. This was the result of the state of natural restorations that the land-use managers had imposed upon the site. In some areas, native shrubs and plants were amassing their leaves and tendrils around whatever native trees and yerba maté trees had persisted from the previous plantation. The herbaceous biomass was calculated to have 7.8 Mg C ha^{-1} in this parcel while 3.4 Mg C ha^{-1} was calculated from the litter layer samples. In Turvo, Brazil, the Kauchnaki land parcel boasted the most herbaceous biomass due in part to its mature growth state, having dense native plants and shrubs nestled beneath the much larger trees composing the canopy, and also due in part to the cultivated medicinal plants and native shrubs which the farmers themselves maintained as a part of their land-use plans. The herbaceous biomass was calculated to contain 14.3 Mg C ha^{-1} while the litter layer sample had a value of 3.2 Mg C ha^{-1} was attained. In Paraguay, portions of the agroforestry parcel in Finca #470 were at various disturbancy stages on their recovery to agroforestry conditions. Of the 16 hectare agroforestry parcel, 4 hectare of forested land had been thinned out within the year and the density of yerba maté seedlings had been increased to 800 stems per hectare. The remainder of the 16 hectare agroforestry parcel was composed of 3 sections of 4 Ha each which were converted to an agroforestry system 3, 4 and 5 years previously. Due to the various stages of recovery, herbaceous and litter layer biomass was more heterogeneous, 9.2 Mg C ha^{-1} and 4.7 Mg C ha^{-1} respectively across the entire 16 Ha parcel. To juxtapose values, a monoculture yerba maté parcel in Andrecito, Argentina was found to have 1.1 Mg C ha^{-1} in its herbaceous biomass and 0.6 Mg C ha^{-1} in its litter layer biomass. The large differences in total vegetation carbon among man-made land classes demonstrate the vulnerability of this particular carbon pool to land-use change processes.

Carbon in Roots

Using root/shoot ratios (Table 1), estimates of the carbon stocks in roots were obtained. The stocks ranged between 8.1 Mg C ha^{-1} in Turvo to 4.25 Mg C ha^{-1} in Argentina and 9.7 Mg C ha^{-1} in the agroforestry parcel of Finca #470 in Paraguay.

Carbon in Soil

Soil carbon was analyzed at three depths, 0– 20 cm, 20-40 cm, and 40–60 cm, in each land parcel (Table 3). Within the first 20 cm of depth, the carbon content was highest in the agroforestry parcels, showing it highest concentration in the agoforestry parcels with more mature canopies such as in Turvo, Brazil and Finca #470, in Paraguay with 23.3 Mg

C ha^{-1} and 33.2 Mg C ha^{-1} respectively. Analysis done at the same depth on soil gathered from a monoculture in Andrecito showed 14.7 Mg C ha^{-1}.

In the second depth, very little difference in carbon concentration between the three agroforestry sites was observed with Turvo Brazil, Andrecito and Finca #470, showing 27.1, 26.4 and 26.3 Mg C ha^{-1}, respectively. The monoculture carbon concentration at the same depth was 22.3 Mg C ha^{-1}.

Finally, the third depth – from 40 to 60 cm – in Turvo, Andrecito and Finca #470 showed 38.4, 49.1 and 42.3 Mg C ha^{-1}, while the conventional monoculture showed 44.6 Mg C ha^{-1}.

Table 3. Total carbon content (Mg C ha^{-1}) in soil at three depths in 3 agroforestry parcels and one conventional parcel of the Atlantic Forest eco-region.

Location	Depth (cm) (n=3)			Total carbon
	0-20	20-40	40-60	Soil
Turvo, Brazil				
Agroforestry				
Mean	23.2 ± 3.8	27.1 ± 4.3	38.4 ± 9.1	88.7 ± 8.7
Range	19.8-27.3	24.2-32.1	28.3-46.1	72.3-105.5
Comandante Andrecito, AR				
Agroforestry				
Mean	19.5 ± 4.9	26.4 ± 7.5	49.1 ± 2.6	95 ± 14.2
Range	14.6-24.3	19.1-34.1	47.1-52.1	80.8-110.5
Conventional Monoculture				
Mean	14.7 ± 7.4	22.3 ± 8.2	44.6 ± 14.3	81.6 ± 16.2
Range	6.2-19.3	15.2-31.2	28.1-53.6	49-5-104.1
Finca #470, Paraguay				
Agroforestry				
Mean	33.2 ± 11.9	26.3 ± 5.3	42.3 ± 8.3	101.8 ± 10.4
Range	19.5-40.6	23.2-32.4	33.6-50.1	76.3-123.1

n = number of sites in one land parcel. Mean = carbon content average per parcel sampled. Range = minimum and maximum value of carbon content by parcel sampled. ± = Standard error

Table 4. Total carbon (Mg C ha^{-1}) in 3 agroforestry parcels measured during 2007 and 2009 and the calculated sequestration rate (Mg C ha^{-1} yr^{-1}).

Location	Total Carbon 2007	Total Carbon 2009	Sequestration Rate
Turvo, Brazil	154.7	172.7	9.0
Comandante Andrecito, AR	148.1	153.5	2.7
Finca #470, Paraguay	154.9	163.2	4.15

Total Carbon Concentration and Sequestration Rates

Field samples were gathered and measured in field sites in November of 2007 and again in November of 2009 (Table 4). Differences between total values suggest an increase in carbon fixed. The difference in stored carbon values over the 2-year period suggest a rough annual fixation rate for the isolated agroforestry parcels in Turvo, Andrecito and Finca #470 of 9, 2.7 and 4.15 Mg C ha^{-1} yr^{-1}, respectively. A relative carbon concentration bar graph for each field site can be seen in Figure 18.

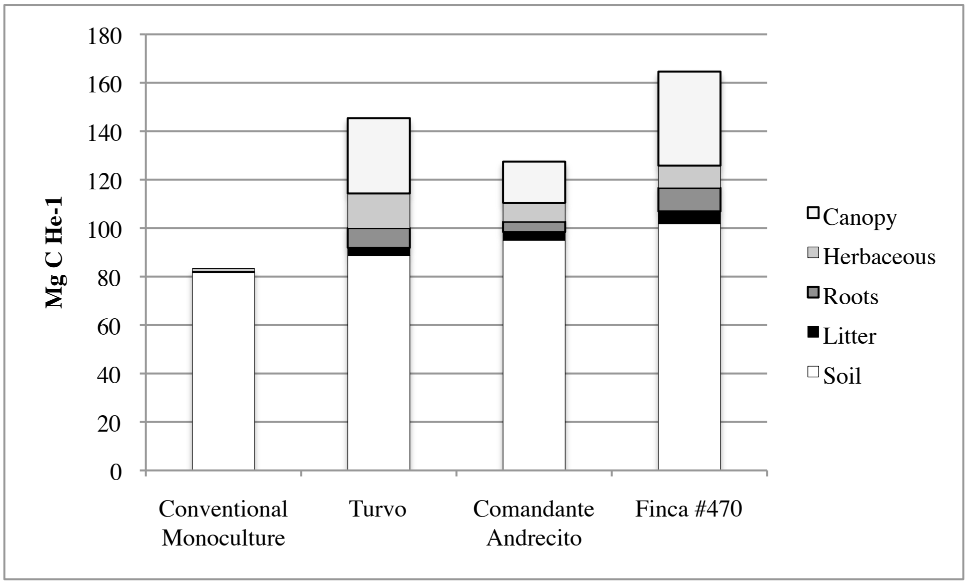

Figure 18. Total carbon content by stock in each parcel.

Vegetation Cover

Land parcels' vegetation cover is presented as 'percent cover value'. Figures 19 (A-D) shows aerial maps of land parcels after an unsupervised classification of the various shades representing the degree of vegetation cover. Such a classification allowed for resembling shade within the satellite imagery to be grouped together and later to be classified. Using these particular aerial maps to determine cover, sections of land were on occasion poorly interpretable due to shadow, and therefore the total percent cover values are approximate. In the agroforestry parcels of Turvo, Andrecito and in Finca #470, 11.73, 19.02 and 20.33 % of the areas were respectively unknown due to shadow. The monoculture in Andcrecito had 0.85 % shadow in its aerial photo. Average vegetation cover values for the entire agroforestry parcels in Turvo, Ancrecito and Finca #470 were 73.10, 51.39 and 44.46 % respectively, while the Andrecito monoculture had an average vegetation cover value of 24.23 % as displayed in Figure 20 (A). Figure 20 (B) shows proportions of percent cover range across land parcels and then relates these values to each land parcel. The most notable trend observed was in the increased proportions of percent cover in the agroforestry sites, with Turvo, Andrecito and Finca #470 showing values of 17.90, 24.83 and 18.08 % of total area being covered with 90-100 % vegetation, versus the conventional monoculture parcels in Andrecito showing a value of 4 % of total area being covered with 90-100 % vegetation.

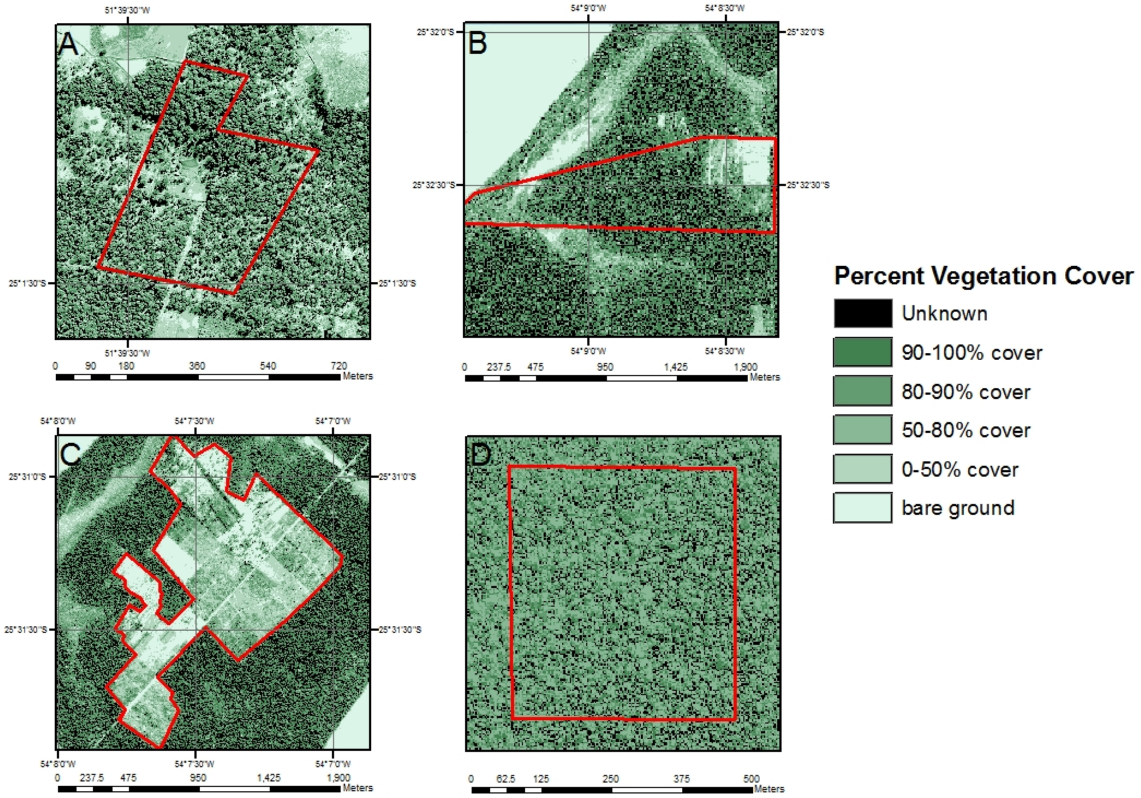

Figure 19 (A-D). Unsupervised classification of infrared and red wavelength aerial imagery of land parcels A-D (A: Kauchnaki farm in Turvo, Brazil, B: The Guayaki parcel in Andrecito Argentina, C: The monoculture yerba maté farm in Andrecito, Argentina, D: The Kuetuby agroforestry plantation in Paraguay) Pixels characterized according to remote sensing of percent cover.

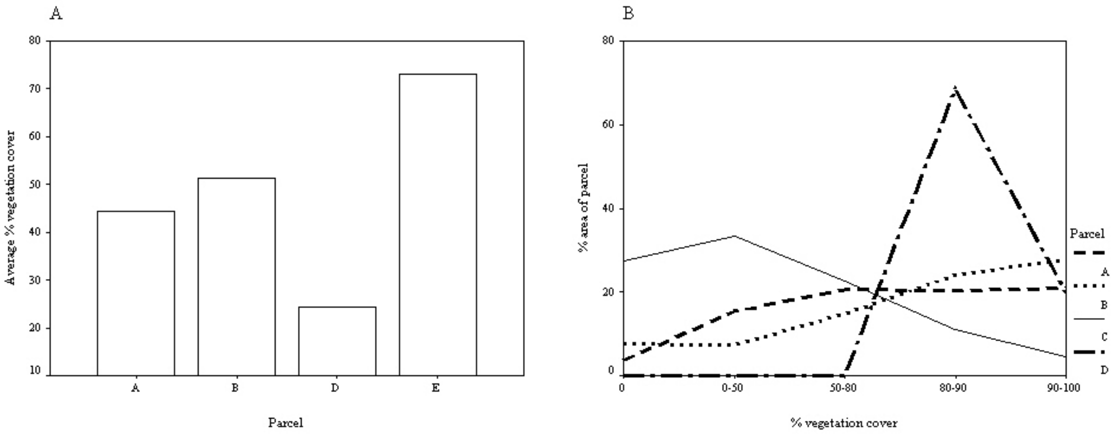

Figure 20 (A-B). The average percent vegetation cover values of each of the field sites A-D (A). The percent area of land and its relative percent vegetation cover at each field site A-D (B).

Chaper VI. Valuation of Ecosystem Services

Introduction

This chapter is based on previous efforts to place economic values on ecosystem services associated with landscapes (Bingham *et al.* 1995; Chee *et al.* 2004; Holmes *et al.* 2004; Ludwig 2000; Nunes 2001; Pattanayak 1996; Turner *et al.* 2000; Zhongxin and Xinshi 2000; Villa *et al.* 2002; Wilson and Howarth 2002; Robbins 2004). The ranges in values of ecosystem services in tropical environments are based on payments from already established "payment for ecosystem services" transactions.

Valuation of Ecosystem Services

Valuation ultimately refers to the contribution of an item to meeting a specific goal. For example, a member of a team is valuable to the extent that he contributes to the team's success. Costanza (2000 p 3) claims that "in ecology, a gene is valuable to the extent it contributes to the goal of survival of the individuals possessing it and their progeny". In conventional economics, a commodity is valuable to the extent that it contributes to the goal of individual welfare as assessed by willingness to pay. One cannot state a value without stating the goal being served. Costanza *et al.* (1998) insist that when valuating commodities within an economic system, it is crucial to "efficiently consider all resources…both marketed and nonmarketed resources, especially ecosystem services" (p 3). In this sense, the consideration of ecosystem services inherent in any economic and hence ecological system is considered foundational to sustainability of that system. Fair consideration of ecosystem services "must be satisfied in an integrated fashion to allow human life to continue in a desirable way" (Constanza *et al.* 1998 p 4). We cannot avoid the valuation issue, because as long as we are forced to make choices we are doing a comparative evaluation. But we need to be as comprehensive as possible in our valuations and choices about ecosystems and sustainability and properly recognize the relationships between goals and values.

The discussion around valuation is based on quantitative analysis of the functioning and provisions of ecosystem services. As stated previously, ecosystem services were selected for analysis in this particular study based on their marketability. That is to say, on their likelihood of receiving funds from offsetting, conservation and payment for ecosystem service programmes (Table 5). This condition provides various reference points for dollar amounts, because marketability is a direct indication of previous studies assessing "willingness to pay" (WTP) and records of "price of sale" being accessible (Kramer and Mercer 1997, Pattanayak and Kramer 2001). The process of valuation, therefore, for this particular study reviews previous ascertained values and transactions for lands in tropical regions and suggests ranges in value for the ecosystem services in question.

Table 5. Global Market Value of Environmental Markets

Environmental Market	Market (billion $US) (2008)
Regulated Carbon	117
Water Quality	9.25
Biodiversity	2.9
Voluntary Carbon	0.7
Forest Carbon	0.037

Sources: World Bank. "States and Trends of Carbon Markets: 2010." Ecosystem Marketplace Reports "Building Bridges: State of Voluntary Carbon Markets 2010" and "State of Biodiversity Markets: Offset and Compensation Programs Worldwide."

Table 6 is a summary of previous studies suggesting values which access pharmaceutical speculative value of the biodiversity within intact ecosystems and the inherent value of intact tropical forest ranges to both users and non-users assessed through a "willingness to pay” proposition. Further, Watershed conservation payment schemes are assessed in communities with tropical watersheds. The values surmised a general range between $\$$ 0.9-107 ha^{-1} yr^{-1} for biodiversity conservation and $\$$3-35 ha^{-1} y^{-1} for watershed conservation. Table 7 summarizes current carbon prices Mg^{-1} that are currently trading on the international carbon market and project prices in the next 10 and 100 years. Prices that are currently relevant range between $\$$45- $\$$150 Mg^{-1} of carbon. Projections suggest a likelihood in the market that prices will increase as high as $\$$200 in the next 10 years and by up to $\$$800 in the next 100 years.

Table 6. Summary of economic valuations for biodiversity and watershed conservation programmes.

Biodiversity	$\$$ ha^{-1} yr^{-1}	Source:
Pharmaceutical worth		
Worldwide	0.9-1.32	Mendelsohn and Balick, 1995
Tropical Forests	1-90	Adger *et al*. 1995
Atlantic Forest South America	4.4	Simpson *et al*. 1996
Existence Value of Tropical forests (WTP for tropical conservation)		
Intrinsic Worth	1.2-64	Adger *et al*. 1995
Scenic non-use value	9-107	Adger *et al*. 1995
WTP measure from U.S residents	4	Kramer and Mercer, 1997
Range	0.9-107	
Watershed		
Prevention of soil erosion and nutrient loss		
Guatemala	12	Ammour *et al*. 2000
Malaysian	4-15	Kumari, 1996, Mohd Shahwahid *et al*. 1997
Cameroon	3-24	Ruitenbeck 1992, Yaron 2001
Indonesia	3-35	Pattanayak and Kramer, 2001
Range	3-35	

Table 7. Current and projected prices for $Mg^{-1}C$.

	$ MgC^{-1}	Source:
Current price	45-150	Sohngen and Sedio 2006
Projected price 2020	150-200	Tavoni et al 2006
Projected price 2105	100-800	Tavoni *et al*. 2006

Suggested Value Ranges for Agroforestry Parcels

Table 8 displays the suggested range of values for the ecosystem services being conserved on visited agroforestry plantations. Ranges were attained by multiplying area conserved by the price range which is a ha^{-1} value. Values for Carbon are based simply on the sequestration rate. No values were associated with already stored quantities above- and below-ground. Biodiversity and watershed values were subjected to reductions because in all three sites, optimum values of biodiversity and watershed maintenance were not reflected. Total range in value of the agroforestry properties in Turvo, Finca #470 and Andrectio were $ 467.5 – 3 606 yr^{-1}, $ 249.15 – 2 878.5 yr^{-1} and $ 449.1 – 12 333 yr^{-1} respectively.

Table 8. Valuation of agroforestry parcel ecosystem services.

Agroforestry Parcel	Ecosystem Service ($ yr^{-1})			Total value on property ($ yr^{-1})
	Biodiversity	Carbon	Watershed	
Turvo	14 – 1700	405 – 1350	48 – 560	467 – 3606
Finca #470	14 – 1700	187 – 623	48 – 560	249 – 2879
Andrecito	76 – 9000	122 – 405	252 – 2940	449 – 12333

Chapter VII. Discussion of Thesis

Within this chapter the implications of the measurement of ecosystem services are discussed in reference to their location and the land-use methodology under which they are subjected. Incorporated into the text is the subtle reference to the farmers' relationships with their land, to transition the reader into a summary of the thesis, which incorporates both the qualitative and quantitative conclusions of this study.

Biodiversity

Interpreting flora counts subjected to the Shannon index indicated the trend towards the preservation and enhancement of biodiversity within agroforestry parcels. Most notable is the reduction of biodiversity measured in nearby monocultures compared to that preserved in the agroforestry parcels of yerba maté. Agroforested yerba maté interactions with native canopy coverage boasted both high richness and evenness, levels often exceeding those gathered from over 100 meters into bordering undisturbed canopy of intact Atlantic Forest. The ecosystem services inherent in biodiversity conservation are in many ways less compartmentalized, having regulating and equilibrating effects on population sizes, pathogen outbreaks and various other life support features of an ecosystem. Most significant of these services for the agroforester is the limit of pathogen spread, the increase in insect biodiversity, bird species and specifically, pollinators (Sperber *et al.* 2004; Waltert *et al.* 2004; Klein *et al.* 2004). A meta-analysis performed by Balvenera *et al.* (2006) substantiates the interpretation and goes on to elaborate on additional positive effects of biodiversity on the overall ecosystem processes.

In Turvo, the agroforesty location was a mature property where some of the yerba maté trees were 25-30 years of age beneath a canopy of over 60 years of age, with some massive *Araucaria* 80 or more years old as indicated by Reginal, the property co-owner and steward. Established within this area was an equilibrium between farmer, native plants and surrounding forest; a trilateral dependence which beckoned each party to play its part, the burden of understanding being squarely placed on the shoulders of the farmer, knowing that their faltering would limit the functioning and duties of the others. According to measurements taken in this property, the certainty in predicting the species of an individual chosen randomly was low. In other words, species richness was high, a value of 3.1 out of 3.5, which was higher than the nearby yerba maté monoculture for which species diversity estimated at just 0.92. Evenness, or equal proportionality of species populations was also reflective of high biodiversity in the agroforested parcel versus the monoculture.

In Comandante Andrecito, an area of former monoculture features was in the process of conversion to agroforestry. A team of farmers were working to clear the base of the now over-grown yerba maté trees and to cut back the undergrowth so to restore rows along the trees and allow for better access to pruning. Though this section of land was younger, the majority of the native preserve had been untouched for the last 60 years. The Iguaçu River was inaccessible even though the property line bordered on the river, an indication of the native reserve undergrowth density. Because of the current efforts to convert the overgrown monoculture to an ‘agroforested parcel in transition’, choosing sites for biodiversity measurements was difficult; the sampling locations were randomly selected only in areas having already been thinned as these estimates required the plots be accessible. Furthermore, over 60% of the land was outside of current agricultural use. I did not include this land in my assessment of biodiversity conservation; had it been included, these values would have been much higher. However, these areas were included in my vegetation cover analysis. Gathering samples from only within the agroforestry restoration area still garnered a much larger richness and evenness measure than the yerba maté monoculture only minutes away.

The Ache community on Finca #470 had been working on converting 16 hectares to agroforestry conditions for the last seven years. Slowly, 4-hectare chunks were thinned out by removing debris that lay strewn about on the forest floor, cutting the odd tree to provide more light to the understory and growing yerba maté seedlings from seed in an outdoor nursery nestled within the community, to be transplanted into the forest for cultivation. A snapshot of the process of transition was possible because the 16 hectare plantation had sections in different ages of transition reflecting the ongoing temporal process. As maturity increased, so did richness and evenness, reaching very high values compared to the soyabean monoculture outside the Mbaracayú Reserve.

The impacts of agroforestry on the parceled land where agricultural activities were ongoing is positive concerning biodiversity conservation when compared to the reduction in biodiversity noted in monocultures. However, quantifying and valuing these localized effects does not capture the entire story. Due to the homogeneity by design of monocultures, edge effects of bordering lands are a necessary imposition from the lack of ecosystem equilibriums that are characteristic of monocultures. These edge effects were measured in the bordering forests of monocultures in three sites chosen because of their proximity to agroforestry parcels and, therefore, their shared eco-region. In all sites, species richness and evenness are diminished beneath optimum values. For every hectare of monoculture, arguably another bordering hectare of forested land is impacted by the imbalance that monocultures impose upon their bordering ecosystems. This dynamic is by and large avoided in yerba maté agroforested parcels as indicated by the samples taken in bordering forested land. In general, whatever manipulation of the environmental that agroforest imposes, it has almost an immeasurable effect on the abundance of flora species in bordering lands, a trend noted by Langton (1990), and Cullen *et al.* (2001), whereby edge effects from agroforestry parcels are significantly reduced.

In promotion of agrofroestry, what is illustrated most distinctly by the land-use methodology is not only the abundance of flora species, but the encouragement of their presence and growth, a variable that is absent in the land-use management methodology of the monoculturalist and thus an indication of their political ecology

Carbon

Increased carbon stocks in agroforested parcels are largely due to the conserved canopy (Lindner 2010), but it also reflects how the canopy affects soil ecology such that increases in below-ground carbon are recorded. Measurements of agroforestry parcels in the Atlantic Forest eco-region reveal the enhanced fixation and storage of more carbon than monocultured land parcels with values in total stored carbon ranging between 154.7-172.7 Mg C ha^{-1}, and sequestration rates between 2.7 – 9 Mg C ha^{-1} yr^{-1} in agroforestry parcels compared to values of total carbon stocks in monoculture plantations of 81.3 Mg C ha^{-1}. Schroeder outlines agroforestry well by characterizing it as representing a “link between the local and global scales… from the farmer's perspective, agroforestry can be a way to increase crop yields and the diversity of products grown and to cultivate an integration with the native ecosystem. An additional benefit is the creation of a carbon sink that removes carbon dioxide from the atmosphere. Successful agroforestry systems will also reduce land clearing and maintain carbon in existing vegetation" (Schroeder 1994 p 92).

In the Yerbe maté agroforestry parcels examined within this study, native canopies provide shade to the understory, but by definition, agroforestry systems do not always incorporate a native canopy. More often, they attempt to integrate more than one cash crop, for example cacao trees (*Theobroma cacao* L.) and Coffee (*Coffea Arabica* L.), or Mango (*Mangifera indica* L.) and Acai (*Euterpe oleracea* Mart.). Indeed, rarely is the native canopy so obviously prized as in the yerba maté industry in this part of the world. The benefits from the yerba maté agroforestry system come in the various downstream benefits of biodiversity conservation, However, all agroforestry systems share large carbon sequestering potentials, whether they be long rotation systems such as homegardens and boundary plantings or timber and cattle system which still can sequester sizeable quantities of carbon in plant biomass and in long-lasting wood products. Further, much higher soil carbon sequestration constitutes another realistic option achievable in many agroforestry systems. Albrecht and Kandju (2003) found that agroforests have sequestered between 12 and 228 Mg C ha^{-1} with a median value of 95 Mg ha^{-1}. According to them, based on the available area that is suitable for the practice (585–1215 x 10^{6} ha), 1.1–2.2 Pg[5] C could be stored in the terrestrial ecosystems over the next 50 years.

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[5] Pg: Pentagram, 10^{15} g

Intensified agricultural practices lead to a reduction in ecosystem carbon stocks, mainly due to removal of aboveground biomass as harvest and loss of carbon as CO_2 through burning and/or decomposition. Agroforestry systems promise to increase aboveground and soil carbon stocks and reduce soil degradation, as well as to mitigate greenhouse gas emissions. Further, the potential of agroforestry systems to sequester carbon in the top 60 cm of soil is 93 Mg ha^{-1}. Even in degraded soils of the sub-humid tropics, improved fallow agroforestry practices have been found to increase top soil carbon stocks up to 1.6 Mg C ha^{-1} y^{-1} above continuous maize cropping (Nair *et al.* 2009).

Because of the mitigated impacts of agroforestry, its was recognized as a greenhouse gas–mitigation strategy under the Kyoto Protocol. This has earned it added attention as a strategy for biological carbon sequestration. Further, researchers working on a joint World Agroforestry Centre-United Nation Environment programme project suggest that integrating agroforestry in farming systems on a massive scale could create vital reservoirs for carbon storage. No less than a billion hectares of farmland is suitable for conversion to carbon agroforestry projects according to estimate of the Intergovernmental Panel on Climate Change (Watson *et al.* 2000). The perceived potential is based on the premise that the greater efficiency of integrated systems in resources (nutrients, light, and water) capture and utilization than monoculture systems will result in greater net carbon sequestration. Added carbon sequestration potential in agroforestry land-use methodologies is an underappreciated and underexploited environmental benefit. If programs are to be designed around financing the transition of conventional monocultured lands into carbon fixing agroforested system, it will depend on the forest carbon projects becoming more widely financed by international carbon markets, the price of carbon in the international market, and the cost related to carbon monitoring (Nair *et al.* 2009).

Watershed Values

The vegetation cover of agroforesty parcels of Turvo, Andrecito and in Finca #470 was measured to be between 65-89% cover. These values are inclusive of shadow, which could not be fully interpreted, but likely were cast across covered land. In the Andrecito yerba maté monocultured parcel, vegetation cover was measured to be roughly 25%. However the lower proportions of higher percent cover in this monocultured area was indicative of a larger area being subject to erosion. If vegetation cover is a reflection of watershed intactness, then the high proportions of highly covered land across agroforestry plantations compared to monocultured plantations position agroforestry as a conservative watershed management system.

In a study performed by El-Hassanin (1993) soil loss and runoff data were obtained from plots representing different conditions of vegetation cover. The highest values of soil erosion were obtained under cultivated soils and the lowest values under forest and grass soils. The relationship between runoff and soil loss was linear and significant under various conditions of vegetation cover. Therefore, soil stability is proportional to vegetation cover, and further, carbon accretion is linked to the structural development of the soil, in particular to increasing carbon in water stable aggregates (WSA). In a review by Mutuo *et al.* of agroforestry practices they found that these systems were able to mitigate N_2O and CO_2 emissions from soils and increase the CH_4 sink strength compared to cropping systems (Mutuo *et al.* 2005). Further, agroforestry and grass buffers have been shown to improve soil properties and overall environmental quality. The results of a study performed by Udawatta *et al.* (2008) show that establishment of agroforestry increased water stable aggregates, including soil carbon, soil nitrogen, and enzyme activity by reducing erosion and ultimately maintaining the structural integrity of the watershed. In another watershed study done by Udawatta *et al.* (2002) nitrate loss was reduced by 37% by using agroforestry treatments. Agroforestry management practices effectively reduced nonpoint-source pollution in runoff from the cash crop upon which fertilizers were used. Interestingly, another key benefit to agroforestry systems is that depending on species composition, fertilizer input can often be reduced or eliminated with the incorporation of nitrogen fixing plants (Scroth 2002). These various studies highlight the watershed maintained features of agroforestry as measured using vegetation cover analysis.

Garrity (1999) suggests that successful watershed management requires practical technical innovation and participatory institutional innovation, He then expressed the opinion that agroforestry has a key role to play in both because agroforestry has been seen as a set of technical options applied at the field level but is now increasingly being conceived as a framework for whole landscape management within a community and ecological context. Watershed degradation poses a threat across the planet but past watershed management programs have most frequently been ineffectual. Evidence indicates that it is possible for smallholders to engage in farming and management of natural forest resources in both productive and conservation-effective manner. Agroforestry research and development is creating a much wider array of practical solutions that reduce the tension in achieving both the environmental service functions of watershed and the productivity functions essential to the livelihood of the dense rural populations that inhabit them. As indicated by the Millenium Development Goals (MDGs), a major role for agroforestry also is emerging in the domain of environmental services. This entails the development of mechanisms to reward the rural poor for the environmental services such as watershed protection and carbon sequestration that they provide to society. Agroforestry research and development is contributing to virtually all of the MDGs. But recognition for that role must be won by ensuring that more developing countries have national agroforestry strategies, and that agroforestry is a recognized part of their programs to achieve the MDGs (Garrity 2004).

Significance of Results and Experimental Error

Subjecting my biodiversity sampling results, vegetation cover analysis and carbon data to any statistical analysis of significance was not possible due to the small number of variations in the sample size. I was therefore unable to draw conclusions with any indication of confidence. Further replications of biodiversity surveys in agroforestry sites and carbon samples would be required in order to subject results to any statistical scrutiny. However, it was the purpose of this research to display to the reader what is clearly evident to the visitor of agroforestry sites, that being that biodiversity is clearly higher, vegetation richer and carbon is being stored to a greater degree in the ever present canopy. An important contribution of my research was to demonstrate a simple method supported by academic literature of evaluating ecosystem services in agroforestry land parcels. This process provides an avenue to build on consulting opportunities focused on identifying ideal directions of environmental stewardship with agriculturalists.

Chapter VIII. Conclusions

The assessment of the potential of yerba maté agroforestry to contribute positively to both the social and environmental stability of rural South America is the focus of this thesis. Ecosystem services within agroforestry parcels were compared to those of nearby monocultures. The vegetation cover of agroforestry parcels in various localities of the Atlantic Forest was measured to be between 65-89% while yerba maté monoculture plantations had roughly 25% cover. Further, values of total stored carbon within the agroforestry parcels ranged between 154.7-172.7 Mg C ha^{-1}, compared to total carbon stock in monoculture plantations of 81.3 Mg C ha^{-1}. Finally, as measured using the Shannon index, richness, and evenness, or equal proportionality of species populations was also indicative of greater biodiversity in agroforested parcels than monocultures, with values of richness ranging from 2.7-3.5 in agroforestry parcels compared to 0.9-1.3 in monocultures. Values of evenness ranged between 0.6 and 0.8 in agroforestry parcels, and averaged 0.2 in monocultures.

These findings illustrate that yerba maté agroforestry can contribute as a regional climate change mitigation strategy. Agroforestry can also help restore and maintain above-ground and below-ground biodiversity, and providecorridors between protected forests and watershed and soil integrity. Agroforestry also reduces pressure on natural forests by way of reduced edge effects. The ecosystem services provided by yerba maté agroforestry are valuable to the farmer, as the stewarding of those services promotes a more autonomous political ecology. Furthermore, the provisioning of ecosystem services by agroforestry is valuable to society as made evident by programs assessing the ‘willingness to pay’ of citizens as well as defined global market prices. An analysis of market prices available within the ecosystem marketplace for total ecosystem services being conserved on agroforestry parcels ranged in value between \$16 – \$160 ha^{-1} yr^{-1}.

My research uncovered a subversive political ecology in the agroforester competing with the dominant infrastructure of large-scale agriculture. The dominant agriculture being driven by pervasive forms of neo-liberalized capitalism which fundamentally defines the political ecology of most farmers nowadays in that region. Agroforestry was quite widely practiced prior to the political, economic and social factors that engendered the practice of monoculture to the yerba maté industry of the Atlantic Forest. This shift in the political ecology of sustainable yerba maté agriculturalists transitioned broader ideologies of sustainable land stewardship while interpolating urban subjects as it went. The yerba maté monoculture is now as much a vehicle for the creation and maintenance of social systems that maintain its dominance, as it was a product of those systems. For many farmers this has meant the cessation of production, in many cases the selling of their farm and their subsequent migrating to the towns. However, many remaining small-scale yerba maté farmers have demanded a change in government policy to promote the smaller-scale sustainable yerba maté farming.

Quantitative analysis of the ecosystem services in yerba maté agroforested land parcels provides a basis upon which a discussion around state led ‘payment for ecosystem services’ programs could be promoted. Governmental policy plays an important role in effecting the development of agricultural systems and management techniques of small-scale farmers. To begin a shift towards environmentally sustainable yerba maté production that supports the livelihoods of small-scale farmers, the state must foster policies that create economic and social conditions that invest in agricultural sustainability. The development of strong leaders and social movements that integrate the concepts of economic justice and environmental sustainability in yerba maté production have the potential to further this paradigm shift. The current strength of a social movement in this direction is evidenced by the creation of marketing cooperatives in various locations where small-scale maté producers are struggling to sell their product. Cooperatives are beginning to institutionalize structures and paradigms of economic justice, accountability and transparency. Further, agroforestry and organic production systems are evolving to support environmental sustainability in the yerba maté sector but the expansion of these programs would be heavily supported with coherent government policy incentives to environmental conservation.

Combining coherent agrarian policies that incentivize conservation of ecosystem services with strong leadership and the appropriate social movements to demand these changes has the potential of achieving both a framework for economic justice and environmental sustainability within the yerba maté sector. This evolution could improve the economic status of the regional agroforesters, promote the conservation of the Atlantic Forest ecosystem and help alleviate the effects of global climate change.

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