A Comparison of Land Use Options for the Mbaracayu Biosphere Reserve

FINAL REPORT

March 2007

Prepared for
Alberta Research Council

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Introduction

Unplanned and unsustainable land use has transformed the Atlantic Forests of Paraguay, Brazil and Argentina. Atlantic Forests have been identified as a Biodiversity Hotspot in recognition of the region as a threatened area of species endemism (Myers et al. 2000). As of 2003, only 7.4 percent of the region's forest cover remained intact (Di Bitetti et al. 2003). In Paraguay, almost 90% of the Atlantic Forest has been lost (Weber and Cardozo 2003), largely as a result of slash-and-burn agriculture that has expanded over the past few decades (Di Bitetti et al. 2003).

In 1991, responding to the rapid loss of Atlantic Forest, the Government of Paraguay created the Mbaracayu Forest Natural Reserve (MFNR) and established the watershed surrounding the watershed (Cuenca Alta del Rio Jejui, hereafter referred to as the Cuenca) as a mixed-used protected area. The MFNR is the largest protected area in Paraguay containing Atlantic Forest and one of the few areas supporting macro fauna that are characteristic of the ecosystem. Over 85 species of mammals, 520 species of birds, and 1000 species of arthropods have been identified, and many more likely remain undetected (Weber and Cardozo 2003). The Cuenca provides some of the highest opportunity in Paraguay to conserve Atlantic Forest (Di Bitetti et al. 2003). In 2002, the Cuenca was declared a Man and the Biosphere World Heritage Site in recognition of the potential to achieve a balanced relationship between people and nature in the region.

While the Cuenca provides a perhaps unparalleled opportunity to conserve Atlantic Forest in Paraguay, substantial challenges must be overcome for the opportunity to be realized. The area converted by agriculture in the Cuenca increased 10-fold between 1973 and 2003, and forest outside of the MFNR and indigenous reserves is fragmented (Figure 1). Cattle ranches and soy plantations cover approximately 17 and 3% of the Cuenca, respectively, while small (i.e. <20 ha) farms owned by campesinos cover approximately 15%. Continued expansion of large-scale agriculture can be expected due to an expanding international market for soy (Dros 2004), while continued expansion of small-scale agriculture is likely to occur due to the region's high poverty and growing population. The Canindeyu department within which the Cuenca is located is recognized by the government as being a department where poverty threatens the preservation of natural resources (Weber and Cardozo 2003).

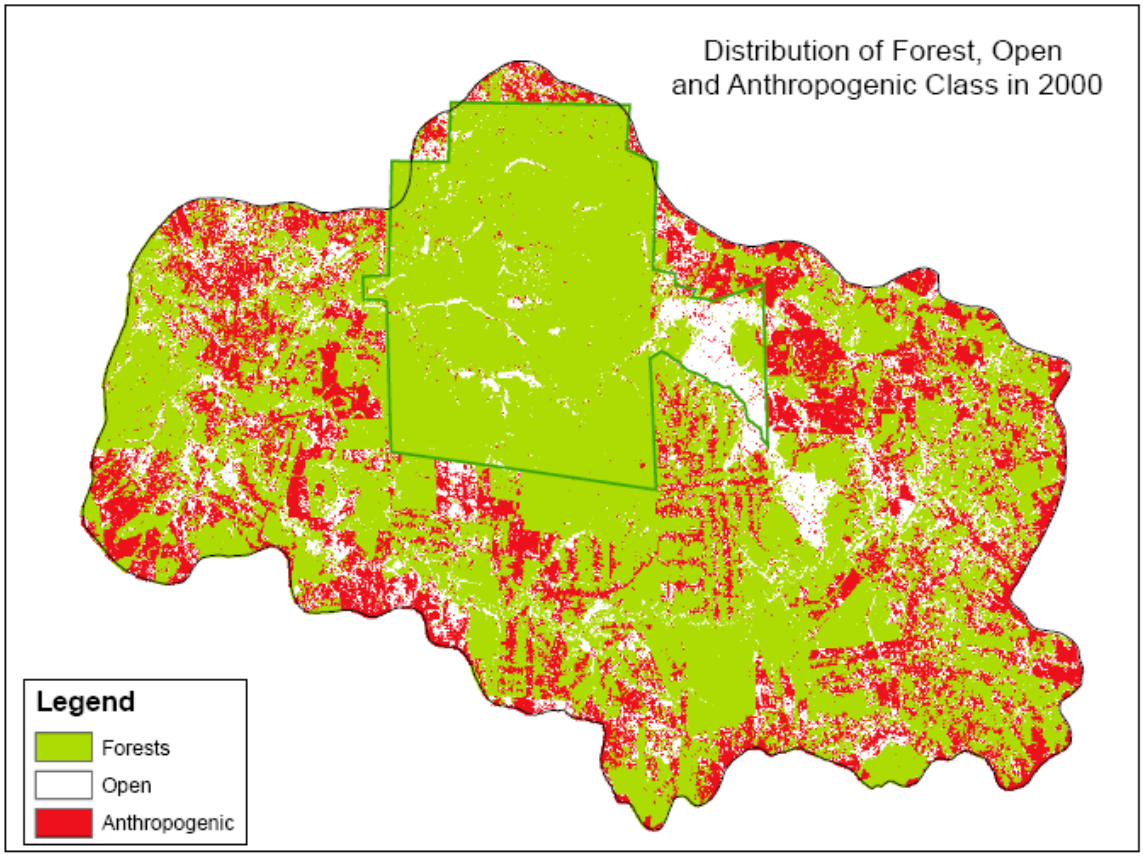

Figure 1: Distribution of natural (forest and open) and anthropogenic landscapes in the Cuenca.

Given the land use pressures facing the region, the future existence of healthy ecosystems within the Cuenca relies on balancing land use with conservation. The Mbaracayu program, run by the Fundacion Moises Bertoni (FMB), seeks to integrate a vision of sustainable and social development in harmony with the conservation of the MFNR (http://www.mbertoni.org.py/ingles/mbarai.htm#objetivo). Objectives of the program include: maintaining the rich biodiversity of MFNR flora and fauna; and, the sustainable use of the natural resources of the basin surrounding the MFNR to improve the well being of local inhabitants. Achievement of these objectives is currently hampered by the lack of a management plan for the Cuenca. A management plan is essential in order to develop a collectively-held vision for the future of the watershed and identify land use and conservation strategies that are consistent with that vision. With the establishment of a management plan, existing development efforts could be better coordinated and new activities could be designed to address socioeconomic and ecological concerns.

In response to the recognized need for a management plan, the FMB is collaborating with the Alberta Research Council on the project "Capacity Enhancement for Community- and Ecologically-based Management in the Bosque Mbaracayu Biosphere Reserve, Paraguay". The project, a joint initiative of the FMB and the Alberta Research Council, has as its purpose to build the capacity of indigenous groups, campesinos, landownsers, and women's groups to identify and implement economically viable and environmentally

sustainable development options for the region. While successful implementation of the project depends in large part upon fostering improved participation by members of communities in planning, also required is capacity to evaluate the current status of the region and explore the consequences of land use options. As such, the project seeks to transfer knowledge, expertise and tools for participatory natural resource management planning.

As part of the project, the land use simulation tool ALCES is being applied to evaluate land use scenarios in the Cuenca. ALCES provides a flexible simulation environment from which the user can explore the future effects of a range of land uses and natural processes to a variety of ecological and socioeconomic indicators. Applying ALCES will contribute to the development of a management plan by informing the identification of sustainable land use options. In March 2006 a series of workshops presented information about ALCES to local communities, government agencies, the FMB, and other nongovernment organizations. As a follow-up to the workshops, this report presents results from an analysis of a small but diverse set of land use scenarios. The report is intended to communicate the ALCES tool and analysis, solicit feedback, and inform training of FMB staff to apply ALCES in the Cuenca.

Methods

Application of ALCES to explore the future effects of land use scenarios required the following steps: 1) adapting ALCES to build capacity to simulate tropical agriculture systems; 2) defining the current composition of the Cuenca; 3) identifying indicators and defining indicator relationships; and 4) defining and simulating management scenarios.

ALCES Description

ALCES is one of the few tools capable of examining inter-relationships of different land-use sectors and exploring their environmental and socioeconomic consequences (Hudson 2002). Due to the tool's capacity to rapidly simulate the long-term effects of a wide variety of land uses and natural disturbances, ALCES has been extensively used to inform land use planning in western North America. By tracking disturbance and resource production associated with land uses, ALCES can explore effects to a variety of indicators including landscape composition, wildlife habitat, socioeconomic indicators, and ecosystem services such as carbon storage and water supply.

The primary land use affecting the Cuenca is agriculture. In addition, forestry could occur given the presence of merchantable timber in the region. Although ALCES is designed to simulate agriculture and forestry, the tool has not previously been applied in a tropical study area. Modification of ALCES was therefore required to add capacity to simulate tropical agriculture. In particular, capacity was added to explore the implications of declining crop productivity over time, including the need to convert natural land to cropland as existing cropland becomes uneconomical to farm.

Current Composition of the Cuenca

The current composition of the Cuenca was estimated using a 2004 land cover data set (Figure 2). The data set interpreted Landsat data using a vegetation classification scheme

based on Ache traditional ecological knowledge (Naidoo and Hill). The land cover data set divides the Cuenca into five forest types, five non-forest types, and three agriculture types (Table 1). Included in the data set was 15000 ha classified as an unspecified agricultural type. The unspecified agriculture area was divided amongst the three agriculture types in proportion to the abundance of each type.

A coverage indicating the location of indigenous reserves[1], the Mbaracayu reserve, and private reserves[2] in the Cuenca was obtained from FMB and used to calculate the proportion of each cover type that is protected from land use (Table 1). Overall, 28% of the Cuenca is protected, including the 64 268 ha MBFR, 14 067 ha in indigenous reserves, and 4 453 ha in private reserves. The private reserves were assumed to be protected from agriculture, but available for forestry (FMB, pers. comm.). All other reserves were assumed to be protected from agriculture and forestry.

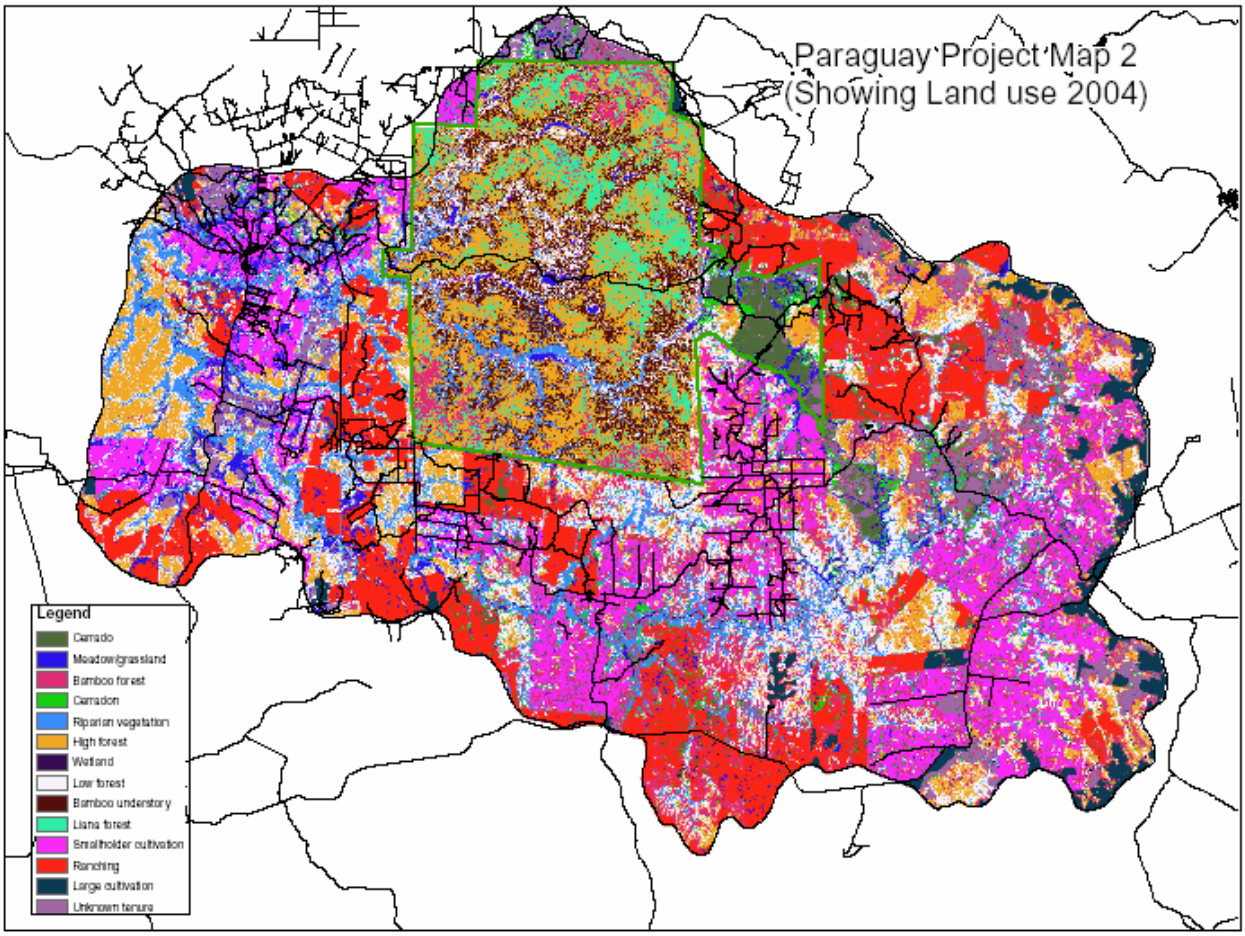

Figure 2: The composition of the Cuenca based on 2004 Landsat data.

[1] Indigenous reserves included in the analysis were Mboi Jagua, Ao Bandera, Comunidades indigenas, Kuetuvy, and Chupa Pou.
[2] The private reserves included in the analysis were Don Marcelo-1, Don Marcelo-2, Don Marcelo-3, Felicidad, Rama-31, and Rama-32.

Cover type	Category	Area (ha)	Percent protected
Bamboo forest	Forest	32303	20%
High forest	Forest	44541	61%
Low forest	Forest	43943	28%
Liana forest	Forest	9067	100%
Bamboo understory forest	Forest	13475	100%
Cerrado	Non-forest	8003	33%
Meadow	Non-forest	10225	22%
Cerradon	Non-forest	7772	17%
Riparian vegetation	Non-forest	21290	37%
Wetland	Non-forest	1496	13%
Smallholder cultivation	Agriculture	42709	0
Ranching	Agriculture	49037	0
Large cultivation	Agriculture	8102	0
Total		291962	28%

Table 1: The composition of the Cuenca based on 2004 Landsat data.

Indicators

The list of indicators evaluated in the scenario analysis should be the simplest set capable of conveying the fundamental tradeoffs associated with land use options. As such, the indicator set should include variables that are meaningful to the local community and communicate both the ecological and socioeconomic implications of land use. The November 2005 FMB workshop "Desarrollando Indicadores de Sotenibilidad Conjuntamente" identified priority indicators. It was not possible to simulate the full suite of indicators identified at the workshop, however, due to lack of information for many of the indicators. The following list of indicators was decided upon for the ALCES simulations based on consideration of workshop outcomes, availability of data, and capacity of ALCES: income generated by natural resource production, agriculture inputs (pesticides and fertilizer use), landcover and wildlife habitat (Peccary and Tapir). In addition, forest carbon storage was included as an indicator to explore the consequences of land use to this important ecosystem service. Forests store carbon in the form of biomass, and removal of forest vegetation results in the release of carbon dioxide to the atmosphere, which subsequently contributes to climate change. The removal of vegetation through land use is the second largest source of carbon emissions after the burning of fuel (Intergovernmental Panel on Climate Change 2000). Due to its role in climate change, carbon storage is potentially of economic value.

Scenarios

The analysis compared the consequences of seven scenarios (Table 2) that differed with respect to agriculture and forestry land use practices and the level of protection.

Three types of farming were simulated: smallholder farms, soy plantations, and cattle ranches. Simulations explored two general types of agriculture: conventional and conservation. Conservation agriculture refers to a suite of practices intended to increase the sustainability of agricultural production including zero tillage, direct seeding, crop rotations, green manure cover crops, and crop residue management (Lange 2005). In addition, under the conservation agriculture scenario all soy cropland expansion occurred into ranchland rather than forest in order to reduce forest loss.

Agricultural activity in the Cuenca has increased over the past 30 years (Figure 3), and land use simulations assumed that agriculture production will continue to expand. Agricultural expansion is constrained by the availability of unprotected land. All simulations assumed that the MFNR and existing indigenous and private reserves are unavailable for conversion to land use. In addition, Paraguay's Forest Law of 1973 stipulates that 25% of any forested property must be maintained as a forest reserve, although the law is rarely practiced (Glastra 1999). A scenario explored the effect of implementing the 25% forest protection law. The scenario assumed that the new protected areas are distributed across natural cover types in proportion to the relative abundance of the cover types in the currently unprotected landscape.

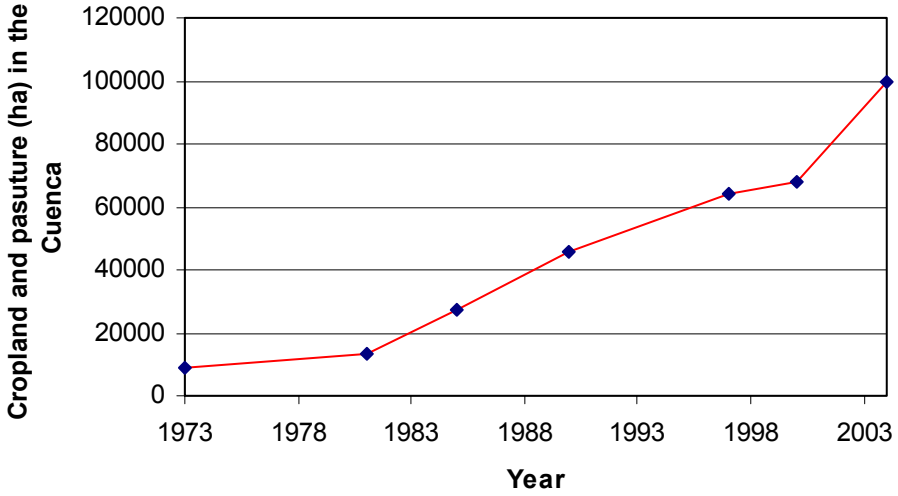

Figure 3: Trend in the area of cropland and pasture within the Cuenca.

Currently, forestry activity within the Cuenca is limited to salvaging some of the timber that is cut when forest is cleared for agriculture and unorganized timber extraction that degrades forest in affected areas (Laura Rodriguez, pers. comm.). A scenario explored the implications of implementing sustained yield timber harvest within the Cuenca. The scenario assumed that all non-protected merchantable forest is dedicated to sustained-yield timber production and is therefore not available for conversion to agriculture. As described in the assumptions section, two types of forestry were simulated: conventional logging and reduced impact logging. Reduced impact logging refers to practices designed to reduce incidental forest disturbance caused by forestry operations. An

additional scenario explored the implications of converting 500 ha of ranchland each year to eucalyptus plantations for timber production.

Scenario Name	Protection Strategy	Land Use Practices	Forestry
Conventional	Existing reserves	Conventional agriculture	Salvage only[1]
Protection law	Existing reserves & 25% of forested property	Conventional agriculture	Salvage only
Conservation	Existing reserves	Conservation agriculture[2]	Salvage only
Conservation and protection law	Existing reserves & 25% of forested property	Conservation agriculture	Salvage only
Natural forestry	Existing reserves	Conventional agriculture and conventional logging	Sustained yield
Conservation and natural forestry	Existing reserves	Conservation agriculture and reduced impact logging	Sustained yield
Eucalyptus forestry	Existing reserves	Conventional agriculture and conventional logging	Expanding eucalyptus plantations

Table 2: Land use scenarios simulated in the study.
[1]The salvage only forestry scenario may underestimate the current rate of timber extraction and forest disturbance in the region because unmanaged timber harvesting from forest is not included. Unmanaged timber harvest from forest was not included because the rate of this activity is not known.
[2]The conservation agriculture scenario was simulated with and without conversion of ranchland to soy production. This was done to isolate the effect of ranchland conversion.

Assumptions

The assumptions used in the simulations were, as much as possible, based on empirical findings from the region. Four categories of assumptions are now summarized: agriculture, forestry, forest carbon, and wildlife.

Agriculture Assumptions

Three types of farms were included in land use simulations: smallholder farms, which are small (under 20 ha) farms that produce a variety of crops; soy plantations, which are large farms used for the industrial production of soy; and ranches, which are large farms used to raise livestock. A number of assumptions were required to simulate ecosystem disturbance and agriculture production associated with each type of farm. These include future trends in production, net-income generation, pesticide and fertilizer use, current use of conservation practices, and the location of future agriculture expansion.

Agiculture Production Trend

Growth in smallholder farm production was assumed to be driven by human population growth in the region. Accordingly, the annual growth rate in smallholder farm

production was assumed to equal 1.5%, which was the population growth rate between 1990 and 2000 in Ygatimi, the most populous district within the Cuenca (UNDP 2003).

Growth in industrial soy plantations was assumed to be driven by international demand for soy. Based on recent growth and projections by government and industry, soy production in Paraguay is predicted to grow each year by 6.1% of the current level of production (Dros 2004). It is assumed that soy production in the Cuenca will grow at this rate.

Future ranchland growth is assumed to be zero based on a lack of directional trend in cattle population in the Canindeyu region over the past decade (Figure 3).

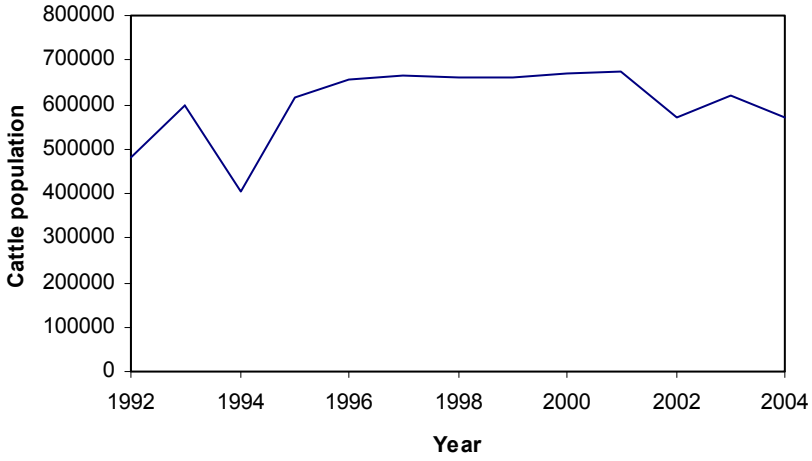

Figure 3. Cattle population in the Canindeyu region from 1992 to 2004.

Location of Agriculture Expansion

Expansion of agriculture in the Cuenca is nonrandom with respect to location. Based on an analysis of historical agricultural expansion in the Cuenca, Naidoo and Adamowicz (2006) determined that soil type, land tenure, and topography were the strongest explanatory variables for location of agricultural expansion and developed spatially explicit conversion probabilities for three agricultural land use types (smallholder, soy plantation, and ranchland). For this study, the spatially explicit conversion probabilities were combined with landcover data to derive relative probabilities of each cover type being converted to each agriculture land use (Table 3). The relative probabilities were used in the simulations to direct where land use expansion should occur. Simulations also explored the effect of converting existing agricultural land from one type to another. In particular, scenarios explored the implications of accommodating future soy production growth by converting ranchland to soy plantations.

Cover Type	Relative conversion probability		
	Smallholder	Soy Plantation	Ranchland
Bamboo forest	0.138	0.126	0.103
High forest	0.103	0.320	0.112
Low forest	0.117	0.101	0.109
Bamboo understory	0	0.005	0.048
Liana forest	0	0.035	0.129
Cerrado	0.178	0.095	0.085
Meadow	0.132	0.061	0.111
Cerradon	0.125	0.074	0.102
Riparian Vegetation	0.11	0.044	0.112
Wetland	0.097	0.139	0.089

Table 3: Relative conversion probabilities for future agricultural expansion.

Net Income

Smallholder Farms Net Income

Results from two studies of farms in nearby districts were used to parameterize net income trajectories for smallholder farms. Florentin et al. (2001) present 20-year crop productivity trends under smallholder conventional agriculture from the Department of San Pedro for cotton, maize, beans, and peanut. Lange (2005) reports the results of a shorter-term (5 year) comparison of conservation and conventional agriculture from smallholder farms in the Departments of San Pedro and Edelira. Results from these studies were used to estimate crop mix, long-term trends in productivity, farm costs, and crop prices associated with conventional and conservation agriculture.

A typical smallholder farm grows a variety of crops and raises livestock. Farms from the two regions studied by Lange (2005) differed with respect to crop mix. On Edelira farms, soybean, yerba mate, and tung accounted for an average of 66% of farmland used for crop production (i.e. not including forest and pasture). San Pedro farms, in comparison, did not grow these crops. On San Pedro farms, cotton, cassava, and maize were the dominant crops covering 75% of farmland used for crop production. To make use of the long-term crop productivity data from Florentin et al. (2001), which do not include soybean, yerba mate, and tung, I assumed that farms within the Cuenca have a crop mix that is equivalent to the San Pedro farms. I calculated the average crop mix across years (1998 and 2003) for San Pedro farms, calculating a separate crop mix for conventional and conservation agricultural strategies. I adjusted the crop mix by removing the peanut, bean and other crop types. Peanut and bean were removed because, although Florentin et al. (2001) provide long-term crop yield data for these crops, they cover only 6% of the farmland on the San Pedro farms. Other crop types were removed because long-term crop yield data were not available. This resulted in conventional and conservation agriculture crop mixes as presented in Table 4.

	Assumed (and actual) Crop Mix	
Crop Type	Conventional Agriculture	Conservation Agriculture
Cotton	24% (21%)	4% (3%)
Maize	20% (17%)	18% (14%)
Cassava	21% (18%)	43% (33%)
Peanut	0% (3%)	0% (3%)
Cowpea	0% (3%)	0% (3%)
Other[1]	0% (7%)	0% (16%)
Pasture/sugarcane	35% (30%)	36% (28%)

Table 4: Crop mix assumptions for conventional and conservation smallholder farms.
[1]Other includes pineapple, different food crops, watermelon, rock melon, calabash, pumpkin, and tobacco.

The 20-year crop productivity trends from Florentin et al. (2001) were combined with crop price and farming cost data presented by Lange (2005) to estimate conventional agriculture net income trajectories for maize and cotton (Figure 4). Using data from Edelira and San Pedro farms, and averaging across years (1998 and 2003), average crop price and conventional agriculture cost were $0.52/kg and $388.32/ha, respectively, for cotton and $0.11/kg and $178.72/ha, respectively, for maize. When calculating net income trajectories, crop price and agriculture cost was assumed to be constant while productivity declined as described by Florentin et al. (2001). I assumed that land used to produce cotton and maize would be abandoned when productivity is no longer sufficient to cover farming costs, i.e. when net income is below zero. This occurred after 26 years for cotton and 12 years for maize.

After abandonment of cropland, soil fertility gradually recovers as forest grows back on the site. Recovery of soil fertility, however, is slow. Metzger (2002) estimated that 11 years of fallow are required after each year of crop production to maintain crop productivity in the northeastern Brazilian Amazon. This, however, is likely an underestimate of the time required to restore soil fertility in the Cuenca because soil fertility will be severely reduced after multiple consecutive years of crop production. I was not able to find information in the literature regarding the length of time needed to restore soil fertility after consecutive years of crop production. Forest recovery after cropland abandonment can take more than 50 years (NASA 1998), suggesting that recovery of soil fertility will be slow. In the absence of better information, I have assumed that 30 years are required after abandonment of cropland prior to recovery of soil fertility. Time required for soil fertility recovery after abandonment is a key uncertainty that should be the focus of future research. After soil fertility is recovered, I assumed that a site is capable of supporting a native vegetation community or being cleared once again for cropland. In simulations, distribution of recovered land across native cover types is equivalent to the initial distribution of unprotected land across native cover types.

In contrast to cotton and maize, cassava productivity does not decline over time under conventional cultivation (Lange 2005). Cassava net income under conventional cultivation was estimated to be $353.85/ha (Figure 4). This was calculated by averaging across years the net income generated by farms studied by Lange (2005) using conventional agriculture. Because cassava productivity does not decline, land used for cassava production is not abandoned in the simulations.

Net income trajectories for conservation agriculture (Figure 4) were estimated using data collected by Lange (2005) during the first year of conversion to no-tillage agriculture (1998) and five years later (2003). None of the farms studied by Lange (2005) grew cotton using conservation agriculture in both 1998 and 2003, although 3 farms grew cotton using conservation agriculture in either 1998 or 2003. Therefore, temporal trend in net income under conservation agriculture could not be estimated for cotton. Instead, net income for cotton under conservation agriculture was assumed constant and equal to the average net income across the three San Pedro farms ($447/ha). While multiple farms grew maize under conservation agriculture in both 1998 and 2003, all farms used intercropping and the crop mix often varied across years. Only two farms used the same intercropping mix across years (one with cowpea, and the other with velvetbean). Net income trends for these two farms were averaged. Maize productivity beyond 5 years was assumed to be constant. Data from Lange (2005) was also used to estimate cassava net income under no tillage ($552.25/ha).

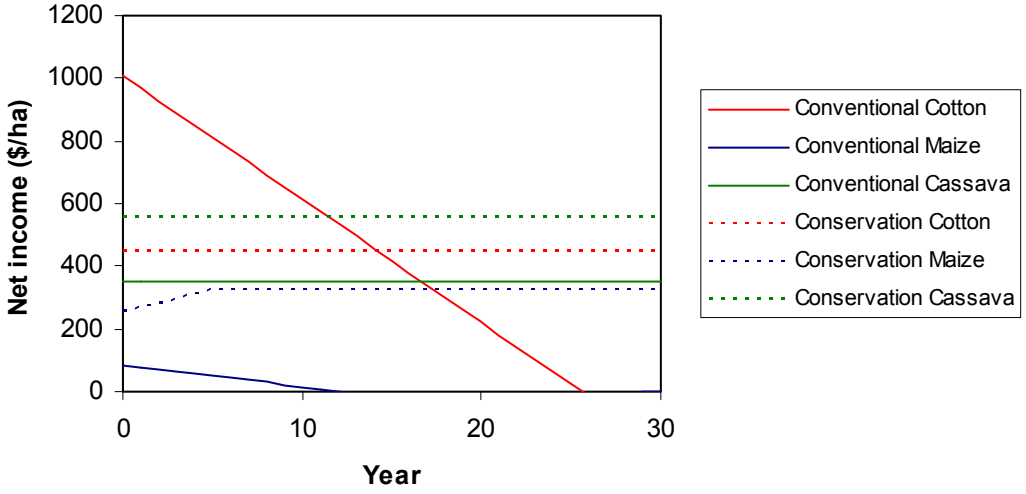

Figure 4: Net-income trajectory assumptions for smallholder crop production using conventional and conservation practices. Tables presenting net-income trajectories are available in Appendix 1.

Under conventional and conservation agriculture, pasture on smallholder farms was assumed to generate a net income of $52.50/ha, which represents the average net income generated by pasture across all farms and years studied by Lange (2005). No consistent trend in net income was apparent across years (i.e. 1998-2003).

Soy Plantations Net Income

Using data collected from 135 ha soy farms in the nearby districts of San Pedro and Itapua, Sorrenson (1997) estimated 10-year trends in net income that can be achieved under conventional and conservation agriculture (Appendix 1). These trajectories were used in the simulations, and farm abandonment was assumed to occur when net income was less than 0, which occurred after 7 years under conventional agriculture (Figure 7). As explained previously for abandoned corn and cotton cropland, recovery of soil fertility on abandoned soy plantations was assumed to require 30 years.

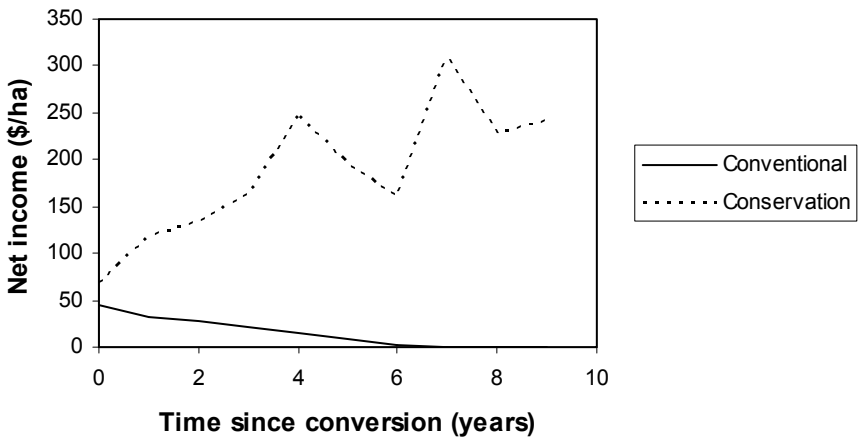

Figure 5: Net-income trajectory assumptions for soy production under conventional and conservation practices. Trajectory data are from Sorrenson (1997). A table presenting net-income trajectories is available in Appendix 1.

Ranch Net Income

Estimating net income generated by ranchland required the following information: stocking rate, current cattle population, average weight at slaughter, and cattle population turnover rate. Stocking rate was estimated using cattle population and pasture area data from for the Canindeyu region from the Paraguay Agriculture Census. The most current data available (2003) indicate 618,304 cattle were raised on 412,239 ha[3] of natural and cultivated pasture[4]. This results in a stocking rate of 1.5 cattle/ha. Although data from 1993-2002 were also available, the 2003 estimate was used due to an apparent gradual decline in stocking rate over time (Figure 6). A recent survey found an average stocking rate of 1.7 cattle/ha on 4800 ha of cultivated pasture in the Cuenca (Laura Rodriguez, pers. comm.). Given that cultivated pasture supports a higher stocking rate than natural pasture, which is the dominant type of ranchland in the Cuenca, the survey suggests that 1.5 cattle/ha is a reasonable estimate of the stocking rate in the region. Based on a stocking rate of 1.5 cattle/ha and 49 037 ha of ranchland, the current population of cattle in the Cuenca is estimated to be 73 556. The derived stocking rate estimate of 1.5 cattle/ha was lower than the 2-3.2 cattle/ha reported by Simpson and Fretes (1971) for an experimental station in Chaco, Paraguay. I assumed that the stocking rate is constant

[3] This includes natural and cultivated pasture, but not montes and esteros. Therefore, ALCES only tracks the area converted to pasture but not the additional area (e.g. forest and wetlands) used for raising cattle.
[4] It is assumed that all pasture is used for cattle production (i.e. and not other types of animals).

over time. This assumption is optimistic; the stocking rate will likely decline over time due to pasture degradation (FAO 2006). However, I was not able to locate data to estimate the rate of decline in net income/ha that can be achieved through cattle production.

Of the cattle raised in Canindeyu, 95% were raised for meat; the simulations assume all cattle are raised for meat. The average weight at slaughter for cattle in Paraguay in 2002 was 175.1 kg, while the population turnover rate for cattle in Paraguay in 2002 was 12.6% (FAO 2005). The price of slaughtered cattle in Paraguay during the week of August 17 2006 was 0.90 US$/kg. The price was reduced to 0.70 US$/kg to account for the cost of getting cattle to market (Laura Rodriguez, pers. comm.). Using these parameters, net income generation from ranchland was assumed to equal $23.17/ha.

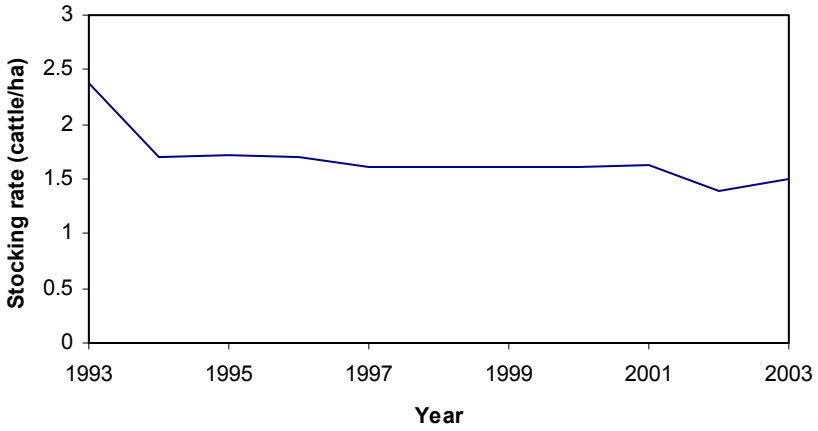

Figure 6. Stocking rate in the Canindeyu region from 1993 to 2003, as derived from pasture area and cattle from the agriculture census.

Pesticide and Fertilizer Use

Smallholder farm pesticide and fertilizer use

Pesticide and fertilizer use on smallholder farms was based on data reported by Lange (2005). For the land use simulations, pesticide use for each crop (cotton, maize, corn) is assumed to equal the average rate, weighted by crop area, across San Pedro and Edelira farms for each of the crops included in the simulations (Table 5). The assumed pesticide use rates are substantially higher for conservation than for conventional agriculture. However, if only the San Pedro farms are considered, pesticide use decreases with the adoption of conservation agriculture. Pesticide use on smallholder farms should be regarded an uncertainty that requires further research.

Of the 73 ha of cotton, maize and cassava crops studied by Lange (2005), fertilizer was applied to only 1 ha. Therefore, fertilizer use on smallholder farms is assumed to be 0.

Crop	Pesticide use (L/ha)

	Conventional	Conservation
Maize	0	2.44
Cassava	0	1.57
Cotton	1.88	3.60

Table 5: Assumed pesticide use rates on smallholder farms using conventional and conservation agriculture. See text for details.

Soy plantation pesticide and fertilizer use
Pesticide application rate on soybean crops is assumed to be 7.5 L/ha, based on a 5 to 10 L/ha pesticide application rate on soy cropland in South America reported by Bickel and Dros (2003). Bickel and Dros (2003) also report that using no-tillage (i.e. conservation agriculture) increases pesticide application rate for soybean. However, I was not able to find data comparing pesticide application rates for tillage and no-tillage soybean production. Therefore, pesticide application rate is assumed to be 7.5 L/ha on tillage and no-tillage soybean crops. Pesticide use on soy plantations is a key uncertainty that requires further research.

Fogel and Riquelme (2005) also report that soy farming in Paraguay accounts for 75% of fertilizer use. These ratios were applied to national fertilizer use data (FAOSTAT) and combined with national soy production data (FAOSTAT) to estimate fertilizer use per hectare of soy production. Between 2000 and 2002, average fertilizer use for soy production was estimated to be 52.7 kg/ha.

Initial Use of Conservation Agriculture
Data describing the level of adoption of conservation practices in the Cuenca or surrounding region were not available. Therefore, initial use of conservation agriculture was assumed equivalent to the level of adoption of these practices in Paraguay. Sorrenson et al. (1998) report that, of the 1,700,000 ha of smallholder farms in Paraguay, approximately 6,500 ha (0.004%) are farmed using conservation agriculture. In 2002, 60% of the total mechanized medium and large farms in Paraguay used conservation agriculture (Lange 2005).

Forestry Assumptions

Simulations explored the implications of harvesting natural forest and harvesting eucalyptus plantations. The assumptions for each are now explained in turn.

Timber harvest from natural forest
High forest, bamboo understory forest, and liana forest are assumed to contain merchantable wood, following the assumption that big bamboo and low forest types do not contain valuable timber species (Naidoo, pers. comm.). Because almost all bamboo understory forest and liana forest is protected, essentially all forest available for timber harvest is high forest. For each of the merchantable forest types, mature wood volume is assumed to equal the forest inventory from Ubiratam Vendramini, a property in the Cuenca. The forest inventory, which is for high forest, classifies inventoried species into

two quality groups: A[5] and B[6]. When estimating the volume of mature forest stands, I only included trees with diameter greater than or equal to 40 cm based on a minimum harvest diameter of 40 cm. Total volume for trees greater than 40 cm diameter was 14.790 m^3r/ha for commercial quality group A and 21.910 m^3r/ha for commercial quality group B, for a total of 36.701 m^3r/ha.

There are two situations in which natural forest could be harvested in the study area: clearing of land for agriculture and harvesting forest that is dedicated to timber production. When land is cleared for agriculture in the region, typically only a small percentage of the usable wood is utilized and the majority is burned. Grauel (1996) estimates that in Itapua, 5-10 m^3 of the estimated 150-300 m^3/ha is utilized, and that 70-80 percent of the useful wood is burned. Bozzano and Weik (1992) estimated that 1.8% of the wood (by volume) that is disturbed every year is utilized for saw logs and that an additional 5.6% is used for fire wood. The remainder is burned to clear land for agriculture. I assumed that 5/150 (3.33%) of the inventoried volume of wood is used for saw logs when forest is cleared for agriculture.

Based on growth rates observed during the first 8 years after logging Brazilian Amazonia, Carvalho et al. (2004) estimated that the forest would reach a stock available for harvesting 30 years post harvest. Vanclay (1994) reports 20-40 years as a typical tropical forest cutting cycle. Jackson et al. (2002) report a 30 year cutting cycle for a forest area in the Santa Cruz state of Bolivia. I assumed that natural forest can be reharvested every 30 years.

Not all trees selected as viable for harvest from inventory are harvested due to defects, not being found by the sawyer, or being missed by the skidding team. In Holmes et al. (1999), 397 of the 726 trees (54.7%) selected as viable for harvest from inventory lists were skidded to the log deck under conventional logging (CL) while 328 of the 670 trees (49.0%) were skidded to the log deck under reduced impact logging (RIL). In addition, some merchantable timber was wasted by cutting stumps too high, splitting logs, and due to improper bucking practices. Under CL 3.12 m^3r/ha were lost due to these causes while under RIL, 1.26 m^3r/ha were lost. The standard harvest volume was 25.36 m^3 in both cases. If wood waste had not occurred, harvest volume would have been 28.48 m^3/ha and 26.62 m^3/ha under CL and RIL, respectively. Therefore, under CL 11% of the harvest volume is wasted (3.12/(25.36+3.12)). Under RIL, 4.7% of the harvest volume is wasted. This equals 12.3% and 5.0% of the final harvest volume, respectively[7]. When missed merchantable trees and wood waste are both accounted for, CL utilizes 47.97% of

[5] Species in Commercial Quality Group A are Cedro, Guatambu, Incienso, Lapacho, Peroba, Peterevy, Taperyva guasu, Urunde'y mi, and Yvyra'ro.
[6] Species in Commercial Quality Group B are Alecrin, Cancharana, Colita, Guaviju, Inga, Katigua, Kirandy, Kupa'y, Kurupa'y ra, Laurel sayju, Timbo, Yva'ro, Yvyra pere, and Yvyra pyta.
[7] Based on the results from this study, I will calculate harvested volume as (inventory volume) * % of viable trees skidded to deck * (1-%wood waste). Under conventional logging, % skidded equals 54.7% and %waste equals 12.3%. Under RIL, % skidded equals 49.0% and % waste equals 5%. Therefore, under conventional, harvested volume equals 17.61 m3/ha. Under RIL, harvested volume equals 17.01 m3/ha. These harvest estimates are similar those reported by Jackson et al. (2002) (12.1 m3/ha).

available merchantable timber from natural forest and RIL utilizes 46.55% of available merchantable timber.

Based on a 30-year rotation age and timber productivity and utilization rates reported above, the annual allowable cut from natural forest in the Cuenca is 10,120 m^3r under CL and 9,820 m^3r under RIL. RIL achieves a lower AAC because the wood utilization rate is lower.

The value of harvested wood was estimated based on approximate prices paid for standing timber in the Cuenca (Laura Rodriguez pers. comm.): 7-9 US$/$m^3r$ white wood and 10-13 US$/$m^3r$ primary species. The mid point of each of these ranges is applied when calculating timber value (i.e. 11.50 US$/$m^3r$ primary species and 8 US$/$m^3r$ white wood). Commercial quality group A (see above) is assumed to be equivalent to primary species, and commercial quality group B is assumed to be equivalent to white wood. Therefore, value of standing timber in a mature stand is estimated to be 11.50 US$/$m^3r$ * 14.790 m^3r/ha = 170.09 US$/ha for primary species and 8 US$/$m^3r$ * 21.910 m^3r/ha = 175.28 US$/ha for white wood for a total value of 345.37 US$/ha. The average value of timber per m^3r is then 345.37 US$/ha/36.701 m^3r/ha = $9.41/$m^3r$.

<u>Timber harvest from eucalyptus plantations</u>

The life-history of eucalyptus and silvicultural strategies allow eucalyptus plantations to achieve higher growth rates than natural forest. As a result, plantations can be harvested at a younger age. FAO (2001a) reports a mean annual increment of 18-20 $m^3ha^{-1}yr^{-1}$ and a rotation length of 8-10 years for eucalyptus grown in South America. In the eucalyptus plantation scenario, I assumed a mean annual increment of 19 $m^3ha^{-1}yr^{-1}$ and a rotation length of 9 years. Eucalyptus plantations expanded from 0 ha at the start of the simulation at a rate of 500 ha each year by converting ranchland. All plantations that are of a merchantable age are harvested in any given year. I assumed that all timber volume in a harvested eucalyptus plantation was utilized, and that plantations were replanted immediately after harvest. Capacity of plantations to grow eucalyptus remained constant over successive rotations (FAO 2001b).

Carbon Storage Assumptions

Aboveground forest carbon storage assumptions for undisturbed forest types are as follows (Robin Naidoo pers. comm.): bamboo understory = 83.60 t/ha (biomass = 167.196 t/ha), vine undergrowth = 106.37 t/ha (biomass=212.744 t/ha); high forest = 63.01 t/ha (biomass=126.018 t/ha); big bamboo = 44.89 t/ha (biomass=89.787 t/ha); and low forest = 31.71 t/ha (biomass=63.413 t/ha). Aboveground carbon stored by eucalyptus plantation forest was assumed to be dictated by the mean annual increment. Aboveground carbon storage in natural non-forest types (wetland, meadow, riparian, cerrado, and cerradon) was assumed to equal 25 t/ha (biomass=50 t/ha) based on an assumed carbon storage value for swamp in the study area (Robin Naidoo pers. comm.). In simulations, carbon can be affected by conversion of forest to agriculture and by timber harvest. Both of these activities remove timber (through harvest or burning) and create coarse woody debris from which carbon is eventually released to the atmosphere through decomposition.

Agriculture conversion is assumed to remove all aboveground forest carbon. Although wood debris may remain on cropland and pasture for years after conversion, I assume the debris is committed emissions, i.e. C that will eventually be lost from the site via decomposition or combustion. When cropland is abandoned, forest biomass gradually grows back. Using Zarin et al.'s (2001) equation for the rate of biomass accumulation on abandoned agricultural land with sandy soils in the Amazon, I assumed that biomass would accumulate at a rate of 2.93 tonnes/ha/year. Once soil fertility of abandoned cropland has recovered, which was assumed to require 30 years, the land was assumed capable of supporting biomass equivalent to that supported by native vegetation. Biomass accumulation was equivalent regardless of the previous agricultural activity based on Zarin et al.'s (2001) finding that re-growth did not differ between abandoned pasture and slash-and-burn agriculture.

Conversion of forest to agriculture can also impact soil carbon storage. Although Hughes et al. (2002) discovered that land use did not cause an impact on soil C, decrease in soil organic matter through erosion and biological decomposition caused by conventional tillage agriculture can reduce soil carbon storage over time (Bayer et al. 2000). No-tillage agriculture, on the other hand, can increase soil carbon storage over time (Amado et al. 2006). A study in central Brazil determined that, relative to native savanna vegetation, conventional tillage decreased soil carbon by 0.4 Mg/ha per year whereas no-tillage and crop rotation increased soil carbon by 2.18 Mg/ha per year (Corazza 1999). Currently, soil carbon is not included in simulations due to uncertainty surrounding the size of the soil carbon store and the rate at which the soil carbon store changes in response to land use.

Timber harvest of a eucalyptus plantation was assumed to remove all aboveground forest carbon. Timber harvest of a natural forest was assumed to impact aboveground carbon through the export of logs, and through coarse woody debris (CWD) accumulation caused by tree damage associated with tree felling, and through road, deck and skid trail creation.

To estimate the amount of carbon exported as logs from natural forest, harvest volume (17.61 m^3r/ha for CL and 17.01 m^3r/ha for RIL) was first converted to biomass. To convert logged volume to biomass, a wood density ratio of 0.6 tonne/m^3r was applied, which is representative of typical wood density for tropical tree species in the Americas (Brown 1997). Biomass was then converted to carbon by assuming that biomass is 50% carbon. Therefore, carbon exported as logs was 5.283 t/ha under CL and 5.103 t/ha under RIL.

In their study of disturbance caused by reduced impact logging (RIL) in southern Amazonia, Feldpausch et al. (2005) determined that CWD carbon created by logging natural forest was 2.4 times the carbon exported as logs. Of the CWD created by RIL, Feldpausch et al. (2005) reported that 75% was due to tree felling, 13% to roads, 3% to decks, and 9% to skid trails. Although harvest intensity in Feldpausch et al. (2005) (6.4-15.0 m^3/ha) is lower than harvest intensity being assumed for the Cuenca (17.01-17.61

m^3/ha), CWD generation per volume harvested is not affected by harvest intensity (Feldpausch et al. 2005). Therefore, results from Feldpausch et al. (2005) were used to estimate CWD carbon created by RIL for the simulations.

To estimate CWD carbon created by conventional logging (CL) of natural forest, RIL findings from Feldpausch et al. (2005) were converted using findings from comparisons of forest disturbance caused by RIL and CL in Amazonia. Canopy gap disturbance caused by felling under CL was found to be approximately twice that caused under RIL (Table 6). Comparisons of road, deck, and skid trail disturbance caused by CL and RIL (Table 7) determined that forest disturbance from roads, landings, and skid trails by CL was 1.6, 1.9, and 1.9 times greater, respectively, than by RIL. Applying these ratios to Feldpausch et al.'s (2005) findings for CWD created by RIL, I estimated that CWD carbon created by CL was 4.6 times the carbon exported as logs (Table 8).

Study	Ratio comparing canopy disturbance caused by CL and RIL	Canopy disturbance measure
Pereira et al. (2002)	2.45	Canopy gap fraction
Asner et al. (2004)	1.84	Canopy gap fraction
Johns et al. (1996)	1.94	Trees damaged per hectare
Average	2.08	

Table 6. Results from studies comparing canopy disturbance caused by CL and RIL.

	Asner et al. 2004		Pereira et al. 2002		Average	
	CL	RIL	CL	RIL	CL	RIL
Roads	1.43%	1.1%	1.6%	0.8%	1.52%	0.95%
Landings	0.87%	0.57%	1.4%	0.65%	1.14%	0.61%
Skid trail and bole zone manoevering	7.63%	4.3%	7.05%	3.25%	7.34%	3.78%

Table 7: Percent of harvest area disturbed by roads, landings and skid trails.

	Ratio comparing forest carbon lost due to CWD creation and log export.	
	RIL	CL
Tree felling	1.8	3.6
Road	0.312	0.499
Deck	0.072	0.137
Skid	0.216	0.410

Table 8. Ratios used to calculate incidental CWD caused by logging. RIL ratios are from Feldpausch et al. (2005). CL ratios were calculated using RIL ratios and comparisons of forest disturbance caused by RIL and CL (see Tables 5 and 6).

Based on the carbon assumptions described above, total aboveground carbon loss caused by forestry (exported logs and CWD creation) in natural forest is estimated to be 17.349 t/ha under RIL and 29.829 t/ha under CL. Soil carbon is assumed to not be impacted by forestry (Hughes et al. 2002).

As the forest regenerates post-logging, carbon stores are replaced. However, if regeneration is insufficient to recover biomass to pre-harvest levels, carbon storage of regenerated stands will be less than primary forest. Unfortunately, studies aimed at assessing biomass regeneration after logging in South America are limited (Skole et al. 1999), and no suitable studies were identified. However, a simulation study of south-east Asian dipterocarp forests estimated that aboveground biomass approached preharvest levels after approximately 60 years (Pinard and Cropper 2000), which is equivalent to one harvest cycle in that region. Therefore, preharvest aboveground biomass was assumed to regenerate over one harvest cycle for natural forest, which equals 30 years in the Cuenca (Figure 7).

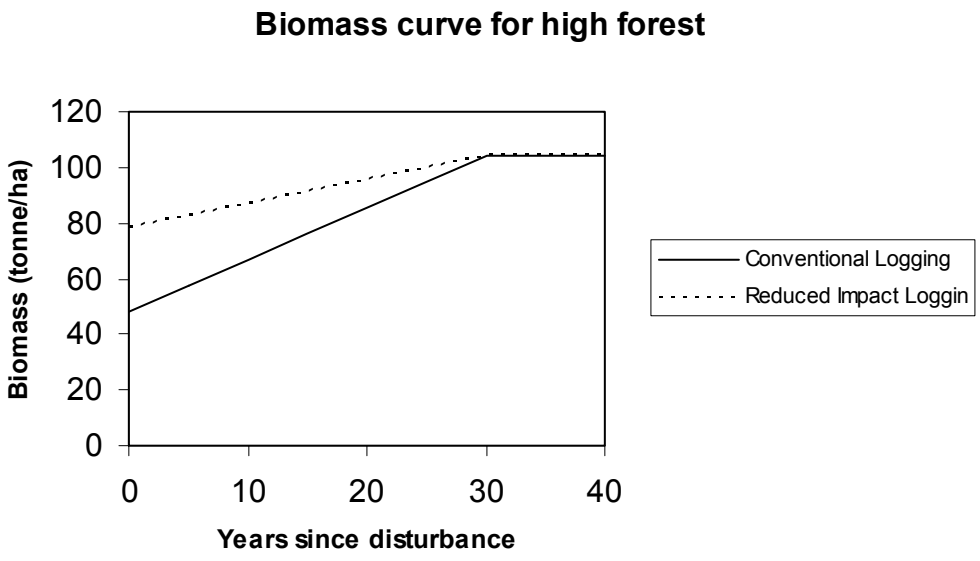

Figure 7: Biomass accumulation curve for high forest.

Wildlife Habitat

The consequences of land use scenarios to wildlife habitat were evaluated by applying results from a 7-year study that investigated the effect of hunting and habitat relationships for the 10 most hunted species in MBNR. Using data collected from MBNR, Hill et al. (2003) developed multiple logistic regression models of encounter probability for each species using the following covariates: forest type, hunting pressure, elapsed time, and the interaction between time and hunting pressure. Based on the model coefficients, Hill et al. (2003) concluded that hunting did not have a significant negative effect on wildlife, with the exception of red brocket deer.

I used the coefficients to calculate encounter probabilities for each habitat type for each species (Appendix 2). ALCES applied the encounter probabilities to calculate the effect of simulated changes in landscape composition to habitat availability using the equation

Habitat availability = $\sum(P_i * E_i)$

where P_i is the proportion of the Cuenca within the ith natural cover type and E_i is the encounter probability for the ith natural cover type. ALCES calculated habitat availability for 2 of the 10 species: white-lipped peccary and tapir. White-lipped peccary is a generalist species, while tapir is a species that prefers the habitat types low forest lianas and small bamboo which are both well represented in protected areas (Figure 8).

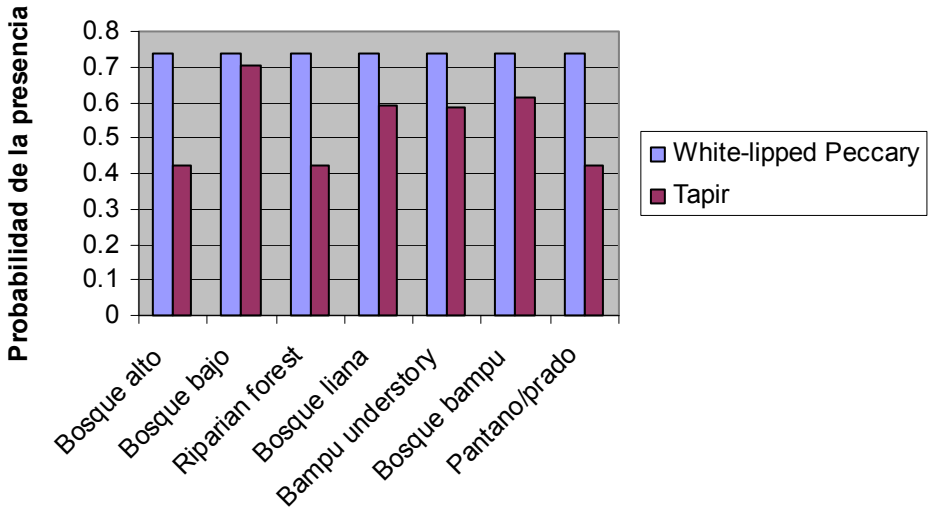

Figure 8: Probability of encountering white-lipped peccary and tapir in a variety of forest types. Probabilities were calculated using logistic regression coefficients from Hill et al. (2003).

Several caveats must be considered when interpreting the simplistic calculation of habitat availability. First, habitat availability evaluates wildlife habitat based on changes in the overall amount of natural cover types, which ignores the potential impact of fragmentation. If forest becomes less useful to wildlife as fragmentation increases, the impact of future land use to wildlife habitat will be overestimated by the simulations. This would be the case, for example, if patches of forest below a threshold size (e.g. home range size) are not occupied by wildlife. On the other hand, habitat availability as calculated in the simulations also assumes that agricultural land and eucalyptus plantations are not utilized by wildlife (i.e. are not wildlife habitat). In addition, simulations assumed that wildlife does not utilize abandoned agricultural land until the land is reclaimed (i.e. after 30 years). If these types of landscapes do contribute to wildlife habitat, habitat availability will be underestimated by the simulations.

The second caveat is that, in addition to the amount of habitat (i.e. habitat availability), wildlife may be impacted by habitat quality. For example, habitat quality may be altered

by the presence of roads, the age of the forest, the percent of the landscape that is cultivated, and human population density. Habitat quality was not investigated here because information was not available on the effects of attributes such as human population and forest age-class to wildlife.

The third caveat is that, while Hill et al. (2003) collected wildlife data from MBNR, I applied results from their research across the entire Cuenca. To evaluate whether extrapolation of results beyond MBNR was valid, I reviewed the literature to determine whether selective logging of natural forest negatively affects the wildlife species studied. Although I was not able to find data for all species, data that I did find suggest that selective logging does not have a negative affect. Johns (1986) found identical encounter rates for red brocket deer in logged and unlogged forests. Johns (1986) also recorded more collared peccary sightings in logged forest than in unlogged forest. The tapir's generalist diet suggests that the species will respond well to changes in habitat following logging (Davies et al. 2001). Johns (1991) reported higher capuchin monkey densities in logged forest than unlogged forest. Based on these reports of non-negative or positive effects of logging on wildlife, I determined that the results from Hill et al. (2003) could be applied to the entire study area. It is important to note, however, that Putz et al.'s (2000) review of the effect of forestry on tropical ecosystems concluded that increased hunting pressure caused by forestry roads is one of the greatest impacts caused by forestry, and that the impacts of commercial hunting are several magnitudes greater than subsistence hunting. However, I was not able to identify research that could be used to approximate a relationship between forestry and decline in wildlife habitat due to hunting pressure.

Results

Land cover dramatically changed over time in the simulations. In the conventional scenario, area within natural cover types decreased whereas area within agricultural cover types increased (Figure 9). Declines in natural cover types occurred in all scenarios, but the magnitude of the decline varied (Figure 10). Under the conventional scenario, natural land decreased to a low of 84521 ha by year 35 such that virtually the only natural land remaining was protected (82788 ha). Thereafter, natural land area oscillated to a small degree as abandoned agricultural land was reclaimed (thereby increasing natural land) and subsequently converted to agriculture (thereby decreasing natural land). Under the protection law scenario, natural land remaining after 50 years was greater because more land was protected (110103 ha). Similarly, the natural forestry scenario maintained more natural land than the conventional scenario because merchantable forest types were not available for conversion to agriculture.

Decrease in natural land was smallest in the conservation scenario as a result of two strategies that reduced the need to convert natural land for agricultural production: a) conservation farming practices that eliminated the need to abandon cropland (Figure 11); and b) all soy plantation expansion occurred into ranchland rather than natural land. The area of natural land conserved by these two strategies was approximately equivalent, with sustainable farming practices conserving 30006 ha and ranchland conversion conserving 33535 ha compared to the conventional scenario. However, some conversion of natural

land still occurred in the conservation scenario in order to increase smallholder farm production at a rate of 1.5% per year (Figure 10). Combining conservation practices with protection or natural forestry did not substantially improve capacity to maintain natural land (Figure 12). This is because expansion of cropland in the conservation scenario was not large enough to use up the non-protected natural landscape. If the simulation period was long enough to convert all available land to agriculture, implementing the protection law or natural forestry in combination with the conservation practices would be effective at conserving more natural land.

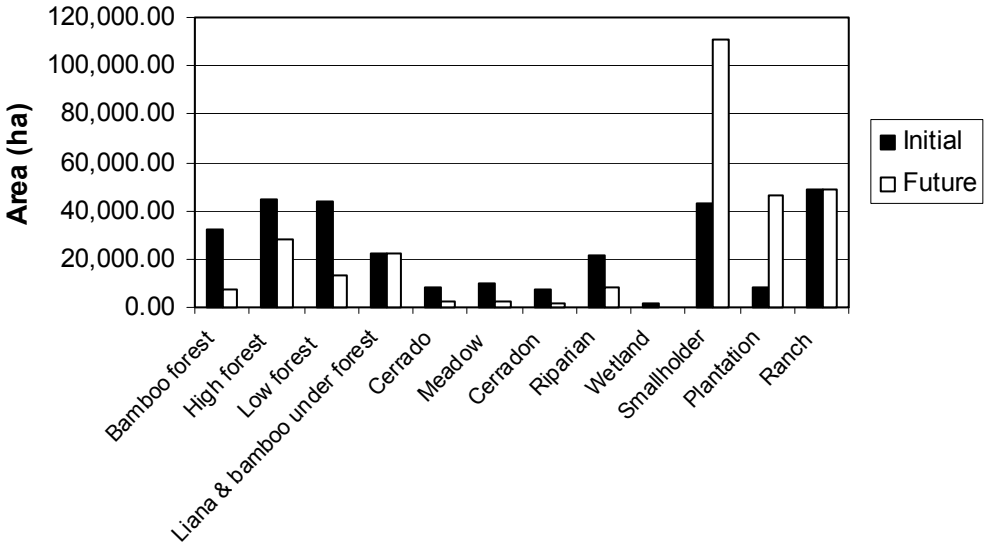

Figure 9. Area in each cover type in the Cuenca at the beginning and end of the conventional scenario.

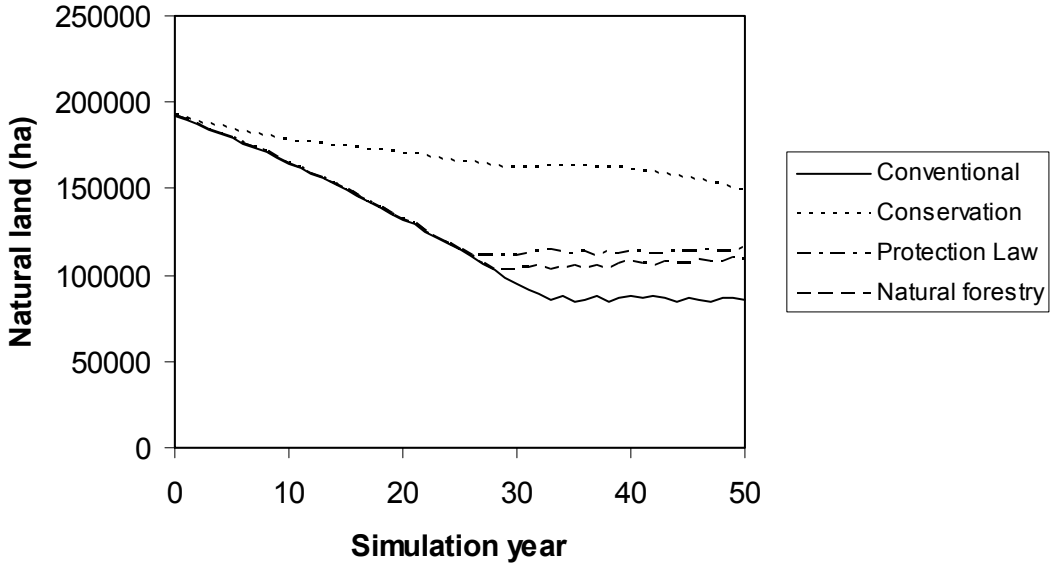

Figure 10. Simulated change in natural land in the Cuenca for four land use scenarios.

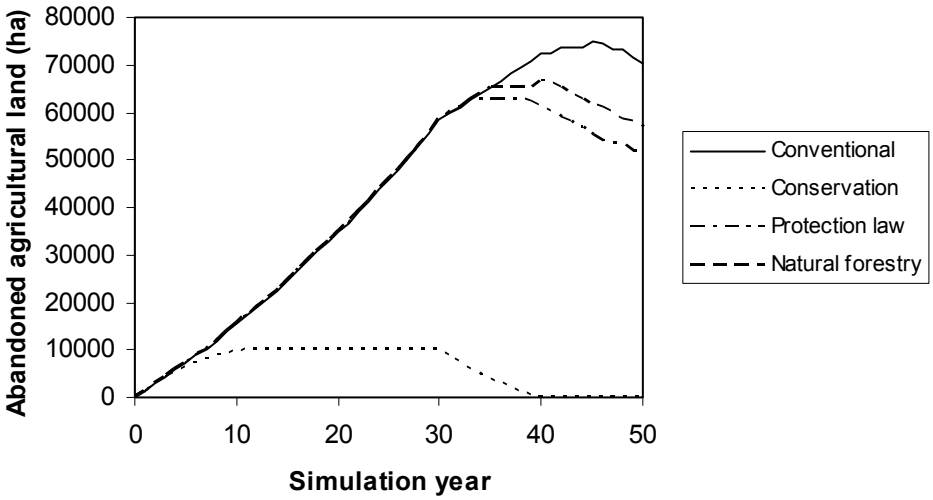

Figure 11. Simulated total abandoned agricultural land in the Cuenca for four land use scenarios.

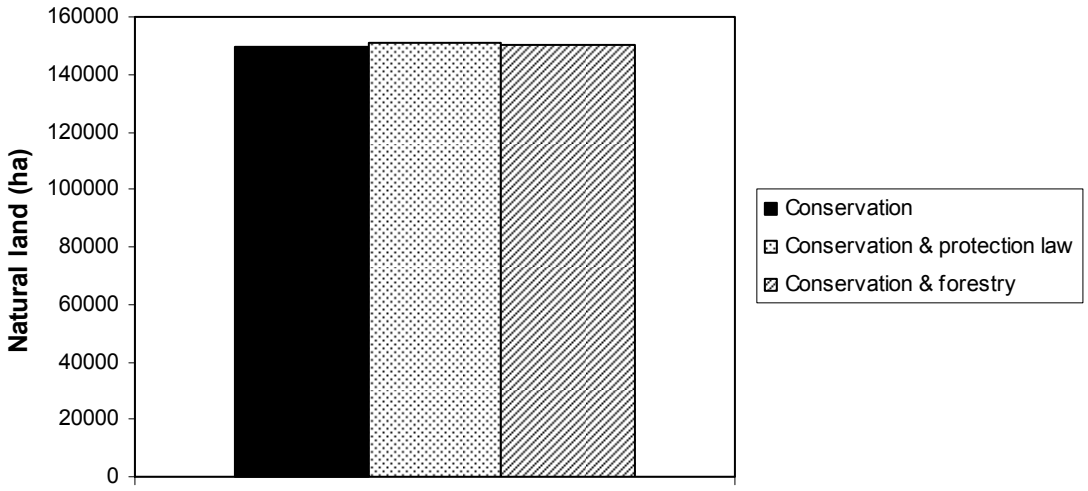

Figure 12. Natural land within the Cuenca at the end of a 50-year simulation when conservation practices are used alone and in combination with the protection law and forestry. Forestry refers to natural rather than plantation forestry.

Net income also changed over the course of the simulations. In the current scenario, net income generated by smallholder farms and soy plantations initially increased in response to expanding crop production (Figure 13). After year 33, however, unprotected natural land for conversion to agriculture was limited to abandoned cropland that had been reclaimed. For the remainder of the simulation, net income generated by smallholder farms declined and then oscillated because insufficient reclaimed cropland was available to permit continued expansion in agriculture production. Soy plantations were not as susceptible to net income decline because of the greater use (60% vs 0.004%) of conservation agriculture practices.

Over the long-term, the conservation scenario achieved the greatest net income (Figure 14) because agriculture productivity was maintained through the use of conservation agriculture practices, thereby eliminating cropland abandonment (Figure 11). In the first few years, however, the conservation scenario achieved lower net income. The reason is that conversion from conventional to conservation agriculture reduces the amount of agricultural land dedicated to cotton production. Cotton is a valuable crop compared to other crops such as cassava, and reduced cotton production causes initially reduced net income generation by the conservation strategy. Cotton productivity rapidly declines under conventional agriculture, however, such that the conservation scenario achieves higher net income over the long-term.

Net income decline in the protection law and natural forestry scenarios occurred earlier compared to the current scenario (Figure 14) because less natural land was available for conversion to cropland. Although net income generated by natural forestry is sustainable, the amount generated per ha is substantially lower compared to agriculture (Figure 15). Converting ranchland to eucalyptus plantations achieved a small increase in net income

(Figure 14), indicating that eucalyptus plantations may be capable of generating more net income than cattle ranching.

Although net income increased under all scenarios, net income per capita did not (Figure 16) because the human population in the Cuenca increased from 30000 to 62224 during simulations following an assumed 1.5% growth rate. Scenarios that used conventional agricultural practices resulted in decreased net income per capita by the fourth decade after all unprotected natural land had been converted to agriculture. Under conservation agriculture, however, net income per capita increased due to the increased and sustainable production achieved by conservation agriculture practices. It is important to note that per capita net income only accounts for income directly generated by the sale of agriculture and timber production, and therefore does not represent all economic production in the region. In addition, net income per capita can not be interpreted as an average wage because it averages across the entire population rather than just the workforce.

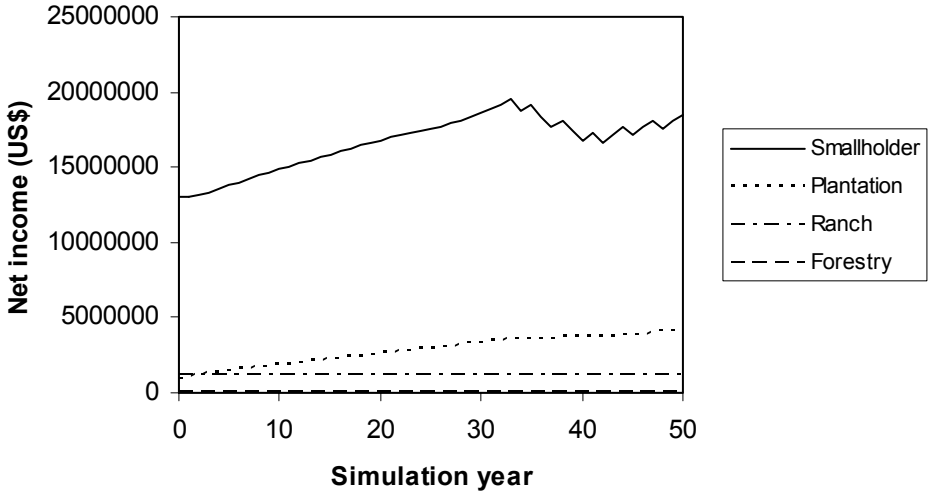

Figure 13. Simulated net income contributed by four land uses in the Cuenca for the conventional scenario.

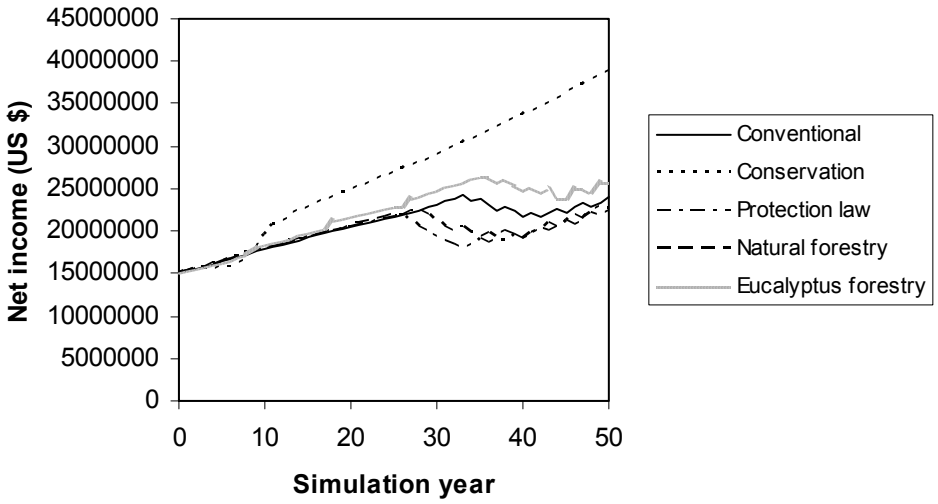

Figure 14. Simulated net income in the Cuenca generated by five land use scenarios.

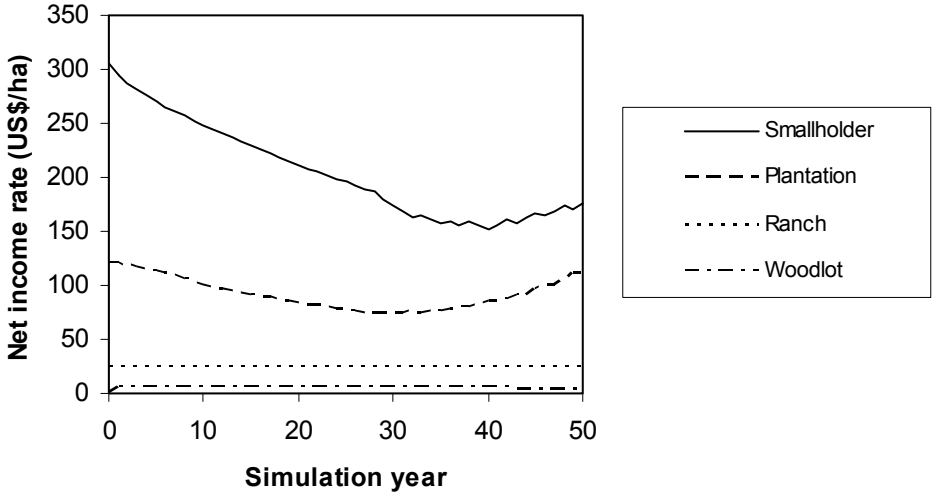

Figure 15. Simulated net income contributed by four land uses in the Cuenca for the natural forestry scenario.

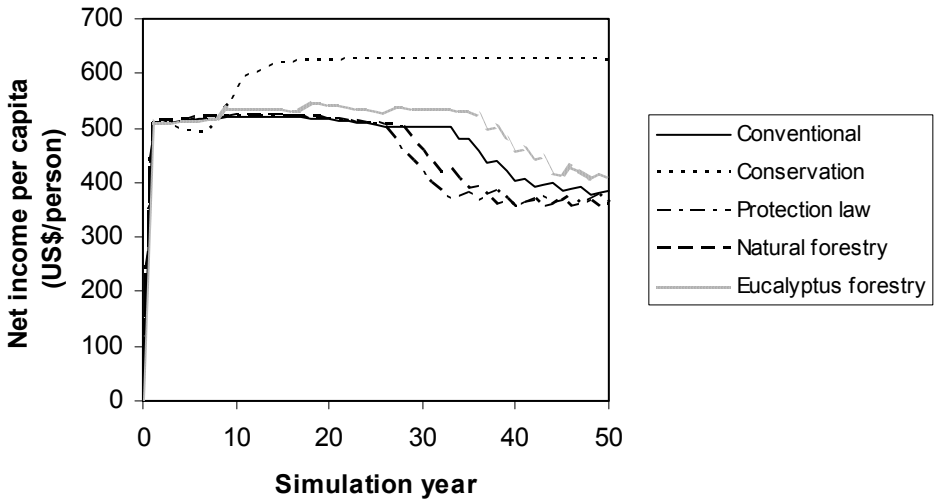

Figure 16: Simulated net income per capita in the Cuenca generated by five land use scenarios.

Carbon storage in the Cuenca declined over the course of simulations due to conversion of forest to farm land (Figure 17). Conservation practices were best able to maintain carbon storage, which again is the result of eliminating cropland abandonment. Compared to the conventional scenario, protection and forestry scenarios maintained more carbon storage because more forest remained in the Cuenca (either as protected areas or woodlots). Similar patterns were evident for the wildlife habitat indicators (Figures 18 and 19).

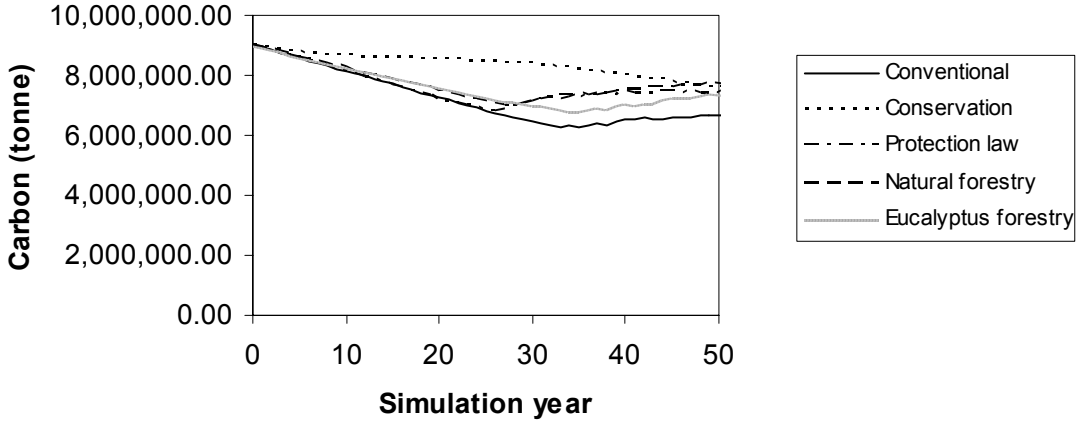

Figure 17. Simulated aboveground carbon storage in the Cuenca for five land use scenarios.

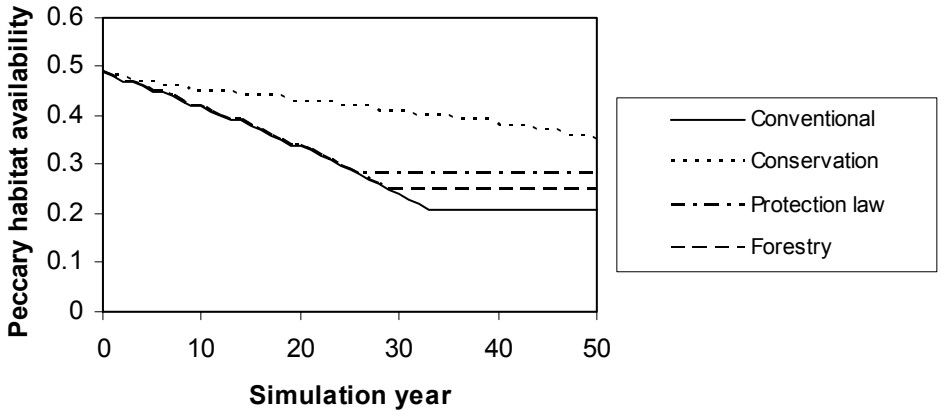

Figure 18. Simulated white-lipped peccary habitat availability in the Cuenca for four land use scenarios. Forestry refers to the natural forestry scenario.

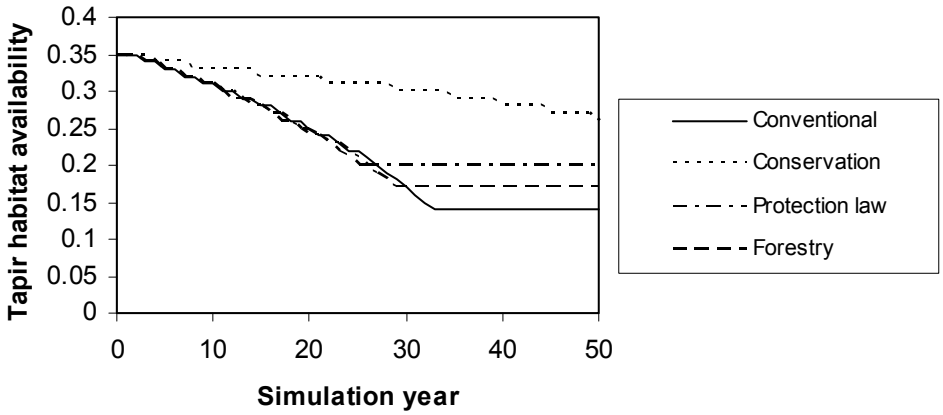

Figure 19. Simulated tapir habitat availability within the Cuenca for four land use scenarios. Forestry refers to the natural forestry scenario.

Pesticide and fertilizer use initially increased during the simulations as the area of cropland increased (Figures 20 and 21). Pesticide and fertilizer use declined towards the end of the simulations, however, in response to declining active cropland area due to abandonment. The exception was the conservation scenario for which pesticide and fertilizer use increased for the entire simulation because expansion in active cropland continued throughout. Conservation practices also resulted in immediately higher pesticide use at the start of the simulation because of the higher pesticide use intensity on smallholder farms under conservation practices as compared to conventioned practices.

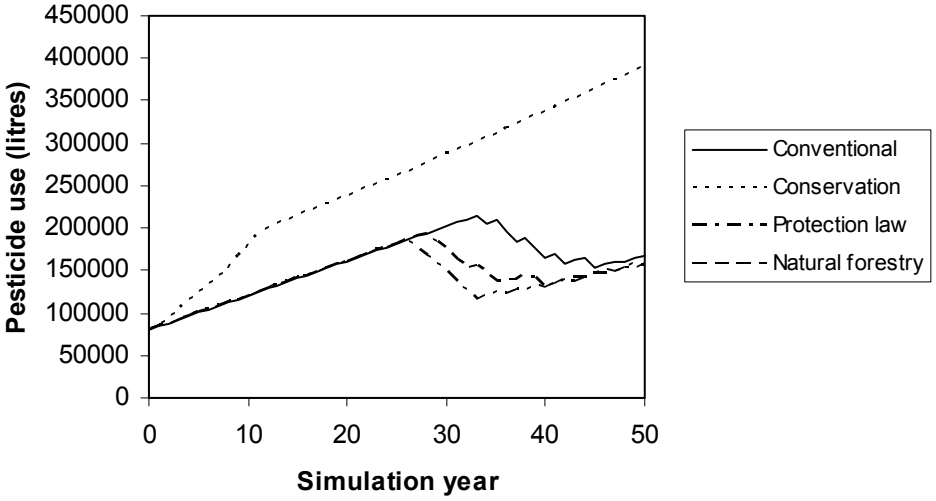

Figure 20. Simulated pesticide use within the Cuenca for four land use scenarios.

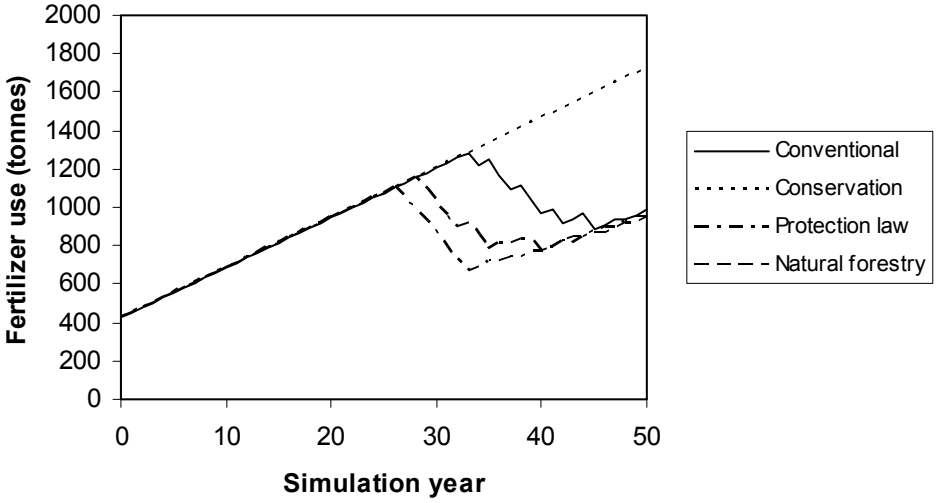

Figure 21. Simulated fertilizer use within the Cuenca for four land use scenarios.

Discussion

Management Implications

The scenario analysis reported here suggests a grave future for the Cuenca if current land use practices continue. In the conventional scenario simulation, continued use of unsustainable farming practices caused abandonment of a vast amount of cropland. As a result, conversion of natural land to agriculture occurred rapidly. In just over 3 decades, all unprotected land was converted to agriculture to the detriment of wildlife habitat and carbon storage. Pesticide and fertilizer use also increased, which could increase the risk of exposure of local inhabitants to toxins. While agriculture production initially grew to accommodate local population growth and rising demand for soy, production declined

after natural land was no longer available to support continued agriculture expansion. Although not included in the simulation, it seems likely that the ecological integrity of protected areas would also decline as local residents would be forced to expand agriculture into these areas to maintain food production. To summarize, the simulation predicts that in 50 years the Cuenca will be a region of severe poverty and ecological degradation if current practices continue.

Fortunately, the scenario analysis indicates that economic and ecological ruin need not occur. Conservation agricultural practices, in particular, have the potential to support the Mbaracayu program's goal of supporting both biodiversity and the well-being of local inhabitants. Wide-scale adoption of conservation practices such as no-tillage and crop residue management may not only maintain but actually increase the productivity of farmland within the region. As a result, abandonment of farmland would no longer be necessary and pressure to convert natural land would decrease. In the conservation scenario simulation, about 60% of the unprotected natural land that currently exists remained after 50 years. At the same time, economic growth was sustained for the full 50 year simulation.

Simulation of the forest protection law demonstrated that protection can improve ecological values such as wildlife habitat and carbon storage, but to the detriment of socioeconomic factors such as net income. Increased protection reduced the area available for cropland such that net income decline occurred earlier. When the forest protection law was implemented in conjunction with sustainable agricultural practices, however, sufficient land was available to support increased protection and a tripling of net income over the 50-year simulation period. This suggests that protection will be most effective when implemented in concert with sustainable management of the working landscape. Sustainable practices can achieve long-term economic productivity and reduce pressure on ecosystems, while protection can ensure the long-term persistence of natural ecosystems.

Sustained-yield harvest of natural forest in the Cuenca resulted in the maintenance of more forest to the benefit of wildlife habitat and carbon storage. However, net income generated by forestry was less than $6 per hectare, a small fraction of that generated by agriculture. Therefore, if the underlying assumptions are accurate, net income generated by forestry is insufficient to make-up for the decline in agricultural production that is foregone to dedicate land to forestry. The implication is that, as with protection, net income declines earlier when forestry is implemented as compared to the conventional scenario. Reasons for the poor economic performance of forestry include low merchantable volume, the need to wait 30 years between harvests (as compared to annual crop harvest), and low wood prices. Even if forestry on natural forest land was economically viable, widespread implementation in the Cuenca is not possible because unprotected merchantable forest only covers an estimated 9% of the region.

Rapid growth of eucalyptus plantations for timber harvest achieved increased net income and carbon storage. The increase was only moderate because of a low (assumed) wood price and because the carbon stored on eucalyptus plantations fluctuates in response to

harvest and re-growth. However, the result does imply that tree plantations may be more economically and environmentally beneficial than ranching. As a result, it is worthwhile to more carefully evaluate the viability of converting farmland to tree plantations in the region.

The poor economic performance of protection and forestry scenarios is at least partially due to the fact that the economic value of maintaining natural land is not considered. Natural land provides a variety of valuable services including carbon storage, food (i.e subsistence hunting), medicinal plants, and water. Degradation of these services by conversion of natural land negatively affects the quality of life of local communities (e.g. subsistence hunting) and beyond (e.g. carbon storage), and this degradation should be considered in economic analyses. For example, forest carbon trading is growing in prominence due to carbon's role in climate change. Recent prices per ton of CO_2 equivalent[8] include US$4.05 (Chicago Climate Exchange), US$9.42 (New South Wales Greenhouse Gas Abatement Scheme), and US$20.80 (European Union Emissions Trading Scheme) (www.ecosystemmarketplace.com). The aboveground carbon stored by natural areas in the Cuenca is currently 32.6 million tons of CO_2 equivalent. If the carbon could be traded at a mid-range value of $10/ton, it would be worth $US 326 million which might make alternatives to agriculture such as protection and forestry economically attractive. Trading of ecosystem services such as carbon is in its infancy and implementation is not straight forward. However, the new Law of Environmental Services (Nº3001/06) in Paraguay provides a regulatory framework that is expected to motivate the trading of forest ecosystem services in the near future. The law is intended to establish a payment mechanism for the environmental services of carbon storage and water and soil conservation that are provided by forest. The law states that agriculture producers that do not have 25% of their property forested are required to undertake reforestation or provide payment to a producer that has a surplus (i.e. more than 25%) of forested land on their property. The law is expected to take effect this spring and the National Forest Service has requested property owners to register their lands to be eligible to receive compensation for forest conservation (Laura Rodriguez, pers. comm.). Further research is required to consider how policy can be structured to provide the economic incentives necessary to motivate conservation of the region's valuable ecological integrity.

Given the combined economic and ecological benefits of conservation agricultural practices, it is perhaps surprising that such practices were used by only 0.004% of Paraguayan smallholder farms in the late 1990's (Sorrenson et al. 1998). Factors that have likely contributed to low implementation of conservation practices include the initial cost of switching from conventional to conservation practices and capacity. Initially, conversion to conservation crop production decreased net income generation because production of economically valuable cotton declined. Even though the productivity of cotton crops is short-lived, farmers may be reticent to forego short-term economic gain in favor of long-term sustainability given that poverty in the region currently leaves 40-50% of families without access to education and healthcare (Weber and Cardozo 2003). In addition, introduction of conservation agriculture requires initial

[8] 1 ton of carbon equals 3.7 tons of CO_2 equivalent.

investment in machinery and other inputs such as green manure cover crop (gmcc) seeds (Lange 2005). Although these costs can be rapidly paid back due to the increased productivity achieved by conservation agriculture, they provide an additional financial barrier to the implementation of conservation practices that were not considered in simulations. As recommended by Lange (2005), a financial mechanism to provide farmers with credit to implement conservation agriculture is needed. Also needed is education of farmers to increase recognition of the benefits of conservation agriculture and capacity to implement the practices. Farmers may understandably perceive implementation of a new agricultural strategy as an economic risk. Education, demonstration projects, and provision of inputs such as gmcc seeds should help overcome this reluctance.

Assumptions and Next Steps

I strove to use the best information available in the scenario analysis. However, as with any simulation study, it is important to reflect upon the realism of the simulations. One way to do this is to compare historical behaviour to that predicted by the simulations. The most important dynamic in the simulations is the conversion of natural land to agricultural land. A reasonable question, therefore, is whether the historic rate of conversion is similar to that predicted in simulations. As shown in Figure 22, the growth in agricultural land predicted by the conventional scenario simulation closely matches growth that has occurred since 1973.

Due to the impact of agricultural expansion on both economic and ecological indicators in the simulations, it is important to consider the sensitivity of scenario results to assumptions underlying the growth of agricultural land. In the simulations, expansion of agriculture land was caused by declining net income rates due to declining crop productivity, the amount of time required to reclaim abandoned cropland, and growth in total agricultural production. Net income trajectories were based on productivity, crop price, and agricultural cost data from the nearby districts of San Pedro, Edelira, and Itapua. The time required to reclaim abandoned cropland was an informed guess based on minimal information. The assumed rate of growth in agricultural production was based on the recent regional population growth rate in the case of smallholder farms, and on government and industry projections in the case of soy plantations. The resulting net income rate, reclamation and production trajectories are bound to be inaccurate to some degree. For example, if productivity, crop price, or agricultural cost typical of the Cuenca differs from these regions due to biophysical factors (e.g. soil type, precipitation, etc.), agricultural practices, or economic factors, then the assumed net income trajectories will be inaccurate. The assumed time required to reclaim abandoned cropland should be viewed as especially suspect due to the lack of information available to inform the assumption.

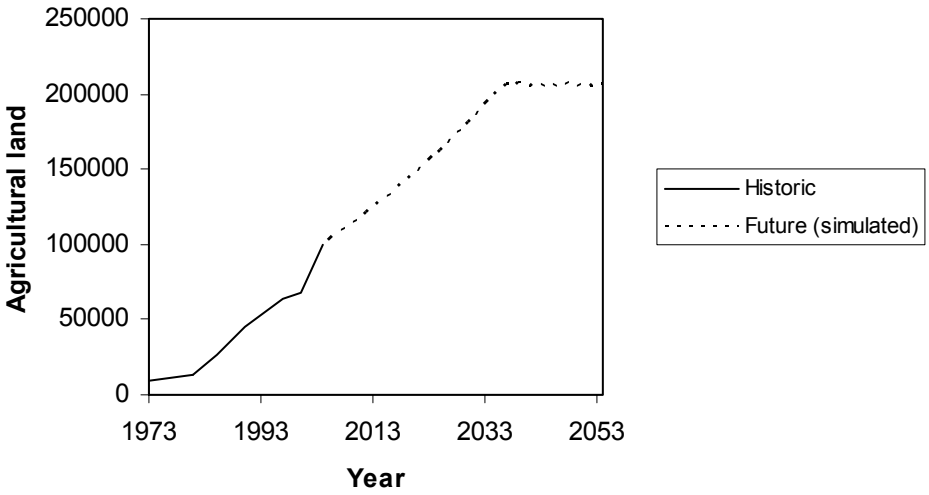

Figure 22. Growth in agricultural land in the Cuenca according to historic data and the conventional scenario simulation.

To explore the sensitivity of scenario results to net income, reclamation and production trajectory assumptions, the current scenario was simulated while varying these trajectories. The rate of decline in net income, the time required for reclamation of abandoned cropland, and the rate of growth in agriculture production were increased and decreased by 25%., Figures 23 and 24 display results from the sensitivity analysis. Not surprisingly, simulated agricultural land and total net income responded to changes in the assumptions. This underscores the importance of implementing research in the Cuenca to explore long-term trends in crop productivity under different agricultural strategies, and the fate of abandoned agricultural land. However, it is also important to note that the overall patterns were largely unaffected by changes in the trajectories. Net income increased until all available land had been converted to agriculture and declined thereafter. As long as productivity of soil declines over time following conversion of natural land to farming, as long as reclamation of soil fertility is slow, and as long as agricultural production in the Cuenca continues to grow, the long-term environmental and economic sustainability of the region will be threatened.

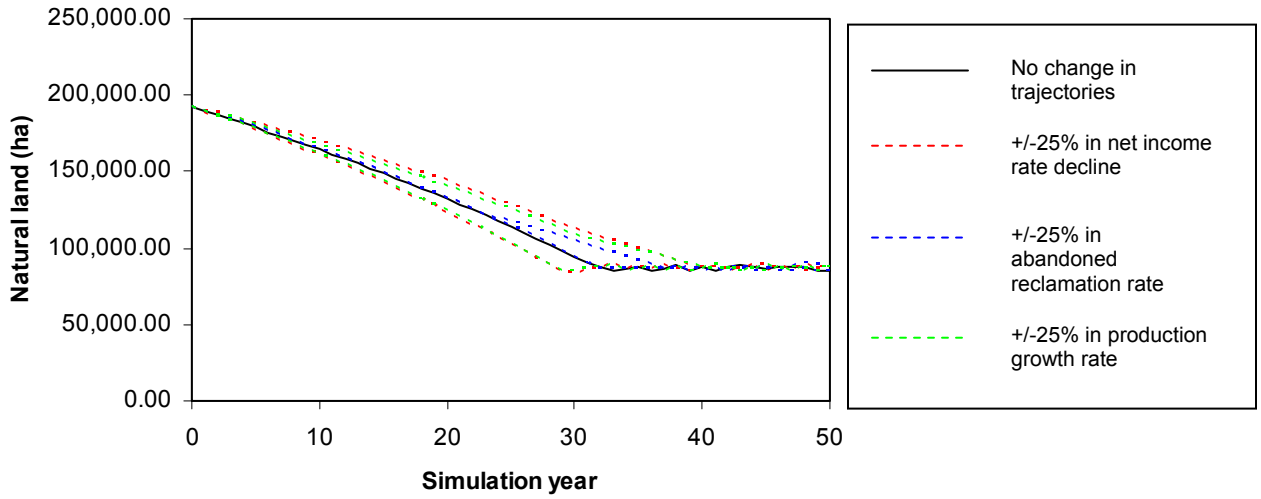

Figure 23. Sensitivity of simulated natural land area under the conventional scenario to changes in net income rate, abandoned cropland reclamation rate, and agriculture production growth rate assumptions.

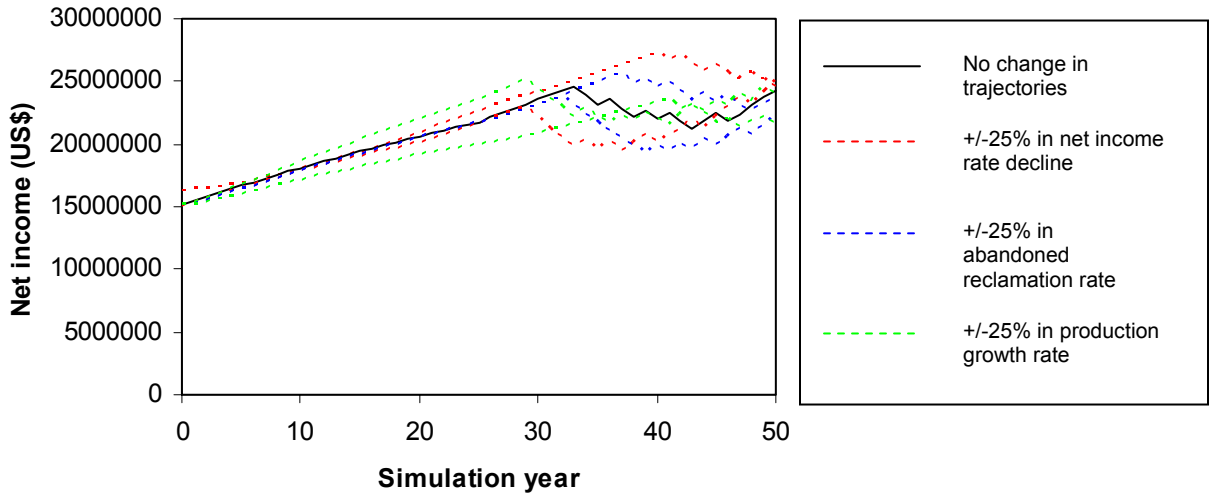

Figure 24. Sensitivity of simulated net income under the conventional scenario to changes in net income rate, abandoned cropland reclamation rate, and agriculture production growth rate assumptions.

Uncertainty also surrounds parameters used to explore consequences to wildlife, pesticide and fertilizer use, and carbon storage. As explained previously, wildlife habitat availability calculations made a number of assumptions including that fragmentation has no effect and that abandoned cropland is not utilized by wildlife. Little data were available to inform assumptions made about pesticide and fertilizer application rates, especially for soy plantations. As a result, simulations assumed that conventional and conservation soy agriculture applied equivalent amount of pesticides. In reality, pesticide

use may increase under conservation agriculture (Bickel and Dros 2003). Data to inform carbon storage projections were also lacking, especially for soil carbon. Soil carbon was excluded from scenarios because quantitative data were not identified. As a result, the simulations may overlook a benefit of conservation agriculture. No-tillage has been shown to substantially increase soil carbon storage (Amado et al. 2006), and therefore conservation agriculture may be capable of increasing the capacity of the Cuenca to store carbon. Wildlife habitat relationships and the response of pesticide and fertilizer use and soil carbon storage to agricultural practices are key uncertainties that warrant future research.

The simulation did not consider the impact of variability in productivity and crop prices. For example, between 1985 and 2005 the international price of cotton displayed a coefficient of variation of 20%[9]. Between 1993 and 2003, cotton yield in the Canindeyu department also varied[10]. Although income volatility presents a challenge to farmers, it should not affect the general conclusion that conservation agriculture strategies are beneficial. In fact, productivity variation may decrease under conservation agriculture (http://www.fao.org/AG/ags/AGSE/conservation.htm), which represents an additional benefit of implementing conservation agriculture in the Cuenca. The sensitivity of simulation results in a given year to stochasticity as well as to minor changes in assumptions emphasizes that the simulations should not be considered accurate predictions of the future. As with any analysis, it is not possible to predict the future with certainty. The simulations are useful, however, for comparing the likely implications of a range of land use options to identify which options are most consistent with the desired future for the Cuenca.

In the simulations, net income generated per hectare differs markedly across land uses. At the start of the simulation, smallholder farms generated almost twice as much net income per hectare compared to soy plantations, and almost ten times as much compared to ranches. The higher net income generated by smallholder farms seems counterintuitive given the poverty experienced by such farmers and the relative wealth of plantation and ranch owners. However, the high productivity of small farms compared to industrial agriculture has been noted elsewhere. Lange (2005) states that despite representing only 14% of the total agricultural land of Paraguay, such farms generate 35% of the total value of agricultural production. The comparative wealth of large landowners therefore appears to be the result of controlling a large land base rather than generating high value per hectare farmed. Perhaps most surprising is the low net income generated by ranchland. However, the trend towards converting ranchland into soy cropland that has been observed in recent years (Laura Rodriguez, pers. comm.) is evidence that ranches are indeed poor economic performers.

The small-set of indicators evaluated by simulations were able to convey important economic and ecological implications of land use. However, many indicators that are important to the local community could not be simulated. Maintaining surface and ground water quality, for example, is a high priority. Water quality is likely to be

[9] Based on the Cotlook A (NE) Index. Data obtained from www.cotlook.com.
[10] Based on Paraguay Agriculture Census data.

negatively affected by pesticide and fertilizer use and positively affected by no-tillage due to reduced sedimentation of streams and rivers. Although the use of pesticides, fertilizers, and no-tillage were included in simulations, I was not able to identify quantitative relationships that could be used to include an explicit water quality indicator. The effect of land use on water quality is a key uncertainty that requires further research.

Other important indicators that were not included in simulations mainly relate to social conditions. Examples of social indicators that are very important to local communities are levels of education, food availability, access to services, income per household, rates of employment, and participation of women in governance and employment. As with water quality, the simulations tracked indicators that likely influence these social indicators. For example, net income and the availability of productive agricultural land will affect food availability, employment, and income per household. However, just as important as total net income is how the net income is distributed across the community. If net income is monopolized by a few wealthy landowners, benefits to the broader community may not occur. The relationship of total net income to social indicators is another key uncertainty that requires further research. Of greatest importance is identification of strategies to maximize the social benefit of economic production.

As with indicators, the scenarios considered in this study demonstrate important implications of land use but are by no means conclusive. Future analyses should expand the suite of scenarios to consider a broader range of land use options. For example, the effect of alternative agriculture strategies such as agroforestry could be explored and additional protected area options could be evaluated. Reforestation scenarios are of special interest due to the new Law of Environmental Services that, as discussed previously, will require property owners to implement reforestation or else face financial penalties. Additional stressors could also be added to the analysis. For example, the spread of invasive species has the potential to alter both agricultural and natural areas. The expanding network of roads is another example of a stressor that, although not included in the simulations, is altering ecosystems in the Cuenca. Lack of information prevented the inclusion of these stressors in the scenario analysis. For example, temporal data to calculate the rate of expansion in roads was not available, and the effect of roads to indicators such as wildlife is poorly understood.

Scenario analysis should not be considered a project with a fixed completion date, but rather a tool that the local community can apply to continue to explore the implications of land use options in the region. In addition to broadening the suite of scenarios that are investigated, ongoing application of ALCES is necessary to incorporate new information about indicators as knowledge improves. Fundacion Moises Bertoni scientists have been trained to use ALCES so that scenario analysis can continue.

Acknowledgements

The scenario analysis would not have been possible without the contributions of numerous individuals. Fudacion Moises Bertoni staff provided a wealth of knowledge as well great hospitality. In particular, I thank Laura Rodríguez, René Palacios, Yan Speranza, Evaristo Mendoza, and Luis Antúnez. I also thank farmers and park wardens

from the Cuenca for the insight they provided. Several individuals from the Alberta Research Council contributed to the project. In particular I thank Ross Mitchell, Laureen Elgert, and Jennifer Karpyshyn for their guidance and assistance, and Daiyuan Pan for completing geospatial analyses. I thank Robin Naidoo for providing land cover and conversion probability data and Rolf Derpsch for providing information about conservation agriculture in the region. I thank Brad Stelfox for his assistance with ALCES.

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Appendix 1: Agriculture Net-income trajectories

	Net income ($/ha)		
Year	Maize	Cotton	Cassava
0 (conversion)	84.8	1006.5	353.9
1	77.8	967.2	353.9
2	70.9	927.9	353.9
3	63.9	888.6	353.9
4	57.0	849.3	353.9
5	50.1	810.0	353.9
6	43.1	770.1	353.9
7	36.2	731.4	353.9
8	29.2	692.1	353.9
9	22.3	652.8	353.9
10	15.3	613.5	353.9
11	8.4	574.2	353.9
12	1.4	534.9	353.9
13	0 (abandoned)	495.6	353.9
14		456.3	353.9
15		417.0	353.9
16		377.7	353.9
17		338.4	353.9
18		299.1	353.9
19		259.8	353.9
20		220.5	353.9
21		181.2	353.9
22		141.9	353.9
23		102.6	353.9
24		63.3	353.9
25		24.0	353.9
26		0 (abandoned)	353.9

Table 1: Net income trajectory for maize, cotton, and cassava under conventional agriculture.

	Net income ($/ha)		
Year	Maize	Cotton	Cassava
0 (conversion)	255	447	552.3
1	268.9	447	552.3
2	282.8	447	552.3
3	296.7	447	552.3
4	310.6	447	552.3
5+	324.5	447	552.3

Table 2: Net income trajectories for cotton, maize, and cassava under conservation agriculture.

	Net Income ($/ha)	
Year	Tillage	No tillage
0	45.3	67.9
1	33.0	117.3
2	27.1	135.3
3	21.1	161.5
4	15.1	246.3
5	9.1	194.3
6	3.1	159.4
7	-2.9 (abandoned)	310.0
8	Abandoned	227.6
9+	Abandoned	240.2

Table 3: Average trend in net income generated by 135 ha soybean farms in San Pedro and Itapua.

Appendix 2: Probability of species encounters

Calculated using multiple logistic regression coefficients from Hill et al. (2003)

Species	**Probability of species encounter**						
	High forest	Low forest	Riparian forest	Liana forest	Bamboo understory	Large bamboo	Meadow swamp
Nine-banded armadillo	0.461	0.461	0.461	0.377	0.502	0.461	0.358
Capuchin monkey	0.723	0.723	0.723	0.723	0.723	0.802	0.579
Coatimundi	0.126	0.126	0.126	0.126	0.126	0.126	0.126
Collared peccary	0.719	0.719	0.719	0.719	0.719	0.719	0.719
Rusty-margined Guan	0.546	0.662	0.546	0.711	0.546	0.546	0.546
Paca	0.945	0.945	0.945	0.809	0.945	0.867	0.945
Tegu lizard	0.701	0.701	0.701	0.701	0.701	0.701	0.701
Red brocket deer	0.741	0.370	0.741	0.741	0.660	0.811	0.741
White-lipped peccary	0.741	0.741	0.741	0.741	0.741	0.741	0.741
Tapir	0.423	0.703	0.423	0.593	0.584	0.615	0.423