DISSERTATION

THREE ESSAYS ON THE ECONOMICS OF RESTORING DEGRADED LAND: FROM GLOBAL TO LOCAL

Submitted by

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ABSTRACT

Restoring the productivity of the world's degraded land has become a global priority. Daily (1995) brought the idea of restoration to prominence with a publication in *Science* entitled 'Restoring value to the world's degraded lands', which was a call to reverse the trend of degradation by restoring degraded land for the wellbeing of humanity. Nearly twenty years later decision makers have started to act on this call by creating high-level restoration initiatives designed to catalyze restoration activities in dozens of countries across the world. So called Great Green Wall projects in China and the Sahel region of Sub-Saharan Africa are planting billions of trees on land that has been barren for decades in an effort to stop the advance of the Gobi and Sahara deserts. In Brazil, the Mata Atlantica Restoration Pact is mobilizing support to restore more than 37 million acres of deforested and degraded forest in Brazil's Atlantic Forest and the United States Forest Service, through the Collaborative Forest Landscape Restoration Program, is working with a large number of forest stakeholders to restore the ecological, economic, and social benefits created by America's forests. These projects are only the best known examples of restoration activities that are taking place across the world, but several hundred more grassroots efforts are attempting to restore the functionality of degraded ecosystems through the process of restoration.

In an effort to coordinate these activities, the Bonn Challenge, a global effort to begin restoring 150 million hectares of degraded forest and agricultural landscapes by 2020, was launched by the International Union for the Conservation of Nature and the Government of Germany in 2011. To date 12 countries have committed to restore more than 60 million hectares (148 million acres) of

degraded land as part of The Bonn Challenge. This effort was reinforced in 2014 through the New York Declaration on Forests, which called for countries to begin restoring 350 million hectares (864 million acres) of degraded forest by 2030.

As global decision makers, governments, and communities join the effort to restore degraded land new questions about the economics of restoration have emerged. Critics argue that the time horizons are too long, the costs are too high, and the benefits are too few to justify public or private expenditures on restoration. At the country level, governments are questioning whether the restoration activities proposed and promoted by government officials and non-governmental organizations are likely to be adopted by the people who manage agricultural and forest landscapes. At the site level, policy makers are interested in the effects that payments for ecosystem services would have on rotational farming systems.

The objective of this dissertation is to address these questions. The specific contributions are described in detail in the body of each chapter. The work contained in the body of this manuscript reflects the combined experience and knowledge gained over four years of working with global decision makers, national governments, and local communities on restoration issues across the world and across multiple institutions.

The first chapter of this dissertation estimates the net benefit of achieving the Bonn Challenge by restoring 350 million hectares of degraded forestland. The chapter presents one estimate of the potential net benefits of fulfilling the target using a novel methodology that combines a global database of forest values with an ecological model of forest restoration. The net benefit of meeting Bonn Challenge is estimated with and without the value of public ecosystem goods and services, to argue that failures in the market for public environmental goods and services reduce the incentive for countries to invest in restoration and that fulfilling the goals of initiatives like the New York Declaration and the Bonn Challenge depend in part on internalizing these external values. In additional, the net benefit is estimated under different social discount rates, reflecting different social discounting philosophies, to show that our assumptions about the future and about how future generations will discount the future have a direct impact on how much land will be restored under the initiative. The results suggest that achieving the Bonn Challenge would generate a net benefit of between 0.7 and 9 trillion USD over a two-hundred year time horizon. The results show that restoration can create benefits that exceed its costs and that the benefits of restoration are greater when the social discount rate is lower. The social discount rate drives the predicted land restoration rate and suggests that the Bonn Challenge is only likely to be met when the benefits and costs of restoration are discounted at the lowest rate found in the literature.

The second chapter of this dissertation turns its attention to evaluating two restoration activities that have been proposed in Rwanda, a country that has committed itself to restore 2 million hectares of degraded forest and agricultural land as part of the Bonn Challenge. Agroforestry and improved woodlot management have been proposed to restore the ecological and economic productivity of agricultural and forestland in Rwanda, but the activities have not been evaluated in terms of their financial profitability, profitability risk, or ecological impacts despite being significant factors that influence the adoption decisions of smallholder landowners who occupy the majority of land in the country. The chapter presents a methodology combining enterprise budgets, biological production functions, and Monte Carlo analysis in an expected utility framework to investigate the financial profitability, financial risk, and ecological impacts of the activities in a smallholder context in four provinces of Rwanda.

The chapter accounts for financial risk by characterizing the uncertainty over financial and ecological outcomes, including profitability, erosion, and carbon storage. The distributions of net present values of each activity under market and ecological uncertainty are estimated and compared using Stochastic Dominance and certainty equivalence criteria in order to rank the activities from the perspective of risk-averse smallholders. The results show that only moderately risk-averse smallholders would be likely to adopt agroforestry and very risk-averse smallholders would prefer agriculture. Internalizing the value of public ecosystem services does not change the result. The results also show that smallholders always prefer current woodlot management practices and that risk preferences do not influence the ranking. The rankings over woodlot management practices do not change when the value of public ecosystem services are internalized from the perspective of smallholders.

The third chapter of this dissertation develops a methodology to estimate the effect of a Payment for Ecosystem Service (PES) mechanism on the incentive to enhance soil fertility on a smallholder managed cultivation-fallow system. The soil fertility on smallholder managed agricultural land in Sub-Saharan Africa is declining because smallholders are abandoning the practice of fallowing in favor of continuously cultivating land. Researchers have suggested that PES mechanisms could be an effective policy tool for encouraging smallholders to use more efficient forms of fallowing to maintain soil fertility because fallowed land provides a number of public benefits that are not accounted for in private decision-making. Yet, the effect that PES would have on a smallholder incentives for managing a cultivation-fallow system are not well understood and the ability of PES to enhance soil fertility has not been studied in the economic literature. To address this gap this chapter develops an analytic model of the cultivation-fallow system to provide some intuition behind the incentives facing smallholders managing such a system and it extends this analysis by developing a numerical model to provide further evidence on the effect PES would have on a cultivation-fallow system in a real-world setting using a case study from Zambia. The results show that payments for fallowing lead to higher soil fertility levels in all cases.

ACKNOWLEDGEMENTS

This dissertation is the culmination of a seven and half year process to push my understanding and application of environmental and natural resource economic theory. It also represents the end of the single longest endeavor I've ever embarked upon: earning a Ph.D. During this time I've had two relationships, experienced the failure of a research project, lived and worked in Switzerland, traveled the world, wrote two dissertation proposals and a dissertation, and gained full-time employment with an international conservation organization and left said employment to finish this dissertation.

Nothing about the process has been easy and I've faced serious moments of self-doubt and existential crises. If I were left to my own devices I almost certainly would not have finished.

While I did the all of the work in this dissertation, the motivation and courage to carry on came from all of my family, friends, and professional colleagues who continued to believe in me and support me during my most difficult moments. I am eternally indebted to all of you.

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CHAPTER 1: A COST BENEFIT ANALYSIS OF THE BONN CHALLENGE RESTORATION TARGET

1.1 Introduction, Problem Statement, and Contribution

Restoring the productivity of the world's degraded land has now become a global priority. Several United Nations conventions have adopted goals specifically focused on restoration.[1] The Bonn Challenge, a global initiative to begin restoring 350 million hectares of degraded forest and agricultural land by 2030 (IUCN, 2015), was created as a vehicle to achieve a myriad of restoration targets. The Bonn Challenge allows commitments made under the Bonn Challenge to count toward the sustainable development goals of the CBD, RIO +20, and the UNFCCC (UNFCCC, 2015). Since 2011 countries have committed to restore approximately 60 million hectares per year under the Bonn Challenge. If this trend continues until 2030 it puts the Bonn Challenge within reach of its 350 million hectare restoration target (IUCN, 2015).

Targets like the Bonn Challenge show that some governments are beginning to look for opportunities to invest in large-scale restoration. However, many countries that have large areas of degraded land have not made restoration commitments because the costs of restoration are thought to outweigh the benefits (MEA, 2005). Restoration is costly, but many of the conclusions that have been drawn about its net benefits are based on incomplete accounting. In many cases, cost-benefit analysis of restoration activities is based solely on financial values instead of a broader value set that reflects additional benefits restoration would create (De Groot et al., 2013; Barbier, 2007).

[1] The Convention on Biological Diversity's Aichi target 15 requires signatories to restore 15 percent of degraded ecosystems by 2020 (CBD, 2011); The UN Framework Convention on Climate Change has also adopted the global goal to slow, halt, and reverse forest cover and carbon loss; and the UN Convention to Combat Desertification is focusing on restoring unproductive lands (UNFCCC, 2013) (UNCCD, 2013).

As the international interest in restoration continues to grow it, is increasingly important to evaluate the efficiency of targets like the Bonn Challenge so that restoration's role in mitigating climate change and supporting sustainable development goals is properly documented (Rey Benayas et al., 2009). To support this effort this paper makes three primary contributions to the literature on the economics of restoration. First, it presents a methodology for valuing the net benefits of large-scale ecosystem restoration initiatives by estimating the net benefit of achieving the Bonn Challenge. Second, this paper shows why it is important to evaluate restoration decisions in a social welfare framework that values public goods, and that uses inter-generational time horizons and low social discount rates. The third contribution of this paper is to estimate the net benefit of achieving the Bonn Challenge restoration target under different social discounting regimes, different valuations of public goods, and different time horizons to see how they affect the argument for investing society's scarce resources in restoration.

1.2 Background and Literature Review

1.2.1 Restoration and the Failure in Market for Public Goods

Land degradation is a physical process leading to the long-term loss of ecosystem functions and productivity. Worldwide, land degradation affects between 15 and 43% of the Earth's land area to varying degrees (Daily, 1995; Archard et al., 2007; WRI, 2011; Bai et al., 2008). More than 15% of the Earth's land area is so degraded that it will not recover without direct intervention (Archard et al., 2007).[2]

Degradation has been linked to a number of social and environmental problems including poverty traps, food insecurity, greenhouse gas emissions, and other negative externalities

[2] See (Oldeman, Hakkeling, & Sombroek, 1991) estimated there are 1.5 billion hectares of degraded land, (Bai, Dent, Olsson, & Schaepman , 2008) estimated that there are 3.5 billion hectares, and (Daily G. C., 1995) suggested there could be as much as 5 billion hectares of degraded land.

(Barbier, 1997; Barbier, 2010; Bai et al., 2008). Global estimates suggest that these problems cost society between \$6.3 and \$10.6 trillion per year (ELD, 2015). As the global population grows the demand for productive land will increase. Restoring the productivity of degraded forestland can meet some of this demand while also keeping existing areas of forestland from being degraded or deforested (Costanza et al., 2014).

While the productivity of degraded land can be restored, failures in land-use markets and differences between private and public decision makers reduce incentives to restore it. In some cases land is left degraded because the sizeable public benefits of restoration are not captured by land managers or accounted for by decision makers (Daily, 1995; Nkonya et al., 2011). Indeed, the restoration literature has shown that a landowner's decision to restore degraded land depends on capturing the value of public ecosystem goods and services, like carbon sequestration (Schiappacasse et al., 2012; Hofer, 2010; Goldstein et al. 2008; Birch et al., 2010).

By definition, public goods are non-rival and non-excludable. This means that one person can consume the good without affecting another's ability to do so. It also means no one can be stopped from consuming the good. These characteristics reduce the incentive for private producers to create public goods because there is no mechanism to recover the costs of production. As a result, the values of public goods are rarely included in private land-use decision-making, leading to excessive land conversion and exploitation at a global scale (MEA, 2005).

The failure in the market for public goods leads to inefficient levels of deforestation and removes incentives for forest restoration, as shown in Figure 1.1. The marginal net social benefit of environmental public goods provided by a forest ecosystem of area S is given by the curve

MSB_F. The values of these services, measured as the aggregated willingness to pay for them less the cost to produce them, is given by the area underneath the MSB_F curve. The marginal net social benefits of marketed outputs of alternative land uses, such as agriculture, are given by the curve MSB_A. Assuming the forest ecosystem is completely intact, a social planner in charge of maximizing social welfare would convert an area of S^* to alternative land use, which is the point at which the marginal benefit of alternative land uses equals the marginal social cost of lost public environmental goods and services from the forest ecosystem.

This outcome assumes that the marginal social benefit of environmental public goods from the forest ecosystem are valued. If the social planner does not know or does not value the social benefit of forest ecosystem goods and services then only the marginal benefits of marketed outputs of alternative land uses will be considered in the land use decision. As Figure 1.1 implies, this situation would lead to the entire forest ecosystem being converted to alternative land uses.

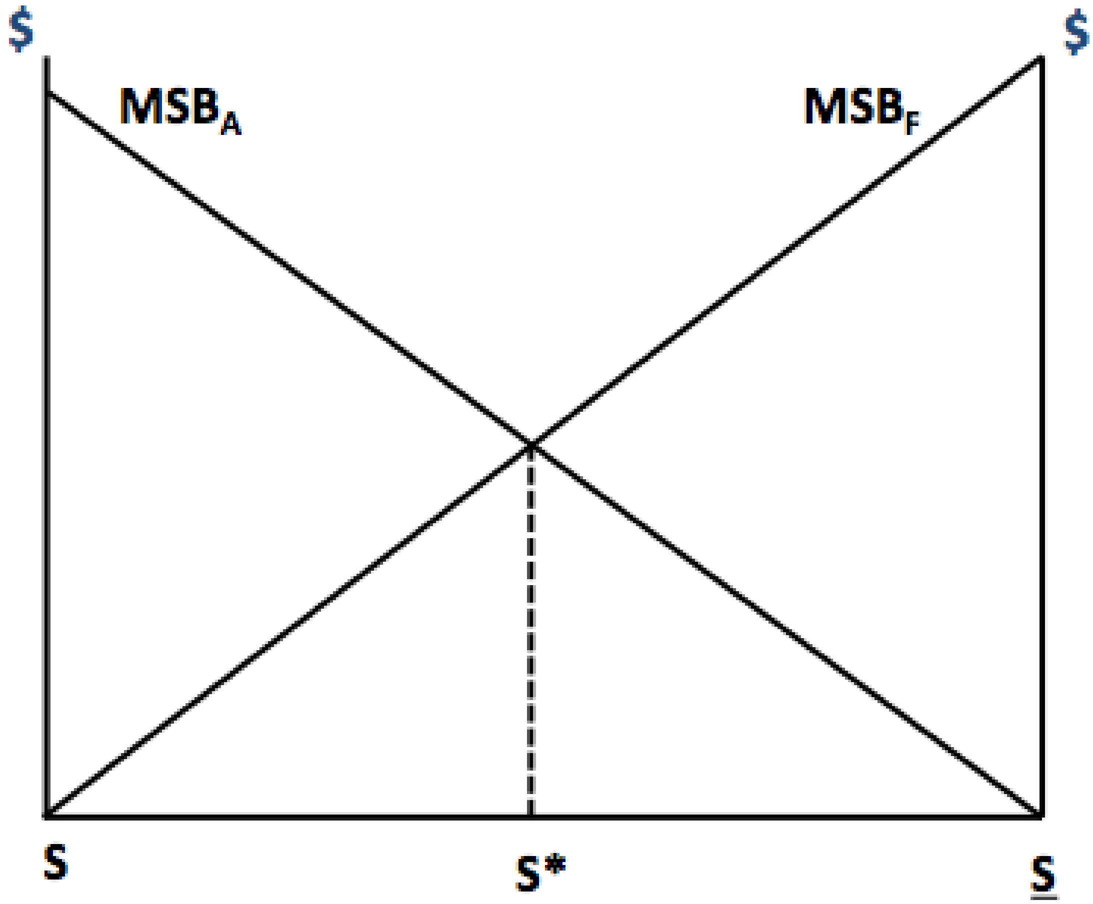


Figure 1.1: Failures in the market for public goods leads to inefficient levels of deforestation and removes incentives for forest restoration. Adapted from Barbier (2007).

When land managers fail to account for and value the public goods and services provided by forest ecosystems too little land will be restored and too much land will be degraded. If the forest ecosystem in Figure 1.1 had been completely converted to an alternative land use there would be no incentive for the private landowner or the social planner to restore any of the land area back to forest. However, if the value of public goods and services provided by the forest ecosystem were accounted for, then a social planner would maximize social welfare by restoring an area of $(\underline{S} - S^{*})$ from the alternative land use back to forest.

1.2.1 Restoration, Time Horizons, and Social Discount Rates

Land can also be left degraded because restoration decision-makers use time horizons that are too short and discount rates that are too high to produce the outcomes that society would prefer. It is well-documented in the climate change literature that over short time horizons there are few, if any, incentives to invest in climate mitigation since the benefits lag behind mitigating actions by at least 30 to 40 years (Palstev et al., 2013). In this regard, restoration is not substantially different. The benefits of restoring degraded land are received over periods of ten to more than two hundred years while the costs of restoration are often paid upfront. As a result, restoration decisions are particularly sensitive to the choice of the time horizon and the social discount rates that are used to evaluate them (Daily, 1995).

The economic literature on climate change has accounted for the ecological reality of climate change by treating it as a multi-generational investment problem. Studies estimating the impacts of climate change use social discount rates to account for the inter-generational nature of the investment decision. The social discount rate reflects the rate at which society would trade consumption in year t for consumption today (Arrow et al., 2012). There has been significant debate around the correct social discount rate because small changes in its value have large impacts on the welfare of future generations.

Social discount rates are based on the Ramsey formula are shown in Equation [1.1]:

$$\rho_t = \gamma + \eta * g_t \qquad [1.1]$$

Which equates the social rate of discount (ρ_t) with the sum of the social rate of time preference (γ) and the product of the marginal elasticity of utility (η) and the percentage increase in per capita consumption (g_t) (Lind 1982).

The value of the social rate of time preference parameter (γ) in the Ramsey formula is debated in the climate change literature. Its assigned value determines how the well-being of future generations should count relative to the well-being of current generations and this makes its value contentious (Goulder and Williams, 2012). Some economists (e.g. Stern, 2007 and Heal, 2003) argue that γ should equal near-zero on the grounds that the welfare of future generations should only be discounted by the small probability of extinction. Some economists argue that values of γ equal to near-zero imply that current generations should make excessive capital investments for the benefit of future generations who are likely to be wealthier than the current generation (Arrow, 1999). And others have argued that the social rate of time preference should be determined by empirical observations of the returns on alternative investments (Nordhaus, 2007). Indeed, empirical evidence of investor behavior suggests that the value of γ ranges between 2 and 5%, but there is no agreement that the preferences of investors and consumers should reflect intergenerational preferences of investments that take place over time horizons greater than fifty years (Pindyck, 2013).

A vein of empirical literature on social discounting has emerged that attempts to solve these disagreements. This literature uses results from behavioral economics to suggest that people use declining discount rate structures (DDR) to discount inter-generational streams of costs and benefits (Hepburn, 2006; Gollier et al., 2008). DDR schedules allow decision makers to put relatively more weight on investments that improve the welfare of future generations (Gollier et al., 2008). They are also appropriate for evaluating social policy decisions because they can give more representation to future generations without reducing the representation given to current generations (Pearce et al., 2003). The debate over the appropriate form and value of the social discount rate and the value of society's social rate of time preference continue to this day without reaching any consensus. It is outside the scope of this paper to formulate a position on which social discount rate value is ‘correct’ or which estimation methods are best. However, it is clear that the choice of social discount rate and its implied moral judgments about inter-generational equity will impact the decision to invest in restoration due to the long periods of time required to restore degraded forest land. While this paper does not take a position on these issues it contributes to the literature on the economics of restoration by exploring the impact that different social discounting philosophies have on the decision to invest in restoration.

1.3 Costs and Benefits of Restoration

Noting that the failure in the market for public goods and the private framing of restoration decisions incentivizes too little land to be restored (leaves too much land degraded), the question is whether or not investing public resources in restoration increases social welfare by improving the net flow of ecosystem goods and services (Pagiola, 2004). Several global studies have demonstrated the significant and measurable economic value of natural ecosystems (Costanza et al., 1997; De Groot et al., 2013; Costanza et al., 2014; Chiabai et al., 2011).

Costanza et al. (2014) used benefit transfer methods to estimate the value of ecosystem goods and services that have been lost due to the conversion of ecosystems. The authors estimated that ecosystem conversion from 1997 to 2011 resulted in the loss of between $4.3 trillion and $20.2 trillion worth of ecosystem services per year. Chiabai et al. (2011) used benefit transfer methodologies to construct a global database of forest ecosystem service values, including wood forest products (WFPs), carbon, recreation, and passive values for different world regions to estimate how social welfare could change under different amounts of deforestation. They found that between 2000 and 2050 the net change in ecosystem service values could range between +$270 trillion USD and -$1,180 trillion USD depending on the land use scenario.

Two global level studies have estimated the net benefits of restoration activities, but there are no studies in the literature looking at the net benefit of global restoration initiatives like the Bonn Challenge. Daily (1995) used an ecological model of restoration to conduct the first global study estimating the benefits of restoring degraded land. She used the model to estimate the percentage of potential direct instrumental value (PDVI) - which she defined as the yields of direct benefits like agriculture, forestry, industrial, and medicinal products - that could be recovered by restoring the world's degraded land over a twenty-five year time horizon. The study estimated that 5% of the global PDVI value that was lost due to land degradation could be recovered through restoration, but the estimate did explicitly value the costs and benefits of such an effort.

De Groot et al. (2013) conducted a cost-benefit analysis of restoration activities for nine major biomes using meta-analysis to create a global database of restoration costs and ecosystem service values (\$ ha^{-1} $year^{-1}$) for pristine biomes. They used the reference ecosystem service values to estimate the value of restoring degraded biomes by assuming restoration could restore 30%, 60%, and 75% of the original ecosystem service values of the pristine biome over a twenty-year time horizon. They discounted the costs and benefits using social discount rates of -2% and 8% to reflect pessimistic and optimistic future economic outlook, respectively. The negative discount rate was used to reflect the possibility that future environmental conditions will deteriorate to such a point that the marginal value of ecosystem goods and services in the future will be greater than they are today. They found that the benefit-cost ratio for restoration ranged from 0.05 to 35 depending on the biome and scenario.

Despite the interest in the value of public ecosystem services, to date no global level studies have evaluated the net benefits of restoration targets accounting for their public ecosystem service benefits. As the above studies show, the benefits of restoration are typically not differentiated according to whether they are private or public, which masks the failures in the market for public ecosystem goods and services that prevent investment in restoration (Blignaut et al., 2014). Even at smaller geographic scales few studies in the restoration literature have estimated both the costs and the ecosystem service benefits of restoration because the predominant trend has been to evaluate restoration projects with ecological rather than economic indicators (Aronson et al., 2010; Wortley et al., 2013). When costs and benefits are estimated they have been inconsistently reported in the literature, creating a lack of comparable values across restoration studies (De Groot et al., 2013). In the literature, discount rates have been applied arbitrarily without any theoretical or philosophical discussion. This can lead to potentially erroneous conclusions about the efficiency of restoration because the benefits are received over decades and centuries while the costs are typically paid upfront (Daily, 1995).

1.4 Model

This chapter reports on a cost-benefit analysis (CBA) of the Bonn Challenge target of restoring 350 million hectares of degraded land. Forest ecosystem service values from Chiabai et al. (2011) are combined with an ecological model of restoration from Daily (1995) to estimate forest stock values for degraded and restored *managed* and *natural* forests in eleven world regions.[3]

[3] The FAO defines natural forests as forests of native species, where there are no clearly visible indications of human activities and the ecological processes are not significantly disturbed, whereas managed forests are defined as forests predominantly composed of trees established through planting and/or deliberate seeding.

Following Chiabai et al. (2011) the stock value of *managed forests* is defined to include wood forest products and non-wood forest products (WFPs and NWFPs) and carbon sequestration. The stock value of *natural forests* is defined to include recreational values, passive use values, and the value of carbon sequestration.

WFPs and NWFPs are most closely related to pure private goods in that they are both rival and excludable. Carbon sequestration and passive use values can be thought of as public goods in the sense that once carbon is sequestered or passive use values are created the benefits extend to everyone and no one can be excluded from them. Recreation is regarded as a private good. Public land management agencies have begun charging user fees to recreationists in an attempt to recover the costs of their management activities and these changes have moved recreation toward the category of a private good by introducing an element of excludability (Quinn, 2002). For example, in many parts of the world ski resorts have turned public forest into rival and excludable resources. This analysis proceeds by regarding recreation as a private good that may not be rival, but it certainly excludable in that it is possible for land managers to charge a fee for access to the resource.

The analysis assumes property rights are assigned and enforced so that the value of excludable goods is completely captured by land managers. The net present values (NPV) of degraded and restored forests in each world region are calculated by multiplying the stock values by the area of degraded forest type. The mode is used to estimate the NPV of achieving the Bonn Challenge as well as the amount of land that could be profitably restored. The sensitivity of the results to different types of discounting is also tested following the inter-generational discounting literature.

Restoration targets imply an assumption that current land use allocations are inefficient and that adopting new land uses that restore the services that have been lost will enhance social welfare. CBA provides a framework for testing these hypotheses and the NPV metric allows direct comparisons to be made between alternative scenarios (Boardman et al., 2006). If the scales of each scenario are equal, larger NPVs are considered to be more efficient than alternatives with smaller NPVs so long as the benefits and costs can be distributed amongst stakeholders in a way that improves the welfare of some without reducing the welfare of others (weak Pareto criterion). Following the restoration literature this analysis assumes that land will be restored whenever the benefits of restoring it are greater than the costs of maintaining it in a degraded state (Daily, 1995, De Groot et al., 2013).

1.4.1 Scenarios

The stock values for managed and natural forests are estimated for four categories of degradation (light, moderate, severe, extreme) and four categories of restoration (restored from light, moderate, severe, and extreme degradation) based on the following four scenarios:

1. Baseline: Costs and benefits are discounted at a social discount rate of 4.3% following Nordhaus (2011), and the stock values of degraded and restored forests include both private and public benefits.
2. Private goods only: Costs and benefits are discounted at a social discount rate of 4.3% following Nordhaus (2011), and the stock values of degraded and restored forests include only private benefits.
3. Stern: Costs and benefits are discounted at a rate of 1.3% following Stern (2007), and the stock values of degraded and restored forests include both private and public benefits.

4. Declining discount rate: Costs and benefits are discounted using a declining discount rate profile following Gollier et al. (2008), and the stock values of degraded and restored forests include both private and public benefits.

Based on the Bonn Challenge target, as much as 350 million hectares of degraded forest in 12 world regions and 6 biomes could be restored if the benefits outweigh the costs.

1.4.2 Calculating Benefits of Restored Forests

Each category of degradation is associated with an expected loss in productivity. Daily (1995) reported that degraded land lost 10%, 25%, 50%, and 100% of its annual productive value depending on whether it was lightly, moderately, severely, or extremely degraded, respectively. The stock values of degraded forests are estimated by discounting the annual ecosystem service flow values from pristine forests from Chiabai et al. (2011) by the expected loss in productivity reported by Daily (1995) and shown in Table 1.1. The figures in parentheses in Table 1.1 are the estimated lengths of time, measured in years, needed for each category of degraded land to completely recover its productivity.

Chiabai et al. (2011) calculated the annual values of four ecosystem services for 11 world regions as shown in Table 1.2. The values of wood forest products (WFPs) are defined as industrial roundwood, wood pulp, recovered paper, sawnwood, wood-based panels, paper and paper board, and wood fuel (Chiabai et al., 2011).

Table 1.1: Degradation's effect on the time needed to restore potential direct instrumental value		
	Forest Type	
Degradation status	Natural	Managed
Lightly degraded	10% (3-10 years)	10% (3-10 years)
Moderately degraded	25% (10 - 20 years)	25% (10 - 20 years)
Severely degraded	50% (50 - 100 years)	50% (50 - 100 years)
Extremely degraded	100% (> 200 years)	100% (> 200 years)
Restored	0% (N/A)	0% (N/A)

Notes: The first number is the percent reduction in productivity compared to pristine forest and the figures in parentheses are the estimated number of years it will take to restore the land to its full productive potential. All estimates from (Daily, 1995).

Non-wood forest products (NWFPs) are defined as “all goods of biological origin, as well as services, derived from forest or any land under similar use, excluding wood in all its forms” (Chiabai et al, 2011; FAO, 1999). The authors estimated the value of WFPs and NWFPs ha^{-1} $year^{-1}$ by dividing the country totals reported by FAOSTAT by the forest area of each country.

Recreational and passive use values were estimated with meta-regression analyses of 59 and 27 estimates respectively. The studies that supplied the estimates were retrieved from the EconLit, EVRI, and IUCN databases on forest studies and represented mostly European based research. The meta-regressions modeled WTP values for recreation and passive uses as a function of income, forest size from the study area, population, temperature, a dummy variable for whether or not the forest was in the Boreal, and a constant term. The results from the meta-regression model were transferred to the world regions contained in this paper by fitting the model to data representing each world region.

Carbon values were taken from Myneni et al. (2001) who estimated the carbon stocks for forests in Canada, North America, China, Japan, Russia, Finland, Sweden, Eurasia, and South Eastern Asia and Gibbs et al. (2007) who reported carbon stock estimates for forests in Brazil, Latin

America, Sub-Saharan Africa, and Central and Northern Asia. The price of carbon comes from Nordhaus (2011) who estimated the social costs of carbon at \$43.46 ton^{-1}.

The flow values in Table 1.2 are combined with the estimates of the expected losses in productivity shown in Table 1.1 to estimate the stock values of degraded forests in each world region following equation [1.3]:

$$PVF_{ifd} = \sum_{t=0}^{200} \delta^t \left[Pro_d * (wfp_{if} + P_c * C_{if} + rec_{if} + passive_{if})\right] \qquad [1.3]$$

where PVF_{ifd} is the present value of degraded forest per unit area. The indices i, f, and d, represent world regions, management type, and degradation category, respectively.

Table 1.2: Forest Ecosystem Service Flow Values for Pristine Forests from Chiabai et al. 2011

	Ecosystem good or service \$ ha^{-1} $year^{-1}$			
Region	WFPs & NWFPs	Recreation	Passive use	Carbon sequestration
AFR	\$2,814	\$652	\$971	\$317
ANZ	\$2,729	\$9	\$349	\$144
BRA	\$1,406	\$118	\$1,288	\$304
ECA	\$426	\$135	\$428	\$84
EUR	\$533	\$955	\$1,329	\$98
JPK	\$2,841	\$4,504	\$3,174	\$120
NAM	\$2,252	\$725	\$3,418	\$103
OAS	\$3,176	\$964	\$250	\$124
OLC	\$1,033	\$359	\$281	\$141
RUS	\$164	\$65	\$218	\$84
SOA	\$1,577	\$3,844	\$1,160	\$245

Degradation category is defined as lightly degraded, moderately degraded, severely degraded, and extremely degraded. The discount factor is represented by δ. Pro_d represents the loss in productivity associated with each category of degradation following 1.2. The variables wfp_{if}, c_{if}, rec_{if}, and $passive_{if}$, are the annual flow values of wood forest products, carbon sequestration, recreation and cultural services associated with forest type f in world region i. P_c is defined as the social cost of carbon, which is set equal to \$43.46 ton^{-1} based on estimates from Nordhaus (2011). All values are held constant over the time horizon of the analysis. For managed forests, rec_{if}, and $passive_{if}$ equal zero following the definition of managed forests discussed above. Similarly, the value of wfp_{if} is equal to zero for natural forests.

The stock values of restored natural and managed forests are calculated for each world region and degradation type following equation [1.4].

Where $PVRF_{ifd}$ denotes the present value of restored forest. The variable $cost_{dt}$ is defined as the upfront fixed cost needed to restore the forest and it is defined so that $cost_{dt=1}$> 0 and $cost_{dt>1} = 0$.

$$PVRF_{ifd} = \sum_{t=0}^{200} \delta^t \left[Pro_{dt}(wfp_{if} + P_c * C_{if} + rec_{if} + passive_{if} - cost_{dt}\right] \quad [1.4]$$

Equation [1.4] is similar to equation [1.3] except that Pro_{dt} becomes a function not only of the degradation category, but also of time, to reflect restoration's ability to restore ecosystem function over time. During the restoration time-horizon Pro_{dt} assumes the value for the category of degradation following equation 1.1. During the restoration time horizon, Pro_{dt}, follows a linear recovery path along the restoration time-horizon. Once the land is restored, Pro_{dt} equals one. Daily (1995) defines Pro_{dt} for lightly degraded land is equal to 0.9 and can be restored to 1 over a period of 7 years, after which the productivity of the land is fully restored. Under this example Pro_{dt} for t = 1,2,..,6 would equal [0.90, 0.92, 0.95, 0.96, 0.98, 0.99] while for t=7,8,…,200 Pro would equal 1.

The net present value of restoring degraded forest is estimated by:

$$NPV_{ifd} = PVRF_{ifd} - PVF_{ifd} \tag{1.5}$$

Equation [1.5] represents the increase in the stock value of forests per hectare net of both fixed costs and opportunity costs. The estimates from equation [1.5] are multiplied by the areas of degraded forest in Figure 1.2 below to estimate the net benefit of achieving the Bonn Challenge target.

1.4.3 Restoration Costs

Restoration requires raw materials, such as tree seedlings, fencing, and fertilizer. Additionally, restoration requires labor to prepare the sites, raise and transport the tree seedlings, and take care of other activities including forest extension and support services. The total cost of these inputs depends on how degraded a site is and how difficult it will be to restore. Additionally, costs vary according to geography, degradation category, the objectives and contexts of specific restoration activities, and the types of restoration methods that are used (TEEB, 2009).

Estimates of restoration costs are lacking in the restoration literature because a standard for reporting does not exist (Blignaut et al., 2014). Previous studies have dealt with the lack of information using very simple approximations of restoration costs. The Economics of Ecosystems and Biodiversity (TEEB) project estimated the average costs ($\$ ha^{-1}$) of restoring forests in different ecosystems by adding 20% to restoration cost values found in a literature review (TEEB, 2009). The study assumed what it called ‘typical’ forest restoration costs varied between \$2390 ha^{-1} and \$3450 ha^{-1}. De Groot et al., 2013 estimated the range of restoration costs

(\$ ha^{-1}) for forest ecosystems based on a survey of 32 studies reporting restoration cost estimates. The authors used that information to estimate the net benefit of restoring forest ecosystems under scenarios where 100% and 75% of the maximum observed restoration cost was required to restore the ecosystem. Neither study attempted to represent restoration costs for different degrees of degradation or different geographies.

This study overcomes the lack of cost estimates by constructing a sample of restoration costs from the World Bank project database and the TEEB report (TEEB, 2009). The World Bank project database was queried for projects that included the word 'restoration' and the cost estimates from the TEEB report were reported from the peer-reviewed literature. 1.3 presents the estimated restoration costs from the two sources. The reported per hectare restoration costs varied from a low of \$214 ha^{-1} to \$3,790 ha^{-1}. The average across the sample in Table 1.3 was \$1,276 ha^{-1}.

Acknowledging the complex nature of restoration costs, but lacking precise cost estimates for each category of degradation in each world region, the following cost structure is used to represent the fixed costs (e.g. materials and labor) of restoration displayed in Table 1.4: the average cost from Table 1.3 is assumed to represent the cost of restoring moderately degraded land since it is the most prevalent degree of degradation found throughout the world and it would seem likely that this fact makes it the most likely type of land to be restored. The cost of restoring lightly degraded land is assumed to be equal to the average cost minus one Standard Deviation, while the cost of restoring severely and extremely degraded land is assumed to be equal to the average cost plus one and two Standard Deviations respectively.

Table 1.3: Selected Forest Landscape Restoration Cost Estimates

Country	Forest Interventions	Source	Cost Ha^{-1}
Albania	Fencing, supplemental seeding, and silviculture	World Bank Project Database	$406
Australia	Re-establishment of native eucalyptus trees after intensive grazing - Active restoration	Dorrough and Moxham, 2005	$728
Australia	Re-establishment of native eucalyptus trees after intensive grazing - Passive restoration	Dorrough and Moxham, 2005	$214
Benin	Community management of forest resources, sustainable fuelwood production	World Bank Project Database	$1,362
Brazil	Reforest unmanaged grasslands native trees	World Bank Project Database	$714
Brazil	Restoration of Atlantic Forest	Instituto Terra, 2007	$1,950
Canada	Thinning treatments and conifer planting to restore coastal riparian forests	Anon, 2007	$1,650
Columbia	Silviculture	World Bank Project Database	$2,042
Columbia	Reforestation	World Bank Project Database	$2,102
Kazakhstan	Irtysh pine forest reforestation	World Bank Project Database	$3,790
Kazakhstan	Planting of dry Aral seabed	World Bank Project Database	$905
Madagascar	Plantation establishment in Masoala Corridors	Halloway et al., 2009	$279
Madagascar	Restoration of rainforest corridors by sourcing and planting trees	Halloway and Tingle, 2009	$938
Maldova	Afforestation/reforestation	World Bank Project Database	$1,122
Peru	Restoration and revegetation of Polylepis forest	Jameson and Ramsey, 2007	$570
Ukraine	Reforestation	World Bank Project Database	$1,651
Average			$1,276
Standard Deviation			$887

Source: Area and cost sources can be found at http://www.worldbank.org/projects and Teebweb.org. All costs are reported in 2015 dollars and restoration activities occurred between 2005 and 2011.

Table 1.4: Estimated Restoration Costs for Different Categories of Restoration			
Extreme	Severe	Moderate	Light
\$3,051	\$2,163	\$1,276	\$389

1.4.4 Discounting forms

The NPVs of the above scenarios are estimated with social discount rates from two parameterizations of the Ramsey formula with values from Nordhaus (2011) and Stern (2007), shown in Table 1.5.

Table 1.5: Parameters Used in Ramsey Formula for Exponential Discounting			
Author	$\boldsymbol{\gamma}$	g	$\boldsymbol{\eta}$
Nordhaus (2011)	3%	1.3%	1
Stern (2007)	0.1%	1.3%	1

The parameter values reflect different views about the utility rate of discount and lead to two different social discount rates. Stern (2007) argues that the marginal utility of future generations should only be discounted by the probability that the human race dies out. In contrast, Nordhaus (2011) adopts a more conservative approach where the marginal utility of current and future generations is discounted at 3%. Both authors agree on the values of utility rate of discount ($\boldsymbol{\eta}$) and the rate of growth between the end of the time horizon and the present (g_t). The sensitivity of the results to a DDR schedule is estimated using the results from Gollier et al. (2008) as discussed in section 1.2.2. The global average certainty-equivalent declining discount rate profile from Gollier et al. (2008) is shown in Table 1.6.

Table 1.6: Global Weighted Average Declining Discount Rate Profile

	Weight type	
Year	GDP	Population (2005)
1	0.04226	0.04362
20	0.03252	0.03848
40	0.02957	0.03578
60	0.02694	0.03302
80	0.02479	0.03041
100	0.02305	0.02795
150	0.02003	0.02295
200	0.01833	0.01979

Note: Schedule from Gollier et al. (2008)

The certainty-equivalent DDR profiles are slightly different depending on which weighting function is used. The population weighted DDR profile is always higher than the GDP weighted DDR profile, but the differences between the two values generally amount to less than 0.5%. In both cases the discount rate starts at approximately 4.2% to 4.3% and gradually declines to a terminal value of 1.8% to 1.9% over a two hundred year time horizon. This analysis adopts the GDP weighted DDR profile because, as Gollier et al. (2008) explain, it is more reliable since the nine countries in the study account for about 46.8% of the world's GDP and only 28.34% of the world's population.

1.5 Area estimates

The distribution of the 350 million hectares of degraded managed and natural forests across all six biomes are estimated as follows. Chiabai et al. (2011) report the distribution of all managed and natural forest by biome for twelve world regions for the year 2000. The distribution of degraded forest is estimated by combining the data from Chiabai et al. (2011) with data from Bai et al. (2008) who report the ratio of degraded to non-degraded land for each world region.[4] Combining these two data sets provides an approximate estimate of the total area of degraded managed and natural forest. Degraded land is assumed to be equally distributed across land cover types (e.g. agriculture, forestry, etc) and biomes. Therefore, if 20% of a world region's land area is degraded, then 20% of managed and natural forests are degraded in each biome.

Table 1.7: Percent of degraded area in different extents of degradation by world region

	Category of degradation as percent of total degraded area			
World region	Light	Moderate	Strong	Extreme
South Asia and India	39.0%	46.0%	14.0%	1.0%
Africa	35.1%	38.9%	25.0%	1.0%
Latin America and Caribbean	43.1%	46.6%	10.3%	0.0%
North America	18.0%	81.0%	1.0%	0.0%
Europe	27.7%	66.0%	4.9%	1.4%
Australia, New Zealand, and Pacific Region	94.0%	4.0%	2.0%	0.4%

Note: Data from Olderman et al. (1991)

Once the total area of degraded forest is estimated, the Global Assessment of Soil Degradation (GLASOD) database is used to estimate the total area of degraded managed and natural forest affected by each category of degradation (i.e. light, moderate, severe, and extreme) in each world region (Olderman et al., 1991). The GLASOD database reports the percent of degraded land in each category of degradation across six world regions. The percent of degraded land under each category of degradation is multiplied by the estimated area of degraded managed and natural

[4] To estimate the area of degraded forest in each management type and biome, data from (Bai, Dent, Olsson, & Schaepman , 2008) was used to estimate the averge degraded area of each world region.

forests in each forest biome and world region to estimate the distribution of degradation across forest management and biomes.[5] Table 1.7 shows the percentage of degraded land (as a percentage of total degraded area) in each category of degradation six world regions.

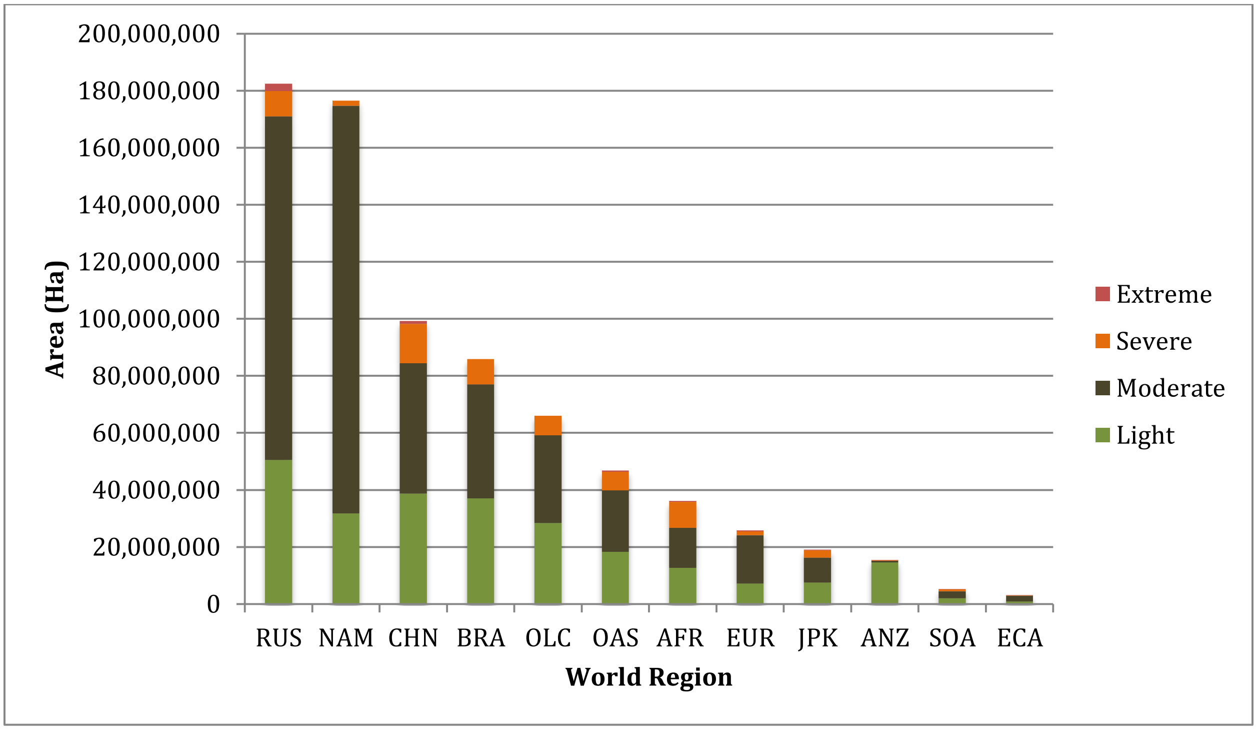


Figure 1.2: The distribution of degraded land potentially restored under Bonn Challenge is divided into four types of degradation that range from light to extreme. An estimated 761 million hectares of managed and natural forests are degraded to some degree. Across all world regions light and moderate degradation are the most common degradation states. Note: Distribution calculated with data from Chiabai et al. (2011); Olderman et al. (1991); Bai et al. (2008).

Figure 1.2 shows the areas of degraded natural and managed forests in each world region that would be restored in pursuit of the Bonn Challenge 2.0 target. In total, an estimated 761 million hectares of managed and natural forests are degraded to some degree. To achieve the Bonn Challenge target of restoring 350 million hectares, approximately 46% of degraded natural and

[5] The GLASOD survey reports are of degradation by severity for six world regions and Chiabai et al. (2011) reports forest area for 12 world regions, including the six world regions reported in the GLASOD survey. The additional regions reported in Chiabai et al. (2011) are a finer categorization of world regions reported in GLASOD and therefore the data are combined by assuming the SOA world region in the GLASOD data represent the SOA, JPK, CHN, and OAS world regions in Chiabai et al. (2011) and also that the OLC in GLASOD represents OLC and BRA in the Chiabai et al. (2011) data and that EUR in the GLASOD data represent EUR, RUS, and ECA in the Chiabai et al. (2011) data.

managed forests would need to be restored. This represents slightly more than a tripling of targets such as the CBD's Target 15, which calls for the restoration of 15% of degraded land in each biome. The majority of degraded forests are either lightly or moderately degraded. Only small fractions are severely or extremely degraded.

1.6 Results

Table 1.8 shows the net present value of restoring a hectare of degraded managed and natural forest for each world region under the baseline scenario. The results show that both managed and natural forests could be efficiently restored in pursuit of the Bonn Challenge target on net. The benefit-cost ratio of restoring 350 million hectares of degraded forest is 7.45 and social welfare would increase by $2.28 trillion. The largest NPVs of restoration are in the natural forests of Japan (JPK), South Asia and India (SOA), and North America (NAM), respectively. In fact, the NPV of restoring natural forests is greater than that of managed forests for every world region except Africa (AFR) and Australia and New Zealand (ANZ), which have relatively large WFP values compared to other regions. The forests in other world regions produce relatively large recreation and passive use values, which make natural forests more valuable than managed forests.

However, severely and extremely degraded land in many world regions, including Brazil (BRA), Australia and New Zealand (ANZ), and Russia (RUS), would cost more to restore than the benefits it would generate. As a result land managers may be unwilling to restore this land. If restoration is limited to the world regions and management types where the NPV of restoration is greater than zero, 255 million hectares of land would be restored under this scenario. If land managers were provided with a subsidy to make them indifferent between leaving the land degraded and restoring it, the total subsidy would cost $23.9 billion.

Table 1.8: NPV of Restoration by World Region and Management type (Public and private goods, 4.3% discount rate)

Region	Management Type	Average NPV Per Hectare				Hectares						
		Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration	Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration	Net Benefit ($ Billions)	Total Cost ($ Billions)	Benefit Cost Ratio
AFR	Managed	$2,685	$6,071	$3,204	$676	245,286	271,841	174,705	6,988	$2.87	$0.71	
AFR	Natural	$2,580	$5,847	$3,012	$533	5,582,884	6,187,299	3,976,413	159,057	$62.64	$16.12	
ANZ	Managed	$3,912	$8,706	$5,467	$2,365	193,347	8,228	3,291	823	$0.85	$0.09	
ANZ	Natural	-$2	$305	-$1,748	-$3,019	6,489,545	275,922	110,460	27,615	-$0.20	$3.04	
BRA	Managed	$219	$779	-$1,341	-$2,715	226,704	245,114	54,222	0	$0.17	$0.42	
BRA	Natural	$2,133	$4,887	$2,187	-$82	16,775,025	18,137,266	4,008,881	0	$133.20	$31.15	
CHN	Managed	$2,732	$6,173	$3,291	$741	1,545,213	1,822,559	554,692	39,621	$17.33	$3.51	
CHN	Natural	$7,408	$16,209	$11,911	$7,173	16,245,990	19,161,936	5,831,894	416,564	$503.41	$36.95	
ECA	Managed	-$170	-$55	-$2,058	-$3,250	54,688	130,303	9,674	2,764	-$0.05	$0.17	
ECA	Natural	$399	$1,165	-$1,009	-$2,468	333,209	793,927	58,992	16,855	$0.96	$1.05	
EUR	Managed	$164	$660	-$1,444	-$2,792	901,551	2,148,101	159,612	45,604	$1.21	$2.85	
EUR	Natural	$3,924	$8,730	$5,488	$2,380	2,379,656	5,669,940	420,950	120,271	$61.43	$7.53	
JPK	Managed	$5,070	$11,192	$7,602	$3,958	450,370	531,205	161,671	11,548	$9.50	$1.02	
JPK	Natural	$14,952	$32,402	$25,818	$17,551	2,969,046	3,501,952	1,065,811	76,129	$186.72	$6.75	
NAM	Managed	$3,863	$8,601	$5,377	$2,298	1,412,112	6,354,505	78,451	0	$60.53	$6.71	
NAM	Natural	$7,725	$16,890	$12,496	$7,610	13,188,497	59,348,238	732,694	0	$1,113.44	$62.65	
OAS	Managed	$3,582	$7,997	$4,858	$1,910	637,158	751,520	228,723	16,337	$9.43	$1.45	
OAS	Natural	$1,737	$4,036	$1,456	-$628	7,756,690	9,148,916	2,784,453	0	$54.45	$16.89	
OLC	Managed	$1,025	$2,508	$143	-$1,607	342,904	370,750	81,947	0	$1.29	$0.64	
OLC	Natural	$573	$1,538	-$689	-$2,229	12,724,397	13,757,701	3,043,383	0	$26.36	$23.63	
RUS	Managed	-$467	-$693	-$2,605	-$3,659	985,472	2,348,056	174,325	49,807	-$2.72	$3.12	
RUS	Natural	-$168	-$52	-$2,055	-$3,248	22,264,708	53,049,484	3,938,522	1,125,292	-$18.27	$70.46	
SOA	Managed	$2,520	$5,717	$2,900	$449	254,275	299,915	91,278	6,520	$2.62	$0.58	
SOA	Natural	$9,514	$20,730	$15,793	$10,070	678,114	799,826	243,425	17,388	$27.05	$1.54	
Total										$2,254	$299	7.54

Table 1.9 shows the net present value of restoring a hectare of degraded managed and natural forest for each world region under the baseline scenario without including the value of public goods. The results show that both managed and natural forests could be efficiently restored in pursuit of the Bonn Challenge target on net. The benefit-cost ratio of restoring 350 million hectares of degraded forest is 1.88 and social welfare would increase by $562 Billion. The largest NPVs of restoration are still in the natural forests of Japan (JPK), South Asia and India (SOA), and North America (NAM), respectively. In this scenario every world region except China (CHN), Japan (JPK), and India (SOA), has land in some category of degradation where the costs of restoring that land exceed the benefits. If land in this scenario were only restored when the benefits outweighed the costs only 197 million hectares would be restored. If land managers were provided with a subsidy to make them indifferent between leaving the land degraded or restoring it, the total subsidy would cost $130.3 billion or $566 ha^{-1}.

Table 1.9: NPV of Restoration by World Region and Management type (Private Goods Only, 4.3% discount rate)

Region	Management Type	Average NPV Per Hectare				Hectares				Net Benefit	Total Cost	Benefit Cost Ratio
		Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration	Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration			
AFR	Natural	$567	$1,526	-$700	-$2,237	5,582,884	6,187,299	3,976,413	159,057	$9.47	$16.12	
ANZ	Managed	$3,882	$8,640	$5,411	$2,323	193,347	8,228	3,291	823	$0.84	$0.09	
ANZ	Natural	-$745	-$1,292	-$3,119	-$4,042	6,489,545	275,922	110,460	27,615	-$5.65	$3.04	
BRA	Managed	$194	$724	-$1,388	-$2,750	226,704	245,114	54,222	0	$0.15	$0.42	
BRA	Natural	-$523	-$813	-$2,709	-$3,736	16,775,025	18,137,266	4,008,881	0	-$34.37	$31.15	
CHN	Managed	$2,712	$6,130	$3,254	$714	1,545,213	1,822,559	554,692	39,621	$17.20	$3.51	
CHN	Natural	$4,192	$9,306	$5,983	$2,750	16,245,990	19,161,936	5,831,894	416,564	$282.47	$36.95	
ECA	Managed	-$184	-$86	-$2,084	-$3,269	54,688	130,303	9,674	2,764	-$0.05	$0.17	
ECA	Natural	-$489	-$741	-$2,647	-$3,689	333,209	793,927	58,992	16,855	-$0.97	$1.05	
EUR	Managed	$143	$615	-$1,482	-$2,820	901,551	2,148,101	159,612	45,604	$1.08	$2.85	
EUR	Natural	$1,187	$2,857	$444	-$1,384	2,379,656	5,669,940	420,950	120,271	$19.05	$7.53	
JPK	Managed	$5,040	$11,127	$7,546	$3,916	450,370	531,205	161,671	11,548	$9.45	$1.02	
JPK	Natural	$8,437	$18,419	$13,809	$8,589	2,969,046	3,501,952	1,065,811	76,129	$104.92	$6.75	
NAM	Managed	$3,837	$8,545	$5,328	$2,261	1,412,112	6,354,505	78,451	0	$60.13	$6.71	
NAM	Natural	$717	$1,847	-$424	-$2,031	13,188,497	59,348,238	732,694	0	$118.74	$62.65	
OAS	Managed	$3,561	$7,952	$4,819	$1,882	637,158	751,520	228,723	16,337	$9.38	$1.45	
OAS	Natural	$1,206	$2,897	$478	-$1,358	7,756,690	9,148,916	2,784,453	0	$37.19	$16.89	
OLC	Managed	$995	$2,443	$88	-$1,649	342,904	370,750	81,947	0	$1.25	$0.64	
OLC	Natural	-$30	$244	-$1,801	-$3,058	12,724,397	13,757,701	3,043,383	0	-$2.51	$23.63	
RUS	Managed	-$484	-$731	-$2,638	-$3,683	985,472	2,348,056	174,325	49,807	-$2.84	$3.12	
RUS	Natural	-$631	-$1,045	-$2,908	-$3,884	22,264,708	53,049,484	3,938,522	1,125,292	-$85.32	$70.46	
SOA	Managed	$2,457	$5,583	$2,785	$363	254,275	299,915	91,278	6,520	$2.56	$0.58	
SOA	Natural	$7,089	$15,524	$11,323	$6,734	678,114	799,826	243,425	17,388	$20.10	$1.54	
Total										$562	$298	1.88

Table 1.10: NPV of Restoration by World Region and Management type (Private and public goods, 1.3% discount rate)													
Region	Management Type	Average NPV Per Hectare				Hectares				($, Billions)		Benefit Cost Ratio
		Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration	Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration	Net Benefit	Total Cost	
AFR	Natural	$9,485	$22,857	$30,179	$28,136	5,582,884	6,187,299	3,976,413	159,057	$318.86	$16.12	
ANZ	Managed	$13,924	$33,373	$45,060	$42,411	193,347	8,228	3,291	823	$3.15	$0.09	
ANZ	Natural	$881	$2,473	$1,334	$465	6,489,545	275,922	110,460	27,615	$6.56	$3.04	
BRA	Managed	$1,617	$4,217	$3,801	$2,832	226,704	245,114	54,222	0	$1.61	$0.42	
BRA	Natural	$7,995	$19,327	$25,184	$23,344	16,775,025	18,137,266	4,008,881	0	$585.62	$31.15	
CHN	Managed	$9,991	$24,055	$31,874	$29,762	1,545,213	1,822,559	554,692	39,621	$78.14	$3.51	
CHN	Natural	$25,573	$60,969	$84,113	$79,873	16,245,990	19,161,936	5,831,894	416,564	$2,107.56	$36.95	
ECA	Managed	$321	$1,147	-$543	-$1,335	54,688	130,303	9,674	2,764	$0.16	$0.17	
ECA	Natural	$2,217	$5,637	$5,812	$4,761	333,209	793,927	58,992	16,855	$5.64	$1.05	
EUR	Managed	$1,431	$3,777	$3,179	$2,235	901,551	2,148,101	159,612	45,604	$10.01	$2.85	
EUR	Natural	$13,962	$33,462	$45,186	$42,532	2,379,656	5,669,940	420,950	120,271	$247.09	$7.53	
JPK	Managed	$17,783	$42,514	$57,997	$54,821	450,370	531,205	161,671	11,548	$40.60	$1.02	
JPK	Natural	$50,713	$120,527	$168,394	$160,723	2,969,046	3,501,952	1,065,811	76,129	$764.36	$6.75	
NAM	Managed	$13,761	$32,986	$44,512	$41,886	1,412,112	6,354,505	78,451	0	$232.53	$6.71	
NAM	Natural	$26,630	$63,473	$87,656	$83,272	13,188,497	59,348,238	732,694	0	$4,182.45	$62.65	
OAS	Managed	$12,823	$30,764	$41,369	$38,870	637,158	751,520	228,723	16,337	$41.39	$1.45	
OAS	Natural	$6,674	$16,197	$20,754	$19,095	7,756,690	9,148,916	2,784,453	0	$257.74	$16.89	
OLC	Managed	$4,300	$10,574	$12,797	$11,462	342,904	370,750	81,947	0	$6.44	$0.64	
OLC	Natural	$2,796	$7,009	$7,752	$6,622	12,724,397	13,757,701	3,043,383	0	$155.59	$23.63	
RUS	Managed	-$669	-$1,198	-$3,861	-$4,519	985,472	2,348,056	174,325	49,807	-$4.37	$3.12	
RUS	Natural	$326	$1,158	-$527	-$1,319	22,264,708	53,049,484	3,938,522	1,125,292	$65.16	$70.46	
SOA	Managed	$9,283	$22,379	$29,502	$27,487	254,275	299,915	91,278	6,520	$11.94	$0.58	
SOA	Natural	$32,591	$77,595	$107,640	$102,442	678,114	799,826	243,425	17,388	$112.15	$1.54	
Total										$9,230	$298	30.94

Table 1.11: NPV of Restoration by World Region and Management type (Private and public goods, Declining discount rate)													
Region	Management Type	NPV Per Hectare				Hectares				Net Benefit ($ Billions)	Total Cost ($ Billions)	Benefit Cost Ratio
		Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration	Light Restoration	Moderate Restoration	Severe Restoration	Extreme Restoration			
AFR	Natural	$4,399	$10,389	$10,731	$8,370	5,582,884	6,187,299	3,976,413	159,057	$132.84	$16.12	
ANZ	Managed	$6,548	$15,288	$16,690	$13,723	193,347	8,228	3,291	823	$1.46	$0.09	
ANZ	Natural	$234	$893	-$821	-$2,006	6,489,545	275,922	110,460	27,615	$1.62	$3.04	
BRA	Managed	$590	$1,705	$167	-$1,118	16,775,025	18,137,266	4,008,881	0	$41.49	$31.15	
BRA	Natural	$3,678	$8,745	$8,730	$6,573	226,704	245,114	54,222	0	$3.45	$0.42	
CHN	Managed	$4,644	$10,947	$11,409	$8,980	16,245,990	19,161,936	5,831,894	416,564	$355.50	$36.95	
CHN	Natural	$12,188	$28,145	$32,330	$27,771	1,545,213	1,822,559	554,692	39,621	$89.16	$3.51	
ECA	Managed	-$37	$275	-$1,573	-$2,681	333,209	793,927	58,992	16,855	$0.07	$1.05	
ECA	Natural	$880	$2,367	$972	-$395	54,688	130,303	9,674	2,764	$0.36	$0.17	
EUR	Managed	$500	$1,500	-$82	-$1,342	2,379,656	5,669,940	420,950	120,271	$9.50	$7.53	
EUR	Natural	$6,566	$15,330	$16,741	$13,768	901,551	2,148,101	159,612	45,604	$42.15	$2.85	
JPK	Managed	$8,416	$19,547	$21,871	$18,377	2,969,046	3,501,952	1,065,811	76,129	$118.15	$6.75	
JPK	Natural	$24,359	$55,893	$66,082	$58,088	450,370	531,205	161,671	11,548	$52.02	$1.02	
NAM	Managed	$6,469	$15,108	$16,471	$13,526	13,188,497	59,348,238	732,694	0	$994.02	$62.65	
NAM	Natural	$12,700	$29,312	$33,749	$29,045	1,412,112	6,354,505	78,451	0	$206.84	$6.71	
OAS	Managed	$6,015	$14,073	$15,212	$12,395	637,158	751,520	228,723	16,337	$18.09	$1.45	
OAS	Natural	$3,038	$7,286	$6,956	$4,980	7,756,690	9,148,916	2,784,453	0	$109.60	$16.89	
OLC	Managed	$1,889	$4,667	$3,770	$2,118	342,904	370,750	81,947	0	$2.69	$0.64	
OLC	Natural	$1,161	$3,006	$1,749	$303	12,724,397	13,757,701	3,043,383	0	$61.45	$23.63	
RUS	Managed	-$517	-$818	-$2,902	-$3,875	22,264,708	53,049,484	3,938,522	1,125,292	-$70.67	$70.46	
RUS	Natural	-$35	$280	-$1,566	-$2,675	985,472	2,348,056	174,325	49,807	$0.22	$3.12	
SOA	Managed	$4,302	$10,166	$10,459	$8,127	678,114	799,826	243,425	17,388	$13.74	$1.54	
SOA	Natural	$15,585	$35,891	$41,752	$36,234	254,275	299,915	91,278	6,520	$18.77	$0.58	
Total										$2,203	$298	7.38

Table 1.10 and 1.11 show the net present value of restoring a hectare of degraded managed and natural forest for each world region under the Stern and DDR scenarios, respectively. The results show that both managed and natural forests could be efficiently restored in pursuit of the Bonn Challenge target on net in both scenarios. The benefit-cost ratio of restoring 350 million hectares of degraded forest is 30.94 for the Stern scenario and 7.38 for the DDR scenario. Social welfare would increase by $9.23 trillion in the Stern scenario and $2.20 trillion in the DDR scenario. In both scenarios there are far fewer areas of forestland where the benefits of restoration are less than the costs. If restoration were limited to the world regions and management types where the NPV of restoration is greater than zero, 341 million hectares of land would be restored under the Stern scenario and 318 million hectares would be restored under the DDR scenario. If land managers were provided with a subsidy to make them indifferent between leaving the land degraded or restoring it, the total subsidy would cost $680 million ($75 ha^{-1}) under the Stern scenario and $1.6 billion under the DDR scenario ($50 ha^{-1}).

Framing restoration as a social welfare problem is important for achieving targets like the Bonn Challenge as Table 1.12 shows. Several authors have argued that investments in restoration, like those to ameliorate climate change, should use longer time horizons and lower social discount rates on moral grounds (Stern, 2007; Gollier et al., 2008). Scenario analysis shows that longer time horizons and lower discount rates make restoring forestland more likely because more weight is given to benefits that occur in the future. This can be seen by the larger areas of land that would be restored under both management types (Table 1.12).

Table 1.12: NPV of Scenarios Under Different Time Horizons, Public Good Values, and Discount Rates				
	Scenario			
Area (Hectares)	Baseline	Baseline without Public Goods	Stern	DDR
Managed Forest (200 year horizon)	20,469,251	20,469,251	20,913,679	20,653,776
Natural Forest (200 year horizon)	235,270,361	177,233,300	320,331,403	297,911,765
Total (200 year horizon)	255,739,612	197,702,551	341,373,231	318,712,454
Managed Forest (25 year horizon)	19,507,152	17,920,952	19,583,796	19,523,961
Natural Forest (25 year horizon)	228,182,019	154,021,594	233,267,936	231,279,821
Total (25 year horizon)	247,689,171	171,942,546	252,851,732	250,803,782

The largest area of land that would be restored occurs when the value of public goods is included under a 200 year time horizon and a social discount rate of 1.3%. Under this scenario the benefits of restoring forestland would outweigh the costs on approximately 342 million hectares of land. Under the declining discount rate (DDR) scenario the benefits of restoring forestland would outweigh the costs on approximately 318 million hectares of land. When the value of public goods is excluded from the decision making process, the time horizon is shortened to 25 years and the social discount rate is increased to 4.3% to reflect the baseline scenario without public good values, only 171.9 million hectares of land are restored. This is approximately 49% of the land area called for under the Bonn Challenge and New York Declaration on Forests.

Table 1.13: Sensitivity Analysis of Baseline Scenario to Changing Discount Rates

Time Horizo	Metric	Discount Rate		
		10%	15%	20%
25 years	NPV (Billions, USD)	$222.68	$109.38	$53.63
	Land area restored (Ha)	191,189,194	169,827,478	157,816,807
200 years	NPV (Billions, USD)	$342.96	$121.49	$56.64
	Land area restored (Ha)	198,537,229	177,584,168	157,816,807

The Baseline scenario's sensitivity to different discount rates was tested under both time horizons and reported in Table 1.13. The results from the sensitivity analysis show that the NPV of achieving the Bonn Challenge is very sensitive to changes in the discount rate. For example, moving from a 10% to 15% discount rate decreases the NPV of achieving the Bonn Challenge from $222.68 billion to $109.38 billion under a twenty-five year time horizon. The land area that would be restored is much less sensitive to the change in the discount rate. The same move from a 10% to 15% discount rate changes the land area that would be restored from approximately 191 million hectares to 170 million hectares over a twenty-five year time horizon. Moving from a 15% discount rate to a 20% discount rate further reduces the NPV of achieving the Bonn Challenge from approximately $109 million to $53 million and reduces the area that would be restored from approximately 170 million hectares to 158 million hectares over a twenty-five year time horizon. The same sensitivity effects are present when the time horizon is expanded to two hundred years.

1.7 Discussion and Conclusion

As the area of productive land that makes modern life possible continues to shrink through conversion and degradation more thought must be given to the idea of restoring degraded land at large scales. Conventionally, countries have viewed restoration as a cost to be paid rather than an investment that returns tangible benefits (Bullock et al., 2011). In part, this is due to the public-good nature of restoration and the inter-generational dimension of restoration investments, which leads to incorrect conclusions about the benefits of investing in restoration when viewed through a financial accounting lens. As this analysis has shown, restoration can increase the flow of ecosystem goods and services in excess of the costs required to do so and the benefits of restoration are greater when the social discount rate is lower.

Still, making restoration profitable from a financial accounting perspective remains difficult. Evaluating international restoration targets like the Bonn Challenge strictly on their financial benefits leads to recommendations that undervalue the full set of benefits that come with achieving restoration targets. The international case for restoration has been made through arguments stating that open-access, missing markets, failures in the market for public goods, externalities, and other market failures have allowed landowners and management agencies to ignore the value of public goods, resulting in too much land being deforested and degraded. It should come as no surprise that evaluating restoration targets with the same financial values that led landowners and land managers to convert too much forestland to agriculture or degrade too much forest through mismanagement is not likely to favor restoration. As shown here, when the value of these goods and services are accounted for the cost-benefit analysis shows that restoring degraded land on large scales is beneficial net of both fixed and opportunity costs. What is needed, then, are policy approaches such as payments for ecosystem services (PES) that transfer the benefit of public goods to landowners, thereby creating an incentive for them to adopt land uses that are more in-line with societal values.

The results also show that governments need to think about restoration through an inter-generational lens since the time horizons to completely restore the functionality of degraded land can span multiple centuries. When the welfare of future generations is discounted at high rates too little restoration occurs relative to the amount called for by international targets like the Bonn Challenge. As the discount rate is reduced, either by adopting a DDR schedule or a more generationally equitable assumption about the social rate of time preference the area that will be restored approaches the level called for by the international community. These results also suggest that private investment, which is widely discussed as being an important part of achieving the Bonn Challenge, will play a limited role in fulfilling international targets because of the high rates of discount associated with private investment decisions.

CHAPTER 2: USING EXPECTED UTILITY THEORY TO EXAMINE THE PREFERENCES OF RISK AVERSE SMALLHOLDER FARMERS FOR TWO RESTORATION ACTIVITIES IN RWANDA

2.1 Introduction, problem statement, contribution

In 2011, the country of Rwanda committed to begin restoring the economic and ecological productivity of 2 million hectares of degraded land by 2020 under the Bonn Challenge and as part of a broader development strategy designed to secure livelihoods, reduce poverty, and promote economic development (IUCNa, 2015). As part of the country's commitment several land use activities have been recommended to restore the productivity of agricultural and forest land. To date the activities have not been evaluated in terms of their financial profitability, financial risk, or ecological impacts despite being significant factors that influence the adoption decisions of risk-averse smallholders who occupy the majority of land in the country (Clay et al., 1998; Jacobson and Petrie, 2009; Bidogeza et al., 2015).

Historically, activities have not been evaluated because there is a lack of financial and environmental data to support such an effort. To address this problem rapid, rigorous, and objective approaches need to be developed. The methodologies should be able to evaluate activities in terms of their financial profitability, financial risk, and ecological impacts from the perspective of risk-averse smallholders without the need for large data collecting exercises. Combining enterprise budgets, biological modeling, and Monte Carlo analysis in an expected utility framework is one way to address this challenge. This approach characterizes both the likely and extreme financial and ecological outcomes of adopting the activities.

As a result, this approach allows restoration activities to be evaluated across a range of potential outcomes and also to ask whether risk-averse smallholders would be likely to adopt the activities given the outcomes that would be expected.

This paper contributes to the literature on the economics of restoration in the following ways. Previous studies evaluating conservation and restoration activities in the East African region have accounted for the effects of restoration activities on crop yields. This paper extends their work by accounting for a broader set of ecological outcomes, including erosion, carbon sequestration, and timber production (Rosenstock et al., 2014). Additionally, while previous studies have accounted for risk created by variability in market, prices this paper accounts for the risk created by the variation of climactic, ecological, and market price variables associated with each activity. As a final contribution, this paper demonstrates how expected utility theory can be used to evaluate the variability in key variables in a way that allows the activities to be ranked from the perspective of a risk-averse smallholder.

2.2 Background

The Republic of Rwanda is a densely populated developing country of 10.5 million people with an annual Gross Domestic Product (GDP) of 1,302 billion Rwandan Francs (RwF) (NISR, 2012; NISR, 2014)[6]. The country occupies a land area of 2.4 million hectares, of which approximately 2 million hectares are under cultivation or permanent pasture (Habiyambere et al., 2009). The livelihoods of most of Rwanda's population are organized around subsistence agriculture and fuelwood energy production.

[6] 1 USD = 750 RwF

An estimated 90% of the population and 70% of the country's land area are devoted to subsistence agricultural production, while a further 16% of land area is allocated to fuelwood and timber production to meet the country's energy needs (Habiyambere et al., 2009).

The agricultural sector in Rwanda is primarily based on low input non-mechanized cultivation of food crops for home consumption (NIS, 2008).[7,8] Drought, nutrient mining, and eroded soils are cited as the primary causes of food insecurity in Rwanda after lack of land (Habiyambere et al., 2009). Approximately 98% of cultivated land is rain-fed, exposing farmers to significant climate-related food security risk from droughts and variability in rainfall. Soil nutrient mining is also a problem since many farmers cultivate land continuously to compensate for the low productivity of farm operations. Land scarcity also drives many farmers to cultivate steep slopes with inadequate ground cover to prevent erosion. As much as 40% of cultivated land in Rwanda is at risk of severe erosion and requires anti-erosion investments before cultivation begins. Estimates suggest that as much as 20 short tons of soil per hectare is lost each year, flowing directly into rivers and streams that are not adequately protected (Habiyambere et al., 2009).

The forest cover of Rwanda has been shaped by the country's food and energy needs in recent years. The area of natural forests inside of national parks and forest reserves has declined since 1990, largely as a result of increased demand for agricultural land and fuelwood plantations (Musahara, H., 2006). The government has protected the remaining areas of intact natural forest and has even led efforts to increase their size through afforestation activities. While the expansion of timber and fuelwood plantations has been necessary to meet demand for energy and

[7] Only 4% of cultivated land is devoted to cash crops, such as coffee or tea, while 67% of cultivated land is dedicated to the production of food crops (NIS, 2008).

[8] In 2008 16% of households purchased inorganic fertilizers and only 10% of households purchased improved seeds, while 75% purchased traditional seeds (Habiyambere et al., 2009).

timber, most plantations are considered inefficient due to poor management practices (Belgian Development Agency, 2012).

Several activities have been recommended by government ministries, non-governmental organizations (NGOs), and development agencies to improve the productivity of smallholder-owned land in the country (Belgian Development Agency, 2012; WVI, 2015). The activities are generally designed to increase crop and timber yields, reduce soil erosion, increase forest cover, and conserve biodiversity. Most recently the Rwandan Natural Resource Authority in partnership with the International Union for the Conservation of Nature (IUCN) and the World Resources Institute (WRI) led a stakeholder centered process to identify 'restoration' activities that would be suitable on smallholder owned land in Rwanda. At the conclusion of the process two land-use activities were recommended to improve the production of different services on agricultural and forestland. The Rwandan Natural Resource Authority is considering promoting the technologies to smallholders through extension and outreach campaigns, but their financial profitability, financial risk, and ecological impacts have not been evaluated to date.

Organizations in Rwanda have a long history of promoting conservation and restoration activities. Still, smallholder adoption of new activities has been low despite the fact that the activities have been tested on research sites with positive economic and ecological effects reported on average (Clay et al., 1998; Bidogeza et al., 2015). Smallholders in Rwanda have been shown to be risk averse, meaning that they consider both the mean financial return and its variation in their land use decision making. Risk-averse smallholders could prefer activities with lower average returns if the variation of the returns was smaller than alternative land uses (Jacobson and Petrie, 2009; Bidogeza et al., 2015). Clay et al. (1998) support this view and suggest that household adoption decisions in Rwanda are a function of four primary factors: 1. Financial returns; 2. Riskiness of financial returns; 3. Physical returns to investment; and 4. Capacity to invest. It is important to evaluate new activities in terms of these factors because failing to do so can lead to the promotion of risky activities that have low probabilities of adoption or lead to poor ecological and financial outcomes for risk-averse smallholders (Rosenstock et al., 2014).

2.3 Theory of decision making under risk and expected utility

The theory of decision-making under uncertainty assumes smallholder agriculturists face choices among risky agricultural activities. The goal of decision-making under uncertainty is to find activities that risk-averse smallholders would prefer to alternative activities. The expected utility framework therefore attempts to understand how smallholders make trade-offs among risky alternatives (Anderson, 1974).

Risky agricultural activities can produce a number of financial outcomes. From the perspective of the smallholder it is uncertain which outcome will occur at the time they have to decide which activity they will adopt. Such risky activities are often thought of as lotteries (Mas-Colell, 1995). A lottery is an offering of distinct financial outcomes, $X = (x_1, x_2, \ldots, x_n)$, with the probability of achieving those outcomes given by $\Pi = (\pi_1, \pi_2, \ldots, \pi_n)$. If only one outcome is possible each time the lottery is played then it follows that $\sum_{i=1}^{n} \pi_i = 1$. This simply says that if X is a set that contains all possible outcomes and Π is a set that contains the probabilities associated with each outcome then one of those outcomes must occur. The average payoff of a lottery is described by the lottery's expected value, which is equal to $\sum_{i=1}^{n} \pi_i x_i$. The expected value is simply a weighted sum of the financial outcomes where the weights are the probabilities that a given outcome occurs. The expected value of a lottery is the size of the financial outcome that a smallholder would receive, on average.

Economic theory suggests that individuals do not evaluate lotteries based on the expected values. Instead smallholders are concerned with the expected utility that a given lottery would provide. Von Neumann and Morgenstern (1953) introduced expected utility theory to account for this important distinction in how individuals make decisions under uncertainty. Expected utility theory is based on four axioms of rational behavior under uncertainty: 1. Preferences are continuous; 2. Preferences are complete; 3. Preferences are transitive; and 4. Preferences are independent. The first axiom simply says that individual preferences, or ordering, over risky alternatives are not affected by small (i.e. marginal) changes in probabilities. The second axiom says that faced with a choice between several lotteries individuals can place them in an order from their most preferred option to least preferred option. The third axiom states that the preferences of individuals are consistent in that if A $\succcurlyeq B$ and B $\succcurlyeq C$ then A $\succcurlyeq C$. The fourth axiom states that an individual's preference over any two lotteries is independent of the outcomes with other possible lotteries.

These axioms lead to the expected utility theorem, which says that if an individual's preferences satisfy the above axioms their preferences can be represented by a utility function of the expected utility form given by:

$$L_1 \succcurlyeq L_2 \leftrightarrow \sum_{i=1}^{n} \pi_i^1 \, u(x_i^1) \succcurlyeq \sum_{j=1}^{n} \pi_j^2 \, u(x_j^2) \qquad [2.1]$$

Where L_1& L_2 are the lotteries associated with different risky agricultural activities and $u(\cdot)$ is a utility function where $u\colon x \rightarrow \Re$. If an individual's preferences satisfy the above utility form then there exists a $U\colon \mathcal{L} \rightarrow \Re$ such that $L_1 \succcurlyeq L_2 \leftrightarrow U(L_1) \succcurlyeq U(L_2)$ for all L $\in \mathcal{L}$. This utility function

U is said to be the so-called von Neuman Morgenstern expected utility function and is represented by:

$$U(L) = \sum_{i=1}^{n} \pi_i \, u(x_i) \ \forall \ \text{L} \in \mathcal{L} \qquad [2.2]$$

The von Neumann-Morgenstern expected utility function is unique in that it can order the preferences of risk-averse smallholders over different risky agricultural activities. To understand what this means in terms of the consequences for decision making think of competing agricultural activities as monetary lotteries that create payoffs ($ ha^{-1}) given by X. The activities can be described by cumulative density functions $F: \mathfrak{R} \rightarrow [0,1]$. That is, for any x ∈ X, F(x) is the probability that the realized payoff is less than or equal to x. If the activities have a probability density function f(x) then F(x) = $\int_{-\infty}^{x} f(t)d(t)$ for all x.

Under the expected utility theorem utility values u(x) for non-negative amounts of money are assigned with the property that any $F(\cdot)$ can be evaluated by a utility function $U(\cdot)$ of the form $U(F) = \int u(x)dF(x)$. $U(\cdot)$ is the von Neuman-Morgenstern expected utility function that represents the mathematical utility expectation over the realizations of x and the values of $u(x)$, which are defined as the utility received from receiving a specific amount of money with certainty (Mas-Colell, 1995). This functional form represents a utility function that is sensitive to both the mean and variance of financial outcomes associated with a given activity. It is important to note that the analytical power of the expected utility representation of smallholder preferences depends on specifying the functional form of $u(\cdot)$.

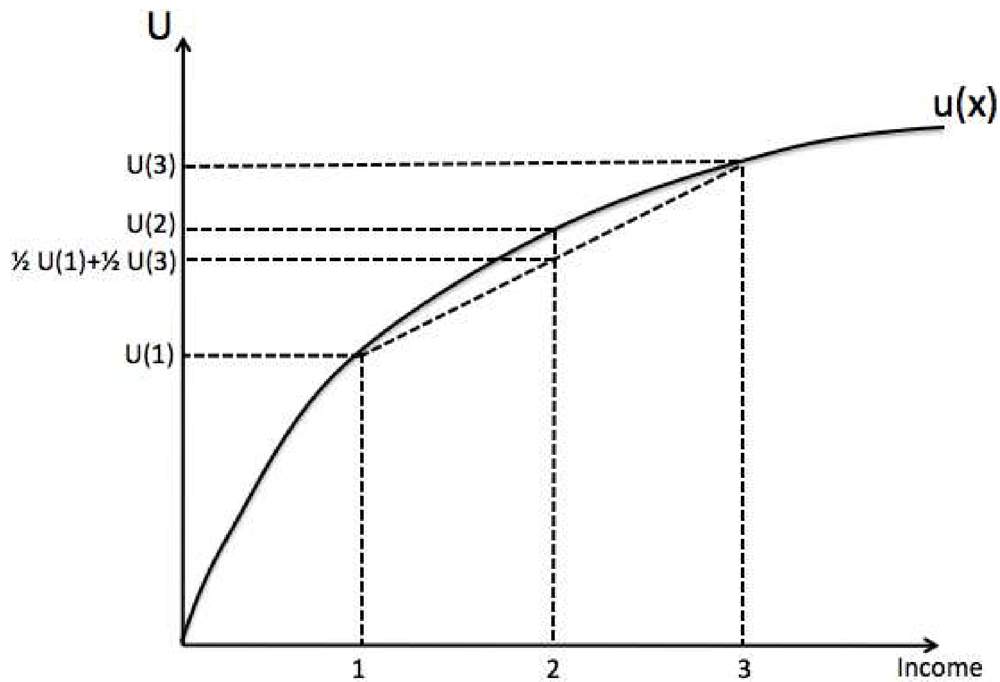


Figure 2.1: In the figure the von Neumann Morgenstern utility is shown for a risk-averse individual comparing the utility or receiving \$2 with certainty or taking a gamble that pays \$1 with probability ½ and \$3 with probability ½. The expected value of the gamble is \$2, which is the same value of the income being offered with certainty, but the expected utility of the gamble is less than \$2 because the individual's utility function is concave meaning the individual is averse to risk and would prefer an income with certainty over the same income with risk.

Under this functional form an individual is said to be risk-averse if and only if:

$$u(\int x \, dF(x)) \geqslant \int u(x) dF(x) \text{ for all } F(\cdot) \qquad [2.3]$$

This identity is known as Jensen's inequality and says that the utility of the expected value of a lottery received with certainty is greater than the expected utility of the lottery. The inequality is a statement that an individual does not like risk. It is also equivalent to saying that the utility function $\boldsymbol{u}(\cdot)$ is concave or that marginal utility is decreasing with income (Mas-Colell, 1995). For risk-averse individuals the marginal utility of receiving an additional dollar of income is smaller than the absolute value of the change in utility that would result from losing a dollar of income. It follows that if an individual with a certain income of $2 was offered the opportunity to participate in a lottery that offered the possibility of gaining or losing $1 of income with equal probability they would not take it as shown in Figure 2.1. In this case the von Neumann-Morgenstern utility associated with this gamble is strictly less than the utility of the certain income.

Figure 2.1 also shows why it is important to characterize the distribution of payoffs associated with risky restoration activities and why those activities need to be evaluated within an expected utility framework. Restoration activities that have higher average payoffs (i.e. higher expected values) compared to current land uses may not be strictly preferred by risk-averse smallholders if the distribution of the payoffs comes with more uncertainty than current land uses.

2.4 Literature Review

In Sub-Saharan Africa (SSA) few economic studies of restoration activities have specifically accounted for risk from the perspective of smallholder landowners. Previous studies of restoration activities in SSA smallholder land use systems have used partial or full enterprise budgeting approaches to compare the marginal benefit of the activities with current land uses. Profitability studies of agroforestry from Zambia used partial and full enterprise budgets to show that agroforestry practices on farm test plots increased crop yields and profitability (Ajayi et al., 2008; Franzel et al., 2004). In a study of a conservation agriculture system designed to restore soil fertility on smallholder farmland in Kenya, the authors concluded that risk-averse farmers would prefer the conservation agriculture system because its discounted net present value ($ ha^{-1}) was higher than those from current land management practices (Guto et al., 2011). Indeed, the literature on restoration activities in SSA often uses the average financial returns to infer that restoration activities are likely to be adopted by risk-averse smallholders (Franzel et al., 1997).

However, these types of arguments only reflect the central tendency of the payoff distribution, which is one dimension of risk. They fail to account for higher distributional moments, which are also necessary for fully characterizing risk. Moreover, only risk neutral or risk seeking smallholders would strictly prefer restoration activities with larger average financial returns as compared to current land uses. Risk-averse smallholders, in contrast, would consider both the mean and variance of financial returns of restoration activities (Mas-Colell et al., 1995).

In response to these shortcomings, recent studies have begun accounting for risk through the use of enterprise budgets and Monte Carlo simulations to characterize both the central tendencies of distributions as well as their variance. Studies of conservation agriculture from East Africa (Rosenstock et al., 2014) and ecological restoration in South Africa (Crooks et al., 2013) used Monte Carlo simulations to characterize the risk from variation in crop yields, water yields, and livestock production in terms of Standard Deviations and Coefficients of Variation of the NPV. However, their approaches did not provide guidance on how risk-averse smallholders would rank the different activities.

Only one study has combined Monte Carlo simulations with an expected utility framework to characterize risk in a way that would allow restoration activities to be ranked by risk-averse smallholders. Djanibekov and Villamor (2014) used Monte Carlo simulations to characterize the financial risk created by the variability of timber yields and output prices for smallholder rubber plantations in Indonesia. The authors asked how Payments for Ecosystem Services (PES) for carbon and biodiversity habitat could decrease financial risk. Their study only characterized the risk from variations in market prices instead of doing a more complete accounting of risk that would have taken ecological variability into account.

This paper builds on previous studies evaluating restoration activities by combining enterprise budgets, biological modeling, and Monte Carlo analysis in an expected utility framework. This approach allows the two proposed restoration activities to be ranked from the perspective of risk-averse smallholders. Combining enterprise budgets with biological production functions can provide a more holistic picture of the impacts of a restoration activity and Monte Carlo simulations can characterize the risk that the impacts do not occur. Framing the results in an expected utility framework makes it possible to rank the activities from the perspective of a representative risk neutral or risk-averse smallholder agriculturist.

2.5 Methods

2.5.1 Identifying restoration activities in Rwanda

Beginning in June 2013 the Rwandan Natural Resource Authority (RNRA), in partnership with the International Union for the Conservation of Nature (IUCN) and the World Resources Institute (WRI), led a stakeholder-centered process to identify restoration activities that would be suitable on smallholder owned land in Rwanda (RNRA, 2014). Through the partnership, stakeholders from communities, NGOs, and government came together at four regional workshops held in the northern, southern, eastern, and western parts of the country to discuss what they hoped to achieve through the restoration process. Workshops were held in October 2013 in Kibuye (Western Province), Nyanza (Southern Province), Kigali (Kigali Province), and Ruhegeri (Northern Province). Approximately 30 – 50 officials from local government and local farmers and foresters participated in each workshop.

During the workshops stakeholders worked together to create a short list of the most relevant and feasible restoration activities for agriculture land and woodlots. Stakeholders described restoration activities by defining which tree species could be used, what their planting density would be, which crops would be used in agroforestry, and which management practices would be used on both agroforestry and improved woodlot sites. Government ministries, including the Rwandan Natural Resource Authority, the Ministry of Agriculture, the Rwandan Environmental Management Authority, and the Rwandan Development Board, and the Rwandan Ministry of Finance and Economic planning, helped to characterize the current land use management practices to establish baselines against which to compare the restoration activities.

In total, the stakeholder process identified two degraded land uses that would benefit from restoration:

1. Degraded maize agriculture
2. Poorly managed eucalyptus woodlots and plantations

Stakeholders also identified two restoration activities that could be used to improve the ecological and economic productivity of the above degraded land uses:

1. Agroforestry with maize
2. Improved management of existing woodlots for fuel wood and structural wood with spacing and erosion and fire-prevention best practices

Based on the current land uses and restoration technologies the following restoration transitions were identified:

1. Degraded maize agriculture → Agroforestry with maize

2. Poorly managed eucalyptus woodlots and plantations → Improved management of existing woodlots with spacing and erosion and fire-prevention best practices

The financial and ecological effects of each restoration transition were modeled by creating enterprise budgets and biophysical models of two ecosystem services based on the stakeholder characterizations of the land use practices discussed below.

2.5.2 Data collection

Data for the biological production functions were taken from a number of sources. Estimates of the mean annual increment of timber growth for Grevillea robusta, a common tree species used in agroforestry systems, were taken from Kalinganire (1996), while estimates for Eucalyptus tereticornis, the most common tree species used in fuelwood plantations, were taken from the Belgian Development Agency (2012). Provincial level monthly precipitation data from 2007 to 2009 was sourced from Meteo Rwanda. Provincial level soil erodibility and soil cover values were retrieved from a GIS database provided by the Rwandan Natural Resources Authority (RNRA). Provincial level slope estimates were taken from the 2008 Rwandan Agricultural Survey. Provincial level crop yield data for maize and beans for 2007 to 2009 were sourced from RNRA.

Data for the enterprise budgets were taken from a number of sources. During the regional workshops stakeholders reported the most likely tree species, stocking densities, crop types, and management practices associated with each degraded land use and restoration activity. Officials at the Rwanda Ministry of Agriculture and Animal Resources and the Rwanda Natural Resource Authority reported the average market prices for maize and fuelwood. The price of carbon was taken from the 2015 report on the state of the voluntary carbon market (Hamrick et al., 2015).

Stakeholders reported the cost of labor during the stakeholder workshops and information from the Ministry of Agriculture and Animal Resources suggests that labor does indeed have an opportunity cost because labor shortages are becoming more common when help is needed to prepare fields for cultivation during peak agricultural seasons (MINAGRI, 2010).

2.5.3 Biological production functions

This paper uses biological production functions to account for the ecosystem service effects of each restoration activity. The biological production functions are used to estimate the impact of the activities on timber and crop yields, erosion, and carbon sequestration.

2.5.3.1 Timber

This paper estimates the mean annual increment of timber growth for 1-hectare of agroforestry and woodlots using mean annual increment data for Grevillea robusta from (Kalinganire, 1996), and for Eucalyptus tereticornis from (Belgian Development Agency, 2012). Stakeholders reported that Grevillea robusta is the most common agroforestry species adopted by farmers and eucalyptus species are the most commonly grown species on fuelwood plantations (Belgian Development Agency, 2012). Annual timber yields were estimated by multiplying the mean annual increment for a single tree of each tree species by the stocking density of trees for each current land use and restoration activity following Table 2.1.

Table 2.1: Mean Annual Increments (MAI) for Grevillea Robusta and Eucalyptus Tereticornis for Representative Stocking Densities

Species	Single tree	300 trees ha^{-1} yr^{-1}	1100 trees ha^{-1} yr^{-1}	1600 trees ha^{-1} yr^{-1}	Source
Grevillea robusta	0.0048 (0.002)	1.44 (0.6)	-	-	Kalinganire, 1996
Eucalyptus tereticornis	0.0065 (0.001)	-	7.15 (1.1)	10.4 (1.6)	Belgian Development Agency, 2012

Notes: Standard Errors are in parentheses. Grevillea robusta was only considered in an agroforestry context with 300 trees ha^{-1}.

Each land use and restoration activity was assigned a stocking density for the most common tree species associated with the land use or restoration activity. Stakeholders widely reported that an additional 300 trees per hectare could be planted as part of an agroforestry enterprise. Eucalyptus woodlots are currently stocked at a density of 1,100 trees per hectare, but an improved stocking regime could increase the density to 1,600 trees per hectare (Belgian Development Agency, 2012).

2.5.3.2 Carbon sequestration

The annual rate of carbon sequestration is calculated for each current land use and restoration activity following Equation 2.3 from the Intergovernmental Panel on Climate Change's (IPCC) Good Practice Guidelines (IPCC, 2003):

$$C0_2e\ (tonnes) = (AGB + RBDM) * 0.49 * 3.67 \qquad [2.3]$$

$$RBDM = e^{(-1.805+0.9256*\ln(AGB_i)} \qquad [2.4]$$

Where 0.49 is the factor used to convert short tons of dry matter to carbon and 3.67 is the factor used to convert carbon to $C0_2$ equivalent (IPCC, 2003). The variable AGB refers to above ground biomass and it is calculated by multiplying the timber volume estimates from Table 2.1 by biomass conversion expansion factors reported by the IPCC for each climate zone and forest type (IPCC, 2003). The variable RBDM represents root biomass dry matter or belowground biomass and it is calculated using Equation 2.4, which is a function of AGB reported in Table 2.1 (IPCC, 2003).

2.5.3.3 Erosion

Annual rates of erosion are calculated using the Universal Soil Loss Equation (USLE) (Bishop and Allen, 1989). The USLE was developed as a tool to easily quantify the long-term average annual soil loss of cultivated land under various conditions so farmers and policy makers can select land use practices that keep erosion to acceptable levels following Equation 2.5:

$$Erosion = R * K * LS * C * P \qquad [2.5]$$

Where
R = Rainfall erosivity factor
K = Soil erodibility factor
LS = Plot length and slope factor
C = Soil cover factor
P = Support practice factor

The rainfall erosivity factor, R, measures the energy delivered during each precipitation event. The soil erodibility factor, K, represents the ability of soil to be moved by rainfall and runoff. The plot length and slope factor, LS, represents soil erodibility due to combinations of slope length and steepness relative to a standard unit plot, which is defined as a plot where LS, C, and P are all equal to 1. The soil cover factor, C, represents the effect of plants, soil cover, below-ground biomass, and cropping activities on soil erosion and the support practice factor, P, represents the effects of practices like contouring, strip cropping, and terracing, but is usually set to equal 1 (Renard et al., 2011).

Table 2.2 displays the information used to estimate the annual rate of erosion for each degraded land use and restoration activity.

Table 2.2: Parameter values used to estimate erosion						
Land Use	USLE Parameter					Estimated Annual Erosion Short Tons Ha^{-1} Yr^{-1}
	R	K	LS	C	P	
Degraded maize agriculture	332	0.12	1.5	0.3	1	18
Agroforestry with maize	332	0.12	1.5	0.1	1	6
Poorly managed woodlots	431	0.15	1.5	0.15	1	19
Well managed woodlots	431	0.15	1.5	0.1	1	15

Annual precipitation data was converted into an estimate of energy intensity, R, by dividing total precipitation by the average number of annual precipitation events and assuming each event lasts an average of 3 hours. The soil erodibility index, K, and soil cover factor, C, were queried from a GIS database provided by the Rwandan Natural Resources Authority for each land use and restoration intervention. Plot lengths were estimated from the 2008 Rwandan Agricultural Survey by taking the square root of the average plot size for each province (NIS, 2008). The support practice factor, P, reflects the effects of practices that would reduce the amount and rate of the water runoff thereby reducing erosion. However, P is often assigned a value of 1 unless specific management practice information is available (Renard et al., 2011).

2.5.3.4 Crop yields

Production risk is one of the defining features of smallholder agricultural systems in SSA (Di Falco and Chavas, 2009). Crop production in Rwanda is largely rain-fed, with more than 1 million hectares relying solely on rain. Less than 5,000 hectares are irrigated and fertilizer is applied to less than 3% of cropland (Habiyambere et al., 2009). Variation in precipitation, specifically a lack of rain, can severely reduce yields and cause significant income losses and negative impacts on the livelihoods of smallholders. In order to account for this risk the relationship between annual precipitation and the average annual per hectare crop yields have to be estimated. This analysis relies on a panel data set of crop production, seasonal precipitation, and annual total planted area from 2007 to 2009 at the district level from RNRA and Meteo Rwanda, respectively, to estimate crop production functions that account for the relationship between crop yields and precipitation.

The data create two specific challenges for this analysis. First, the omission of relevant variables can potentially introduce bias into the coefficient estimates of regression equations (Greene, 2003). Omitting region-specific predictor variables like average annual temperature, altitude, and soil fertility can bias the coefficients of variables that are included in the regression analysis. For example, if average annual temperature and annual precipitation are negatively correlated omitting the temperature variable could over-estimate the effect annual precipitation has on average annual crop yields. The second challenge posed by the data is that the relationship between average annual per hectare yields and the right-hand-side (RHS) variables that influence those yields may be determined simultaneously, leading to simultaneous equation bias (Greene, 2003). While the theoretical existence of this potential bias is well documented in the smallholder production function literature (Griliches and Mairese, 1995), the empirical literature has not made a concerted effort to account for its potential impacts in the estimation of production functions (See Di Falco and Chavas, 2009; Chirwa et al., 2007; Neumann et al., 2010; Sherlund et al., 2001).

This paper accounts for the effect of unobserved time-invariant region-specific variables by using a fixed effects model following the smallholder agricultural production function literature (See Di Falco and Chavas, 2009; Chirwa et al., 2007; Neumann et al., 2010; Sherlund et al., 2001). Region specific variation can also be controlled with Random Effects models if there is no reason to think that the error term of the model is correlated with any of the explanatory variables (Gujurati, 2003). However, there is reason to believe that the error term is correlated with the explanatory variables because it is easy to imagine that total planted area for a specific region is correlated with unobservable region-specific characteristics like innate soil fertility and labor availability. Additionally, Fixed Effects models are preferred when there is a reason to believe the observational units are not random drawings from a larger sample (Gujurati, 2003). It was not possible to include additional and potentially relevant time varying ecological or economic data mentioned above because these data were not available for the time span of the agricultural data at the time of the study.

Simultaneity between the dependent and independent variables in the regression equation can be addressed in a number of ways. Due to data limitations these approaches were not pursued. For example, indirect least squares or two-stage least squares (i.e. instrumental variable approach) can all overcome the violation of classical assumptions created by the simultaneity between the dependent and independent variables so long as the data is available to specify the reduced form or instrumental variable equations, respectively. The limited availability of data in Rwanda forecloses both options. As a result, the model presented below addresses one form of potential bias through the use of a fixed effect model.

Without an approach to control the simultaneous equation bias there is no way to guarantee that the parameter estimates are consistent and extra caution should be used interpreting the results.

The model of annual per hectare regional crop production is conceptually similar to those found in previous studies of low-input farming systems. The model assumes that crop production is a functions of precipitation, land area devoted to the crop, and region specific characteristics such as soil fertility and labor supply that are considered time-invariant (Chirwa et al., 2007; Sherlund et al., 2001). While the empirical literature has used a variety of functional forms to estimate smallholder production functions, the Cobb-Douglas form has been extensively documented because of its ability to estimate returns to scale (Bravo-Ureta and Pinheiro, 1993; Bravo-Ureta and Evenson, 1994; Battese and Coelli, 1995). Following this literature the production function for maize in region i during time period t is defined as a deterministic Cobb-Douglas production function of the form:

$$\text{average per hectare yield}_{\text{it}} = \text{planted land area}_{\text{it}}{}^{\beta_1} + \text{annual precipitation}_{\text{it}}{}^{\beta_2} \qquad [2.6]$$

Where $\text{average per hectare yield}_{\text{it}}$ is the average per hectare maize yield in region i at time t, $\text{planted land area}_{it}$ is the total land are planted for maize in region i at time t measured in hectares, and $\text{annual precipitation}_{it}$ is the total annual precipitation for region at time t measured in millimeters. Equation [2.6] cannot be estimated with Ordinary Least Squares (OLS) because it is not linear in the parameters, but it can be transformed to an estimable econometric model by taking the log of both sides to create a so-called log-log model of the form:

$$\ln(\text{average per hectare yield}_{\text{it}}) =$$

$$\{\beta_1 \ln(\text{planted land area}_{\text{it}}) + \beta_2 \ln(\text{annual precipitation}_{\text{it}}) + \beta_i Z_i + \varepsilon_{\text{it}}\} \qquad [2.7]$$

Where Z_i is a vector of region specific dummy variables that account for region-specific time-invariant unobservable variables that influence the average per hectare maize yields in each region. ε_{it} is the deviation from the conditional mean for region at i at time t. Equation [2.7] is linear in the parameters and can be estimated with OLS if the potential simultaneity bias is overlooked. Table 2.3 reports the means of the data used to estimate equation [2.7] with OLS.

Table 2.3: Means of data used in maize yield regression

Variable	Definition	Mean	Std. Dev.
Average per hectare yield	Average yield per hectare in a given region in a given year (Short Tons Ha^{-1} Yr^{-1}).	1.96	2.11
Planted land area	Total land area planted in a given region in a given year (Ha^{-1} Yr^{-1}).	2,851	1,622
Annual Precipitation	Total precipitation received during the year in a given region (mm Yr^{-1}).	600	177

Table 2.4 shows the results from the regression analysis. The coefficient estimates for precipitation and land area are both significant at the 99% level and the signs of both parameter estimates are consistent with expectations. More growing season precipitation is correlated with higher crop yields, on average. The negative sign of the Land Area Planted coefficient reflects the fact that as cultivation of maize is expanded more marginal land is used.

Table 2.4: Crop yield regression results

Variable	Maize
Precipitation (mm Yr^{-1})	0.49** (0.14)
Land Area Planted (Ha^{-1} Yr^{-1})	-0.46*** (0.11)
Sample size (N)	108
R^2	0.42
F-Value	2.95

Notes: Standard errors in parenthesis. *** = P < 0.001

The results of the regression equation reported in Table 2.4 are used to estimate the crop yields that could be achieved with agroforestry by combining them with data from Dreschel et al. (1996). The authors reported the impacts of agroforestry systems on maize yields in Rwanda as a percentage of degraded agricultural yields. The authors found that the yield response could range from between -35% to + 65% compared to degraded maize yields (Dreschel et al., 1996). To estimate the yield of agroforestry, this paper first calculates the yield of degraded agriculture using Equation 2.6 and the estimated coefficients in Table 2.4 and multiplies that value by the expected crop yield response.

2.5.4 Enterprise Budgets

Enterprise budgets were created for each degraded land use and restoration activity based on the information provided by stakeholders during the four regional workshops. The data presented in the enterprise budgets are based on stakeholder's consensus over values and can therefore be considered as approximate averages. Stakeholders validated the final versions of the enterprise budgets during a fifth workshop held in Kigali in March, 2014. Tables 2.5 and 2.6 display the enterprise budgets for degraded agriculture, agroforestry, poorly managed woodlots, and well-managed woodlots, respectively.

Table 2.5 shows the cost and revenue structure for degraded agriculture and agroforestry operations. Agriculture in Rwanda is a low-input activity that uses no mechanization and relies on very few inputs because most farmers cannot afford to make investments (Habiyamere et al., 2009). Labor and farm equipment, like hoes and shovels, are the most costly farming inputs (NIS, 2008).

Degraded agricultural systems use approximately 3000 Kg of organic fertilizer, but one of the advantages of agroforestry systems is that they do not use organic fertilizer because the tree roots bring soil nutrients from deep below ground closer to the surface where crops can take advantage of them (Sanchez and Palm, 1996).

The fixed costs of agricultural systems are very low because only basic materials like a hoe and shovel are required. The fixed costs associated with agroforestry are higher because tree seedlings have to be purchased at a cost of 1,000 RwF (approximately $1.25) per seedling. The average revenue from crop yields in agricultural systems ranges between 267,000 RwF per year for beans to 287,000 RwF per year for maize. In agroforestry systems the average revenue from crop yields can range between 309,000 RwF and 330,000 RwF. Agroforestry systems also generate additional revenue of 1,068,200 RwF from the sale of timber at the end of a twenty-year rotation interval.

Table 2.6 shows the cost and revenue structure for poorly managed woodlots and well managed woodlots with best practices observed. The productivity of most woodlots in the country is low because they are established on marginal land, and landowners use poor management practices during planting, thinning, and harvesting (AFF, 2011). Planting material is the largest cost of establishing a woodlot whether it is poorly managed or well managed. In both instances between 1100 and 1600 eucalyptus seedlings need to be bought at a cost of 100 RwF per seedling. The site is prepared before planting by clearing bush at a cost of 13,200 RwF ha^{-1} for twenty days of labor.

Table 2.5: Enterprise Budget for Degraded Agriculture and Agroforestry in Rwanda

			Degraded Agriculture (Maize)		Agroforestry	
	Unit	Price (RwF)	Quantity	Value (RwF)	Quantity	Value (RwF)
Variable costs						
Crop Seed	Kg	90	40	3,600	40	3,600
Labor (crops)	Days	660	221	145,860	199	131,340
Labor (trees)	Days	660	-	-	44	29,040
Organic fertilizer	Kg	2	3,000	6,000	-	-
Capital costs	-	660	-	660	-	660
Fixed costs						
Tree seedlings	Seedlings	1,000	-	-	300	300,000
Small agricultural equipment	-	-	-	1,900	-	1,900s
Revenue						
Crop yields	Kg	350 - 515	910	318,500	578 - 2260	202,300 – 1,163,900
Timber yields (Year 30)	m^3	10,900	-	-	98	1,068,200
Ecosystem Services						
Carbon	Short Tons $Ha^{-1}Yr^{-1}$	2,500	-	-	0.9 - 2.7	2,250 – 6,750
Erosion	Short Tons $Ha^{-1}Yr^{-1}$	1,350	18		6	

Table 2.6: Enterprise Budget for Poorly Managed and Well Managed Woodlots in Rwanda						
			Poorly managed woodlots		Well managed woodlots	
Items	Unit	Price (Rwf)	Quantity	Total cost (RWf)	Quantity	Total cost (RWf)
Variable costs						
Digging	Hole	10	1,100	11,000	1,600	16,000
Planting material	Seedling	100	1,100	110,000	1,600	160,000
Seedling transport	Seedling	10	1,100	11,000	1,600	16,000
Planting	Seedling	5	1,100	55,000	1,600	8,000
Beating up (15% - 30%)	Seedling	50	330	16,500	240	12,000
Pruning	Tree	25	1,100	25,250	1,600	40,000
Thinning (after 4th year)	Tree	30	250	7,500	250	7,500
Coppicing (every 7 years)	Tree	25	1,100	27,500	1,600	40,000
Fixed costs						
Bush clearing	Days	660	20	13,200	20	13,200
Trench establishment	Meter	125	-	-	300	37,500
Fire lane creation	Meter	125	-	-	300	37,500
Fire lane maintenance	Year	5,000	-	-	-	5,000
Trench maintenance	Meter	50	-	-	300	15,000
Remove old stumps	Ha	100,000	-	-	1	100,000
Revenue						
Poles	Pole	1500	250	256,130	250	256,130
Fuelwood	Stere	2400	52	124,800	73	171,865
Ecosystem Services						
Carbon	Short Tons $Ha^{-1}Yr^{-1}$	2,500	11	-	16.32	13,300
Erosion	Short Tons $Ha^{-1}Yr^{-1}$	1,350	19		15	

In well-managed woodlots the site is prepared in a number of other ways. The stumps of old eucalyptus trees are removed to maximize the plantable area for the new seedlings. Stakeholders reported that trenches and fire lanes were needed to limit erosion and reduce the risk standdestroying wildfires. Trenches and fire lanes also require annual maintenance. After the site is prepared the seedlings are transported to the site at a cost of 10 RwF per seedling and planted at a cost of 5 RwF per seedling. Once the seedlings are planted they have to be pruned of excessive branches at a cost of 25 RwF per tree. After the first year seedlings that did not survive are replaces through a process known as ‘beating up.’ In poorly managed woodlots the average seedling replacement rate is 30% while it is 15% for well-managed woodlots (Belgian Development Agency, 2012). At the end of the fourth year the stand is thinned by removing approximately 250 trees, which are sold as poles at market price of 1,500 RwF per pole. Every seven years the stand is coppiced and the timber is sold as fuelwood at a price of 2,400 RwF per stere (i.e. m^3).

2.5.5 Repeated random sampling (Monte Carlo simulations)

This paper uses repeated random sampling (i.e. Monte Carlo simulations) to account for the variability of financial revenue and ecosystem service values. The simulations draw parameter values from their probability distributions to determine the variability of the associated outcomes. This paper accounts for the variability in market prices for crops and fuelwood as well as the variability of precipitation, tree growth rates, and the impact of agroforestry tree species on crop yields.

Table 2.7 lists the assumptions and data sources used to characterize the distributions of each variable included in the Monte Carlo simulations. Data characterizing the mean annual incremental (MAI) growth rate of *Grevillea robusta* was taken from Kilinganire (1996). The authors assessed the growth of the species through a random sample of 67 farms in seven ecological zones in Rwanda for different aged stands of trees.

The study reported the average MAI across ecological zones and stand ages and this study used that information to calculate the average MAI and its Standard Deviation across ecological zones and stand ages.

Table 2.7: Distributional assumptions for economic and biological variables used in the Monte Carlo analysis

Variable	Distribution assumptions	Draws	Source
Grevillea robusta MAI	MAI~N(1.44,0.6)	N=1,000	Kalinganire, 1996
Eucalyptus tereticornis MAI	MAI~N(7.15,1.1); MAI~N(10.4,1.6)	N=1,000	Belgian Development Agency, 2012
Impact of agroforestry tree species on crop yields	Impact~Tri(-0.35, 0.3, 0.6)	N=1,000	Dreschel et al., 1996
Market prices	Fuelwood~Tri(380,2400,4700) Maize~Tri(160,250,450)	N=1,000	Rwanda Ministry of Agriculture and Animal Resource, 2013
Growing season precipitation	Precip~ Bootstrapped	N=1,000	Meteo Rwanda, 2013

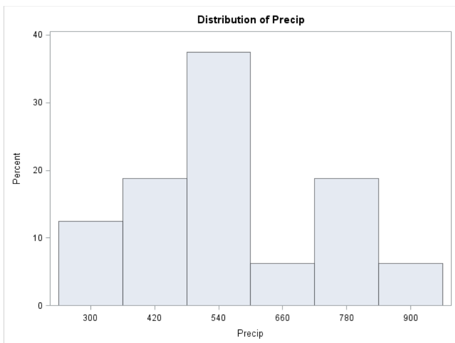
Data characterizing the MAI and Standard Deviation of the growth rate of a single tree of *Eucalyptus tereticornis* was taken from the Belgian Development Agency's report on improving woodlot management in Rwanda (Belgian Development Agency, 2012). The single-tree estimate was multiplied by the stocking densities reported for poor and well-managed woodlots to estimate the mean MAI for both management practices. The maize yield impacts of agroforestry were characterized using data from Dreschel et al. (1996) who reported the impact of agroforestry species on maize yields Rwanda. In their study, the authors gathered crop yield data from multiple farmer-managed agroforestry plots from across the country over periods of 1 to 3 years and reported the percentage change in maize yields as compared to standard agricultural maize yields. The authors reported that the yield impacts varied field to field, plot to plot. The range of agroforestry maize yield impacts was between -45 to +60%. Officials at the Rwanda Ministry of Agriculture and Animal Resource’s and the Rwanda Natural Resource Authority reported the minimum, maximum and average market prices for maize and fuelwood. Provincial level monthly precipitation data from 2007 to 2009 was sourced from Meteo Rwanda, which recorded annual precipitation in all four provinces included in the analysis.

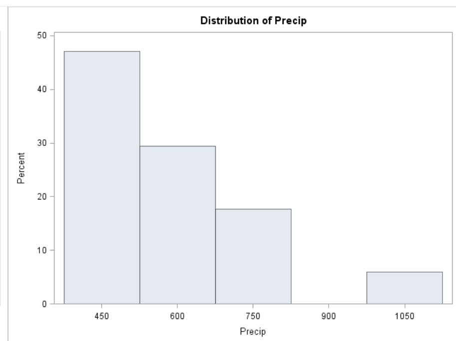
The studies reported above did not characterize the distributions of the data. To overcome this limitation, the data was used to parameterize the most likely probability distribution functions associated with the processes that generated the data. In a Monte Carlo simulations of optimal timber harvesting under uncertainty Van Kooten et al. (1992) used a triangular distribution to model the average annual growth of trees in a boreal forest in northern Canada. The study used the triangular distribution because it can be parameterized when only the maximum, minimum, and mean outcome values were known. In contrast, Moore et al. (2012) assumed annual tree growth rates followed a normal distribution because both the mean annual increment and its standard error could be observed. Since both the mean annual increment and its standard error are observable for both tree species of interest to this analysis the mean annual increments of both species can be characterized by a normal distribution.

The effect that agroforestry trees would have on crop yields is only reported for Rwanda in terms of the maximum, minimum, and mean impacts as a percentage of average yield so a triangular distribution is used to approximate the data generating process. Data on market prices for fuelwood and maize were only reported as maximum, minimum, and means without standard errors so their distributions were characterized also triangular distributions. Histograms of annual precipitation, shown in Figure 2.2, revealed that the distributions of the data did not fit any known distributions.

The distributions of that data were approximated with repeated random sampling with replacement (i.e. bootstrapping).

While this method approximates the empirical distributions of the data its major limitation is that it does not draw values that have not already been observed in the sample. As a result, extreme precipitation values may be absent from the simulation.





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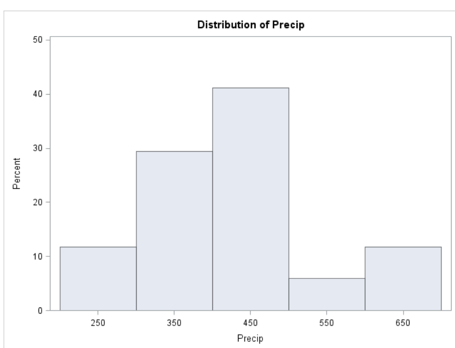

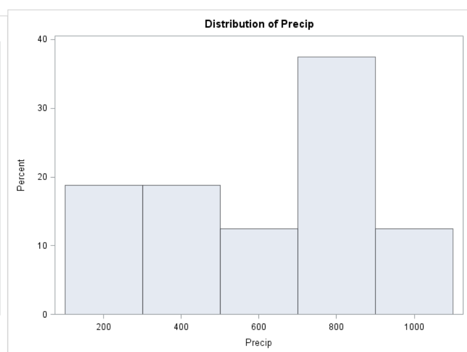

C D

Figure 2.2: Histograms of total annual precipitation (mm^{-1} year^{-1}) for Kigali (A), Western (B), Southern (C), and Northern (D) Provinces from 2003 to 2009

Exactly 1,000 random samples of each variable were drawn from the distributions presented in Table 2.7. Each draw was composed of 30 annual observations that were used to calculate the NPV of the land use enterprise.

In total, the data set contained 1,000 observations of NPVs calculated over a thirty-year period. The data were used to estimate the distribution of NPVs for each current land use and restoration activity.

2.5.6 NPV Decision Metric

This paper uses the Net Present Value (NPV) metric to compare the financial attractiveness of current land uses and restoration activities. The net present value metric is calculated by subtracting the summed and discounted stream of costs from the summed and discounted stream of benefits for each enterprise over a thirty-year time horizon:

$$NPV = \sum_{t=0}^{30} \delta^t (B_t - C_t) \qquad [2.8]$$

Where B_t are the financial benefits and costs at time t, respectively, and δ^t is the discount factor. Enterprises with larger NPVs are considered to be more efficient than alternatives with smaller NPVs as long as the benefits and costs can be distributed amongst stakeholders in a way that improves the welfare of some without reducing the welfare of others. The enterprises budgets from Tables 2.5 and 2.6 are used with the data from the Monte Carlo simulations to estimate the NPV of agriculture, agroforestry, poorly managed woodlots, and well managed woodlots. A 16% rate of discount, which is the average rate of interest charged to farmers by the National Bank of Rwanda, was used in the analysis (Maniriho and Bizoza, 2013).

2.5.7 Risk Analysis

Comparing the expected utility functions of smallholders under different agricultural and restoration activities is data and or assumption intensive. Expected utilities reflect individual preferences for income and risk that have to be characterized through elicitation of risk-preferences or through assumptions. This is problematic because operationalizing expected utility theory depends on either 1. Collecting information on smallholder's preferences, or 2. Making assumptions about smallholder preferences (Andersen, 1974).

One way to navigate this challenge without information on individual risk-preferences and with minimal assumptions about the shape of individual utility functions is to compare the distributions of payoffs from different activities rather than comparing the expected utilities that would be achieved under those distributions (Mas-Colell, 1995). The pay offs of different distributions can be compared in two ways that are consistent with expected utility theory. First, the distributions can be compared by their expected returns. Second, they can be compared by the dispersion of those returns. The goal is to be able to look at two distributions from different activities and unambiguously say that $F(\cdot)$ has higher returns than $G(\cdot)$ and that $F(\cdot)$ is less risky than $G(\cdot)$. These ideas are known as First and Second Order Stochastic Dominance, respectively.

2.5.7.1 First Order Stochastic Dominance

Smallholders with a non-decreasing expected utility functions prefer more to less. First Order Stochastic Dominance (FOSD) compares distributions of payoffs in a way that makes it possible to say that every utility maximizer who prefers more to less would prefer $F(\cdot)$ to $G(\cdot)$. A distribution is said to display First Order Stochastic Dominance over another distribution if for every non-decreasing function $u: \Re \rightarrow \Re$ that $\int u(x) dF(x) \geqslant \int u(x) dG(x)$. It can be shown that this is true if and only if $F(x) \leq G(x)$ for all x (Mas-Colell, 1995). That is, smallholders with a utility function, U(x), such that U(x)>0 will prefer a FOSD distribution to one that is dominated.

2.5.7.2 Second Order Stochastic Dominance

Second Order Stochastic Dominance (SOSD) captures the idea that risk-averse smallholders receive diminishing marginal utility from increasing amounts of income.

If a distribution demonstrates Second Order Stochastic Dominance over another distribution then every risk-averse individual would prefer that distribution to the distribution that is dominated.

A distribution can be said to display SOSD over another distribution if and only if:

$$\int_{-\infty}^{x}[G(\cdot) - F(\cdot)]dt \geq 0 \; for \; all \; x \qquad [2.9]$$

Foster and Sen (1997 cited by Formby et al., 1999) have proposed an alternative approach to assess SOSD for distributions with different means. This approach, known as Normalized Stochastic Dominance (NSD) compares the CDFs of normalized distributions. For a probability distribution function of NPV $f(\cdot)$ with mean u_f its normalized PDF, $f(x^n)$, is defined as the PDF of the normalized NPV, $x^n = x/u_f$. The corresponding normal CDF is $F(x^n)$. The distribution $F(x^n)$ dominates the distribution $G(x^n)$ if and only if:

$$F(x^n) \leq \; G(x^n) \text{ for all } x^n \qquad [2.10]$$

That is, $F(x)$ SOSD $G(x)$ if and only if $\int_{-\infty}^{x} U(x^n)dF(x^n) \geq \int_{-\infty}^{x} U(x^n)dG(x^n)$ for all x^n.

2.5.7.3 Certainty Equivalence

In theory Stochastic Dominance is a good way to rank and compare distributions in an expected utility framework because it requires making minimal assumptions about the shape of individual utility functions. However, in practice it is not always useful because a large number of empirical distributions cannot be ordered (Meyer, 1977). One way around this obstacle is to estimate the Certainty Equivalents (CE) of each distribution. The CE of a lottery is the amount of money an individual would have to be paid with certainty to be indifferent between the payment and participating in the lottery. For risk-averse smallholders CEs are always less than the expected monetary payoff of a lottery. When comparing several different activities a risk-averse smallholder would always prefer the alternative with the largest CE (Mas-Colell, 1995). One drawback to this approach is that it requires specifying a utility function for smallholders. However, the benefit is that it allows for definitive rankings of different activities under specific risk-aversion parameters.

Smallholder preferences over uncertainty are often analyzed by specifying twice differentiable functions like the negative exponential or constant relative risk aversion (CRRA) utility functions (Ngwira et al., 2013; Di Falco et al., 2007). Previous studies from SSA have shown that smallholder behavior is best represented by CRRA utility functions with relative risk aversion coefficients between 1 and 5. A relative risk aversion of 3 is considered to be moderately risk averse (Binswanger, 1981; Chavas and Holt, 1996; Gollier, 2001). CRRAs are also attractive because they imply that absolute risk aversion is decreasing with wealth. This suggests that relatively poor subsistence smallholders would be much more sensitive to downside risk than relatively wealthier smallholders (Pratt, 1964).

The CRRA utility function is represented by:

$$U(X) = X^{1-\gamma}/(1-\gamma) \qquad [2.11]$$

Where γ is the measure of relative risk aversion and is assumed to be $\gamma \geq 1$. A value of $\gamma = 0$ reduces equation 2.11 to $U(X) = X$, which would reflect a risk-neutral smallholder. If $\gamma = 1$ then equation 2.11 becomes $U(X) = \ln(X)$. Higher values of γ correspond to smallholders with more aversion to risk.

Under a CRRA utility function the CE of a given lottery can be determined by finding the value of CE that solves the following equation:

$$U(CE;\gamma) = \sum_i \pi_i ({X_i}^{1-\gamma}/(1-\gamma)) \qquad [2.12]$$

Where π_i is the probability weight associated with observing payoff X_i. In this study the values of π_i for each realization of NPV (i.e. X_i in 2.12) were calculated using Proc Freq in SAS 9.2, which returned an empirical probability distribution of the NPV realizations and their corresponding frequencies. The frequencies were used as estimates of the probabilities. As the value of $\boldsymbol{\gamma}$ increases (i.e. the smallholder becomes more risk-averse), the value of the CE decreases relative to the expected value of the lottery. The difference between the CE and the expected value is known as the 'risk premium.' It reflects the amount of money a smallholder would be willing to give up to avoid the risk of the lottery. In a study of smallholder risk in the Ethiopian Highlands, Di Falco et al. (2007) assumed the smallholders had a relative risk aversion equal to 3. The CE for each activity in this study are solved for values of γ from 0 to 9, reflecting smallholders with no, low, moderate, and high levels of risk aversion, respectively (Di Falco et al., 2007).

2.6 Results

2.6.1 Stochastic Dominance

The results for the Stochastic Dominance analysis are shown in the figures below. The analysis looked at the CDFs for the transitions from degraded maize agriculture to agroforestry with maize and from poorly managed to well-managed woodlots for all four provinces considered in this study. The Stochastic Dominance analysis was also carried out by including the values of ecosystem services in the partial enterprise budgets to see if internalizing their value would alter the preferences of smallholders. The results showed that including the values did not change the preferences of smallholders so those figure are included in the appendix for brevity.

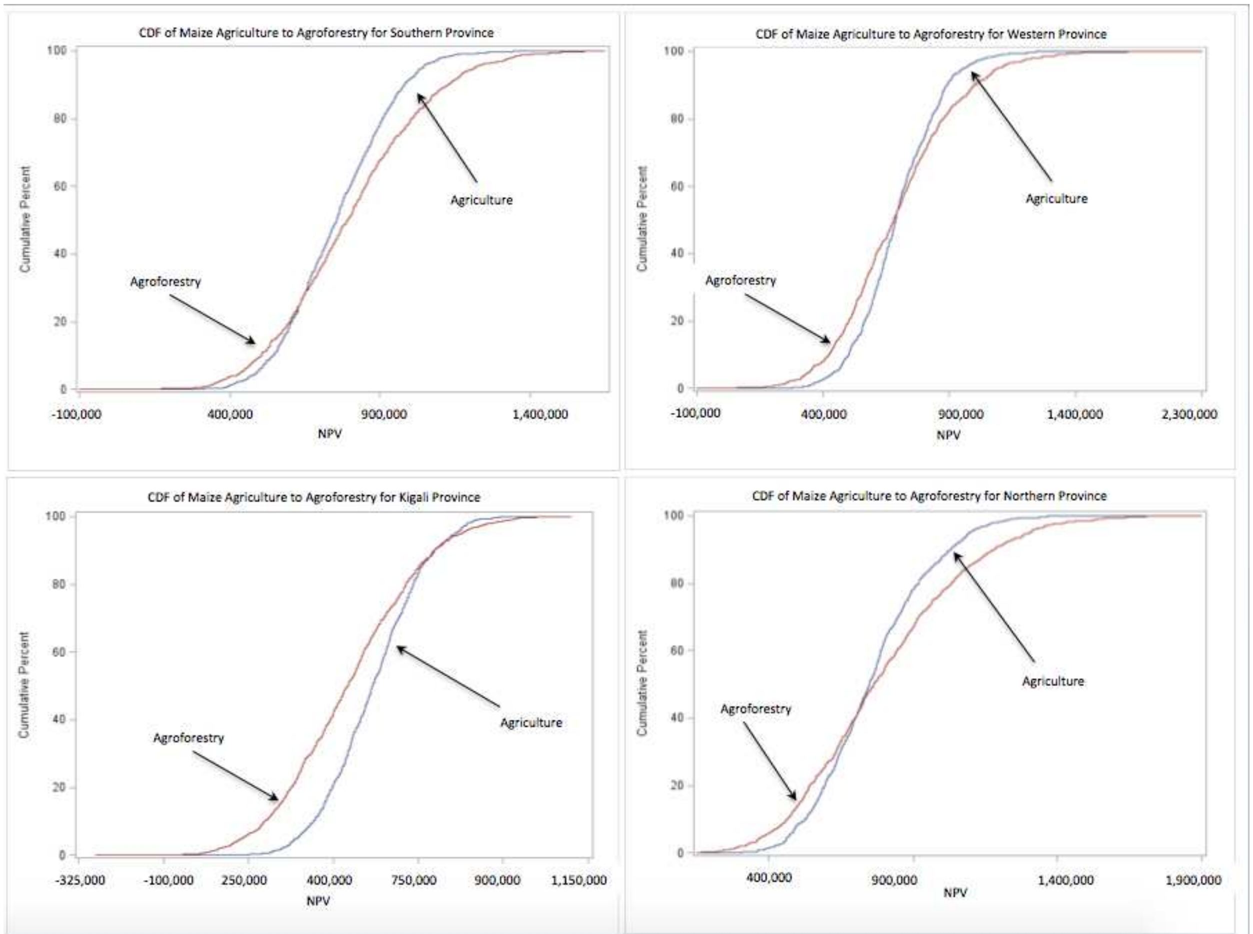

Figure 2.3: The CDFs of NPV for degraded maize agriculture to agroforestry with maize for four provinces in Rwanda. A non-overlapping CDF to the right of an alternative CDF is said to demonstrate First Order Stochastic Dominance the alternative. Smallholders who prefer more to less will always choose an activity with a CDF that strictly dominates another CDF. In the example above none of the CDFs dominate the others as shown by the crossing graphs of the CDFs in every province.

For the transition from degraded maize agriculture to agroforestry with maize neither activity displays First Order Stochastic Dominance (FOSD) over the other. The CDFs of NPV show that the returns from agroforestry are more likely to be smaller than the returns from maize. This can be seen in Figure 2.3 where the CDFs of the NPV for both activities cross each other in every province. The definition of FOSD is that one distribution FOSD another if and only if $F(x) \leq$

$G(x)$ for all x (Mas-Colell, 1995). As Figure 2.3 shows neither transition dominates the other and this suggests that for this transition there is not an unambiguously dominant activity for smallholders who strictly prefer higher financial returns.

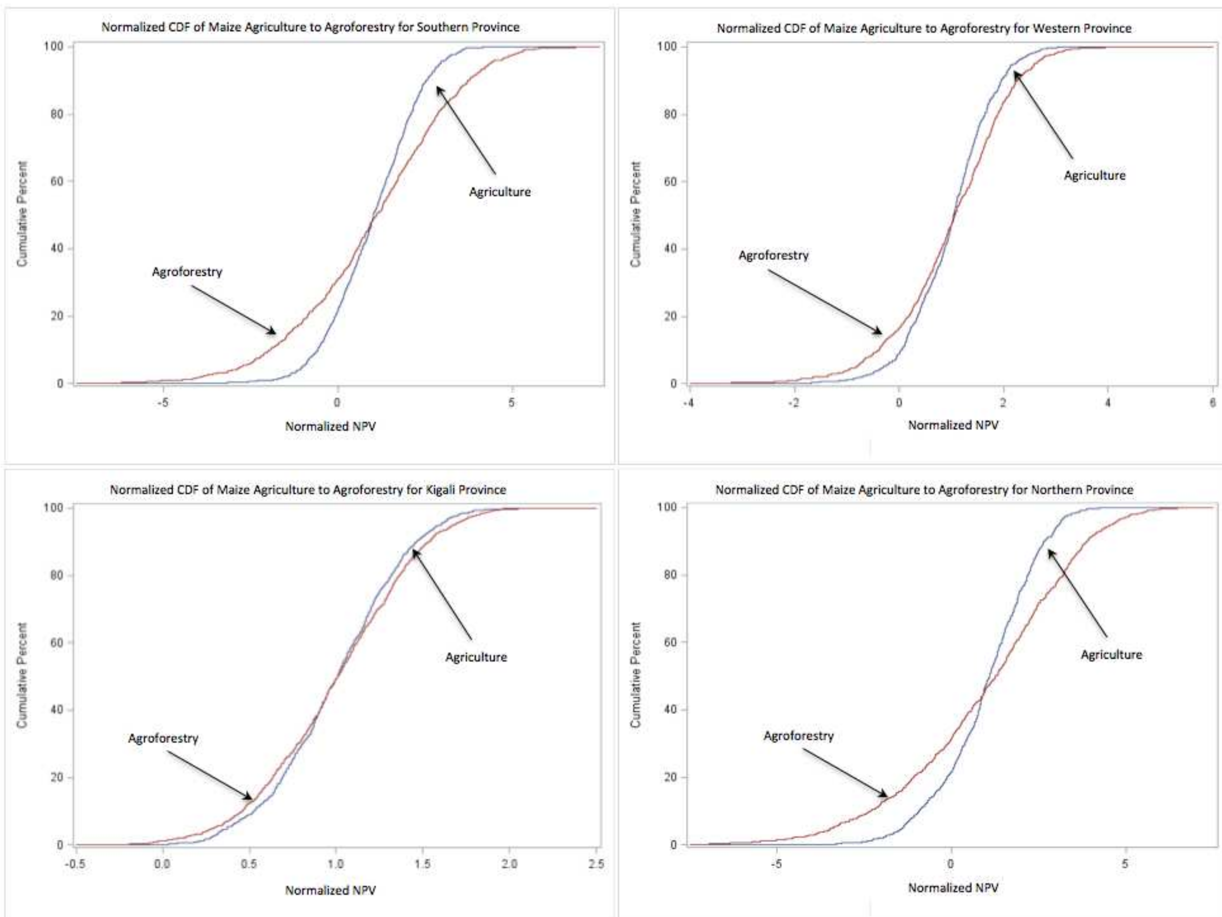

Figure 2.4: The normalized CDFs of NPV for degraded maize agriculture to agroforestry with maize for four provinces in Rwanda. A non-overlapping normalized CDF to the right of an alternative normalized CDF is said to demonstrate Second Order Stochastic Dominance over the alternative. Smallholders who are risk averse will always choose the activity with a normalized CDF that demonstrates Second Order Stochastic Dominance. In the example above none of the normalized CDFs dominate the others as shown by the crossing graphs of the normalized CDFs in every province.

The test for Second Order Stochastic Dominance (SOSD) is designed to find activities that unambiguously result in lower variability of net returns than alternative or competing activities. Risk-averse smallholders would strictly prefer an activity that demonstrates SOSD over alternatives. The results in Figure 2.4 show that neither degraded maize agriculture or agroforestry with maize dominate each other. First Order Stochastic Dominance is a necessary condition for a CDF to SOSD another distribution. This suggests that for this transition there is not an unambiguously dominant activity for smallholders who strictly prefer low variability to high returns or for those who prefer higher returns as well as low variability.

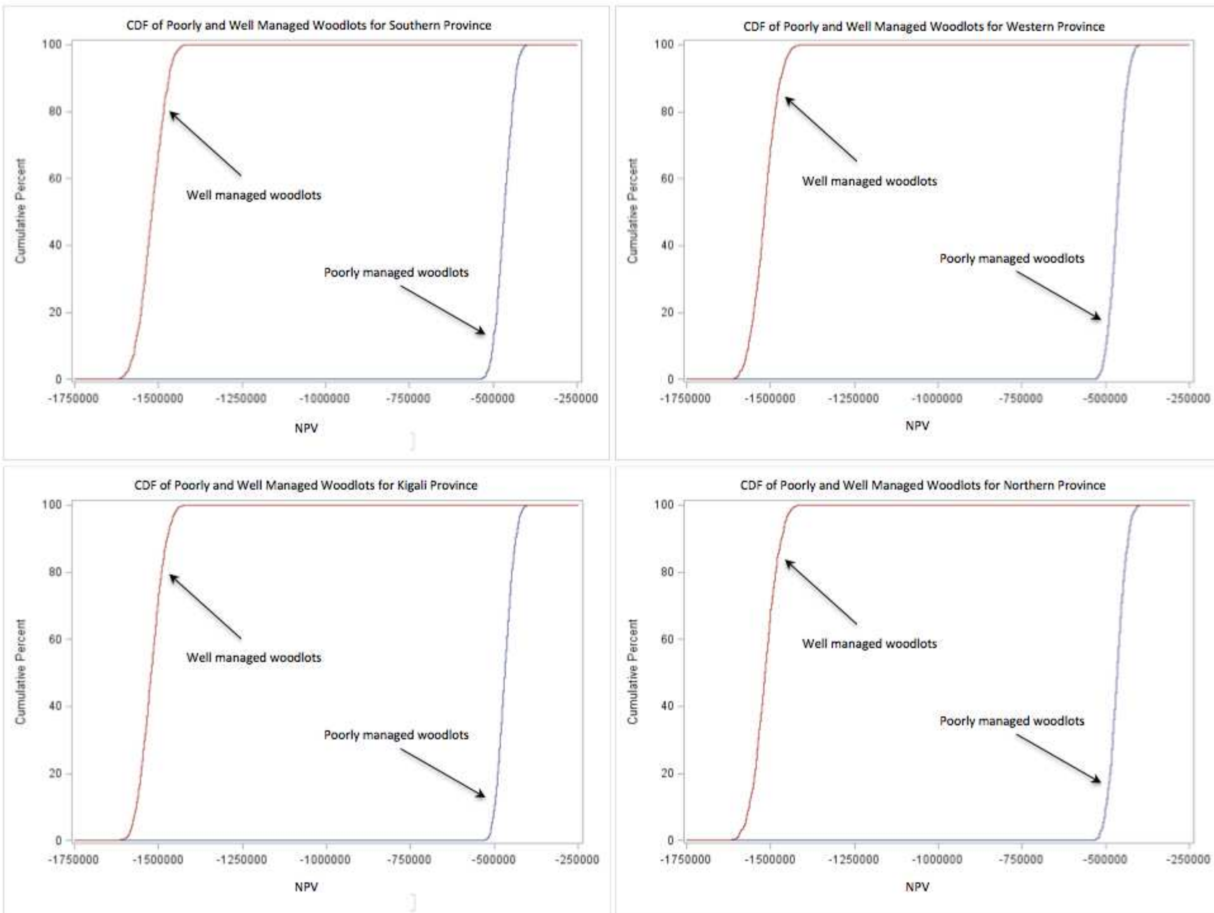

Figure 2.5: The CDFs of NPV for poorly managed woodlots and well managed woodlots for four provinces in Rwanda. A non-overlapping CDF to the right of an alternative CDF is said to display First Order Stochastic Dominance over the alternative. Smallholders who prefer more to less will always choose an activity with a CDF that dominates another CDF. In the example above poorly managed woodlots display First Order Stochastic Dominance over well-managed woodlots in each province.

Figure 2.5 show that poorly managed woodlots display First Order Dominance (FOSD) over well-managed woodlots. In other words, a smallholder with a utility function that strictly prefers more to less will always choose to invest in poorly managed woodlot practices rather than well managed woodlot practices. The CDFs show that poorly managed woodlots are unambiguously less likely to result in larger negative returns than well-managed woodlots. Figure 2.5 shows that the CDFs of the NPV for both activities are strictly non-overlapping. The definition of FOSD is that one distribution FOSD another if and only if $F(x) \leq G(x)$ for all x (Mas-Colell, 1995). As Figure 2.5 shows it is clear that poorly managed woodlots FOSD well-managed woodlots. Figure 2.6 shows that neither poorly managed woodlots or well-managed woodlots display Second Order Stochastic Dominance. This suggests that for this transition there is not an unambiguously dominant activity for smallholders who strictly prefer low variability to high returns or for those who prefer higher returns as well as low variability.

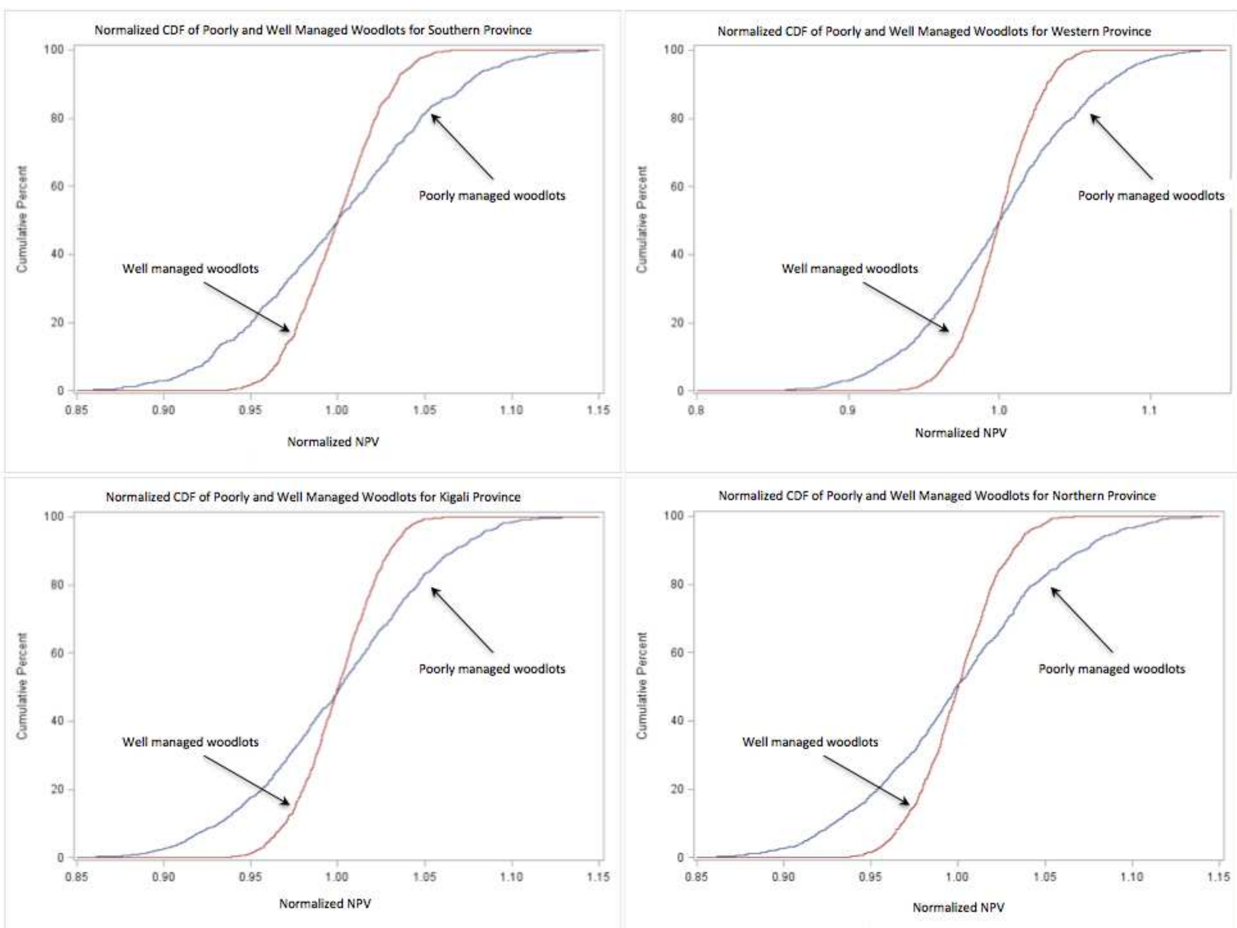

Figure 2.6: The normalized CDFs of NPV for poorly managed woodlots and well managed woodlots for four provinces in Rwanda.

2.7 Certainty Equivalence[9]

Unlike the Stochastic Dominance analysis, the results from the Certainty Equivalence analysis allow precise preference orderings to be made between agriculture with maize and agroforestry with maize. Figure 2.7 shows smallholders with CRRA utility functions prefer agriculture to agroforestry in each province across all values of relative risk aversion coefficients.

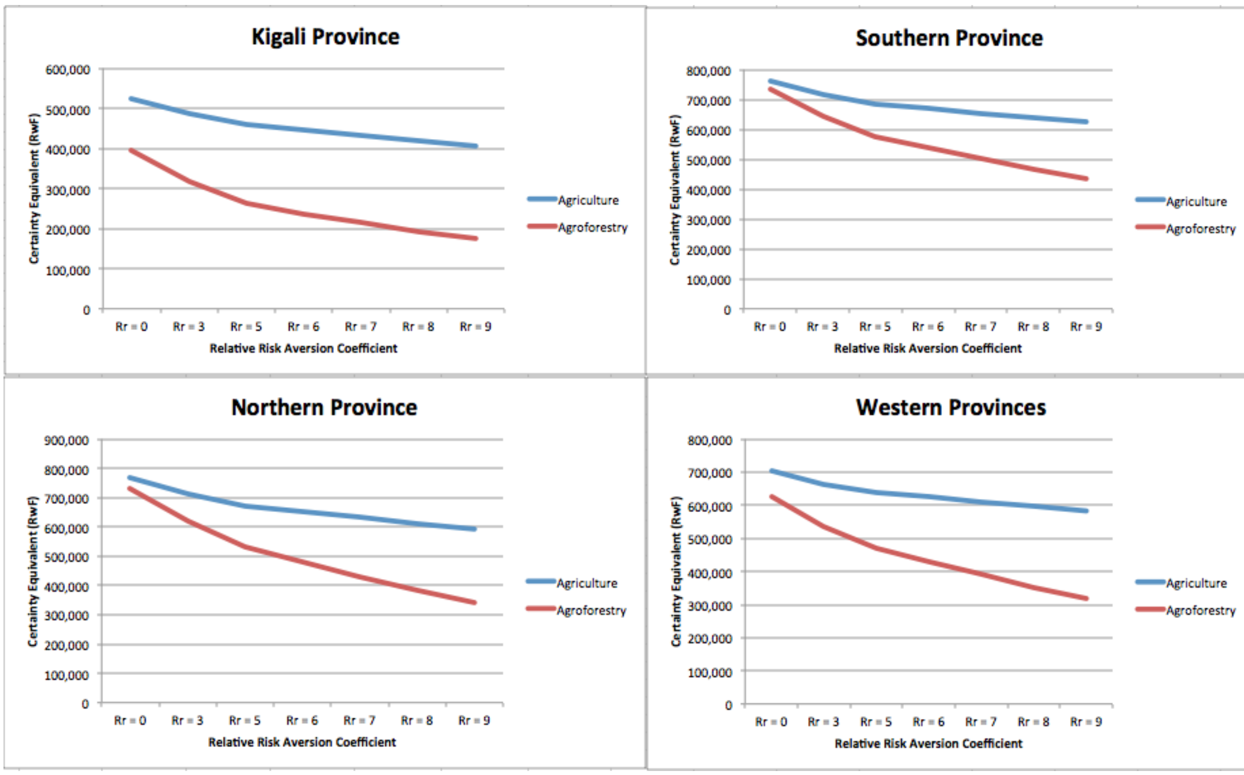

Figure 2.7: Certainty Equivalence Under Different Risk Aversion Coefficients for Agriculture and Agroforestry

For Kigali Province the CE for agriculture under a relative risk aversion coefficient of 3 is 485,965 RwF and for agroforestry it is 316,401 RwF. The results suggests smallholders with moderate risk aversion will prefer agriculture to agroforestry. When smallholders are more risk averse (relative risk aversion coefficient of 5) the CE of agriculture in Kigali province is 459,369

[9] A CE analysis was not done for the woodlot activities because the Stochastic Dominance analysis definitively showed that any risk-averse smallholder would prefer current woodlot management practices to the best practices being proposed.

RwF and 261,969 RwF for agroforestry meaning agriculture is the preferred activity of highly risk-averse smallholders as well. These results are supported by the Stochastic Dominance analysis, which showed that the CDF of agriculture was almost exclusively to the right of the CDF for agroforestry.

In the Northern Province the CE for agriculture under a relative risk aversion coefficient of 3 is 711,307 RwF and for agroforestry it is 618,765 RwF. This suggests smallholders with moderate risk aversion will prefer agriculture to agroforestry. When smallholders are more risk averse (relative risk aversion coefficient of 5) the CE of agriculture in Northern Province is 672,020 RwF and 529,703 RwF for agroforestry meaning agriculture remains the preferred activity of highly risk-averse smallholders. This same pattern is observed in the Southern and Western provinces as well. In both provinces smallholders prefer agriculture because it has lower probabilities of large losses as shown in the Stochastic Dominance analysis.

Policy makers could offer upfront one-time payments to risk-averse smallholders to incentivize adoption. The difference between the curves in Figure 2.7 shows the one-time payments that would be necessary to equate the CE values of agroforestry with agriculture. For moderately risk averse smallholders in Kigali (relative risk aversion coefficient of 3) a one-time payment of 169,563 RwF Ha^{-1} (\$257 Ha^{-1}) would equate the CE values of agriculture and agroforestry. Small holders in Kigali province with a relative risk aversion coefficient of 5 would require a one-time payment of 197,400 RwF Ha^{-1} (\$300 Ha^{-1}) to equate the CE values. Smallholders in Northern, Southern, and Western provinces with a relative risk aversion coefficient of 5 would require 142,317 RwF, 111,127 RwF, and 168,945 RwF in one-time payments, respectively, to be equate the CE values of agriculture with agroforestry. These payments could be justified if the present value of public ecosystem goods and services or other external impacts from agroforestry were less than or equal to the payments that would be necessary to equate the CE values of the two activities.

2.8 Discussion and Conclusion

This study analyzed the financial profitability, financial risk, and ecological impacts of two proposed restoration activities in Rwanda. The study developed a methodology combining enterprise budgets, ecosystem service modeling, and Monte Carlo analysis in an expected utility framework. Employing this type of mixed methodology to analyze the factors that influence smallholder adoption is important because failing to account for these factors can lead to the promotion of risky technologies that have low probabilities of adoption or lead to poor ecological and economic outcomes. Additionally, accounting for both the financial and ecological impacts of land use transitions can provided information to policy makers that could make these land uses more profitable and thus increase the rate of adoption.

The results of the Stochastic Dominance analysis suggest that the preference of smallholder households over degraded maize agriculture and agroforestry were both inconclusive. Even when the values of ecosystem services were internalized from the perspective of the smallholder the results of the Stochastic Dominance analysis were still inconclusive. This result suggests that privatizing the public benefits of ecosystem services associated with agroforestry will not change the adoption decisions of most smallholders on its own.

The Certainty Equivalent analysis extended the Stochastic Dominance analysis by representing the preferences of no, low, moderately, and highly risk-averse smallholders, respectively. The results from this analysis showed that maize agriculture was the preferred activity across all risk preferences. This is particularly problematic because the current agricultural practices are leading to the long-term decline of crop yields and many experts believe wide-scale smallholder adoption of agroforestry is necessary in order to maintain or enhance food security in the country.

Policy makers can take several actions to increase the adoptability of agroforestry. First, agroforestry activities can be made more attractive by reducing the sources of uncertainty that create risk from the perspective of smallholders, especially the risks related to precipitation and market prices. Rwandan smallholders are willing to pay risk premiums to reduce their exposure to climactic risk and a nascent agricultural insurance industry is emerging to meet this demand. However, insurance cannot increase the attractiveness of agroforestry on its own because smallholders will be able to buy insurance regardless of land use. More importantly, the results from this study showed that agroforestry is inherently more risky than agriculture, which means agroforestry risk premiums will be higher. Policy makers could subsidize agroforestry premiums to increase the attractiveness of the activity as long as the social costs of the market distorting effects of the subsidy were less than the social benefits of increased agroforestry adoption.

Providing agricultural extension services to farmers is another way to reduce the risk of agroforestry. One source of risk in this study was the wide-ranging effect agroforestry has been observed to have on crop yields. If following a set of best practices can systematically reduce this variation, extension and outreach programs could work with smallholders to share this information and reduce the risk associated with agroforestry.

Upfront payments that reduce the cost of adopting agroforestry would also improve adoption as shown in the Certainty Equivalence analysis. Payments could be made in cash or in kind. One potential solution would be to offer smallholders vouchers of a certain value that are redeemable at local tree nurseries for a certain number of trees of the smallholders choosing. Similar voucher programs have been put in place to encourage the use of artificial fertilizers and results from Rwanda suggest that smallholders are willing to adopt agroforestry practices if they can choose tree species tailored to their individual household needs.

Another option is to offer safety nets to farmers who adopt agroforestry. A safety net works by providing some sort of assistance, either food or cash, to smallholders in times of adverse weather shocks. If the safety net available contingent on the adoption of agroforestry it would reduce the risk found in the tail of the distribution of payoffs. This would change the risk profile of the activity to make it more competitive with agriculture. This would primarily influence the land use decision of very risk-averse smallholders who are the most sensitive to the risk found in the left hand tail of the pay-off distribution.

Increasing the adoptability of best management practices for woodlots will require reducing their costs. The results of the Stochastic Dominance analysis of woodlot management practices show that current practices, which are said to be poor by environmental authorities in the country, display First Order Stochastic Dominance over well-managed woodlot practices. This analysis showed that best practices are very costly relative to the marginal gains in timber yields they achieve. Smallholders with utility functions that value more over less will strictly prefer to continue with the current woodlot management practices. The second order Stochastic Dominance analysis showed that the best practices also did not lower variability of woodlot returns enough for risk-averse smallholders to prefer this management style over current practices. The results of this analysis suggest that higher stocking densities can improve the productivity of woodlots. However, the suite of best management practices that are also being recommended will not be adopted. Even including the values of ecosystem services does not change the profitability enough for smallholders to adopt best management practices. This suggests that policies designed to incentivize good woodlot management may not be justified from a social benefit perspective.

CHAPTER 3: CAN PAYMENTS FOR ECOSYSTEM SERVICES SCHEMES INCENTIVEZE THE RESSTORATION OF SOIL FERTILITY IN CULTIVATION-FALLOW SYSTEMS: A NUMERICAL ANALYSIS

3.1 Introduction, Problem Statement, and Contribution

Pressure on agricultural land in Sub-Saharan Africa (SSA) is leading to a decline in soil fertility, crop yields, and food security (Franzel, 1999). While farms of all sizes are affected, the problem is especially serious for smallholder farmers (Barbier, 2000). Estimates suggest the soil degradation costs smallholder farmers in SSA \$5.4 billion year^{-1} in lost crop productivity (Dreschel et al., 2001). Traditionally, soil fertility has been maintained on smallholder land through rotational farming systems where land is periodically taken out of production to allow soil nutrients to regenerate. However, these rotational systems have largely been replaced by continuous cultivation practices as population pressure, shrinking smallholder farm sizes, and a growing agricultural land rental market creates incentives for smallholders to cultivate continuously (Jayne et al., 2014). Unlike rotational farming systems, which maintain soil nutrients over time, continuous cultivation gradually reduces soil nutrient levels and leads to the long-term decline of crop yields (Franzel, 1999). As a result, food security continues to be a widespread concern throughout the region (FAO, 2015).

A number of SSA countries have spent considerable amounts of public money supporting inorganic fertilizer subsidies in an attempt to improve soil fertility on smallholder farms. For example, in 2008 Malawi and Nigeria spent approximately 72% and 42% of their respective agricultural budgets on fertilizer subsidies (Dorward and Chirwa, 2010; Mogues et al., 2008). However, evaluations of the effectiveness of fertilizer subsidies in SSA have shown that they suffer from inefficiencies due to high costs, mismanagement of funds, elite capture, and ineffectiveness in reaching smallholder farmers (Umar et al., 2011; Duflo et al., 2008). Moreover, at a biophysical level soil degradation reduces the ability of crops to utilize the nutrients contained in artificial fertilizers (Marenya et al., 2012). As a result, the ability of smallholder farmers to benefit from the application of inorganic fertilizers is inhibited because they are farming on depleted soils that are less responsive to fertilizer application.

So called ‘Improved’ tree fallows, which use nitrogen-fixing trees and shrubs in rotation with crops to restore soil nutrients on agricultural land, have been suggested as an alternative low cost way for smallholder agriculturists to maintain and even improve soil fertility (Franzel, 1999). Improved tree fallows can restore soil fertility more quickly than regular fallows. Improved tree fallows also supply a number of socially valuable ecosystem services. For example, long fallow periods can allow secondary forest to regenerate, providing similar levels of carbon sequestration and other ecological functions as virgin forests (Klemick, 2011). Improved fallows have also been shown to reduce erosion, increase protection of nearby forests, create new habitat for biodiversity, and enhance watershed protection (Pattanayak and Depro, 2004).

Improved fallows also have a number of societal benefits including increasing food security by maintaining higher soil fertility levels and lowering poverty rates by providing additional sources of farm income (Franzel, 1999). While controlled field trials and laboratory experiments have shown improved fallows can create private and public benefits, adoption rates have been low and dis-adoption is a common occurrence. This is partially explained by the facts that the information on the expected benefits of improved fallows are not well documented or communicated to smallholder farmers, who are reluctant to invest scarce financial and temporal resources in untested technologies (Pattanayak and Depro, 2004).

Land use decision making is largely determined by financial incentives and this leads to land uses that emphasize the production of salable commodities rather than a more holistic production sets that include public ecosystem goods and services (Barbier, 2000). The public good nature of the benefits associated with improved fallows suggest that even when smallholders adopt the practice they are likely to allocate less land to or fallow for shorter periods than society would prefer. Researchers have suggested that payments for ecosystem services schemes (PES) could correct the public good market failure and also encourage greater soil fertility conservation (Liu et al., 2011; Franzel, 1999; Wunder et al., 2005).

The theoretical evidence on the direction and the size of the effect that PES would have on soil fertility in cultivation-fallow systems is limited and somewhat ambiguous. Empirical literature on the topic is virtually non-existent. Some authors have used theoretical models to characterize the land use decisions of smallholders in response to changes in wealth, land quality, and output prices (e.g. see Balsdon, 2008; Batabyal and Lee, 2003; Wilassen, 2001; Barett, 1991). These studies did not evaluate how a price mechanism tied to a specific land use practice would impact soil fertility levels over time.

To address these gaps this chapter explores the incentives facing smallholders managing cultivation-fallow systems. The decision making process is first represented by a single period analytical model of a smallholder agriculturists decision to choose the optimal length of time to fallow before cultivating the land. This model is used to explore how payments for fallowing, output prices, discount rates, and transition costs influence the optimal fallow length. Since smallholders have the option of switching from one activity to another at any point in time a more complete analytical model is used to further explore the incentives facing smallholders.

The descriptive limits of this model are quickly reached and other methods are needed to answer the central question of this chapter. A dynamic programming model of a cultivation-fallow system is developed to provide evidence on the effects that payments for ecosystem services would have on soil fertility on smallholder farms using a case study of a cultivation-fallow system in Zambia.

This contribution is important and timely because worldwide more than 300 million smallholder agriculturists are shortening fallow periods, adopting new soil management activities, and intensifying agricultural production (Mareyna et al., 2011). It is increasingly important to understand how policies, including payments for ecosystem services, influence the land management decisions of smallholder agriculturists, especially in rotational land management systems. This paper also contributes to the literature on conservation planning and evaluation by demonstrating how dynamic programming methods can help policy makers and conservation agencies evaluate the efficiency of un-tested policies on dynamic systems.

3.2 Background on PES and fallow-cultivation models

In the general form of the cultivation-fallow problem a smallholder is tasked with maximizing the discounted flow of present and future benefits from a field by choosing the length of time the land is cultivated before it is fallowed and vice versa. The productivity of the land in each time period is determined by the fertility of the soil, which evolves positively or negatively depending on whether the land is being fallowed or cultivated. In each time period the smallholder faces a decision to continue in the current phase of fallow or cultivation or switch to the alternate phase. The decision to remain in the current phase is determined by the value being earned relative to the opportunity cost of remaining in the current phase. If the smallholder starts from a cultivation phase with abundant soil nutrients the opportunity cost of continuing to cultivate is relatively low. The smallholder will continue to cultivate until the current benefits of cultivating the field is less than or equal to the opportunity cost of doing so reflected in the discounted future benefits of fallowing the field. At this point the smallholder will switch to fallow. The fallow phase will continue until the opportunity cost of remaining in fallow (foregone current production) becomes greater than or equal to the benefits of continuing to fallow.

The smallholder's problem is to find the optimal cultivation-fallow strategy that indicates at what soil fertility level to stop cultivation and switch to fallow and at what soil fertility level to stop fallowing and switch to cultivation in order to maximize the net present value (NPV) of the field across the planning horizon. A two period problem is straightforward to solve because the smallholder only needs to account for the impacts of one decision: when to switch from one phase to the other. The solution becomes significantly more difficult to find when the time horizon is infinite because the smallholder needs to account for an infinite stream of costs and benefits for each switching decision.

The theoretical literature on cultivation-fallow systems has focused on evaluating the dynamics of the systems under changing output prices, interest rates, and wealth. No previous study has specifically focused on the effects that payments for adopting improved fallows would have on soil fertility. The theoretical literature has approached the fallow-cultivation problem in two ways. One vein of literature has focused on when to begin either the cultivation or fallow phase of the cultivation-fallow cycle and how that decision is affected by changes in output prices, discount rates, and wealth. The other vein of literature has focused on the holistic view of the problem by asking how both cultivation and fallow lengths change simultaneously in response to changes in output prices and discount rates. Both sets of literature provide insight into an appropriate approach to the problem this paper is exploring: namely can PES schemes increase soil fertility in smallholder managed cultivation-fallow systems.

Balsdon (2008) and Batabyal and Lee (2003) used theoretical optimization models to study the cultivation and fallow sides of the problem in the context of slash and burn agriculture in Southeast Asia. Balsdon (2008) used a theoretical optimization model to estimate the length of time that a field should be cultivated before being abandoned. They also asked how that decision depended upon wealth. The study found that increases in smallholder wealth created an incentive to extract more soil nutrients from a field before abandoning it and clearing a new area, which led to greater degradation of soil nutrients, but less deforestation.

Batabyal and Lee (2003) provided insight into the other half of the problem using a model to solve for the optimal length of time that cleared land should be fallowed before being cultivated. Their study found that as the discounted financial returns to fallowing land increase the optimal length of the fallow period also increases. The results from these studies suggest that PES payments could have ambiguous effects on soil fertility. On the one hand, payments could lead to an increase in cultivation lengths. On the other hand, PES payments could increase the returns to fallowing thereby increasing the length of fallows as suggested by Batabyal and Lee (2003).

Balsdon (2008), Batabyal and Lee (2003), and Barrett (1991) all explored the effects of the discount rate on the cultivation-fallow problem. Balsdon (2008) showed that an increasing discount rate caused smallholders to extract soil nutrients more rapidly. It also caused smallholders to extend the length of the cultivation period to avoid or postpone the cost of shifting fields. Batabyal and Lee (2003) and Barrett (1991) showed that when the only decision is how long to fallow land before cultivating it the optimal response to an increase in the discount rate is to reduce the length of the fallow. Evidence on the effect that an increase in the discount rate would have on the optimal lengths of fallow and cultivation are not documented in the literature. The evidence presented above suggests that under high discount rates one would expect longer periods of cultivation and shorter periods of fallow, all else equal. These findings are particularly important in smallholder settings where discount rates are often high (Barbier, 2000).

While these results provide some insight into the fallow-cultivation problem they do not address the central question of this chapter. Balsdon (2008), Barrett (1991) and Willassen (2001) all considered the effect that increasing output prices would have on average levels of soil fertility. Balsdon (2008) showed that in a slash-and-burn agricultural system higher output prices would increase the marginal cost of delaying the beginning of a new cultivation cycle. Barrett (1991) used his results to qualitatively argue that a change in output price would have no impact on the stock of soil fertility because the increased price of outputs increases both the benefits and costs of a longer fallow period by the same amount. Willassen (2011), in contrast to Barrett (1991), showed that an increase in the output price would create incentives for smallholders to switch from fallow to cultivation at lower fertility levels and to switch from cultivation to fallow at higher fertility levels,

These results suggest that PES would increase the opportunity cost of remaining in a cultivation phase, thereby shortening cultivation periods. All else equal, one would expect the payment for environmental services to increase average soil fertility levels over time. Yet, as no closed form solution to the problem exists it is difficult, if not impossible, to analytically determine the magnitude by which soil fertility would improve. As a result, it is necessary to study the problem through a numeric dynamic programming model. However, first it is useful to present a simple model of a single-shot fallow-cultivation model to gain some basic understanding of the problem.

3.3 Analytic Model

Consider a fallow-cultivation system that is managed by a smallholder to maximize the present value of the current and future flows of net benefits over finite and infinite time-horizons. Unlike the Faustmann (1849) model of optimal forest rotation, where the payoff of growing timber is received at the end of the rotation interval, in the fallow-cultivation system income from fallowing and cultivating is received continuously during the respective fallow and cultivation periods. The smallholder's problem is to choose the respective fallow and cultivation periods that maximize the present value of the land.

Before any decisions are made a smallholder inherits a field with a given soil fertility level that gives rise to an instantaneous rate of productivity, V_t, whose instantaneous net benefit for time t is given by $P * V_t$, where P is the output price. When land is cultivated its soil fertility and the associated productivity it generates for the farmer declines at a constant rate denoted by D. If land with an initial soil productivity of V_0 were put directly into cultivation any period of time, the productivity at any time would be given by equation [3.1].

$$V_t = V_0 e^{-Dt} \qquad [3.1]$$

Faced with a discount rate of r, the discounted presented value of profits earned by the farmer over a period of T years would be given by equation [3.2].

During fallow, soil productivity evolves over time following the non-linear soil productivity equation.

$$\int_0^T \left[P * V_0 e^{-(D+r)t}\right] dt \qquad [3.2]$$

The equation describing the dynamic process of soil depletion developed by Trenbath (1984; 1989) and shown in Figure 3.1:

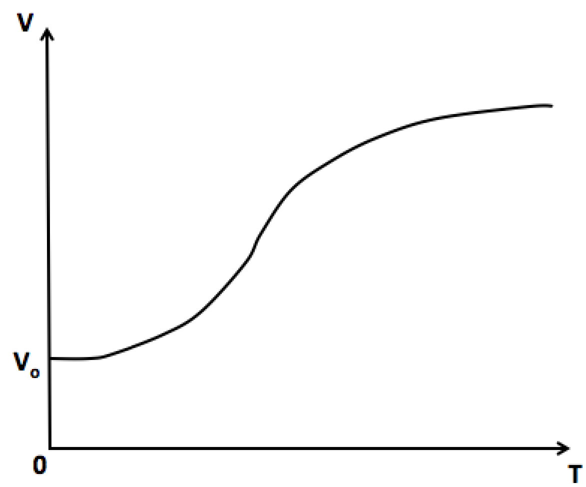

Figure 3.1: Soil productivity growth curve

Figure 3.1 shows that under the assumptions of the model the marginal benefit of fallowing land for an additional year increases when soil productivity levels are low and decreases when soil productivity levels are high. Following from this assumption: If land with an initial soil productivity of V_0 were put into fallow for a period of τ years, its instantaneous cultivation benefit upon termination of the fallow phase would be given by equation [3.3].

Under a PES scheme a conditional annual payment, B, would be made whenever land is left fallow.

$$P * V = P * V(\tau, V_0) \qquad [3.3]$$

The smallholder would bear an annual expense to keep the land in fallow, which is measured by C. The net benefit of fallowing land with a starting soil productivity level of V_0 for a period of τ years followed by a cultivation period of T years would be given by:

$$\int_0^\tau [e^{-rt}(B - C)]\, dt + \int_0^T \left[P * V(\tau, V_0)e^{-Dt}e^{-r(\tau+t)}\right] dt \qquad [3.4]$$

The first term measures the discounted net financial benefit received by the smallholder from the payment for ecosystem services received while the land is under fallow. The second term measures the discounted financial benefit received by the smallholder from the production and sale of crops during the cultivation period with the soil productivity that accumulated during the fallow period.

The smallholder's problem is to choose the fallow and cultivation periods ($\boldsymbol{\tau}$, T) that maximize the discounted present value of the land. Over a single fallow-cultivation cycle the problem can be formally written as:

$$Max_{\tau,T} \int_0^\tau [e^{-rt}(B - C)]\, dt + \int_0^T \left[P * V(\tau, V_0)e^{-\gamma t}e^{-r(\tau+t)}\right] dt \qquad [3.5]$$
$$s.t.$$
$$V_0 = Constant$$

The solution to this problem is given by the first order conditions representing the derivatives of equation [3.5] with respect to τ:[10]

$$\frac{\partial NPV}{\partial \tau} = e^{-r\tau}(B - C) + \int_0^T [P * \frac{\partial V(\tau,V_0)}{\partial \tau} e^{-Dt} e^{-r(\tau+t)}]dt - r\int_0^T [P * V(\tau,V_0)e^{-Dt}e^{-r(\tau+t)}]dt = 0 \quad [3.6]$$

Equation [3.6] implicitly defines the optimal fallow length (τ^*) for the single-shot fallow-cultivation problem. Equation [3.6] can be re-written as the familiar result to the Faustmann model where τ^* should be chosen such that:

$$(e^{-r\tau}(B - C) + \int_0^T [P * \frac{\partial V(\tau,V_0)}{\partial \tau} e^{-Dt} e^{-r(\tau+t)}]dt) / \int_0^T [p * V(\tau,V_0)e^{-\gamma t}e^{-r(\tau+t)}]dt = \mathrm{r} \quad [3.6a]$$

Equation [3.6a] simply says that land should be left fallow as long as its value, measured in terms of current payments for ecosystem services and future crop yields, rises at a rate greater or equal to the rate of discount. In the absence of payments for ecosystem services, equation [3.6] reduces to:

$$(e^{-r\tau}(-C)\int_0^T [P * \frac{\partial V(\tau,V_0)}{\partial \tau} e^{-\gamma t} e^{-r(\tau+t)}]dt / \int_0^T [P * V(\tau,V_0)e^{-\gamma t}e^{-r(\tau+t)}]dt = \mathrm{r} \quad [3.6b]$$

Equation [3.6b] simply says that land should be left fallow so long as the discounted future value of net crop benefits grows faster than the rate of discount.

Comparing [3.6a] and [3.6b] it is apparent that payments for ecosystem services increase the size of the numerator relative to the denominator for a given τ. The denominator is increasing with respect to τ, while both terms in the numerator are decreasing with respect to τ. In order for the identity to hold when payments for ecosystem services are included the numerator must shrink and the denominator must grow. This means that that the optimal τ when payments for

[10] The derivatives of the objective function were derived using the second fundamental theorem of calculus ($if\ F(T) = \int_0^T f(t)dt\ then\ \frac{\partial}{\partial T}F(T) = f(T)$) and Leibniz's rule (if $\mathrm{F}(\mathrm{T},\tau) = \int_0^\tau f(T,t)dt\ then\ \frac{\partial}{\partial T}\mathrm{F}(\mathrm{T},\tau) = \int_0^\tau \frac{\partial}{\partial T} f(T,t)\,dt$).

ecosystem services are included in the objective function is greater than the $\boldsymbol{\tau}$ that is selected when payments are not included. In the single-shot problem once the land is put into cultivation the cultivation length (T) is chosen such that $\frac{\partial NPV}{\partial \tau} = 0$ at the end of cultivation cycle since it is not possible to return to the land to fallow.

The simplified problem represented by equation [3.6] serves the purpose of exposing some of the basic intuition of the farmer's decision making. To do this it assumes that land cannot be followed after it is cultivated. In other words, cultivation has no opportunity cost. Of course as long as the manager has some stake in the future, cultivation of land does have an opportunity cost because it can always be returned to fallow so that its future productivity is greater and this cycle can be repeated an infinite number of times (Barrett, 1991). The more appropriate problem reflecting this additional complication can be written as:

$$Max_{\tau,T} \sum_{0}^{T} [e^{-rt}(B - C)] + \sum_{0}^{T} \left[P * V(\tau, V) e^{-Dt} e^{-r(\tau+t)}\right]$$

$$+ \sum_{0}^{T} \left[e^{-r(\tau+T+t)}(B - C)\right] + \sum_{0}^{T} \left[P * V(\tau, V) e^{-Dt} e^{-r(\tau+T+\tau+t)}\right] + \cdots \qquad [3.7]$$

$$s.t. V = V_0$$
$$V_1 = V(\tau, V_0) e^{-DT}$$

The solutions to this problem are given by the first order conditions representing the derivatives of equation [3.7] with respect to τ and T:

$$\frac{\partial}{\partial \tau} = \sum_{n=1}^{\infty} e^{-rn(\tau+T)}(B-C) + \sum_{n=1}^{\infty} \int_0^T [P * \frac{\partial V(\tau,V_0)}{\partial \tau} e^{-\gamma t} e^{-rn(\tau+t)}]dt - r\sum_{n=1}^{\infty} \int_0^T [P * V(\tau,V_0) e^{-\gamma t} e^{-rn(\tau+t)}]dt = 0 \qquad [3.8]$$

$$\frac{\partial}{\partial T} = \sum_{n=1}^{\infty} P * V(\tau,V_0) e^{-\gamma t} e^{-rn(\tau+T)} = r\sum_{n=1}^{\infty} \int_0^{\tau} [e^{-rn(\tau+T+t)}(B-C)]dt \qquad [3.9]$$

Equations [3.8] and [3.9] are a pair of simultaneous equations that implicitly define the optimal fallow and cultivation periods (τ^*, T^*) for the infinite-horizon fallow-cultivation problem. Equation [3.8] has a similar interpretation to equation [3.6] except that the decision to increase the length of the fallow by one unit of time has an impact on the current and all future periods. The first two terms in equation [3.9] represent the marginal benefit of extending the length of each fallow period by an additional unit of time. The first term represents the discounted net benefit of receiving an additional payment for ecosystem services. The second term represents the benefit of carrying additional soil productivity into each subsequent cultivation phase. The third term represents the opportunity cost of extending the current and future fallow periods by a unit of time, which extends the length of time between fallow periods and the time the soil productivity benefits are realized. Equation [3.8] can be re-organized in a similar manner to [3.6a] such that:

$$\sum_{n=1}^{\infty} e^{-rn(\tau+T)}(B-C) + \sum_{n=1}^{\infty} \int_0^T [P * \frac{\partial V(\tau,V_0)}{\partial \tau} e^{-\gamma t} e^{-rn(\tau+t)}]dt / \sum_{n=1}^{\infty} \int_0^T [P * V(\tau,V_0) e^{-\gamma t} e^{-rn(\tau+t)}]dt = r \qquad [3.8a]$$

Equation [3.8a] says that optimal fallow length, τ^*, should be chosen such that the discounted value of the land rises at the same rate as the rate of discount.

Equation [3.9] recognizes that there is an opportunity cost of cultivating because land can always be fallowed. The equation says that the optimal length of cultivation should be chosen such that the discounted marginal benefit of extending cultivation by one year in the present and all future periods exactly equals the discounted marginal cost of delaying the payments for ecosystem services during fallow periods.

While these equations are useful for gaining some intuition behind the incentives that influence smallholder decision-making, they are analytically intractable in the sense that they cannot be solved for the optimal fallow and cultivation lengths for specific parameter values. To do this we must turn to a dynamic programming model of the system.

3.4 A Numeric Model of the Cultivation-Fallow Decision

The model represents the cultivation-fallow decision making process of a single agriculturist growing maize and representing a case study from Zambia. In Zambia, the main food producers are resource poor smallholders who manage complex rotational farming systems with low inputs, low soil fertility, and low crop yields (Franzel, 1999). Maize is the staple crop of the country and accounts for 60% of agricultural production. Crops are rain fed and produced without any artificial fertilizers, pesticides, or mechanization. Farmers are very remote and lack access to research and extension services, high yield seeds, capital, and credit. Even when such inputs are available farmers are often unable to afford them (Franzel, 1999).

Zambian smallholders have reduced fallow lengths in favor of cultivating continuously without investing in nutrient replacing inputs (Kwesiga et al., 1999). Continuous cultivation gradually reduces soil fertility and leads to lower annual crop yields. Using short duration tree fallows of one or more years can improve the future stock of soil fertility and the resulting flow of crop yields. However, the marginal benefit of an additional year of fallow, in terms of future crop benefits, has been shown to decline with time. This means smallholders face a dynamic tradeoff between allocating land to cultivation or fallow. The rate of soil nutrient loss and nutrient accumulation have both been shown to be non-linearly related to time, complicating the smallholder's problem (Van Noordwijk, 1999).

Trenbath (1984; 1989) proposed a non-linear system of equations depicting the dynamic process of soil depletion and soil restoration during cropping and fallow periods, respectively. In the model crop yields are assumed to be directly proportional to a soil fertility index X $\in [0,1]$. During the cultivation phase field observations of crop yields under continuous cultivation have shown that soil fertility declines each period by a crop-dependent proportion, D (Van Noordwijk, 1999). Soil fertility evolves each period according to:

$$X_t = X_o * (1 - D)^t \qquad [3.10]$$

Where:

X_t = soil fertility at time t, $X_t \in [0,1]$

X_o = soil fertility at time t=0
D = crop specific soil fertility reduction factor (unit-less)

The crop yield associated with each state of soil fertility, X_t, is given by:

$$Y_t = \propto * X_t \qquad [3.11]$$

Where:

Y_t = crop yields at time t (short tons ha^{-1})

$\propto$ = crop specific conversion efficiency factor

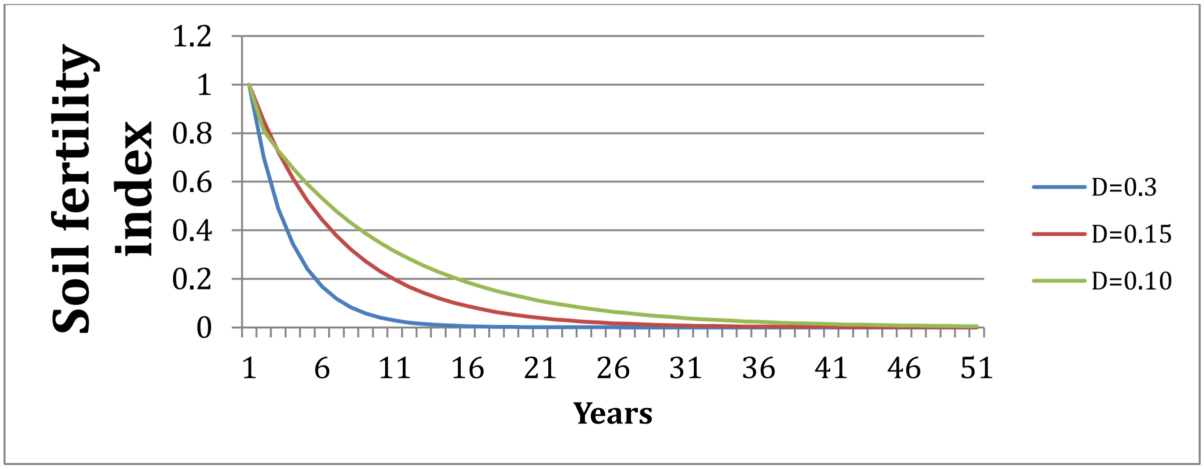

Figure 3.2: Soil fertility as a function of years of cropping for three nutrient reduction factors from (Van Noordwijk, 1999).

Figure 3.2 shows how soil fertility and the associated crop yields decline over time under continuous cultivation. Under a low impact crop (D = 0.1), soil fertility can be maintained at moderate levels for as many as 7 to 8 years. For Medium impact crops (D = 0.15) the fertility of the soil declines at a faster rate that approaches zero after 25 years. High impact crops (D = 0.3) fertility can approach zero after only 10 years.

Trenbath (1984; 1989) showed that soil fertility can be improved during fallow periods and that recovery of soil fertility follows an asymptotic path towards the maximum achievable fertility level according to:

$$X_t = \frac{(X_{max} - X_{t-1})^2}{X_{max}(1+K_f) - X_{t-1}} \quad [3.12]$$

Where:

X_t=soil fertility at end of fallow period

X_{max}=maximum achievable fertility

K_f=fallow efficiency factor

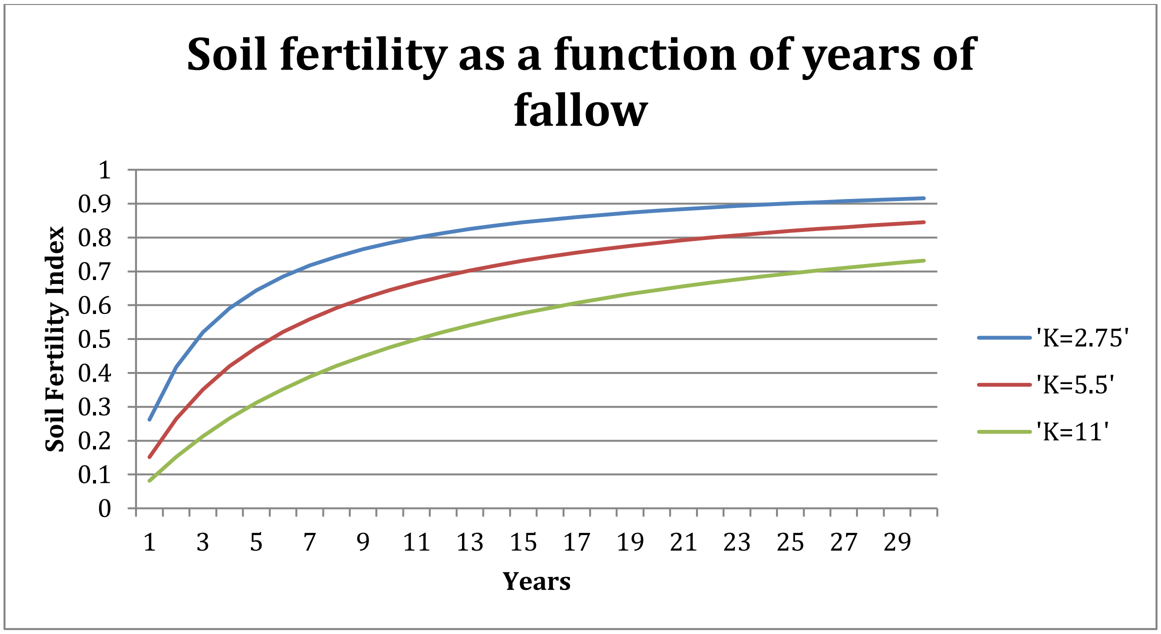

Figure 3.3: Soil fertility as a function of years of fallow under three 'Half-recovery time' parameterizations from (Van Noordwijk, 1999).

Figure 3.3 shows how soil fertility and the associated crop yields improve over time under different fallows of different efficiency. Under a low efficiency fallow (K=11), completely degraded soil can be restored to 50% fertility in approximately 10 years. For medium efficiency fallows (K=4) the fertility of the soil improves at a faster rate that approaches 50% fertility in approximately 4 years, while high efficiency fallows (K=1) can restore soil fertility to 50% in 1 year. In all cases, the change in the marginal soil fertility improvement with respect to time is positive, but decreasing.

Since soil fertility depends on the control chosen by the smallholder the state equation for soil fertility is:

$$X_{t+1} = \begin{cases} X_t * (1 - D) & for\ \gamma_t = 0 \\ \left(X_t + \frac{(X_{max} - X_t)^2}{X_{max}(1 + K_f) - X} \right) & for\ \gamma_t = 1 \end{cases} \qquad [3.13]$$

Where $\boldsymbol{\gamma_t}$ denotes the state of the field (i.e. cultivation $\gamma_t = 0$, fallow $\gamma_t = 1$) at each point in time. The two terms in equation [3.15] track the hectare-level outflows and inflows of soil nutrients as a function of soil fertility and the state of the field (i.e. cultivation or fallow). When the field is being cultivated the stock of soil fertility in the next period would decline following X_{t+1}=$X_t * (1 - D)$. If the field is being fallowed the stock of soil fertility in the next period would be $X_{t+1} = \left(X_t + \frac{(X_{max}-X_t)^2}{X_{max}(1+K_f)-X_t}\right)$.

3.4.2 Objective function

The smallholder's problem is to observe both the soil fertility and state of the land (i.e. cultivation or fallow) each period and maximize the discounted stream of current and future revenues by deciding whether to cultivate or fallow the land subject to an equation of motion governing soil fertility. If the state of the field changes from cultivation to fallow or from fallow to cultivation the smallholder must pay a switching cost. If the field remains in the same state the smallholder only has to pay the management costs associated with that state.

The problem is formally represented by:

$$Max\ \pi_{\gamma(t)} = \sum_{t=0}^{\infty} \delta^t \{(P_C * Y_t(X_t; \alpha) - C_c) * (1 - \gamma_t) + \left((P_{ES} - C_F) * \gamma_t\right) - C_S * \left(\gamma_t - (\gamma_{t-1} * \gamma_t)\right) - C_S * (\gamma_{t-1} - (\gamma_t * \gamma_{t-1}))\} \qquad [3.14]$$

Subject to:

$$X_{t+1} = \begin{cases} X_t * (1 - D) & for\ \gamma_t = 0 \\ \left(X_t + \dfrac{(X_{max} - X_t)^2}{X_{max}\left(1 + K_f\right) - X_t}\right) & for\ \gamma_t = 1 \end{cases}$$

$$\gamma_0 = \tilde{\gamma}$$

$$X_0 = \tilde{X}$$

$$\gamma_t \in [0,1]$$

Where:

P_C = Crop price (\$ ton^{-1})

C_c = Cost of cultivation (\$ ha^{-1} year^{-1})

P_{ES} = Price for fallowing (\$ ha^{-1} year^{-1})

C_F = Cost of fallowing (\$ ha^{-1} year^{-1})

C_S = Cost of switching from cultivation to fallow or fallow to cultivation (\$ ha^{-1})

$\bar{\gamma}$ = Starting phase of the cultivation-fallow system

$\tilde{X}$ = Starting soil fertility level

δ = Discount factor

D = reduction factor of soil fertility from cropping

c = conversion efficiency of soil fertility to crop yield

During the cultivation phase the financial benefits of crop production depend positively on the soil fertility level. However, the financial benefits associated with the fallow phase are positively related to the price paid to smallholders to fallow their land and are determined independently of the soil fertility level. The solution to this problem is a strategy that determines the optimal action to take for any field with a state given by the soil fertility X and phase γ. The optimal cultivation-fallow strategy is defined as a set of two soil fertility levels, X^* or X_*, which together determine when it is optimal to switch from cultivation to fallow (X_*) and when it is optimal to switch from fallow to cultivation (X^*).

3.4.3 Calibration and parameters

The parameter values for the model are listed in Table 3.1. The soil fertility model was calibrated and parameterized using estimates from (Kwesiga and Coe, 1994) who studied the crop yield improvements achieved with *Sesbania sesban* fallows in Chipata, Zambia. Improved fallows were established for 1, 2, or 3 years at densities of 0.5m x 0.5m, 0.7m x 0.7m, and 1m x 1m, respectively. In the experiment the performance of two fallow treatments were compared against the performance of continuously cultivating maize on control plots. In the first treatment, plots were fallowed for 1, 2, and 3 years, the year after which unfertilized maize yields were 2.27, 5.59, and 6.02 t ha^{-1}, respectively. In the second treatment, plots were fallowed for 3 years after which they were fertilized with 112 kg N ha^{-1} during continuous cultivation. Maize yields in this treatment declined from 6.09t ha^{-1} to 4.88 and 4.28t ha^{-1} after three years of continuous cultivation. Maize yields in the control plots were 1.6, 1.2, and 1.8t ha^{-1}, following 1, 2, and 3 years of continuous cultivation. In one plot, maize yields following a 3-year fallow with the application of 112 kg N ha^{-1} were measured to be 7.57 t ha^{-1}.

The model is calibrated as follows. To estimate the crop efficiency parameter, α, the yield achieved following the 3 year improved fallow with N application is assumed to correspond to the highest attainable soil fertility index.

This information is used to solve equation [3.11] for the crop efficiency parameter, α.[11] The baseline soil fertility index starting value, X_0, is calibrated to the yields of the control plots using equation [3.10] and the value of the crop efficiency parameter, α.[12] The data from the second treatment were used to estimate the crop specific soil fertility reduction factor, D.[13]

Table 3.1: Parameter values used to calibrate and solve the model

Parameter	Value	Source
X_0 = Baseline starting value of soil fertility index	0.2	Kwesiga and Coe (1994)
K_F = Fallow efficiency factor	4 (1,10)	Kwesiga and Coe (1994)
D = crop specific soil fertility reduction factor	0.1 (0.2, 0.3)	Kwesiga and Coe (1994)
α = crop specific conversion efficiency factor	7.57t unit of soil fertility^{-1}	Kwesiga and Coe (1994)
X_{max} = maximum achievable soil fertility index	1	
P_c= Crop price	\$66 t^{-1}	(Franzel, 1999)
C_c= Cost of cultivation	\$62 ha^{-1}	(Franzel, 1999)
P_{es}= Payment for services from improved fallow	\$100 ha^{-1} year^{-1} (0, \$200)	(Franzel, 1999)
C_f= Cost of improved fallow	\$60 ha^{-1}	(Franzel, 1999)
Cs = Switching cost	\$100 ha^{-1}	(Franzel, 1999)
Υ= Starting state (fallow)	1	Equation 3.8
δ= Discount factor	0.8 (0.7, 0.9)	

Van Noordwijk (1999) reported a range of unit-less fallow efficiency factors, K_F, which reflect the rate at which for soil fertility recovers under different types of fallow. The values range from

[11] $Y = \alpha * X_t$ is set equal to $7.57 = \alpha * X_t$ where $X_t = 1$ by assumption. The equation reduces to $\alpha = 7.57$.

[12] Maize yields in the control plots were 1.6, 1.2, and 1.8t ha^{-1}, following 1, 2, and 3 years of continuous cultivation. The estimated soil fertility index of the control was found by setting the identify $Y = 7.57 * X_t$ equal to $1.6 = 7.57 * X_t$ and solving for X_t.

[13] The maize yields in this treatment declined from 6.09t ha^{-1} to 4.88 and 4.28t ha^{-1} after three years of continuous cultivation. Using the estimated crop specific conversion efficiency factor to solve for the soil fertility index value, these yields correspond to soil index fertility levels of X = 0.80, X = 0.64, and X = 0.57, respectively. Equation [3.1] is re-arranged such that $d = 1 - X_t/X_0$ and the equation is solved for d using all three data points. For X_0 = 0.80 and X_t = 0.64, D = 0.2 and for X_0 = 0.64 and X_t = 0.57, D = 0.1. I use the lower bound as a conservative estimate of the rate of soil fertility loss due to continuous cultivation.

1 to 11. High values of K_F are inversely related to the efficiency of the fallow in equation [3.11]. I use $K_F = 4$ as a rough approximation of the efficiency of improved fallows following (Van Noordwijk, 1999).

Franzel (1999) reported the financial costs of improved *Sesbania sesban* fallows and maize cultivation in Zambia in 1996 dollars. Since the model treats prices and costs as relative, the unadjusted price and cost estimates from Franzel (1999) are used in the model. He reported that the cost of maize cultivation was \$62.16 ha^{-1} $year^{-1}$, while the cost of *Sesbania sesban* fallows was \$60.51 ha^{-1} $year^{-1}$. He assumed maize seed was applied at a rate of 20kg ha^{-1} at a cost of \$1kg^{-1}. Cultivation was also assumed to require 100 days of labor at a cost of \$0.4 day^{-1}. This represents the average daily return to agricultural labor. Improved fallows require approximately the same amount of labor in addition to incurring nursery costs to produce the *Sesbania* seedlings. Market prices for maize at the time of the study were reported to be \$66 $tonne^{-1}$. The price paid to smallholders for fallowing their land was reflective of their opportunity cost of doing so. The value of their foregone agriculture production was estimated by calculating the average revenue received from continuous cultivation. The average yield of continuous cultivation was 1.5t ha^{-1} over a three year period, which is equal to \$101 of revenue at an average crop price of \$66 $tonne^{-1}$.

3.4.4 Solution method

Equation [3.14] represents an infinite horizon discrete time continuous state discrete choice dynamic programming model (Miranda and Fackler, 2002). In order to solve this maximization problem equation [3.14] is represented with the Bellman equation.

The Bellman equation, $V(X,\gamma)$, is the unknown value function, which represents the maximum obtainable value of current and future crop production along the optimal path given soil fertility X and field phase γ.

$$V(X,\gamma) = Max_{\gamma\in[0,1]}\{f(X,\gamma) + \delta * V(g(X,\gamma), h(\gamma))\} \qquad [3.15]$$

The function $f(X,\gamma)$ measures the per season return from cultivation, $g(X,\gamma)$ describes how inter-season soil fertility evolves, and h(γ) describes which state the field will be in the next period. Combined, these functions represent the smallholder's trade-off between cultivation and fallow in the present and the effect of the decision on all future returns. Smallholders with secure land tenure are expected to maximize the discounted present value of their field over an infinite time horizon rather than maximizing the value of crop production over a single period. The solution to the Bellman equation is therefore the optimal strategy for maximizing the value of a field in any state.

Equation [3.15] is a functional equation whose unknowns are not parameters, but entire functions. Functional equations have no closed-form solution and have to be numerically approximated. Miranda and Fackler (2002) describe the problem as a functional equation problem where one must find a function, V, that satisfies V=TV. Functional equations cannot be solved exactly because the unknown is a function whose domain is defined over an infinite number of points and solving for the functional would therefore require imposing an infinite number of conditions on the solution (Miranda and Fackler, 2002).

Interpolation approximates a real-valued, but analytically intractable function, V, with a computationally tractable approximation, $\hat{V}$. This is done by specifying a linear combination of functions, commonly known as basis functions, which will approximate $V(X,\gamma)$. The basis coefficients, c_j , are determined so that [3.7] is satisfied. Since the value function being approximated is two-dimensional the problem is to approximate the function on the two-dimensional interval I = $\{(x_1, x_2)|\ a_i \le x_i \le b_i, i = 1{,}2\}$ where $x_1 = X$ and $x_2 = \gamma$. Miranda and Fackler (2002) define the basis function for state i as $\{\emptyset_{ij}|j = 1{,}2, \dots, n_i\}$ which is an n_i degree univariate basis function for real valued functions defined on $[a_i, b_i]$ and $\{x_{ij}|\ j = 1{,}2, \dots, n_i\}$ is a sequence of n_i interpolation nodes for each state.

They define an n = $\prod_{i=1}^{d} n_i$ degree basis function defined over the two-dimensional interval I as:

$$\emptyset_{j1,j2,\dots jd}(x_1, x_2, \dots, x_d) = \emptyset_{1_{j1}}(x_1)\emptyset_{2_{j2}}(x_2) \dots \emptyset_{d_{jd}}(x_d) \quad \forall\ i = 1{,}2 \dots, d\ and\ j_i = 1{,}2, \dots, n_i \qquad [3.16]$$

A grid of n interpolation nodes for I can also be created by calculating the Cartesian product of the univariate interpolation nodes for each state. With the basis functions and interpolation nodes in hand the approximation of $V(X, \gamma)$ is given by the tensor product:

$$\hat{V}(X, \gamma) = \sum_{j1}^{n_1} \sum_{j2}^{n_2} \cdots \sum_{jd}^{n_d} c_{j1} \dots c_{jd} \emptyset_{1_{j1}}(x_1)\emptyset_{2_{j2}}(x_2) \dots \emptyset_{d_{jd}}(x_d) \qquad [3.17]$$

In matrix notation this system can be written as:

$$\Phi c = \hat{V} \qquad [3.18]$$

Where $\Phi = \Phi_d \otimes \Phi_{d-1} \otimes \cdots \Phi_1$is an n x n interpolation matrix that is the tensor product of the univariate interpolation matrices and $\hat{V}$ is the n x 1 vector of the approximated values of $V(X, \gamma)$ calculated at the interpolation nodes. This represents a system of n linear equations with n unknowns.

Approximating functional equations requires a specification of the basis functions and collocation nodes. The n basis functions should be linearly independent in order to store as much information about the value function as possible (Miranda and Fackler, 2002). The authors suggest using Chebychev polynomials is a good approach because they are orthogonal to each other. Miranda and Fackler (2002) also show that linearly spaced nodes are not efficient from an approximation perspective because too few nodes are located toward the boundaries of the state-space, where approximation errors can be largest. Chebychev nodes overcome this limitation by placing more nodes toward the edge of the state space in order to add data to areas where uncertainty is greatest.

For the current problem there are i = 2 states representing soil fertility and the state of the field, respectively. Soil fertility is represented by a third degree Chebychev polynomial (i.e. $n_1 = 3$) basis defined on the interval [0,1]. The dichotomous state of the field is represented by second-degree Chebychev polynomial (i.e. $n_2 = 2$) basis defined on the interval [0,1]. The Chebychev nodes for each state are calculated following equation [3.19].

The Chebychev nodes are mapped to the [-1,1] interval for a tighter fit over the state space following:

$$x_{ij} = \frac{a_i+b_i}{2} + \frac{b_i-a_i}{2} * Cos\left(\frac{n_i-j_i+0.5}{n_i} * \pi\right), \forall i = 1,2 \; and \; j_i = 1,2,\dots,n_i \qquad [3.19]$$

$$z_{ij} = 2 * \frac{x_{ij}-a_i}{b_i-a_i} - 1 \qquad \forall i = 1,2 \; and \; j_i = 1,2,\dots,n_i \qquad [3.20]$$

This leads to two matrices of Chebychev nodes, one for each state:

$$z_1 = [-0.865 \quad 0.001 \quad 0.865] \qquad [3.21]$$

$$z_2 = [-1 \quad 1] \qquad [3.22]$$

The univariate basis functions are calculated at the nodes of each state as follows:

$$\emptyset_{11} = 1$$

$$\emptyset_{12} = z_{1j}$$

$$\emptyset_{13} = 2 * z_{1j} * \emptyset_{12} - \emptyset_{11} \qquad [3.23]$$

$$\emptyset_{21} = 1$$

$$\emptyset_{22} = z_{2j}$$

The univariate basis matrices for this problem are then:

$$\emptyset_1(z_1) = \begin{bmatrix} 1 & -0.865 & 2.361 \\ 1 & 0.001 & -0.001 \\ 1 & 0.865 & 0.619 \end{bmatrix} \qquad [3.24]$$

and

$$\emptyset_2(z_2) = \begin{bmatrix} 1 & -1 \\ 1 & 1 \end{bmatrix} \qquad [3.25]$$

The full basis function is the tensor product of $\emptyset_2 \otimes \emptyset_1$:

$$\Phi = \emptyset_2 \otimes \emptyset_1 = \begin{bmatrix} 1 & -0.865 & 2.361 & -1 & 0.865 & -2.361 \\ 1 & 0.001 & -0.001 & -1 & -0.001 & 0.001 \\ 1 & 0.865 & 0.619 & -1 & -0.865 & -0.619 \\ 1 & -0.865 & 2.361 & 1 & -0.865 & 2.361 \\ 1 & 0.001 & -0.001 & 1 & 0.001 & -0.001 \\ 1 & 0.865 & 0.619 & 1 & 0.865 & 0.619 \end{bmatrix} \qquad [3.26]$$

The approximation of the value function is then represented by equation [3.19]. The solution for the basis coefficients that solve the Bellman equation are found through the interpolation procedure described in Miranda and Fackler (2002). First, an initial guess of the coefficient values is put forward. With this guess in place, solve:

$$Max_{\gamma\in[0,1]} \left\{ f(X,\gamma) + \delta \sum_{j1}^{3} \sum_{j2}^{2} c_{j1} c_{j2} \emptyset_{1_{j_1}}(g(x_1,\gamma)) \emptyset_{2_{j_2}}(h(\gamma)) \right\} \qquad [3.27]$$

Equation [3.29] represents the right hand side of equation [3.19] and the initial guesses of the coefficient values multiplied by the full basis function represent the left hand side of the equation. With these guesses in hand the algorithm chooses new coefficient values that minimize the difference between:

$$Min_{c_1 \ldots c_j}\left\{\sum_{j1}^{3}\sum_{j2}^{2} c_{j1}c_{j2}\emptyset_{1_{j1}}(x_1,x_2)\emptyset_{2_{j2}}(x_2)\right\} -$$

$$\left\{f(X,\gamma)+\delta\sum_{j1}^{3}\sum_{j2}^{2} c_{j1}c_{j2}\emptyset_{1_{j1}}(g(x_1,\gamma))\emptyset_{2_{j2}}(h(\gamma))\right\} \qquad [3.28]$$

The sum of the squared differences between the new coefficient values and the old coefficient values is calculated. If the sum of the squared errors is larger than some arbitrary threshold the algorithm uses the new coefficient values to solve equation [3.29] and update the coefficient values as described above. This continues until the sum of the squared differences between the new coefficient values and the old coefficient values is less than or equal to the arbitrarily defined threshold. When this point is reached the approximation $\Phi c = \hat{V}$ is solved for all possible state values.

3.5 Results

3.5.1 Optimal Cultivation-Strategy

The model was solved using GAMS. The baseline parameters in the model assumed a discount rate of 20%, a fallow-efficiency factor of 4, a fertility reduction factor of 0.1, a maize price of \$66 ton^{-1}, and a payment for fallowing of \$100 ha^{-1}year^{-1}. The baseline model was solved for different parameterizations of PES price (\$0 ha^{-1}, \$50 ha^{-1}, \$150 ha^{-}1), discount rate (10%, 30%), price of maize (\$33 ton^{-1}, \$100 ton^{-1}), fallow-efficiency factor (1, 10), and soil fertility reduction factor (0.2, 0.3). For any field with a state given by the soil fertility X and phase γ the optimal cultivation-fallow strategy is defined as a set of two soil fertility levels, X^* or X_*.

Together these variables determine when it is optimal to switch from cultivation to fallow (X_*) and when it is optimal to switch from fallow to cultivation (X^*). Table 3.2 displays the optimal cultivation-fallow strategy for each model parameterization.

For any given state (X, γ) the combined pair (X^*, X_*) defines the management strategy that maximizes the discounted value of the cultivation-fallow system over an infinite time horizon. For example, Table 3.2 shows that the optimal cultivation-fallow strategy when the payment to fallow is \$0 ha^{-1} $year^{-1}$ is to continue fallowing so long as the soil fertility level is less than 0.404.After this point is reached the optimal strategy is to switch from fallow to cultivation.

Table 3.2: Optimal Cultivation-Fallow Strategies from Dynamic Optimization Model

Parameter	Value	PES Price	Discount Factor	Fallow to Cultivation (X^*)	Cultivation to Fallow (X_*)
		\$0	0.8	0.404	0.011
		\$50	0.8	0.573	0.012
		\$100	0.8	0.594	0.110
		\$150	0.8	0.626	0.270
		\$100	0.7	0.522	0.078
		\$100	0.8	0.594	0.110
		\$100	0.9	0.616	0.215
Price of maize	\$33	\$100	0.8	0.798	0.225
Price of maize	\$66	\$100	0.8	0.594	0.142
Price of maize	\$100	\$100	0.8	0.451	0.110
Fertility reduction factor	0.1	\$100	0.8	0.594	0.110
Fertility reduction factor	0.2	\$100	0.8	0.591	0.099
Fertility reduction factor	0.3	\$100	0.8	0.592	0.099
Fallow efficiency factor	1	\$100	0.8	0.668	0.010
Fallow efficiency factor	4	\$100	0.8	0.594	0.110
Fallow efficiency factor	10	\$100	0.8	0.459	0.130

It is optimal to continue cultivating the field until the soil fertility level equals 0.011. The optimal management strategies can be used to see what effect payments have on soil fertility levels by simulating the strategies over a fixed time horizon for a given soil fertility and state of the field (i.e. fallow or cultivation).

3.5.2 PES Impact on Soil Fertility

All the results were estimated with a starting soil fertility index value of 0.20 and a starting state of being in fallow. The model was solved over an infinite time horizon and the results are presented for a fifty-year simulation. Table 3.3 shows the results from the simulations.

The results show that paying smallholders to fallow their fields improves soil fertility in every scenario. Under the baseline scenario with no payment for ecosystem services (i.e. no payment for fallowing) the average ending soil fertility level is 0.162. When the PES payment is set at \$50 $ha^{-1}year^{-1}$ the average ending soil fertility level increases to 0.212. At PES payments of \$100 ha^{-1} $year^{-1}$ and \$150 ha^{-1} $year^{-1}$ average ending soil fertility levels are 0.331 and 0.447, respectively. These results suggest that payments increase the opportunity cost of cultivation. Smallholders respond to the new incentives by cultivating less often and fallowing more often.

Table 3.3: Results from Dynamic Optimization Model Simulations

Parameter	Value	PES Price	Discount Factor	Average Ending Soil Fertility	Fallow Length	Cultivation Length	Ratio of Cultivation to Fallow	Cycle Length	NPV
		$0	0.8	0.162	2	35	17.5	37	$211
		$50	0.8	0.212	4	38	9.5	42	$271
		$100	0.8	0.331	4	16	4.0	20	$426
		$150	0.8	0.447	4	8	2.0	12	$592
		$100	0.7	0.281	3	18	6.0	21	$271
		$100	0.8	0.331	4	16	4.0	20	$426
		$100	0.9	0.417	4	10	2.5	14	$963
Price of Maize	$33	$100	0.8	0.535	11	12	1.1	23	$210
Price of Maize	$66	$100	0.8	0.331	4	16	4.0	20	$426
Price of Maize	$100	$100	0.8	0.284	2	11	5.5	13	$689
Fertility Reduction Factor	0.1	$100	0.8	0.331	4	16	4.0	20	$426
Fertility Reduction Factor	0.2	$100	0.8	0.326	4	8	2.0	12	$347
Fertility Reduction Factor	0.3	$100	0.8	0.350	4	5	1.3	9	$309
Fallow Efficiency Factor	1	$100	0.8	0.227	2	40	20.0	42	$685
Fallow Efficiency Factor	4	$100	0.8	0.331	4	16	4.0	20	$426
Fallow Efficiency Factor	10	$100	0.8	0.298	7	12	1.7	19	$274

The results show that smallholders maintain lower soil nutrient levels under higher discount rates. When the discount rate is 30% the average soil fertility level is 0.281. At a 20% discount rate the average soil fertility level is 0.331 and at 10% discount rate the average soil fertility level is 0.417. At a 30% discount rate the average fallow length is 3 years while the average cultivation length is 18 years. At 20% and 10% discount rates the average fallow length increases to 4 years while the average cultivation length decreases to 16 and 10 years, respectively. When the discount rate goes down the relative benefit of fallowing goes up because the smallholder places more value on future benefits when the discount rate is lower. As a result, smallholders fallow more often and cultivate less leading to higher levels of soil fertility. These results are consistent with the arguments of Balsdon (2008), Batabyal and Lee (2003), and Barrett (1991) who all hypothesized that high discount rates create an incentive to extend the length of the cultivation period to avoid the cost of shifting fields or phases.

Unlike the results of Willassen (2011) who showed that soil fertility is increasing with respect to crop price, the results in Table 3.3 show that when payments are made to smallholders to fallow their fields the relationship between soil fertility and crop prices is inversely related. When the payment for fallow is held constant at \$100 ha^{-1} year^{-1} the results show that the optimal average soil fertility is equal to 0.535 at a crop price of \$33 ton^{-1}. At crop prices of \$66 ton^{-1} and \$100 ton^{-1} the optimal average soil fertility declines to 0.331 and 0.284, respectively. Despite the lower soil fertility levels, the value of the system is increasing with respect to crop prices. The results suggest that to maintain soil nutrient levels over time payments for fallow will have to adjust to crop prices.

A sensitivity analysis was conducted to see how sensitive the model results are to different parameter values in the soil state equation. The results show that specific dimensions of the model are sensitive to changes in the value of the soil fertility reduction factor. As a crop becomes more demanding of soil nutrients the optimal response by smallholder farmers is to maintain soil nutrient levels by decreasing the length of the cultivation phase. Under a low soil fertility reduction factor of 0.1 the optimal average soil fertility level is equal to 0.331 and the optimal lengths of the fallow and cultivation phases are 4 and 16 years, respectively. A soil fertility reduction factor of 0.2 causes the smallholder to maintain soil fertility at 0.326 by keeping the optimal fallow length at 4 years and reducing the optimal cultivation length to 8 years. If the soil fertility reduction factor equals 0.3 the optimal response is to keep the optimal fallow length at 4 years and reduce the optimal cultivation length to 5 years. This maintains an optimal average soil fertility level of 0.35. As the soil fertility reduction factor increases the NPV of the system declines from $426 to $309 largely because more frequent switching between cultivation and fallow phases is necessary to maintain soil fertility under higher soil fertility reduction factors.

Fallow practices that restore soil nutrients faster than others can cause smallholders to change the optimal average soil fertility they maintain on their fields in addition to changing the length of the cultivation-fallow cycle. Under a fallow efficiency factor of 1, which reflects fallows are able to quickly restore soil nutrients, the smallholder will maintain an optimal average soil fertility level of 0.227. The soil fertility level is maintained with relatively short two-year fallows followed by 40 years of cultivation because the efficient fallows can quickly restore even the most depleted soils. When the fallow efficiency factor is increased to 4, reflecting fallows that take a relatively longer time to restore soil nutrients, smallholders respond by increasing the length of fallows to 4 years and reducing the length of cultivation to 16 years. This results in an optimal average soil fertility level of 0.331. If the fallow efficiency factor equals 10 the optimal average soil fertility level is 0.298, which is maintained by 7-year fallows followed by 12 years of cultivation. The NPV of the system is also sensitive to changes in the fallow efficiency factor. As the factor increases from 1 to 10 (i.e. fallows become less efficient) the NPV declines from high of $685 to a low of $274.

The results show that soil fertility is determined by the ratio of cultivation length to fallow length except in the cases where an underlying parameter in the soil fertility model changes. As the ratio decreases it implies that smallholders spend relatively more time fallowing their field and relatively less time cultivating it. The results from Table 3.3 show that the optimal cultivation to fallow ratio is declining with respect to the value of the payment made to smallholders for fallowing. Without any payment the optimal ratio is 17.5 years of cultivation for every year of fallow. As the payment is increased the optimal ratio moves from a value of 9.5 to a value of 4, to a value of 2 for payments of \$50 ha^{-1}year^{-1}, \$100 ha^{-1} year^{-1} and \$150 ha^{-1} year^{-1}, respectively. The results also show that when payments for fallow are in place the optimal cultivation to fallow ratio is increasing with respect to crop prices. At a crop price of \$33 ton^{-1} and a fallow price of \$100 ha^{-1} year^{-1} the optimal cultivation to fallow ratio is 1.09. If the price of crops increases to \$100 ton^{-1} the optimal ratio increases to 5.5. The optimal cultivation to fallow ratio is decreasing with respect to the discount rate. At a 30% discount rate the optimal cultivation to fallow ratio is equal to 6, but at a 10% discount rate the ratio declines to 2.5. These results are notable because they point to levers that policy makers can use to influence soil fertility on smallholder cultivation-fallow systems.

3.6 Conclusion

This chapter developed a dynamic programming model of the cultivation-fallow problem of a representative smallholder agriculturist. The model is parameterized for a one-hectare plot of maize agriculture using data from field trials of improved fallows in Zambia. The results from the dynamic programming model are used to simulate the optimal soil fertility paths for cultivation-fallow systems under different parameterizations in order to demonstrate how soil fertility levels would evolve under different levels.

The results suggest that paying smallholders to fallow their fields can increase soil fertility in cultivation-fallow systems. The results show that payments increase the opportunity cost of cultivating and lead to shorter cultivation periods and longer fallow periods, all else equal. This leads to higher soil fertility levels whenever payments are made. If a smallholder finds him/herself in a fallow phase with low soil fertility the optimal response in every situation is to continue fallowing until the rate of soil fertility restoration becomes so small that the foregone opportunity of using those soil nutrients to produce crops outweighs the benefits of continuing to fallow. During cultivation phases payments for fallow increase the opportunity cost of cultivation (the foregone benefits of fallowing), which creates an incentive for smallholders to switch from cultivating to fallowing at higher soil fertility levels. The same effect is observed by decreasing smallholder's discount rates over the future. However, in order for payments to continue to be effective they must adjust to changing crop prices and the evolution of new farm technologies. The results show that when payments for fallow are in place higher crop prices lead to less fallow and lower soil fertility levels and that the same effect is observed for more efficient fallows.

While payments for fallow can improve soil fertility levels on smallholder agricultural land this study did not explore what conditions are necessary for payments to be a viable policy response to land use challenges. Despite the increasing popularity of PES programs, several authors have noted their limitations, including the need for specific institutional structures and near-perfect information, difficulty measuring services, and high transaction costs (Gottfried et al., 1996; Kroeger and Casey, 2007). However, in the context of improved fallows PES schemes are likely to have some merit because the necessary condition for PES to be effective, as identified by Pagiola and Platais (2007), is that a fraction of the benefits of a particular land use must be viewed as externalities from the land manager's perspective. Engel, Pagiola, and Wunder (2008) further argue that if the number of impacted parties is small, property rights are assigned, and the service is well defined, then PES price mechanism can convey information about the value of an externality and lead to more efficient allocations of resources. They argue that this is especially true when the number of beneficiaries is particularly small or centralized, which reduces transaction costs to a modest fraction of total project value. This suggest that payments for fallow are most likely to succeed in areas where the public benefits of improved fallows are captured by a small number of people, such as downstream water users, public utilities, or nearby neighbors.

Future studies on this topic should attempt to expand the ability of the dynamic programming models to answer other policy relevant questions. For example, the models should be expanded to incorporate the effect of wealth and the ability to augment soil fertility with artificial fertilizer. These dimensions of the problem were beyond the scope of the present study, but in the future increased computing power and more efficient computational algorithms will allow these types of changes to be made without sacrificing the ability to solve the problem. Future research should also expand the model to study the effects of uncertainty since crop yields are dependent on rainfall, which is highly variable. Lastly, future research should explore the effect of different payment types since this is an area that has been largely neglected in the PES literature even though it could have large implications for the success of PES schemes.

CHAPTER 4: CONCLUDING REMARKS

This dissertation explored and contributed to the knowledge and discourse on the economics of restoration by looking at three unique questions being asked by decision makers at different levels of the debate on restoration. First, this dissertation addressed the question of whether or not the time horizons are too long, the costs are too high, and the benefits are too few to justify public expenditure on restoring degraded land and found that the answer depends on which benefits society values and how society chooses to discount the future and accordingly the welfare of future generations.

When the value of public ecosystem goods and services are accounted for and the welfare of future generations is discounted at the lowest defensible rate found in the economic literature the benefits of restoration far outweigh the costs. The results from this scenario suggest a strong case can be made for fulfilling global initiatives to restore large areas of degraded forest land. However, when the value of public ecosystem goods and services are unaccounted for and social discount rates are set based on optimistic projections about the wealth of future generations the case becomes weaker and there is smaller chance of fulfilling the goals of the global restoration initiatives. Choosing the right social discount rate to weight the costs and benefits of public investments in restoration remains more art than science, but at the very least decision makers should always include the value of environmental public goods and services in social cost-benefit analyses. To do otherwise is to discount the preferences of the global community who derive real value from environmental public goods and services.

The second chapter of this dissertation evaluated two proposed restoration activities in Rwanda. This chapter presented a methodology combining enterprise budgets, biological production functions, and Monte Carlo simulations in an expected utility framework to investigate the financial profitability, financial risk, and ecological impacts of agroforestry and improved woodlot management in a smallholder context in four provinces of Rwanda. The results showed that smallholders of all risk-preferences would be unlikely to adopt agroforestry. Internalizing the value of public ecosystem services did not change the result. The results showed that smallholders always prefer current woodlot management practices to best management practices and that risk preferences did not influence the ranking. Internalizing the value of public ecosystem services also had no effect on the preferences of smallholders between the two activities.

The results from the second chapter suggest that the risk-preferences of smallholders are important determinants of adoptability for some restoration activities and thus demonstrated the importance of using evaluation methodologies that can account for risk preferences. In time and data limited settings using enterprise budgets and Monte Carlo simulations provides a timely and cost-effective method to help organizations and decision makers evaluate activities along specific dimensions of risk prior to promoting them to smallholders. Additionally, accounting for both the financial and ecological impacts of activities can provide information to policy makers increase the rate of adoption by giving them information on how to increase the profitability of the activities. However, in the case of the two restoration activities considered in Rwanda including the values of ecosystem services did not change the preferences of smallholders. Policy makers can take steps to change the preferences of smallholders by increasing the profitability and lowering the risks associated with both activities. These steps could include subsidizing crop insurance premiums, providing agricultural extension services, and offering food and financial safety nets to smallholders who adopt the activities.

Future research on this topic should focus on empirically estimating the risk preferences of smallholders and incorporating that information into household production models that take a holistic view of the adoption decisions of households. The second chapter was not able to account for other factors like household wealth and labor availability that are also determinants of activity adoption. Collecting additional information on the household decision making process would provide policy makers with a better understanding of the factors that encourage or prevent the adoption of restoration activities and how these factors are influenced by risk preferences. This understanding would help inform the design of policies to encourage adoption.

The third chapter of this dissertation developed a methodology to estimate the impact that a Payment for Ecosystem Service (PES) mechanism would have on the dynamics of soil fertility for a small cultivation-fallow system using a case study from Zambia. The results showed that payments for fallowing would lead to higher soil fertility levels in all cases. While payments for fallow can improve soil fertility levels on smallholder agricultural land this chapter did not explore what conditions are necessary for payments to be a viable policy response to land use challenges. Future research should focus on identifying areas where PES are most likely to succeed. Future studies on this topic should also attempt to expand the ability of the dynamic programming models to answer other policy relevant questions related to the effect that increasing wealth would have on household production functions. Another interesting dimension of the problem to explore is how smallholders would respond to payments under uncertainty because as chapter 2 showed the effects of uncertainty depends on the risk preferences of smallholders.

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CHAPTER 1 APPENDIX

Table A1.1: World regions used in analysis from Chiabai et al. (2011)

World Region	Description	Countries
AFR	Africa	Angola, Botswana, Comoros, Kenya, Lesotho, Madagascar, Malawi, Mauritius, Mayotte, Mali, Mauritania, Morocco, Mozambique, Namibia, Niger, Réunion, Seychelles, South Africa, Swazilan, Uganda, Tanzania, Zambia, Zimbabwe, Algeria, Burkina Faso, Burundi, Benin, Chad, Djibouti, Egypt, Eritrea, Ethiopia, Libya, Somalia, Sudan, Tunisia, Western Sahara, Cameroon, Cape Verde, Central African Republic, Congo, Democratic Republic of Congo, Republic of Côte d'Ivoire, Equatorial Guinea, Gabon, The Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Nigeria, Rwanda, St. Helena, São Tomé and Príncipe, Senegal, Sierra Leone, Togo
ANZ	Pacific	American Samoa, Australia, Cook Islands, Fiji, French Polynesia, Guam, Kiribati, Marshall Islands, Micronesia (Federated States of), Nauru, New Caledonia, New Zealand, Niue, Northern Mariana Islands, Palau, Papua New Guinea, Pitcairn, Samoa, Solomon Island, Tokelau, Tonga, Tuvalu, Vanuatu, Wallis and Futuna Islands
BRA	Brazil	Brazil
CHN	China Region	China, Hong Kong SAR, Taiwan Province of China
ECA	Eastern Europe and Central Asia	Belarus, Moldova, Occupied Palestinian Territory, Tajikistan, Turkmenistan, Ukraine, Uzbekistan, Kazakhstan, Kyrgyz Republic
EUR	Europe	Albania, Andorra, Austria, Belgium, Bosnia and Herzegovina, Bulgaria, Channel Islands, Croatia, Cyprus, Czech Republic, Denmark, Estonia, Faeroe Islands, Finland, France, Germany, Gibraltar, Greece, Holy See, Hungary, Iceland, Ireland, Isle of Man, Italy, Latvia, Liechtenstein, Lithuania, Luxembourg, Macedonia, Republic of Former Yugoslav, Malta, Monaco, Netherlands, Norway, Poland, Portugal, Romania, San Marino, Serbia, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, United Kingdom
JPK	Asia	Japan, Korea, Democratic People's Republic of Korea
NAM	North America	Canada, Mexico, United States
OAS	Other Asia	Mongolia, Brunei Darussalam, Cambodia, Indonesia, Lao People's Democratic Republic, Malaysia, Myanmar, Philippines, Singapore, Thailand, Dem. Republic of Timor-Leste, Vietnam

OLC	Other Latin American and Caribbean	Anguilla, Antigua and Barbuda, Aruba, Bahamas, Barbados, Bermuda, British Virgin Islands, Cayman Islands, Cuba, Dominica, Dominican Republic, Grenada, Guadeloupe, Guyana, Haiti, Jamaica, Martinique, Montserrat, Netherlands Antilles, Puerto Rico, South Georgia and the South Sandwich Islands, St. Kitts and Nevis, St. Lucia, St. Vincent and the Grenadines, Turks and Caicos Islands, United States Virgin Islands, Argentina, Belize, Bolivia, Costa Rica, Chile, Colombia, Ecuador, El Salvador, Falkland Islands, French Guiana, Guatemala, Honduras, Nicaragua, Panama, Paraguay, Peru, Suriname, St. Pierre and Miquelon, Trinidad and Tobago, Uruguay, Venezuela
RUS	Russia and Caucuses	Armenia, Azerbaijan, Georgia, Russia
SOA	South Asia and India	Rep. of. Afghanistan, Bangladesh, Bhutan, India, Maldives, Nepal, Pakistan, Sri Lanka

CHAPTER 2 APPENDIX

A2.1 Net returns

The results from the Monte Carlo analysis of the partial enterprise budgets are shown in Tables 2.8 and 2.9. The negative financial returns of all of the enterprises considered in this study reflect the low productivity of land-based enterprises in East Africa when the opportunity cost of labor is accounted for. The financial results show that both of the activities considered in this study do not positively increase the profitability of land compared to current land uses.

Maize agriculture in Rwanda has an average NPV of between -139,989 RwF and -382,024 RwF across the four provinces considered in this study. The Standard Deviation of the financial returns of maize agriculture varied from 137,982 RwF to 192,534 RwF. In contrast, the average NPV for agroforestry systems with maize ranged between -102,649 RwF to -460,640 RwF, while the Standard Deviation of the NPV of maize agroforestry ranged between 194,089 RwF and 278,958 RwF.

The results from the Monte Carlo analysis show that the average NPV of maize agriculture and agroforestry systems with maize are not significantly different across provinces as shown by the p-values in Table A2.1. However, the financial uncertainty associated with agroforestry systems with maize are larger for every province considered in the study. In the study, maize agriculture is exposed to risk from variability in precipitation and market prices for maize. Agroforestry systems are also exposed to these sources of risk. However, they are also exposed to additional risk through the variability of timber growth rates, fuelwood prices, and the effect agroforestry trees have on crop yields. In combination these additional sources of uncertainty increase the variability of financial returns from agroforestry as compared to agriculture with maize.

Table A2.1: Financial results from Monte Carlo Analysis of Agroforestry and Agriculture Enterprise Budgets							
Province	Land Use	NPV	Standard Deviation	Maximum	Minimum	Change in NPV	P-Value
Kigali	Agriculture	-382,024	137,982	87,663	-826,778	-78,616	<0.999
	Agroforestry	-460,640	194,089	101,271	-1,009,764		
Northern	Agriculture	-139,989	192,534	810,902	-677,567	28,072	<0.999
	Agroforestry	-111,917	278,958	806,205	-743,461		
Southern	Agriculture	-140,573	172,310	548,523	-585,357	37,924	<0.999
	Agroforestry	-102,649	240,694	715,864	-805,480		
Western	Agriculture	-204,602	157,336	378,317	-637,874	-7,578	<0.999
	Agroforestry	-212,180	232,962	762,202	-880,083		

The results from the Monte Carlo analysis show that the average NPV of woodlots managed with best practices and poorly managed woodlots are significantly different and that best management practices, such as establishing anti-erosion trenches, fire lanes, and removing old stumps do not increase the financial performance of woodlots under the current set of market incentives. Poorly managed woodlots in Rwanda have an average NPV of between -359,120 RwF and -360,383 RwF across the four provinces considered in this study. The Standard Deviation of the financial returns of poorly managed woodlots varied from 14,904 RwF to 16,974 RwF. In contrast, the average NPV for woodlots managed with best practices ranged between -1,204,587 RwF to -1,207,618 RwF while the Standard Deviation ranged between -21,748 RwF and 23,616 RwF.

Table A2.2: Financial results from Monte Carlo Analysis of Poorly Managed and Well Managed Woodlot Enterprise Budgets

Province	Land Use	NPV	Standard Deviation	Maximum	Minimum	Change	P-Value
Kigali	Poorly Managed Woodlot	-360,383	14,904	-323,369	-396,418	-847,236	<0.0001
	Well Managed Woodlot	-1,207,618	21,748	-1,153,470	-1,264,375		
Northern	Poorly Managed Woodlot	-359,120	15,534	-319,451	-399,305	-845,467	<0.0001
	Well Managed Woodlot	-1,204,587	22,846	-1,147,441	-1,267,133		
Southern	Poorly Managed Woodlot	-359,804	16,074	-320,652	-400,160	-846,365	<0.0001
	Well Managed Woodlot	-1,206,169	23,616	-1,149,227	-1,270,233		
Western	Poorly Managed Woodlot	-359,320	15,315	-319,124	-399,338	-846,261	<0.0001
	Well Managed Woodlot	-1,205,581	22,725	-1,144,720	-1,261,601		

The financial uncertainty associated with woodlots managed with best practices is greater than for poorly managed woodlots for every province considered in the study. In the study both types of woodlots are exposed to two sources of risk. Woodlot managers face risk from the variability in tree growth rates and also from the variability in the market prices for fuelwood. Woodlots managed with best management practices have higher stocking densities compared to poorly managed woodlots. As a result the variability in both tree growth rates and market prices is amplified.

A.2 Ecosystem Service Results

The results from the Monte Carlo analysis of the ecosystem service models are shown in Tables A2.3 & A2.4. The ecosystem service results for agriculture and agroforestry with maize show that agroforestry produces improved environmental outcomes compared to agriculture. Despite the variable effect of agroforestry on crop yields, on average agroforestry yields 13 – 15 short tons per hectare more than maize agriculture over a thirty year time horizon and the effect is statistically significant at the 99% level. Additionally, agroforestry enterprises store between 25 – 27 short tons more CO_2e than agriculture enterprises and that effect is also significant at the 99% level. Agroforestry enterprises reduce erosion by 153 to 167 short tons per hectare compared to agriculture and the effect is statistically significant at the 99% level.

Table A2.3: Ecosystem Service results from Monte Carlo Analysis of Agroforestry and Agriculture

Province	Land Use	Crop Yields T Ha^{-1}	Change	P-Value	Carbon T CO_2e Ha^{-1}	Change	P-Value	Erosion T Ha^{-1}	Change	P-Value
Kigali	Agriculture	19	13	<0.0001		26	<0.0001	481	-153	<0.0001
	Agroforestry	31			26			328		
Northern	Agriculture	21	15	<0.0001		26	<0.0001	526	-167	<0.0001
	Agroforestry	36			26			359		
Southern	Agriculture	21	15	<0.0001		27	<0.0001	520	-165	<0.0001
	Agroforestry	36			27			354		
Western	Agriculture	21	14	<0.0001		25	<0.0001	513	-163	<0.0001
	Agroforestry	35			25			350		

The ecosystem service results for woodlot enterprises show that woodlots managed with best practices produce improved environmental outcomes compared to poorly managed woodlots. Woodlots managed with best practices store between 137 and 140 short tons more CO_2e than poorly managed woodlots over a thirty year period. The effect is statistically significant at the 99% level. Additionally, woodlots managed with best practices reduce erosion by 137 to 149 short tons per hectare over a thirty-year period and the effect is statistically significant at the 99% level for every province considered in the study. The increased stocking density of woodlots managed with best practices increases timber production by 96 – 98 cubic meters over a thirty year period compared to the poorly managed woodlots.

Table A2.4: Ecosystem Service results from Monte Carlo Analysis of Poorly Managed and Well Managed Woodlots

Province	Land Use	Carbon T CO_2e Ha^{-1}	Change	P-Value	Erosion T Ha^{-1}	Change	P-Value	Timber M^3 Ha^{-1}	Change	P-Value
Kigali	Poorly Managed Woodlots	305	137	<0.0001	410	-137	<0.0001	211	96	<0.0001
	Well Managed Woodlots	442			273			307		
Northern	Poorly Managed Woodlots	312	140	<0.0001	448	-149	<0.0001	216	98	<0.0001
	Well Managed Woodlots	453			299			315		
Southern	Poorly Managed Woodlots	308	139	<0.0001	443	-148	<0.0001	213	97	<0.0001
	Well Managed Woodlots	447			295			310		
Western	Poorly Managed Woodlots	307	138	<0.0001	436	-145	<0.0001	213	97	<0.0001
	Well Managed Woodlots	445			291			310		

A2.2 Stochastic Dominance Analysis with Value of Ecosystem Services

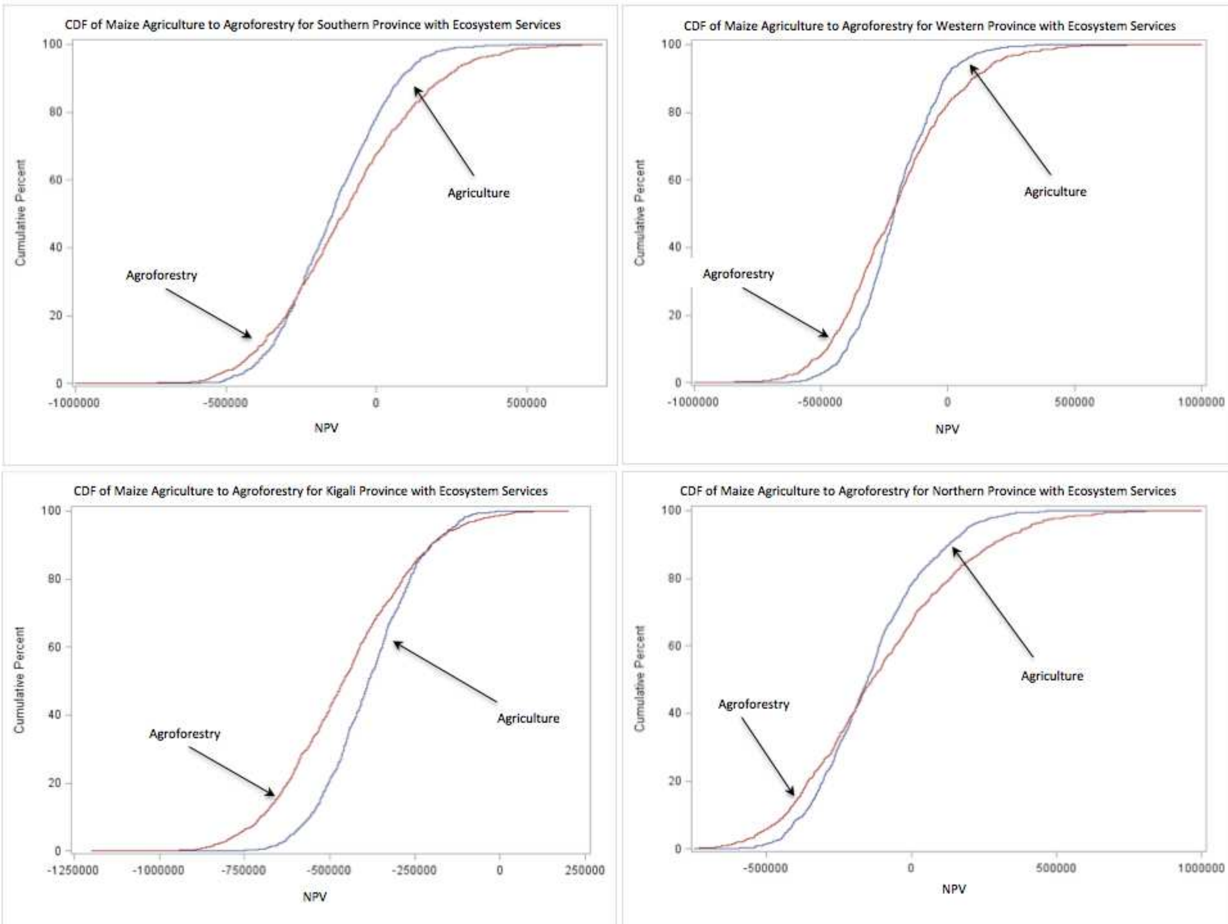

Figure A2.1: The CDFs of NPV for degraded maize agriculture to agroforestry with maize with ecosystem service values included for four provinces in Rwanda. A non-overlapping CDF to the right of an alternative CDF is said to display First Order Stochastic Dominance over the alternative. Smallholders who prefer more to less will always choose an activity with a CDF that strictly dominates another CDF. In the example above none of the CDFs dominate the others as shown by the crossing graphs of the CDFs in every province.

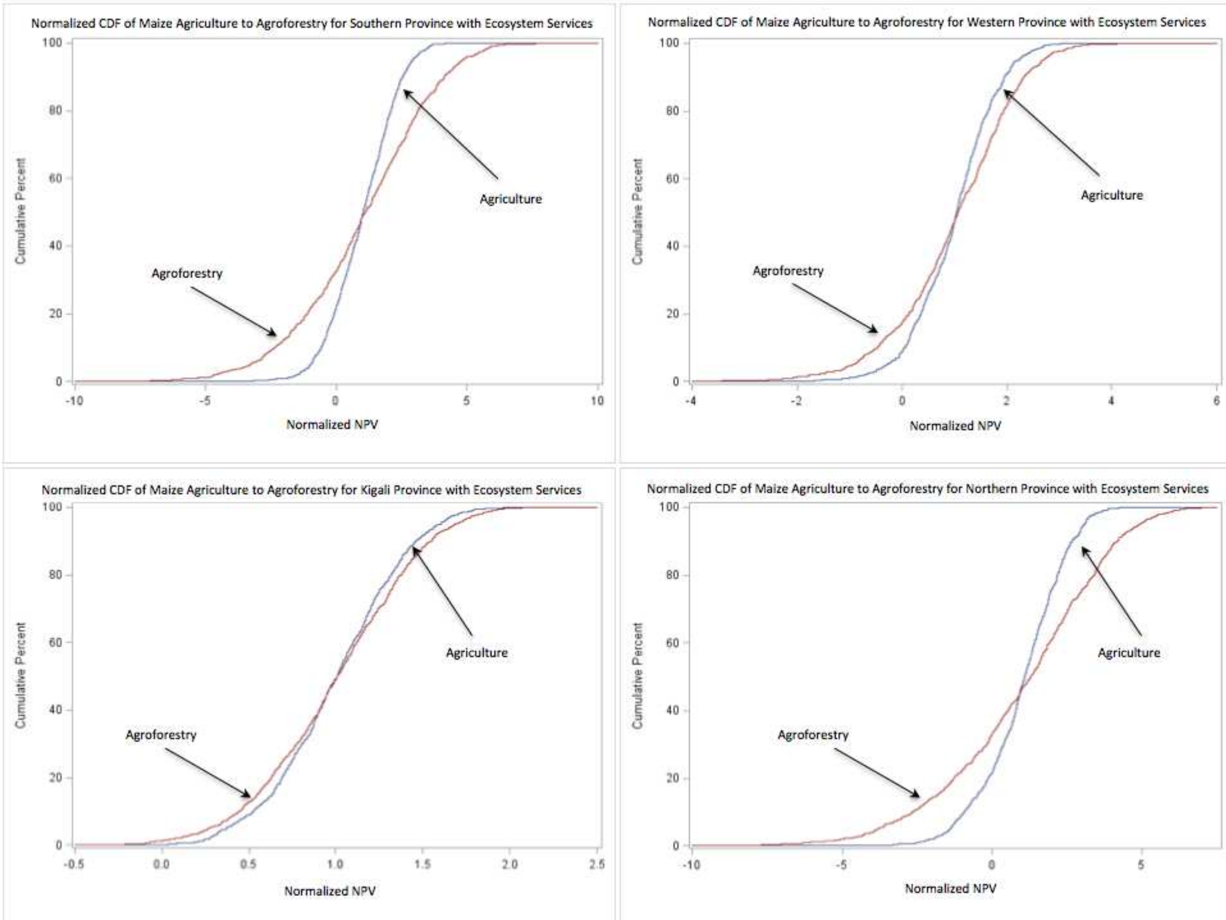

Figure A2.2: The normalized CDFs of NPV for degraded maize agriculture to agroforestry with maize with ecosystem service values included for four provinces in Rwanda. A non-overlapping normalized CDF to the right of an alternative normalized CDF is said to display Second Order Stochastic Dominance over the alternative. Smallholders who are risk averse will always choose an activity whose normalized CDF displays Second Order Stochastic Dominance. In the example above none of the normalized CDFs dominate the others as shown by the crossing graphs of the normalized CDFs in every province.

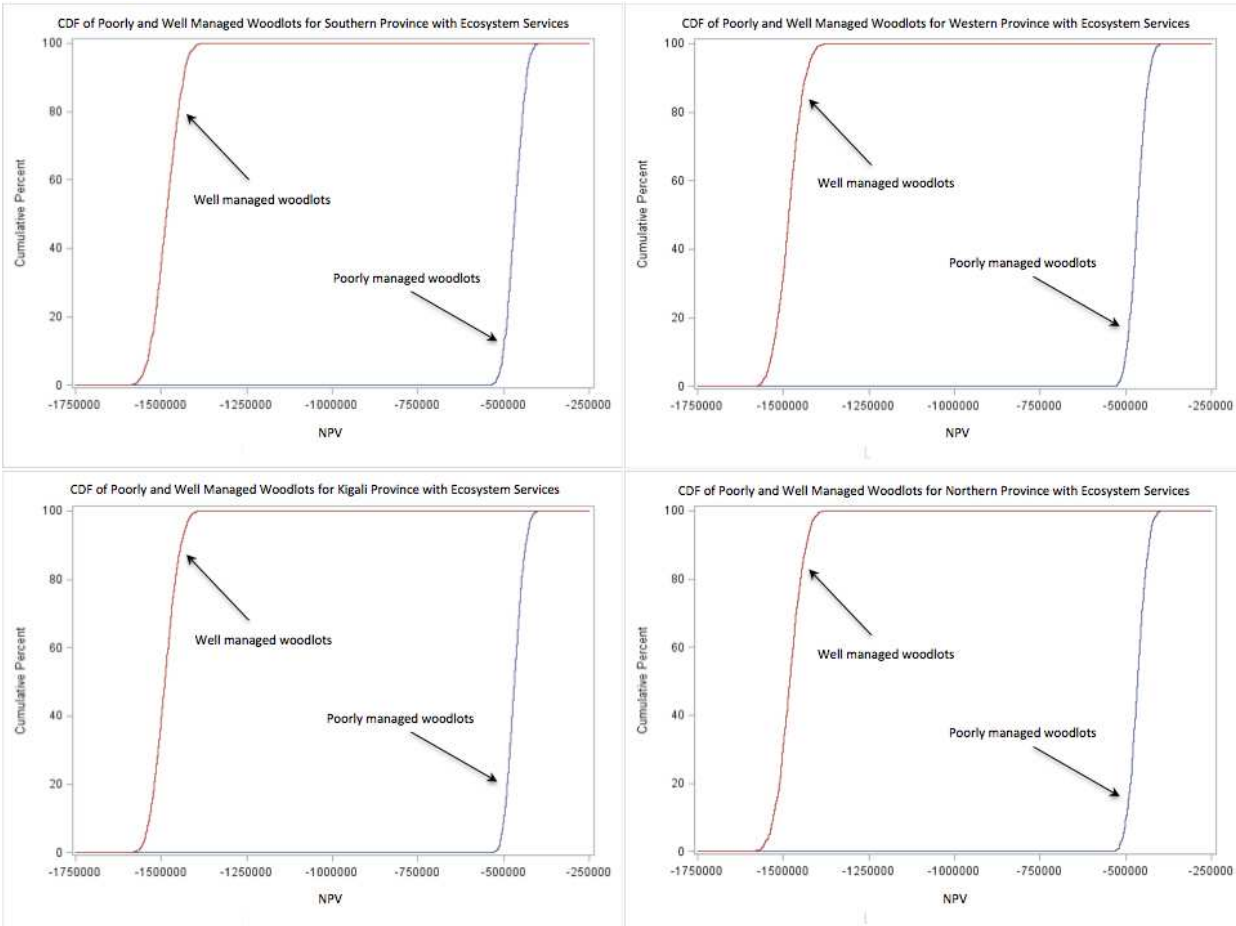

Figure A2.3: The CDFs of NPV for poorly managed woodlots and well managed woodlots with ecosystem service values included for four provinces in Rwanda. A non-overlapping CDF to the right of an alternative CDF is said to display First Order Stochastic Dominance over the alternative. Smallholders who prefer more to less will always choose an activity with a CDF that strictly dominates another CDF. In the example above poorly managed woodlots display First Order Stochastic Dominance over well managed woodlots in each province.

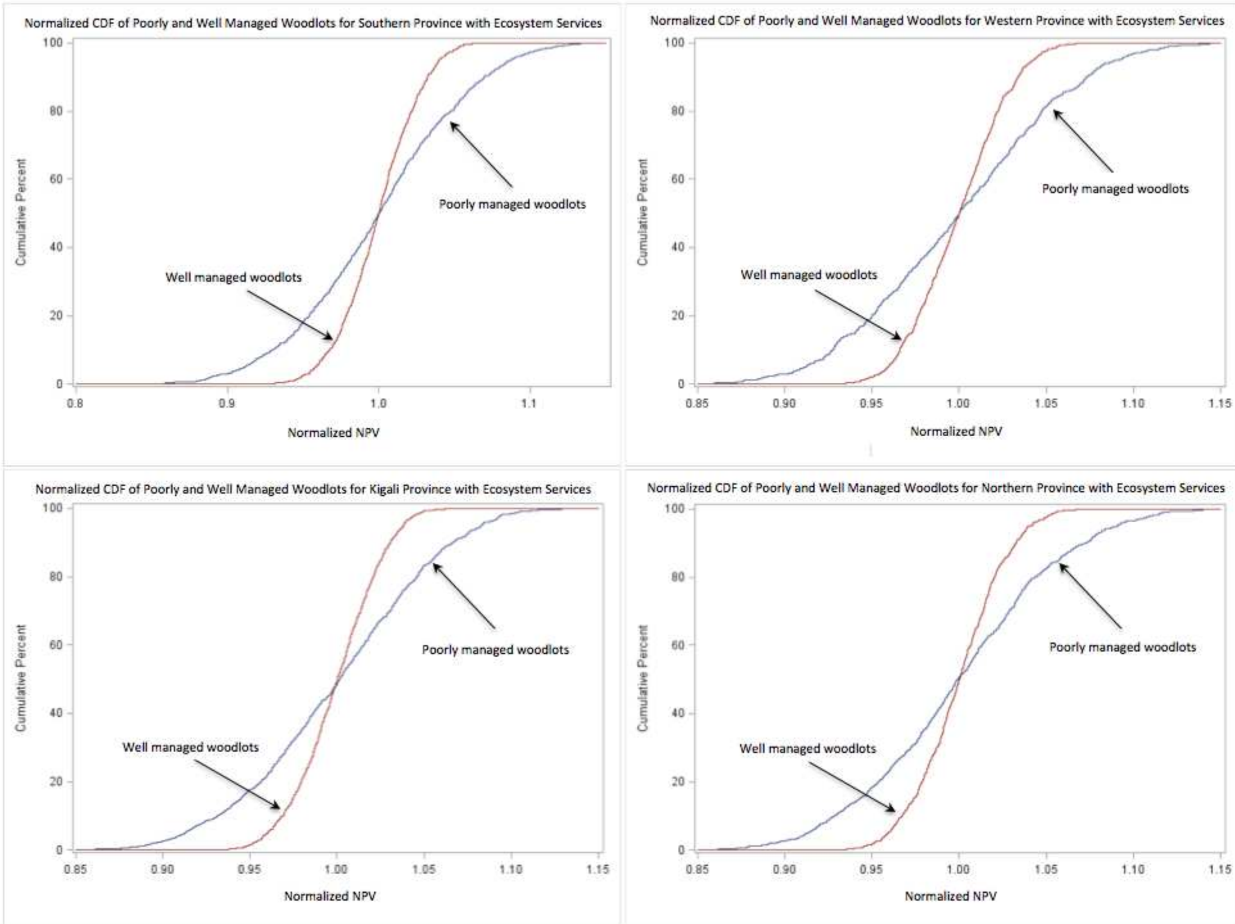

Figure A2.4: The normalized CDFs of NPV for poorly managed woodlots and well managed woodlots with ecosystem service values included for four provinces in Rwanda. A non-overlapping normalized CDF to the right of an alternative normalized CDF is said to display Second Order Stochastic Dominance over the alternative. Smallholders who are risk averse will always choose an activity whose normalized CDF displays Second Order Stochastic Dominance. In the example above none of the normalized CDFs dominate the others as shown by the crossing graphs of the normalized CDFs in every province.