The impact of sustainable intensification on landscapes and livelihoods (SILL)in Zambia

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Through action research and development partnerships, Africa RISING will create opportunities for smallholder farm households to move out of hunger and poverty through sustainably intensified farming systems that improve food, nutrition, and income security, particularly for women and children, and conserve or enhance the natural resource base.

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Abbreviations

BCEF	Biomass Conversion Expansion Factor
CA	Conservation Agriculture
CIDA	Canadian International Development Agency
CF	Conservation Farming
CFU	Conservation Farming Unit
CIFOR	Center for International Forestry Research
COMACO	Community Markets for Conservation
CSO	Central Statistical Office
ECZ	Environmental Council of Zambia
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FRA	Forest Resources Assessment
FSP	Forest Support Project
GART	Golden Valley Agricultural Research Trust
GDP	Gross Domestic Product
GMA	Game Management Area
GRZ	Government of the Republic of Zambia
ICRAF	World Agroforestry Centre
IAPRI	Indaba Agricultural Policy Research Institute
IIED	International Institute for Environment and Development
IFAD	International Fund for Agricultural Development
JCTR	Jesuit Center for Theological Reflection
JICA	Japan International Cooperation Agency
ILUA	Integrated Land Use Assessment
MAC	Maximum Allowable Cut
MACO	Ministry of Agriculture and Cooperatives
MSU	Michigan State University
NP	National Park
REDD	Reducing Emissions from Deforestation and Forest Degradation
REDD+	Reducing Emissions from Deforestation and Forest Degradation, Conservation, Enhancement of Carbon Stocks and Sustainable Management of Forests
SDM	System Dynamics Modeling
SILL	Impact of Sustainable Intensification on Landscapes and Livelihoods
SLCS	South Luangwa Conservation Society
UNEP	United Nations Environment Programme
USAID	United States Agency for International Development
WB	World Bank
WWF	World Wide Fund for Nature
ZAWA	Zambia Wildlife Authority
ZESCO	Zambia Electricity Supply Corporation
ZFAP	Zambia Forestry Action Plan
ZFD	Zambia Forestry Department

1. Introduction

The global population is projected to increase to nine billion by 2050, and rising demand for food to feed a rapidly growing population is promoting both land conversion for agricultural expansion and unsustainable agricultural intensification, reducing the resilience of rural households dependent on agriculture and the landscapes that support their livelihoods. The UN World Population Prospects estimates the current population in Africa at 925 million (United Nations, 2012). Africa has the highest population growth rates in the world, and the continent's population is projected to double to 1.8 billion in the next four decades (Cleland, 2013). Human population in Africa has more than tripled in the second half of the twentieth century, from 230 million in 1950 to 811 million in 2010 (FAO, 2011; World Bank, 2013). Population growth is expected to remain high on the continent, with a projected increase of another half billion people by 2030, and eventually reaching more than 2 billion people by 2050. The population of Zambia is projected to more than double in the next generation, rising from 16.2 million in 2015 to 44 million in 2050. Population growth, along with rising incomes, will have significant implications for agriculture, energy, and food demand.

Agriculture faces tremendous pressure to supply the growing and also wealthier population with more food, fiber and fuel (Tilman et al., 2011). Agriculture is a major source of emissions of greenhouse gases, user of water and land resources, and the greatest threat to biodiversity (Achard et al., 2002; Rudel, 2009; Foley et al., 2011). Rapid population growth in Africa is driving a growing demand for food. Food security remains a long-standing problem throughout sub-Saharan Africa, which has lagged behind other regions of the world in terms of agricultural productivity. Although Africa has previously produced enough to meet its own food needs, the population has grown at a faster rate than agricultural productivity over the past century, which has exacerbated problems related to hunger, malnutrition, and food insecurity (Sanchez, 2002; Sanchez and Swaminathan, 2005; Hazell and Wood, 2008; World Bank, 2008a; Todaro and Smith 2009). The continent has reverted from being a net exporter of agricultural commodities to being a net importer of the same during the last three decades (Rakotoarisoa et al., 2012). In particular, sub-Saharan Africa (SSA) has the greatest level of agricultural imports, which have been increasing at an exponential rate (FAO, 2011; Livingston et al., 2011). The Africa Progress Panel (2014) estimates that African countries spent US$35 billion on food imports (excluding fish) in 2011.

Most African farmers are smallholders and they face an array of challenges related to agricultural productivity, including costly fertilizer inputs, low marginal returns, and declining agricultural productivity (Giller et al., 1997). Intensification of agriculture has been cited as a plausible pathway to food security for poor households in Africa and elsewhere (Angelsen and Kaimowitz, 2001; Snapp et al. 2010; Tittonell et al., 2012; Vanlauwe et al., 2014), but inputs such as fertilizer are often costly and only marginally profitable for the rural poor (Morris et al. 2007; Tittonell and Giller, 2013).

High demand for food coupled with input-intensive conventional agricultural intensification practices that are increasingly common in Africa are potentially leading to agricultural land degradation, land conversion, and exacerbating climate change. National policies often promote input-driven intensification, which strives to increase agricultural output through increased use of inputs. Input-driven agricultural systems will experience an increase in yields and productivity, but this increase is often unsustainable. Unsustainable agricultural practices render the soil unproductive (Giller et al., 1997) and contribute to land degradation (Barbier, 1997; Sanchez et al., 1997; Symeonakis, 2007). Once land becomes degraded and scarce farmers put more pressure on protected areas and marginal hillsides (Headey and Jayne, 2014). Unsustainable practices can also force smallholders to acquire new fertile lands (often marginal forests) through land conversion or extensification. The transition to intensive agriculture may also reduce total soil carbon stocks, and increase emission of greenhouse gases such as carbon dioxide, methane, and nitrous oxide thus exacerbating climate change. The net effect of unsustainable agriculture intensification is a reduction in the food security and resiliency of agricultural households and the landscapes on which their livelihoods depend.

Sustainable intensification interventions on the other hand have the potential to mitigate the impact of agriculture on the landscape by intensifying agricultural production without increasing deforestation or the cultivation of more land and without reducing biodiversity (The Royal Society, 2009; Garnett et al., 2013; Keating et al., 2013). It is common think of intensification in terms of land as the key input and improving yields (productivity of the land) as the key objective. Pretty et al. (2011) define SI as (i) production of more food, feed, fiber, and/or fuel per unit of land, labor, and/or capital used; (ii) maintained and or improved natural resource base, including enhanced ecosystems services; and (iii) resilience to shocks and stresses, include climate change. SI practices could reduce agricultural encroachment into forests, preserving biodiversity and soil carbon stocks (Phalan et al., 2011; Pretty et al., 2011). Thus, SI approaches can potentially minimize environmental and long-term economic costs by increasing the efficiency of agricultural systems and by contributing to household and ecosystem resilience. SI approaches must include higher yields overall because most potentially useable land consists mainly of forests, wetlands, or grasslands, whose conversion would greatly increase emissions of greenhouse gases (Garnett et al., 2013), which is not sustainable in the long term. Increasing the land area in agriculture would also have significant environmental costs in terms of wildlife conservation, carbon storage, flood protection, recreation, and other ecosystem services.

There is much debate about the efficacy (and impacts) of sustainable intensification in practice. While nitrogen fixing agroforestry trees potentially increase farm productivity, especially under conditions of resource scarcity in regions such as sub-Saharan Africa, it is unclear if these systems indeed contribute to low-emission agricultural development (Rosenstock et al. 2013). Conservation agriculture has been vigorously promoted in countries like Zambia, though the impacts (in general) have been hotly debated (Baudron et al., 2012; Rockström et al., 2007). Where SI technologies like agroforestry and conservation agriculture are adopted, it is not clear if the benefits slow land degradation and compel farmers to stay on their farms, easing pressure on forests from land clearing, or if increased agricultural income from adoption of more profitable farming technologies causes farmers to expand their agricultural land holdings through deforestation or attracts new farmers to agriculture. These are some of the issues we explore in the review of literature that follows.

The objective of the Impact of Sustainable Intensification on Landscapes and Livelihoods (SILL) project is to advance the understanding of landscape-level implications of farm-level decisions related to sustainable intensification (SI) on forest and biodiversity conservation. The study sites for this project are Eastern Province and Lusaka Province in Zambia. Funding for this study has been provided by the United States Agency for International Development (USAID), which supports programming in sustainable intensification, climate change mitigation, and biodiversity conservation. This study has two objectives:

1. Provide an evidence base for the links between field and farm-scale sustainable intensification interventions and climate change mitigation and biodiversity conservation in Zambia; and

2. Provide recommendations to inform the design of integrated programs involving agriculture and environmental conservation for USAID programming in Zambia and other regions.

Sustainable intensification in this context focuses on two interventions: conservation agriculture (CA) and agroforestry (AF) and their linkages with forest protection and biodiversity conservation. These linkages were explored through three project activities:

- Review of literature on SI and impacts on the environment;
- Inventory of datasets that provided parameters used to quantify linkages; and
- Development of participatory systems dynamic models that conceptualize the drivers of change.

This study specifically examines these linkages by considering questions around this central theme. Is there evidence of encroachment into forested landscapes and protected areas? How do household, community, or landscape-level decisions impact or influence each other?

Traditional programming approaches most often use farm-level or household-level metrics to measure the outcomes of their programs (e.g., productivity) without considering the impact on the larger landscape. Likewise, community-level decisions on resource use at the landscape scale do not usually consider the impact of such decisions on smallholder farmers or rural livelihoods. This integrated study on the impacts of sustainable intensification on landscapes and livelihoods aims to advance understanding of these linkages in order to design effective integrated projects and interventions to promote sustainable development.

Integrated and multi-functional approaches are required in order to enhance rural food security in sub-Saharan Africa while minimizing habitat loss and greenhouse gas emissions. Such approaches should provide near-term livelihood benefits for smallholder farmers and longer-term environmental benefits for broader landscapes. Although this study is focused on two provinces in Zambia, it is expected that the results will have implications for the design and implementation of integrated conservation-development programs that build resilient households and conserve agricultural landscapes while also contributing to climate change mitigation and protection of biodiversity.

This report is organized as follows. Section Two begins with a review of the literature on landscape-level impacts of SI, and it is followed by a literature review on forest resources in Zambia and the drivers of deforestation. Section Three provides a description of the methods used during the project, including the participatory system dynamics modeling approach and the inventory of datasets. Section Four describes the system dynamics models that were developed at the national and provincial scales and the scenarios that were simulated to consider various policy interventions and hypothetical situations. Section Five presents a summary of conclusions from the project and provides some recommendations for integrated programing and metrics to address the linkages between farm-level decisions and environmental landscapes. The appendices include lists of tree species used in charcoal production and for timber production, and a table representing population trends in GMAs.

2. Agricultural-Environmental Linkages

In this review of literature on agricultural-environmental linkages, the landscape-level implications of SI are examined in terms of the impacts on forest and wildlife conservation. Land conversion from agricultural expansion contributes to both deforestation and habitat loss, so the implications for climate change mitigation and biodiversity conservation are often linked. Deforestation contributes to emissions of greenhouse gases that contribute to global climate change, so efforts to mitigate the effects of climate change have focused on forest conservation. Habitat impairment contributes to biodiversity losses, and the combined effect of habitat loss and illegal poaching in Zambia has taken a toll on wildlife populations and has prompted a focus on wildlife conservation. This study is motivated in part by the question of the potential contribution of SI to both agricultural development and efforts to conserve forest resources and biodiversity in Zambia.

This section begins with a review of the literature on sustainable intensification activities in Zambia, specifically conservation agriculture and agroforestry. We then examine the literature on forest resources in Zambia and forest cover estimates and deforestation rates, and continue with an examination of the various causes and drivers of deforestation and land use change. We then examine the literature on wildlife conservation issues in Zambia, including the threats to certain wildlife species due to poaching for ivory markets or the bushmeat trade. We then examine the evidence base for linkages between SI and biodiversity, specifically the impact of agricultural activities on wildlife. In summarizing these agricultural-environmental linkages, we conclude with evidence of the pressure on the environment from a growing population and a rising demand for energy and food, and we consider the drivers of deforestation and biodiversity losses that are attributed to charcoal production, rural fuel wood collection, the timber trade, and agricultural expansion.

A. Sustainable Intensification Activities in Zambia

Sustainable intensification in the context of this project focuses on interventions to promote two agricultural technologies: conservation agriculture and agroforestry. These are discussed below.

Conservation agriculture

Conservation agriculture (CA) is widely promoted as a solution to low productivity and soil degradation problems in sub-Saharan Africa (Giller et al., 2009; Mazvimavi, 2011). CA is designed to improve farm productivity by combining three principles: (1) minimum soil disturbance; (2) permanent organic soil cover; and (3) crop rotation and crop diversification (FAO, 2012; Andersson and D'Souza, 2014). CA is a set of practices that is a possible avenue for intensifying agriculture to improve food security amid heightened environmental challenges and population increase (Thierfelder and Wall, 2010; Friedrich et al., 2012; Verhulst et al., 2012; Corbeels et al., 2014). CA has also been highlighted as one of the major avenues for climate change adaptation and mitigation in agriculture within the Intergovernmental Panel on Climate Change (IPCC) and has been shown to help stabilize crop yields in variable rainfall areas (Corbeels et al., 2014).

Conservation agriculture has been widely promoted in Zambia since the mid-1990s. The CA technologies practiced in Zambia involve dry season land preparation using minimum tillage or no-till methods (e.g., ripping, planting basins and zero tillage), permanent soil cover through the retention of crop residues, crop rotations and agroforestry (Haggblade and Tembo, 2003; Baudron et al., 2007). The CA practices were initially promoted on the premise that they would improve crop yields because of the potential to rejuvenate soils. But more recently, these practices are increasingly seen as potential adaptation and mitigation options to climate variability (Jain, 2007; Nyanga et al., 2011; Arslan et al., 2014; Kuntashula et al., 2014; Ngoma et al., 2014b). Conservation farming (CF) is a package of CA practices that was introduced by the Conservation Farming Unit (CFU) and the Golden Valley Agricultural Research Trust (GART)[1] in Zambia. CF is primarily promoted to smallholders as hand hoe planting basins and ox-drawn ripping in moderate to low rainfall agro-ecological zones, where cotton is often an important cash crop.

Despite the promotion of CA over the last two decades, overall adoption rates are low and inconsistent in Zambia (Arslan et al., 2014; Ngoma et al., 2014b). Some reports, however, indicate that where promotion has been sustained, adoption levels are significantly higher (Kasanga and Daka, 2013). Many CA adoption studies focus on farmers' use of minimum tillage to prepare the land as the most essential and most observable indicator of adoption. Arslan et al. (2014) use panel data from the Supplemental Surveys (SS) to show that the percentage of households using minimum tillage as the main land preparation method on at least one plot increased from 8 percent in 2004 to 14 percent in 2008 nationwide. Ngoma et al. (2014b) used pooled cross sectional data from Crop Forecast Surveys to show that less than 10 percent of farmers used minimum tillage between 2008 and 2012 in districts with a long CA history in the country. The same authors found that minimum tillage increased from 2.9 percent of farmers in 2008 to 7.4 percent in 2012 in Eastern Province. Another study found similarly low but increasing levels of adoption among cotton farmers, with a high of 15 percent in Eastern Province in 2012 (Grabowski et al., 2015). Conservation agriculture could perform better than conventional agriculture only if farmers applied fertilizer or compost. With current costs and prices, conservation agriculture is unprofitable except on small plots for farmers with low opportunity cost of household labour. These findings suggest that conservation agriculture can improve maize yields but capital and labour constraints limit adoption to small plots in the absence of free or subsidized inputs.

Haggblade and Tembo (2003) estimated that in the 2001-02 season, 20,000-60,000 smallholder farmers used some form of CF, and adoption increased to 120,000 in 2002-03. More recent national estimates of CF adopters range from 160,000 to 270,000 smallholders. They estimated the area under some CF practices to be between 40,000 and 110,000 hectares. The area under CF practices has increased in recent years through the promotion of integrating *Faidherbia albida* trees into annual crop fields under CA practices. CFU reports over 300,000 hectares under Evergreen agriculture planted under minimum tillage (Garrity et al., 2010). More recently the Crop Forecast Survey showed an estimated adoption rate of 0.41 and 0.19 ha per household per annum for ripping and basin planting in Zambia, respectively (Ngoma et al., 2015).

Corbeels et al. (2014) found limited expansion of CA of less than 1 percent of the national cropland (i.e. 35,000 hectares) under CA in Zambia and (Andersson and D'Souza, 2014) found that in many cases the adoption of CA is only partial. Only three percent of CA users practiced the two principles of minimum tillage and crop rotation in 2008 in Zambia (Arslan et al., 2014).

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[1] CFU and GART continue to play the leading roles in the extension of CA to smallholders. Major donors who support their work include the World Bank (WB), Canadian International Development Agency (CIDA), European Union (EU), and the Governments of Norway and Finland. Their efforts are supplemented by those of USAID (partnering with Catholic Relief Services, CARE, World Vision, and Land o'Lakes), FAO, IFAD, and the Ministry of Agriculture and Cooperatives (MACO) of the Government of the Republic of Zambia (GRZ).

Empirical evidence on the impact of CA on smallholder productivity is still mixed (Giller et al., 2009; Andersson and D'Souza 2014; Brouder and Gomez-Mcpherson, 2014). There is limited empirical evidence that CA raises crop yields under smallholder farm conditions since most of the evidence comes from experimental on-station or on-farm trials that involve high input use, weeding, and other skills for crop management (Giller, 2009; Andersson and Giller, 2012). A recent study by Ngoma et al. (2015) found that when tillage is done before the onset of the rains, rip tillage conferred average maize yield gains of 577 kg per ha to 821 kg per ha over conventional plow tillage nation-wide. These increments translate to a relative increase of 42.3 percent over the gains from conventional plow in certain agroecological zones. Additionally, the yield benefits of CA may be realized in the medium to long term (Giller et al., 2009; Thierfelder et al., 2013). Moreover, higher yields combined with high input use do not necessarily raise total farm productivity and income.

CA yield advantages are attributed to improved water infiltration, soil moisture, soil porosity, soil organic matter, and crop management (Thierfelder et al., 2013; Corbeels et al., 2014). Similarly, basins planting increased maize yield of 191 kg/ha over conventional hand-hoe tillage across the country. When tillage is done after the onset of the rains, rip tillage confers no yield gain over conventional plow tillage, whereas basins planting yielded a yield decrement of 168-179 kg/ha from conventional hand-hoe tillage. The positive yield benefits confirm similar trends in the region, with negative to neutral yield advantages in the short-term (Thierfelder et al., 2013; Brouder and Gomez-McPherson, 2014; Corbeels et al., 2014). One important benefit of CA is that it may provide higher yields than conventional agriculture during severe drought years although evidence for this is mostly anecdotal. Currently most poor farmers use basins and average plot sizes are 0.46 ha for all cotton farmers in Eastern Province (Grabowski et al., 2015) although less for poorer farmers. Maize yields under CA are also reportedly more resilient during drought years even when there is complete failure on conventional plots but there is no empirical evidence of this. Another alleged attribute of CA is its potential to bring degraded land back into production, especially for compacted and nutrient depleted soils although we were unable to find empirical studies documenting this feature. There is evidence that CA is associated with long-term soil improvement but from research was conducted on research stations (Thierfelder et al., 2015).

CA is increasingly considered a potentially effective strategy to address low agricultural productivity, while enhancing the capacity of smallholders to mitigate and adapt to climate change effects (Corbeels et al., 2014; Giller et al., 2011). CA practices have been actively promoted at a national scale by the Zambian government and international donors with a target of achieving 40 percent adoption by 2016 (GRZ National Agricultural Investment Plan, 2013) but there is little empirical evidence these efforts are paying off. Because there is no evidence base to demonstrate that conservation agriculture improves yields, there is little evidence of a linkage between CA and biodiversity outcomes. But even if yields can be marginally improved, it can be argued that such a small change is unlikely to affect biodiversity outcomes. From this review, the only evidence base to demonstrate that conservation agriculture improves yields is derived from the benefits of early planting (Ngoma et al., 2015).

Agroforestry

Agroforestry is the deliberate integration of trees and woody shrubs into farms and productive landscapes in order to diversify and increase production, while promoting social, economic and environmental benefits for land users (Leaky, 1996; Garrity et al., 2006). Agroforestry can improve the resilience of agro-ecosystems (Ajayi et al., 2011) by enhancing nutrient supply through biological nitrogen (N2) fixation and nutrient cycling (Barnes and Fagg, 2003), improving soil structure and water infiltration (Chirwa et al., 2007), serving as an erosion control measure (Albrecht and Kandji, 2003) and increasing the efficiency of resource capture and use (Kohli et al., 2008). Agroforestry systems can take a range of forms and provide an array of benefits including fertilizer trees for land regeneration, soil health and food security; fruit trees for nutrition and income; fodder trees that improve smallholder livestock production; timber and fuel wood trees for shelter and energy (Garrity, 2004). Given its multiple uses and benefits, agroforestry is widely considered pertinent to achieving sustained growth in agricultural production of smallholder systems (Garrity et al., 2010), particularly in maize-based cropping systems on depleted soils in Southern Africa (Kwesiga and Coe, 1994). Agroforestry is also considered as an alternative land use strategy that offers solutions to land and forest degradation and to biodiversity loss as well through diversification (Oke and Odebiyi, 2007).

There is a growing recognition of the role of agroforestry in biodiversity conservation (Cassman et al., 2005). The primary agroforestry systems include tree crop-based agroforests (Michon and de Foresta, 1999; Mafongoya et al., 2006; Akinnifesi et al., 2008; Sileshi et al., 2008; Garrity et al., 2010). Agroforestry systems can serve as secondary habitat for certain species and curb the conversion rates of natural habitat (see Schroth et al., 2004a; 2004b) as well as create a more benign and permeable 'matrix' between habitat remnants compared with less tree-dominant land uses. Depending upon the tree species and their management, their incorporation into crop fields and agricultural landscapes may contribute to maintaining vegetative soil cover year-round (Boffa, 1999); enhanced suppression of insect pests and weeds (Sileshi et al., 2006); and effectively conserve both above- and below-ground biodiversity (Scherr and McNeeley, 2008).

Agroforestry systems have been promoted for over two decades in most southern African countries but few quantitative studies on the adoption of individual tree species exist. In the Zambezi Basin Agroforestry Project[2], two-thirds of the 400,000 smallholder farmers reportedly adopted fertilizer tree systems[3] by the end of the project (Schuller et al., 2005; Ajayi et al., 2011). Adoption studies in Malawi and Zambia show an increasing adoption of intercropping of fertilizer trees on farms, including *Faidherbia albida* (Bunderson, 2012). Keil et al. (2005) calculated an adoption rate of 75 percent or almost 50,000 farmers among those who initially tested fertilizer trees in Zambia. Moreover, the average land per household under agroforestry increased from 0.07 hectares in the mid-1990s to 0.2 hectares in 2003 (Akinnifesi et al., 2010).

Innovations using agroforestry species have emerged as a result of agroforestry adoption. Farmers' innovations, such as the use of bare-rooted rather than bagged saplings, planting more than one tree species, and changes to the planting time, have encouraged the adoption of improved fallows by around 70,000 farmers in Eastern Zambia in 2010 (ICRAF, 2013). Approximately 70 percent of farmers who initially tested the fertilizer tree system adopted it. CFU has been promoting the integration of *Faidherbia* with conservation farming practices in Zambia and estimates more than 160,000 Zambian farmers have adopted (Shitumbanuma, 2012), covering an area of approximately 300,000 hectares (ICRAF, 2013). Garrity et al. (2010) estimated that around 200,000 households have tested fertilizer trees through Malawi's Agroforestry Food Security Program.

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[2] The project was implemented in five countries, namely Malawi, Mozambique, Tanzania, Zambia, and Zimbabwe.

[3] Fertlilizer tree systems refer to the following nitrogen-fixing trees and shrub species: *Acacia* spp., *Cajanus cajan*, *Calliandra caolthyrsus*, *Gliricidia sepium*, *Leucaena* spp., *Sesbania sesban*, *Tephrosia candida* and *Tephrosia vogelii* (Kwesiga and Coe, 1994; Akinnifesi et al., 2008).

Several factors influence the adoption or disadoption of agroforestry among smallholders in southern Africa. Ajayi et al. (2007a) classified the key factors into four broad categories: a) factors related to institutions and policies, including land and tree tenures; farm inputs and output prices and fertilizer subsidies; b) attributes of specific technology such as species specific management pattern, the time lag between costs and benefits for biomass and crop yields; c) household and farm characteristics (e.g., resource endowment, size of household, household labor availability; and per capita landholding); and d) location specific biophysical and climatic conditions (e.g., rainfall patterns and variability, soil type and fertility levels) that support tree establishment. A recent study by Place et al. (2012) found that many important policy constraints hinder wider adoption of agroforestry among smallholder farmers in developing countries, both at formulation and implementation levels. Agroforestry can be knowledge-intensive rather than cash-intensive so the sustained adoption by the rural poor requires rural advisory services (Place et al., 2002; Ajayi et al., 2005; Kwesiga et al., 2005).

Agroforestry systems also hold the key to provide rural fuelwood and other household wood requirements on farm (Akinnifesi et al., 2008; Winterbottom et al., 2013). Kwesiga et al. (1999) confirmed the provision of fuelwood from *Sesbania sesban* fallows in Eastern Zambia. Moreover, Govere (2002) found that improved fallows provide 11 percent of firewood consumed by rural households in Eastern Zambia. Among these improved fallows, *Gliricidia specium* is promoted by NGOs like COMACO for both soil fertility replenishment and firewood provision. Trimmings of branches of *Gliricidia sepium* provide firewood to women (Winterbottom et al., 2013). The volume of firewood from these agroforestry systems, however, varies between species. For example, tree species of *Leucaena leucocephala* supply wood at a rate of 20-60 m^3 per hectare per year (Winterbottom et al., 2013). Assuming an average density of 750 kg per m^3 (UNEP, 2015) and an average land of 0.47 ha per household under agroforestry system (ICRAF, 2013), *Leucaena* could produce at least 7,050 kg of fuelwood per year. This quantity of fuelwood should be enough to meet the fuelwood requirement of small rural households of about five members. The on-farm fuelwood provision is of particular benefit to women and children in rural Zambia by reducing their burden of walking long distances to collect firewood due to local scarcity (Neufeldt et al., 2015).

Variants of agroforestry have been combined with other practices in the development of integrated interventions in some countries. For example, Orr et al. (2015) examined the adoption of an integrated food-energy system in southern Malawi, which combined improved cookstoves designed to reduce demand for fuelwood with a thick-stemmed variety of pigeonpea, which was found to increase both food supply and supply of fuelwood from pigeonpea stems. Adoption of the improved cookstove was found to be higher among households that were better off and where women had greater control over decision-making. However, adoption of the integrated system was not associated with reduced demand for fuelwood from forests or reduced frequency of collection, perhaps because adopter households may consume more fuelwood because they are better off. Pigeonpea cultivation did increase food supply for adopter households. Consequently, the integrated food-energy system had mixed results, improving food supply but not necessarily reducing demand for fuelwood (Orr et al., 2015).

B. Forest and Wildlife Resources in Zambia

The indigenous forests in Zambia are rich in biodiversity. They are home to approximately 5,500 species of flowering plants, 146 species of ferns, and 88 species of mosses (GRZ, 1997). The vegetation is dominated by miombo woodlands, which are primarily characterized by savannah grasslands and tree species that provide wood that is suitable for charcoal production (Chidumayo, 1990; Mulenga et al., 2014). These forest resources are sources of energy, construction materials and wild food that support local livelihoods, especially in rural areas in Zambia (Jumbe et al., 2008; Kalinda and Bwalya, 2014; Zulu and Richardson, 2013). A wide range of non-timber forest products are collected from these resources, including raw materials for construction, thatching, crafting, wild foods (e.g., bushmeat, edible caterpillars, fruits, honey, mushrooms, and tubers), as well as medicinal products. Non-timber forest products do not only provide important sources of food and materials for subsistence purpose, they also significantly contribute to rural household income (Bwalya, 2011; Mulenga et al., 2014). Environmental benefits of forests include erosion control and sediment retention, agricultural support services, and carbon storage and sequestration (Basson et al., 2009; CSO, 2012a, 2013; GRZ, 2011).

Zambia has a large standing stock of wood biomass, estimated at about 2,941 million m^3, with a re-growth rate of 568 million per annum (Kalinda et al., 2008; Mukosha and Siampale, 2009; Bwalya, 2011). Over 2,700 million m^3, or 94.7 percent of this national stock are in the forests, with over 70 percent in the miombo forests (Mukosha and Siampale, 2009). Kalinda et al. (2008) further estimated the total above- and below-ground biomass at 3,300.9 million m^3 and the maximum allowable cut (MAC) at 17.5 million m^3 per year.

Estimates of forest cover and deforestation

Zambia is endowed with abundant forest resources compared to most of the other miombo countries (World Bank, 2008b). The nationwide Integrated Land Use Assessment (ILUA 2005-2008) inventory classified the national vegetation into three categories: miombo woodlands; deciduous forests[4]; and shrub thickets (Kalinda et al., 2008). With its relatively small population, the available forested land per capita is estimated at about 3.5 hectares, compared to 1.7 in Zimbabwe, 1.6 in Mozambique, and 0.2 in Malawi (World Bank, 2008b). There is, however, a high level of variation in the estimates of national forest cover in Zambia. Recent estimates of the remaining forest cover range between 39 million (CSO, 2013) and 50 million hectares (ha) (Kalinda et al., 2008). Based on these estimates, forests cover represents between 52 and 65 percent of the total land area, estimated at 75.3 million hectares (World Bank, 2008b).

Discrepancies in forest cover estimates are reported both between years and within the same year. For example, forest cover was estimated to be as low as 30.1 million ha in 2004 (Pohjonen, 2004), and at over 49 million in 2010 (FAO, 2010). For the year 2005, FAO (2005) and ZFD and FAO (2008a; 2008b) reported different estimates at 42.4 million and 46.6 million ha, respectively. Kamelarczyk and Smith-Hall (2014) identified 19 studies, which provide estimates of forest cover between 1962 and 2010 in Zambia. The lack of consistency in forest cover estimates is also reflected in the estimates of the rate of deforestation. Despite the lack of consensus on the extent of forest cover in Zambia, estimates indicate a decline by 39 percent between the mid-1970s and 2013 (FAO, 2006; CSO, 2013). The forest cover declined from 81 percent in the mid-1970s to possibly as low as 42 percent by 2013 (World Bank, 2008b).

[4] From the ILUA, deciduous forests include *Baikiaea* forests, Munga, Mopane, and Kalahari woodlands; Riparian, swamp, Parinari, Itigi, and Lake Basin chipya forests constitute Evergreen forests; termitary associated bushes are the Shrub thickets; and all areas with little tree coverage, comprising riverlines, plains, dambos are either grasslands or wooded grasslands (See Kalinda et al., 2008; p 22).

Deforestation rates in Zambia are alarming (Stringer et al., 2012), and estimates range widely (Kamelarczyk and Smith-Hall, 2014). While Zambia does have a relatively high amount of forested land per capita, annual deforestation rates are ten times higher than most other miombo countries (Dewees et al., 2011). Kamelarczyk (2009) found rates estimated at 298,000 ha per annum in 1996 and at 444,800 ha per annum in 2006. However, similar to forest cover estimates there is a wide range in deforestation rates reported in the literature. Kamelarczyk and Smith-Hall (2014) identified 15 deforestation estimates from 1962 to 2010 with estimates of annual deforestation ranging from about 40,000 ha per year (FAO, 1981) to 980,000 ha per year (Pohjonen, 2004). Recent estimates of deforestation rates range from 167,000 ha or 0.3 percent per annum (FAO, 2010) to 540,000 ha per year (Pohjonen, 2004). But the most recent and widely reported estimates were on the order of 250,000 ha per year to 445,800 ha per year (FAO, 2005). The ILUA reported estimates of 250,000 to 300,000 ha per year, or a relative annual decline of the total forest cover of 0.62 percent (Kalinda et al., 2008; Mukosha and Siampale, 2009).

A critical analysis of existing data suggested that the average rate of forest cover loss from 1969 to 2006 has been about 298,000 ha per year (Kamelarczyk, 2009). Some inventories have even set deforestation rates at above 850, 000 ha per year (FAO 2001, in GRZ, 2006a; Jumbe et al., 2008). This higher rate, reported by the Forest Resource Assessment 2000 (FAO, 2001), is widely cited and used to rank Zambia among the most rapidly deforesting countries in Africa (Henry et al., 2011). For instance, according to Aongola et al. (2009), Zambia has the second highest per capita deforestation rate in Africa and the fifth highest in the world. The FAO (2010) suggests that while there may be a slight forest decrease, the real problem is forest degradation, which is much harder to measure. The aforementioned estimates of forest cover reflect deforestation from land clearing for agricultural expansion and/or human settlements. These rates may not, however, reflect forest degradation attributed to logging and the collection of fuel wood, which affect the remaining forest areas. Evidence suggests that significant degradation from these activities is taking place within the remaining forest area (Kalinda et al., 2008; Gumbo et al., 2013).

Kamelarczyk (2009) suggested a need to verify current rates of deforestation to overcome the high variability of reported estimates. The most recent estimates by UN-REDD and the ILUA provided a rate of deforestation of 300,000 ha per year, or an annual relative decline of 0.33 percent (ZFD and FAO, 2008b). We calculated an average deforestation rate of about 303,000 ha per annum using the most recent estimates between 2005 and 2010. This estimate is similar to that reported in the World Bank development indicators for Zambia (World Bank, 2012).

Drivers of deforestation

Forest loss is mainly driven by wood extraction, agricultural expansion, infrastructure development, and illegal timber extraction (GRZ, 2006b; Kalinda et al., 2008; Chundama, 2009; Vinya et al., 2011). Wood extraction frequently leads to deforestation because thinning for firewood or charcoal facilitates subsequent clearing of land for agriculture (UN-REDD, 2010). Charcoal production has been identified as the largest driver of wood extraction and the primary cause of forest degradation in Zambia (Clarke and Shackleton, 2007, UNDP, 2010; Kutsch et al., 2011; Vinya et al., 2011). Land clearing for agricultural expansion is considered the second largest driver of forest loss in Zambia (Vinya et al., 2011). Timber and wood products are also extracted from forests in Zambia for housing and road construction (UNEP, 2015). Mining and forest fires also contribute to forest loss and degradation in Zambia (Mwitwa and Makano, 2012).

Charcoal is the primary source of energy for cooking among urban households in Zambia, as well as throughout much of sub-Saharan Africa (Zulu and Richardson, 2013). About 5.8 million tons of wood biomass was used to produce charcoal in 2008 (Kalinda et al., 2008). Charcoal production accounted for one-quarter of the annual deforestation rate nationwide (Kalinda et al., 2008), and estimates from Chongwe district in the 1990s and Nyimba district in the 2000s suggest a rate of about 30 percent of total deforestation (Chidumayo, 2001; COMACO, 2010). Based on a conservative estimate of the rate of deforestation of 300,000 ha per year (World Bank, 2008b), the rate of deforestation attributed to charcoal production is estimated at approximately 75,000 ha per year. The rate of deforestation attributed to charcoal varies between provinces and districts (Vinya et al., 2011), but it tends to be localized and closer to urban areas for easier access to markets. Charcoal production may not always lead to deforestation however, because of the ability of many miombo tree species to regenerate quickly after harvesting (Chidumayo, 2004a; Chidumayo and Murunda, 2010).

Firewood is the main source of energy in rural areas, with over 90 percent of rural households depending on forests to meet their energy requirements (Bwalya, 2011). Kalinda et al. (2008) found that firewood is often collected from dead wood and rarely cut down from live trees and estimated the total dead wood biomass to be around 434 million tons in Zambia. If the existing dead wood meets the rural energy demand, the impact of firewood harvesting on forest loss is considered to be negligible (Kalinda et al., 2008). However, evidence shows that commercial harvesting of firewood in some parts of the country leads to high rates of extraction and hence to forest degradation (World Bank, 2008b).

UNEP (2015) found a spatial mismatch between the supply and harvesting of timber, leading to over-exploitation in certain areas. There is a debate about the sustainability of timber extraction in Zambia. Puustjärvi et al. (2005) argued that current levels of extraction are sustainable based on the MAC. Kalinda et al. (2008) estimate the MAC at 17.5 million m^3 per year or about 0.6 percent of the estimated standing stock. Others, such as Gumbo et al. (2013) found that illegal logging in particular is a major cause of deforestation. Kalinda et al. (2008) observed unsustainable harvesting of timber from both protected and open customary forests and evidence of degradation in the remaining forests in Zambia. Over-exploitation has particularly been detrimental to high-value timber species such as the Zambezi teak (*Baikiaea plurijuga*) in Sesheke of Western Province (JICA, 1996, in Gumbo et al., 2013; UNEP, 2000) and Kiaat (*Pterocarpus angolensis*) that resulted to a temporary ban of its export by the forestry department in 2005. These specific cases may contribute to the perception that logging is contributing significantly to forest loss on a wider scale and elevate concerns about deforestation and forest degradation in Zambia (UNDP, 2010).

Estimates of forest cover loss from agricultural expansion are as high as 90 percent in Zambia (Vinya et al., 2011), though such estimates are inconsistent with other estimates of deforestation such as that for charcoal production (Chidumayo, 2001; COMACO, 2010). Forest loss from agricultural expansion is often associated with smallholder agriculture and unsustainable farming practices such as shifting cultivation (Chidumayo, 1987; UN-REDD, 2010, Bwalya, 2011; Campbell et al., 2011). There is also evidence of widespread encroachment into forest reserves and conversion of forest into agriculture. An estimated 78 percent of the total forested land has been openly accessed as a result of poor forest management (Chidumayo and Aongola, 1998; Chidumayo et al., 2001; Bwalya, 2011). About 400 ha of land have been lost to encroachment in Lower Zambezi National Park as a result of new settlements (Kalumba, 1997, in Vinya et al., 2011). More recently, commercial farming has increased pressure on land in some areas of the country (Vinya et al., 2011). There is also increasing pressure as a result of migration. Commercial farmers have migrated to Zambia (most of them in Central Province) from Zimbabwe because of the political and economic instability in their country of origin (UN-REDD, 2010; Vinya et al., 2011).

Infrastructure development is also considered to be a driver of forest loss in Zambia although to a much lesser extent than wood extraction and agricultural expansion (Vinya et al., 2011). In particular, urbanization and industrialization have contributed to deforestation through land clearing for building construction and road infrastructure. Urban settlements and infrastructure are estimated to be expanding at a rate of 3.2 percent per year (ECZ, 2008; Campbell et al., 2011; Gumbo et al., 2013). The amount of deforestation attributed to infrastructure development is likely to increase as demand for housing, roads, energy, water and sanitation, irrigation and communication increase (Foster and Dominguez, 2010).

Mining-related deforestation is mostly caused by land clearing for the construction of mining infrastructure (Chidumayo, 1989; Gumbo et al., 2013) including tunnel support (Mwitwa and Makano, 2012). The economy of Zambia is heavily dependent on copper mining, which comprises over 70 percent of export earnings (CSO, 2012b). The mining sector contributed 12.9 percent to the gross domestic product of Zambia in 2010 (CSO, 2014). The mining sector in Zambia is growing at about 9 percent per year in the last decade and is expected to continue to grow (GRZ, 2011; UNEP, 2015) and to continue to contribute to forest cover loss (Vinya et al., 2011). Vinya et al. (2011) found that the Kalumbila mining concession was developed at the expense of more than 7,000 ha of forest cover in North-Western Province. Miners also use charcoal to meet their energy needs (Chidumayo, 1989; Gumbo et al., 2013).

The two main drivers of forest loss, charcoal production and agricultural expansion have been found to work in tandem in Zambia (UNEP, 2015). Wood extraction for charcoal is often followed by land clearing for agriculture because of the low marginal labor requirements to convert the land after the trees have been removed. Alternatively, the wood biomass from land cleared for agriculture may be used to produce charcoal, generating income for the smallholder farmer. There is however, variation in the levels and causes of deforestation across the country and a poor understanding of causality (Abbot and Homewood, 1999, Chidumayo, 2004b; Fischer and Shively, 2007; Dewees et al., 2011).

Charcoal production and use

Charcoal is the dominant source of cooking fuel in urban areas of sub-Saharan Africa, used by over 80 percent of urban households (Zulu and Richardson, 2013). In Zambia, between 50 and 70 percent of urban low and middle-income households reportedly depend on charcoal for their cooking energy (Kalinda et al., 2008; Bwalya, 2011; Atteridge et al., 2013). There is a wide range of estimates of average monthly consumption of charcoal per household in Lusaka. Estimates range from 49 kg (TechnoShare Associates, 2007) to 58 kg (Chidumayo et al. 2001) to 90 kg (UNDP, 2010) to 114 kg per household per month (Atteridge et al., 2013) for an average household (average household size was 5.26 people in 2014). The widespread use of charcoal is driven by limited access to and high cost of electricity (Kalinda et al., 2008). Few rural households, however, use charcoal as a cooking source because of the loss in efficiency and additional labor required to produce charcoal from wood.

Estimates of the total charcoal use are summarized in Table 1. In 2003, total charcoal consumption in Zambia was estimated to be 1.63 million tons (FSP, 2004). Assuming an average density of 750 kg per m^3 (UNEP, 2015), about 9.04 million m^3 of wood was required to produce this quantity of charcoal (UNEP, 2015). If we apply the estimated rate of 79.37 tons of wood produced per ha of intact forests by Kalinda et al. (2008), over 113,000 ha of forest area would have been used to meet national charcoal demand in 2003. Later estimates were reported at 0.95 million tons (CSO, 2013) and as high as 2.57 million tons (Puustjärvi et al., 2005) for the year 2004. Kalinda et al. (2008) estimated the total charcoal consumption in 2008 to be around 1.39 million tons of charcoal from 7.73 million m^3 of forest. The most recent estimate of charcoal consumption by UNEP (2015) was about 1.25 million tons or the equivalent of 6.97 million m^3 of wood in 2010. By applying the forest area/charcoal production rate calculated by Kalinda et al. (2008), national charcoal consumption in 2010 contributed to the clearing of about 88,000 ha of secondary forests. CSO (2013) reported an additional 48,000 tons of charcoal used in agriculture, industry and mining in 2010, translating into over 266,000 m^3 of wood extracted from 3,350 ha of forest area. Furthermore, Gumbo et al. (2013) estimated legal and illegal charcoal export to be around 3,278 tons per year requiring over 18,000 m^3 of wood biomass and contributing to the clearing of about 230 ha of forest. Charcoal demand is likely to increase in Zambia with its high urban population growth rate of 4.25 percent per year (Arnold et al., 2006; Hofstad et al., 2009; Zulu and Richardson, 2013; World Bank, 2014).

Stable urban demand for charcoal, ease of access to forest resources (partly due to poor enforcement of regulations), and low initial investment costs attract large numbers of people to engage in the production and sale of charcoal. Small-scale charcoal production typically involves cutting big trees into smaller logs and burning them in an earthen kiln, and it is primarily the work of men and older boys in rural villages (Zulu and Richardson, 2013). Commercial production of charcoal involves a greater use of mechanization and employs wage laborers, which is an important source of household income in rural areas.

Table 1: Existing estimates of firewood and charcoal consumption in Zambia

Source	Firewood (million m^3)	[a]Wood for charcoal (million m^3)	Total wood fuel (million m^3)	Charcoal (million tons)	[b]Charcoal (million bags)
PFAP (1997)			19.40		
FPS (2004)	9.49	9.04	18.53	1.63	49.32
Puustjärvi et al./SACOR (2005)	2.34	14.24	16.58	2.57	77.70
Puustjärvi et al./SACOR (2005) Annex 3	16.94	0.54	17.48	0.10	2.93
CSO 2013 (Department of Energy)	12.33	6.31	18.64	0.95	28.70
FAO Yearbook 2004			11.57		
Ng'andwe et al. (2006)			8.80		
Kalinda et al. (2008)	1.76	7.73	9.49	1.39	42.18
Kalinda's adjusted estimates (2010 census data)	2.37	5.64	8.01	1.01	30.73
UNEP (2015) estimates	12.48	6.97	19.45	1.25	38.00

[a] Assuming an average density of 750 Kg per m^3.
[b] A "50 Kg" bag of charcoal weighs on average 33 Kg (Gumbo et al., 2013).
Source: Adapted from UNEP, 2015, p.32.

Clarke and Shackleton (2007) found that charcoal producers practiced selective harvesting of preferred tree species in areas with abundant wood biomass and low demand for charcoal, and clear-cutting harvesting of woodland in areas of high demand for charcoal. The former contributes to forest degradation, whereas the latter directly leads to deforestation. For example, Chidumayo (1991; 1993) attributed the removal of more than 50 percent of woody biomass in miombo woodlands for charcoal production in Zambia. A variety of tree species are selectively cut for charcoal production, with preferred species in miombo ecosystems being of the genera *Brachystegia*, *Julbernadia*, and *Isoberlinia* (Chidumayo, 1993; Dewees et al., 2011). The selection is determined by the density of the wood, and tends to be associated with slower growing trees. Gumbo et al. (2013) found that preferred species are becoming scarcer which has resulted in the use of traditionally protected species such as fruit and medicinal tree species for charcoal production. More than 20 tree species are used in charcoal production (see Appendix 1 for a list of preferred species).

Charcoal is a major source of income and livelihoods, a significant contributor to national energy balances, and a potential renewable energy source capable of powering significant economic growth (Arnold et al., 2006; Sepp, 2010; Zulu and Richardson, 2013). Charcoal production and marketing are lucrative activities for both rural and urban households, and it is also a coping strategy for the rural poor with limited livelihood options. A lack of alternative livelihoods makes the charcoal production business more attractive to young and unemployed people (World Bank, 2009). Demand for transportation and marketing of charcoal creates jobs in both rural and urban areas (Mwitwa and Makano, 2012; Gumbo et al., 2013; Zulu and Richardson, 2013). In 2004 charcoal production created about 145,831 full-time jobs in Zambia (Puustjärvi et al., 2005; CSO, 2013) with another 4,500 people involved in transportation, marketing and distribution of charcoal (Jumbe et al., 2002). Chidumayo (2001) reported that about 9,000 households were engaged in charcoal production in Chongwe district alone in 2002. More recent national estimates have charcoal production at more than 500,000 jobs (Ng'andwe et al., 2006; Mwitwa and Makano, 2012). The value of the charcoal industry was estimated to be US$211 million, with an added value of US$ 126.5 million, or 2.2 percent of Zambia's gross domestic product (GPD) in 2004 (Puustjärvi et al., 2005, in UNEP, 2015).

Alternatives sources of energy are not widely available in Zambia. Electricity is a substitute for charcoal for some urban households, but regular supply disruption and high costs keep adoption relatively low. The supply is limited to only one-fifth of the population across the country (Foster and Dominguez, 2010) since many households are unable to afford the high connection fee, which is about US$ 140 (Haanyika, 2008). In 2010 the Energy Regulation Board authorized the Zambia Electricity Supply Corporation (ZESCO) to raise residential tariffs by 41 percent, which caused many users to reduce electricity consumption (Atteridge et al., 2013). Few of the alternative sources of cooking fuel such as kerosene or liquid petroleum gas are used in Zambia. This may be due to a combination of strong preferences for charcoal and relatively low prices. Small 10-watt off-grid solar systems are available in Lusaka for about US$ 240 (Atteridge et al., 2013) although there is limited information on adoption.

One reason for low adoption of alternative energy sources is the high fixed costs of adoption. Charcoal stoves also known as *mbaulas* cost about US$ 2, have a typical lifespan of about two years (TechnoShare Associates, 2007). The cost of charcoal for a family of five is approximately 270 Zambian Kwacha or US$ 42 per month for two 90 kg bags (JCTR, 2014). More fuel-efficient cook stoves are available through various non-government organizations and have been promoted to households. Electric coil stoves reportedly cost about 65-170 Kwacha (approximately US$ 10-25) and an electric stove sells for about 1,200-1,800 Kwacha (US$ 185-200). The cost of electricity for a family of five is estimated to be about 370 Kwacha or US$ 57 per month (JCTR, 2014). Other energy alternatives include briquettes made with agricultural byproducts (e.g., produced from cotton residues, groundnut hulls, and saw dust) and a gel fuel processed from agricultural waste.

Firewood collection and use

In contrast to the numerous studies on charcoal production, there are few empirical studies on the contribution of firewood collection to deforestation in Zambia. Almost all rural households use firewood for cooking, but consumption of firewood is rarely recorded in national statistics. Kalumiana (1997) estimated that 95 percent of rural households used firewood in the Copperbelt Province in the mid-1990s and estimated per capita consumption at 1,025 kg per year in rural areas and 240 kg per year in urban areas. National estimates indicate that over 90 percent of rural households derive their household energy from forests and woodlands (Bwalya, 2011). Bwalya (2006) used the ILUA datasets to estimate household firewood consumption at 1,562 kg of dry firewood per year nationwide. These estimates are lower than those by rural households in Botswana, estimated at 4,820 kg per year (Chidumayo and Gumbo, 2010) but higher than estimates for eastern Tanzania, estimated at about 1,100 kg per year in the miombo woodlands (Luoga et al., 2000).

In 2003, total firewood consumption was estimated to be 7.12 million tons or 9.49 million m^3 of wood biomass or 119,500 ha of forested land[5] (Puustjärvi et al., 2005). Later studies provided estimates that ranged from 1.32 million tons or 1.76 million m^3 of wood biomass or 22,000 ha (Kalinda et al., 2008) to 9.25 million tons or 12.33 million m^3 or 154,970 ha of wood biomass per year (CSO, 2013). The latest estimate of total household firewood consumption was 9.36 million tons or 12.48 million m^3 of wood biomass or 157,000 ha (UNEP, 2015). Although difficult to quantify, an additional 1.2 million tons of firewood or 1.66 million m^3 of wood biomass was reported to have been used in agriculture, industry and mining in 2010 CSO (2013). This suggests an estimate of total firewood consumption of 10.56 million tons or 14.08 million m^3 of wood biomass used in 2010, extracted from over 177,000 ha of forest or woodland areas in Zambia.

Kalinda et al. (2008) found that firewood is collected from dead wood and rarely cut down from live trees, and there is limited information on deadwood and litter biomass in the different forest types in Zambia. Kamelarczyk (2009) used an IPCC biomass conversion expansion factor (BCEF) to estimate deadwood biomass (using ILUA data) and found a ratio of deadwood biomass to live wood biomass from 0.020 and 0.057. Kalinda et al. (2008) estimated the total dead wood biomass to be around 434 million tons or 2.5 percent of the dead wood stock in Zambia forests and woodlands. These suggest a negligible amount of deforestation can be attributed to firewood collection. However, evidence on commercial production of firewood in some parts of the country suggests over-exploitation and deforestation (World Bank, 2008b, Gumbo et al., 2013).

Unlike charcoal, firewood is not an important source of household income in Zambia, as most households collect the wood for their own use. Based on the estimate for 2004, the trade value for firewood was estimated at US$ 285 million, contributing 0.8 percent the Zambia's GDP in 2004 (Puustjärvi et al., 2005). In addition, firewood created over 6,000 full-time employments in 2004, or 4.2 percent of the wood fuels-related jobs.

Timber and wood products production and use

Timber and wood products consist of industrial timber[6] and non-industrial[7] wood products used for construction and manufacture of wood products for home consumption and industrial uses. Puustjärvi et al. (2005) used data from the mid-1990s to estimate the total household wood production at 647,848

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[5] By applying the forest area/charcoal production rate by calculated by Kalinda et al. (2008).

[6] Industrial timber is sold directly as unprocessed logs or processed as sawn wood, plywood, and veneer.

[7] Non-industrial wood refers to that which is produced locally, mainly for household purposes. It often makes up a larger proportion of wood and timber produced and utilized that does not enter the market but circulates locally, contributing to local economies (construction and building materials, small-scale informal industry, woodcarving and household utensils).

m^3 in 2004, contributing to US$ 18.6 million to the national economy. Logs for sawn timber accounted for three-fifths (400,356 m^3) of production, followed by other wood at 21 percent, and poles at less than 15 percent. These estimates exclude industrial timber produced for either local consumption or export. Ng'andwe et al. (2006) estimated timber production at 777,000 m^3 in 2006, with a gross value added of over US$ 17 million. Using the ILUA household survey data, Kalinda and Bwalya (2014) reported that 19 percent of the 1,680 households source their construction materials from the woodlands nationwide. Total wood used for five household structures (house, homestead and garden fences, livestock pens and granary) was estimated at 47.4 m^3 per household (Chidumayo and Gumbo, 2010). Non-industrial wood is widespread and often makes a larger proportion of wood and timber produced and used locally in construction.

Industrial wood consumption, including saw logs, transmission and fencing poles, was first estimated at 3.1 million m^3 in 1996 (ZFAP[8], 1997, in UNEP, 2015). Ng'andwe et al. (2006) estimated the total production of round wood from all forests types at 1.15 million m^3 in 2006. Their estimate was similar to the corresponding FAO statistic of 1.33 million m^3 in the same year (Puustjärvi et al., 2005). ZFAP (1997) reported an estimate three times higher than other studies, which possibly captures illegal and unreported logging. Much of this trade is in indigenous timber that is exported to neighboring countries such as the Democratic Republic of Congo, South Africa, Tanzania, and Zimbabwe as well as to China.

Forests in Zambia are generally characterized by low timber stocking rates, with the exception of significant pockets of high-value timber species in some areas of the country (ZFD and FAO, 2008a; 2008b). These include a few hardwoods (*Baikiaea plurijuga*, T*ectona grandis* and *Pterocarpus angolensis*), with stocking rates of 0.5 to 2.0 tons per hectare (GRZ, 1997, in Jumbe et al., 2008). (See Appendix 2 for a list of high-value timber species in Zambia.) Mukosha and Siampale (2009) used datasets from the ILUA inventory to estimate the total commercial timber volume at 340 million m^3 in forests, of which 75 percent is sourced from the semi-evergreen forests (Kalinda et al., 2008). This estimate of timber stock, however, rises to 365.8 million m^3 or 12.4 percent of stocks, if all land types are included in the analysis (Mukosha and Siampale et al., 2009). In 2013, a higher total commercial volume of 1,047 million m^3, or 35.4 percent of forests was estimated (FAO 2014, in UNEP, 2015). Based on recent production estimates, the timber sector contributed between US$ 35.9 million in 2004 (Puustjärvi et al., 2005) and US$40.5 million in 2006 (Ng'andwe et al., 2006) to the national economy. The latter estimate accounted for plantations, which contributed 8 percent of the timber value and 28 percent of the value added in manufacturing (Ng'andwe et al., 2006). Timber from indigenous forests accounted for a value added of US$24.45 million in 2006, with an average gross value added of US$ 29.2 per m^3 (UNEP, 2015). In 2003, household timber production was estimated to contribute US$ 32 million, or 0.3 percent to Zambia's GDP (Puustjärvi et al., 2005). The industrial round wood contributed over US$ 40 million to the national economy in 2006 (Ng'andwe et al., 2006). The same authors estimated the value added of wood export at US$ 7 million, or 0.1 percent of GDP in 2006. Later estimates by FAO (2014, in UNEP, 2015) reported the contribution of round wood production, wood processing and pulp and paper at 6.2 percent of GDP. Most recent estimates by UNEP (2015) reported an annual contribution of the US$ 21.5 million, with a total value added of US$ 32 million. The timber sector employs over 10,000 people (Jumbe et al., 2008).

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[8] Zambia developed the Zambia Forestry Action Plan (ZFAP) in 1997. The main components included: 1) identification of key issues affecting conservation and sustainable use of forest resources; 2) developing short- and medium-term National Action Programmes; and 3) increasing public awareness of conservation goals and wise use of forest resources.

The standing stock of commercially valuable timber species was estimated at 1.6 million m^3, or only 9.1 percent of the national MAC (Kalinda et al., 2008). The distribution of total standing stock and associated timber volume vary largely among the provinces (Kalinda et al., 2008) with 60 percent of the total standing stock in three provinces. Lusaka province had the least standing stock, estimated at 88. 8 million m^3, with only 0.5 million m^3 of maximum allowable cut per annum, which was estimated to supply wood fuel for domestic use, rather than timber. This non-existence of timber species in Lusaka province indicates that all the wood need is supplied from extraction in still relatively rich biomass areas. The estimate of standing stock for Eastern province is approximately 264.8 million m^3, or 9 percent of the total standing stock in Zambia and the total MAC was estimated at 1.6 million m^3 per annum. The province is relatively poor in valuable timber tree species, with only 0.1 million m^3 of MAC per year. The remainder was allocated to wood fuel production, supporting charcoal supply to Zambia urban areas as well as neighboring countries (Gumbo et al., 2013).

Agricultural expansion and encroachment

Rising rural populations are rapidly changing the agricultural landscape and urban centers of Africa (Jayne et al., 2014) and this is true of Zambia given the large projected population increase. Urbanization is also happening at a rapid rate and many urban households still depend on farming for some of their livelihood and food security. Urbanization not only raises the demand for existing agricultural land surrounding urban areas but also reduces the supply of arable land that is converted into the urban landscape. With rising land and food prices farming is becoming profitable for those with sufficient capital to acquire good agricultural land and afford modern inputs.

At the same time, crop yields remain very low in Zambia. Without intensification of agricultural production on existing agricultural land farm households will expand their farm size as the population grows. Current agricultural practices characterized by limited response to inorganic fertilizer as a result of constraints related to soil fertility and moisture are unsustainable particularly in densely populated areas (Jayne et al., 2014).

The protected area system in Zambia consists of 20 national parks (NPs) and 36 game management areas (GMAs), covering about 65,000 km^2 and 174,522 km^2, respectively (Simasiku et al., 2008). Together with several other protected area categories, these NPs and GMAs represent about two-fifths of the national total land area in Zambia (GRZ, 2010a). A map of NPs and GMAs in Zambia is presented in Figure 1. NPs are designed for the protection of wildlife, biodiversity and ecosystems and are used for tourism, but human settlement is not permitted (GRZ, 2010a). GMAs were established as buffer zones for national parks, with the primary role of sustaining wildlife use (NPWS, 1998; Simasiku et al., 2008; Namugala, 2009; Banyopahyay and Tembo, 2010; Richardson et al., 2012). Economic activities that are compatible with wildlife protection are allowed (Simasiku et al., 2008; Richardson et al., 2012) and trophy hunting is currently allowed although there have been bans on hunting big cats and elephants as recently as 2014. In addition to large proportions of land allocated to wildlife, Zambia has three transfrontier conservation areas (ZAWA, 2009, 2010; Watson et al., 2014).

Human encroachment poses a greater threat to the protected areas in Zambia than in most other African countries (Pfeifer et al., 2012). Migration to land abundant rural areas is increasing as road infrastructure grows but the lack of infrastructure is a binding constraint in Zambia (Chamberlin et al., 2014). Migration thus presents a threat to biodiversity in Zambia because many of the land abundant areas where migration is happening are GMAs. Human population growth rates were estimated at 2.49 ± 0.18 percent in GMAs (Zambia Central Statistical Office, 2011). The same study indicates that rates in

GMAs are higher than those outside of GMAs (2.31 ± 0.24 percent). According to our estimates based on the 2010 census data, over 200,000 people live in the 7 million hectares of protected areas in Eastern and Lusaka Provinces, and the population grew by at least 133 percent since 2000 (see table in Appendix 3). This is a slightly higher growth rate than the overall population in Eastern province but significantly lower than the overall population growth rate for Lusaka province. Almost 40 percent of GMAs comprised human-modified habitat, representing over half of the rates of disturbance outside of the protected area network (c.f. 71.2 percent) (Lindsey et al., 2014). By contrast, habitat loss in NPs is limited, averaging 2.9 percent.

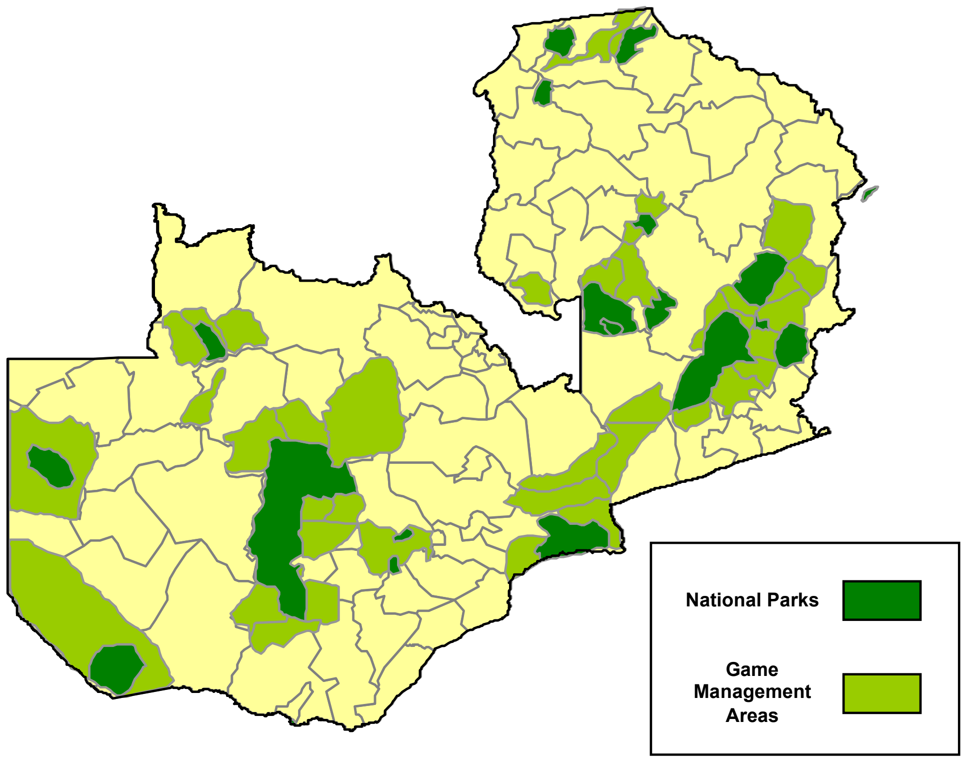

Figure 1: Map of national parks and game management areas in Zambia
Source: Food Security Group, Michigan State University

Encroachment into GMAs may be motivated in part by greater opportunities for wage labor and self-employment, but it also exacerbates human-wildlife conflicts, often leading to crop losses for smallholder farmers (Richardson et al., 2012). Analysis of data from a nationwide survey of rural households in Zambia shows that GMAs are positively associated with household income and crop damage from wildlife conflicts. Gains and damages were greatest among households in GMAs with greater wildlife diversity, with net gains relatively greater for wealthier households. Households in prime GMAs were more likely to participate in off-farm wage and self-employment compared to other rural households, but they were also more likely to suffer crop losses related to wildlife conflicts (Richardson et al., 2012). These conflicts undermine the objectives of SI, and highlight the incompatibility of agricultural development in areas that are managed for wildlife conservation, particularly when it involves the conservation of large mammals.

Encroachment into GMAs has been attributed to a range of activities including in-migration, agricultural expansion, poaching, uncontrolled fires, and mining, among others. Holden (2001) argues that deforestation in miombo woodlands has been driven by "agricultural and technological change" as well as subsequent population growth, migration, and ineffective government policies. Encroachment into forest reserves and charcoal making is responsible for considerable deforestation, and that agricultural expansion and wood fuel harvesting have the strongest correlation to deforestation and forest degradation (Kamelarczyk and Smith-Hall, 2014). Ngoma et al. (2014a) found that 24 percent of smallholder farmers expanded cultivated land into forests, clearing on average 0.65 ha per household with plans to clear on average 1.1 ha more land over the next five years. Simukonda (2011) found evidence of mining in Lukusuzi National Park in Eastern Province. Watson et al. (2014) found that development of national roads has facilitated human encroachment into GMAs at a rate of 2 km per year.

There is no effective mechanism to restrict human development in GMAs (e.g., Lindsey et al., 2014) and this has resulted in increasing human encroachment and high levels of illegal hunting and habitat loss in many of the GMAs (Lewis and Phiri, 1998; Simasiku et al., 2008; Becker et al., 2013; Lindsey et al., 2014). The loss of habitat in GMAs is also exacerbated by resource extraction for charcoal and timber as well as mining (Chemonics International Inc., 2011; Simukonda, 2011; Vinya et al., 2011). The rate of habitat loss attributed to encroachment was estimated at 0.69 percent per year in GMAs, higher than that outside protected areas (0.51 percent per year) and only 0.05 percent per year in protected areas (Watson et al., 2014). Based on trends in Eastern Province, Watson et al. (2014) estimate that about 82 hectares of habitat are lost per hour in GMAs across the country.

Rates of encroachment and habitat loss vary among GMAs from 1 to 99 percent in 2007 (ZAWA, 2008). The Bilili Springs GMA in Southern Province exemplifies the highest intensity of encroachment, from a rate of 3.7 percent to 99 percent over the last 40 years (Watson et al., 2014). Similarly, 84 percent of Namwala GMA (Southern Province) and 54 percent of Mumbwa GMA (Central Province) have been converted for settlement and economic activities (Watson et al., 2014). By contrast, only 134 km^2 or 1 percent have been converted in Lunga Luswishi GMA (North-Western Province), 10 km^2 per annum (or 0.76 percent) in Sandwe GMA (Eastern Province), and 33 km^2 per year (or 0.75 percent) in Lupande GMA (ZAWA, 2011). The rate of encroachment grew from 4 km^2 per year before 1985 to 29 km^2 per year in 2010, representing an average increase of 0.37 percent per annum, nationally.

Encroachment undermines the buffer zone role of GMAs for national parks, jeopardizing the ecological connectivity among protected areas and the concept of trans-frontier conservation areas (Watson et al., 2014). Moreover, encroachment constrains the economic performance of GMAs, impeding the contribution of sustainable wildlife usage to household livelihoods and welfare in rural areas (Richardson et al., 2012). Changes to the classification system for GMAs by the Zambia Wildlife Authority (ZAWA) and an increase in the number of depleted or secondary GMAs from 16 to 24 between 1997 and 2008 illustrates the impact of encroachment on GMAs (Simasiku, 2011).

From consultations with stakeholders, several issues emerged as drivers of encroachment and degradation in rural Zambia. The problem appears to be primarily driven by the growing demand for agricultural land of a rapidly growing rural population. Land allocation policies are poorly understood, in part due to the existence of a dual tenure system of state and customary land that has conflicting objectives regarding land use. Furthermore, the fact that all forest and wildlife resources were vested in the State leaves rural communities with little authority or agency regarding natural resource management. The scale of encroachment and degradation could perhaps be better understood through improved monitoring of forest cover, and better enforcement of forest laws and land use regulations.

Poaching, bushmeat trade and wildlife populations in Zambia

Zambia is endowed with abundant wildlife resources, including hippos, elephants, giraffes, lions, leopards, hyenas, crocodiles, and zebras, most of which can be found exclusively in national parks and GMAs. Many of these areas are under pressure from illegal hunting for bushmeat, ivory, and other animal products, and poaching has led to the decline or depletion of some species in particular areas. GMAs are classified as prime, secondary, specialized, or under-stocked. Prime areas are those GMAs in which wildlife species are diverse, abundant, and can sustain safari hunting. Secondary GMAs are those in which species are less abundant but that can still sustain limited hunting. Specialized GMAs are frequently found in wetland areas and are characterized by the presence of only a few species (usually antelope). In under-stocked GMAs, wildlife populations are sparse and hunting quotas are limited (Richardson et al., 2012).

There is scant evidence that sustainable intensification practices at the farm level will affect wildlife populations in Zambia. The causal relationships are complicated and depend on the species. In terms of wildlife species that are harvested for bushmeat the main pathway is that food and livelihood security theoretically lead to less poaching and less household consumption of bushmeat. Poaching affects wildlife populations directly but becomes a less relevant livelihood when people are economically secure. Tanzania and Zambia are the largest sources of, and transit countries for, Africa's illegal ivory (Wasser et al., 2008; Wasser et al.; 2009) but there is no data on the extent to which Zambians turn to poaching as a coping strategy. The difficulty of collecting this data is likely due to the illegality of the practice and the difficulty in collecting accurate data of this nature. Trophy hunting would also presumably decrease the loss of wildlife although this is regulated to an extent in Zambia. The Zambian government banned the hunting of big cats and elephants in 2013 and then removed the ban in 2015 after the wildlife populations allegedly recovered (GRZ, 2015).

Poaching appears to be reducing elephant populations in some parts of Africa while populations are rising in other parts of the continent (Wasser et al., 2007; Milliken et al., 2009; Styles, 2009; Wasser et al., 2010). The status and distribution of African elephants varies considerably across the species' range and sub-regions from small and fragmented populations in West Africa to vast tracts of elephant range in Central and Southern Africa. Southern Africa accounts for over one-third of the species' total range and is home to 58 percent (320,904 elephants) of the elephant populations in SSA (Blanc et al., 2007). It is unclear however, whether wildlife populations are rising or falling in Zambia for most species. Some believe there has been a steady rise in elephant poaching in recent years, providing ivory to China (Elephant poaching on the rise in Zambia, 2014). According to the Elephant Database (2013), the population of Elephants has decreased dramatically since the 1970s (population of about 200,000) to less than 30,000 elephants in Zambia today. Some elephant populations "may increase at a birth rate of 5.5 percent per year but most populations increase at lesser rates" (van Aarde, 2009) and natural elephant deaths are approximately 1-3 percent (Abel and Blakie, 1986). Elephant poaching was reported to be 5-7 percent in 1986 (Abel and Blakie, 1986) but current estimates of poaching deaths suggest a much lower rate of less than 1 percent (ex. 135 elephants were reportedly killed, which is about 0.6 percent of the reported 21,000 total population in 2012/13).

The lion is primarily threatened by loss of habitat, depleted prey populations, and human–wildlife conflict. In Zambia, lions now mainly occur in and around isolated protected areas, with approximately

156 lions estimated to be in SLNP. Trophy hunting is concentrated on the borders of protected areas, and wire snare poaching was common, particularly in the GMAs where lion hunting occurred (Becker et al., 2013). Estimated loss of adult males per year is an indicator of the potential hunting revenue and/or snaring mortality (8.41 to 9.91 adult males or about 6 percent). Becker et al. (2013) simulate adult males losses at various levels and find the average age of male lost is 9.1 and average age of all adult males to be 6. The average age of adult males lost each year is an indicator of potential trophy quality and the average age of adult males remaining in the population is an indicator of the quality of photographic tourism and potential future hunting revenue.

Given the complex dynamics of wildlife populations, the difficulty in accurately measuring them, along with other causal factors that affect poaching (such as hunting regulations) it is nearly impossible to model the linkages between agricultural activities and wildlife. We can look at how population dynamics are affected by loss of habitat, but this is also very challenging for the reasons discussed below.

Activities such as charcoal and timber harvesting that lead to forest loss and fragmentation threaten the survival of wildlife species, including bird, small herbivore, lion, and elephant species. But the relationship between wildlife populations and loss of habitat through deforestation is unclear. Some species are directly affected by a loss in forested area while other species might experience an increase in their population from a decrease in forested area. For example, woodland animals such as small and large woodland grazers, browsers and mixed feeders other than elephants, woodland birds, and woodland predators would likely be negatively impacted as they depend heavily on forest resources for survival. An increase in their population can also diminish forested area or compound human encroachment into forested areas. Browsers that affect woodlands in Zambia are impala, kudu, bushbuck, rhino, eland and duiker. On the other hand species such as hippos respond to an increase in grassland habitat as woodlands decline and this is also probably also true of other grassland animals (Abel and Blakie, 1986). Other grassland animals such as birds, small and large open grassland grazers, and grassland predators would likely increase their population size as forested area decreases (Baudron and Giller, 2014; Wright et al., 2012).

There is some documentation that encroachment patterns are threatening wildlife populations through habitat loss and poaching (Poulsen et al., 2009). A recent study by Lindsey et al. (2014) found that the biomass of large wild ungulates is lower in GMAs, averaging 212 ± 59 kg per km^2 less than in extensive game ranches (mean 2,424 ± 305 kg per km^2). The same is true for national parks where the estimate for the average biomass is only 0.32-fold (mean 791 ± 240 kg per km^2) that in game ranches. Similar trends were also reported for wildlife diversity in GMAs (mean 4.7 ± 0.58 species), followed by NPs (mean 7.2 ± 0.9 species), and in ranches (mean 11.1 ± 0.86 species). The higher biomass and diversity on private ranches has been attributed to resource availability for anti-poaching.

There are also important interactions between plants and various animal species such as seed predators and seed dispersers (Gautier-Hion et al., 1985; Jordano et al., 2003; Beaume et al., 2013). Maintaining these interactions promotes the health of species, populations, communities, habitats and ecosystems. Human pressure on wildlife, either through hunting or poaching, affects plant reproduction. These ecosystems are shaped by animal-plant interactions and many plant species depend on animals for seed dispersal (Denis, 2007; Forget et al., 2011). The loss of keystone species can impact the integrity of ecosystems and their services (Coppolillo et al., 2004). Wilkie and Carpenter (1999) showed that primary seed dispersers such as primates, bats, and birds are more vulnerable to human predation than seed predators (e.g., small rodents). Elephants (*Loxidonta africana*) play the role of seed dispersers of large trees in the Congo Basin forests (Blanc et al., 2007) and maintain the transition zone separating habitats of genetically distinct savannah and forest elephants (Blake et al., 2009; Lewis et al., 2009). Therefore, their local extirpation may affect plant diversity and the long-term viability of the system (Blanc et al., 2007; Blake et al., 2009).

Recent studies indicate that seed dispersal plays an important role in gene flow, colonization potential and plant migration in response to biodiversity maintenance (Schupp et al., 2010). Wildlife loss leads to changes in both structure and dynamics of habitats concerned (Terborgh et al., 2008). The depletion of wildlife species impacts plant communities in three different but inclusive ways. First, it reduces the amount and efficiency of seed dispersal for plant species whose seed dispersal agents are affected by hunting and/or poaching (Beckman and Muller-Landau, 2007; Wang et al., 2007). Moreover, hunting alters species composition of the seedling and sapling layers (Stoner et al., 2007). Mendoza and Dirzo (2007) found that selective hunting (of medium- and large-sized species) further leads to differential predation on seeds, with more predation on small seeds. A severe depletion in wildlife species in tropical forests might result in changes in biodiversity, species dominance, survival, demography, as well as spatial and genetic structure (Wright et al., 2007).

To further complicate matters, causality runs both ways between household well-being and wildlife population size. Nyirenda et al. (2011) found that elephants were the main culprits of crop depredation, accountable for 67.82 and 98.4 percent of total wet and dry farming crop raiding incidences respectively in Eastern Zambia (maize and cotton were the most affected crops). In the absence of their predators large generalist herbivores can have devastating effects on biodiversity and have transformed the natural habitats in game reserves where they are protected (Cumming et al. 1997). Given the difficulty of measuring wildlife populations and lack of empirical data on wildlife population size and rates of change in Zambia compounded by the complex dynamic feedback between population size and forest cover it is nearly impossible to predict the impact of changes in forest cover on a given species.

Agroforestry and wildlife

Agroforested areas with less intensive management and high canopy cover have high species richness and are more similar to neighboring forest reserves than intensively managed forests with open canopies. Several studies suggest that agroforestry systems have high species richness and resemble neighboring forest reserves in species composition if the forest land was recently converted (e.g., Beukema et al., 2007), the management was less intensive (e.g., Harvey and Villalobos, 2007); and the canopy cover of native trees was high (e.g., Bos et al., 2007).

Agroforestry systems are important for the protection of species and habitats outside formally protected areas. For example, Williams-Guillén et al. (2007) found that shade coffee plantations can serve as alternative habitats for howler monkeys (*Aouatta palliata*) in Nicaragua. Huang et al. (2002) found a positive impact of agroforestry on the biodiversity conservation of nature reserves in Tanzania (conservational functional groups are more diverse in agroforestry systems than in tree monoculture). Schroth and Harvey (2007) found that animal diversity is higher in cocoa agroforests with high plant diversity, structurally complex canopies, and abundant surrounding forest cover.

Agroforestry systems are also found to maintain heterogeneity at the habitat and landscape scales. A study on tree inventories conducted by Kindt et al. (2006) found significant differences in tree species composition and that the choice of shade trees left on plantations contributed to the overall heterogeneity of trees in this agroforestry landscape in Kenya. Similarly, Hemp (2006) found that traditional coffee-banana plantations maintain a multilayered vegetation structure similar to tropical montane forests. Trees in agroforestry landscapes reduce pressure on formally protected forest reserves. Furthermore, recent case studies demonstrate a positive relationship between functional group diversity and pollination or biological pest control in agroforestry systems (Klein et al., 2006; Klein et al., 2003). In general it has been found that high levels of species diversity facilitates overall ecosystem resilience in agroforestry systems (Elmqvist et al., 2003; Loreau et al., 2003; Ricketts et al., 2004; Naidoo and Adamowicz, 2005; Tscharntke et al., 2005).

C. Research Gaps

There is a detailed discussion of forest cover data in the literature review above. A wide variety of methods have been employed to measure forest cover and deforestation rates over time with little agreement. Further, it is difficult to model dynamics of land cover change in Zambia because there has been little spatial work done at the scale required to actually detect land use change. To effectively do this would require advanced analysis that involves triangulation between conflicting MODIS data and LandSat data, which has not been undertaken in Zambia and is beyond the scope of this project. Issues such as detecting forest degradation versus deforestation would need to be addressed. This data gap leaves questions about the extent to which deforestation or encroachment is happening in Zambia, which are central to the research questions posed.

Another data challenge is that it is difficult to quantify the adoption of sustainable intensification practices and CA in particular. CA is a set of agricultural practices that are packaged and promoted together so quantifying CA and CA adoption is complicated since someone can partially adopt CA by adopting some of the practices but not all, and on some part of their farm but not the entire farm. Ngoma et al. (2014b) found very low adoption rates because of this distinction (as opposed to CFU and others who do not make this distinction). Furthermore there is evidence that some aspects of CA (specifically mulching and minimum tillage) can serve as a buffer during periods of drought but this is not enough to say that CA unambiguously raises yields.

There is another gap in the literature in the sense that even if we were able to say that CA (or AF for that matter) raises yields, we would not be able to say whether it directly contributes to agricultural expansion or encroachment. It would take a very targeted study to make this connection between CA and expansion or extensification, and it is likely that a bumper harvest in one period would not directly translate to an expansion of agricultural land since there are other resource constraints to expansion (namely access to additional land and access to the required labor to cultivate a larger farm). One future area of research needed is a behavioral analysis of the implications of higher yields for agricultural expansion, and the linkages to SI practices.

There is theoretical and empirical evidence that promoting CA or AF in or near Game Management Areas has negative implications on wildlife biodiversity though increased human-wildlife conflict. However, the conflict is related to agriculture more generally, not conservation farming, per se. Agroforestry potentially alleviates some pressure on fuelwood harvesting but since this is such a small portion of overall wood energy, it would have little impact on reducing deforestation, relative to the contribution of charcoal production.

Relatively little is known about household behavior and decision-making in sub-Saharan Africa regarding urban household energy choices generally, and rural household participation in charcoal production and trade specifically. In addition, little is known about the institutional and forest management arrangements and the incentive mechanisms to promote and ensure sustainable charcoal production. Yet many African governments, including Zambia need evidence and practical arrangements for sustainable charcoal production and marketing. Thus, it is important to examine the possibilities for sustainable charcoal value chains from the production and institutional dimensions through transportation and marketing to the urban end-user, and the extent to which sustainable charcoal production and trading can potentially contribute to the slowing of deforestation as well as to net poverty alleviation and gender equity.

We also do not have evidence of a linkage between encroachment and wildlife populations because we do not know the extent to which wildlife populations in Zambia are affected by habitat loss. Partially this is because we do not have accurate population data (counts of various species) and partially because some animals (like elephants) can have an ambiguous relationship with habitat loss since they live in forests and grasslands and an increase in their population size has a negative feedback on habitat since they also destroy forest and cropland via depredation. There is evidence that encroachment into GMAs may exacerbate human-wildlife conflicts, often leading to crop losses for smallholder farmers (Richardson et al., 2012), but the drivers of in-migration into GMAs is poorly understood.

D. Summary of Agricultural-Environmental Linkages

Literature suggests that the main drivers of deforestation in Zambia are urban charcoal consumption and agricultural expansion, although less is known about the extent and impact of agricultural expansion compared with charcoal consumption. Reducing charcoal consumption would likely involve coordinated policies that address rural household dependence on charcoal as an income source in addition to policies that seek to reduce urban charcoal demand.

Despite the existing large stock of dead wood at a national level, evidence of over-exploitation in specific areas suggests the need to provide sustainable sources of energy to rural households in forest-depleted areas.

The high demand for specific tree species for timber production may lead to over-exploitation, resulting in species extinction, forest degradation, or even forest loss (Gumbo et al., 2013). A sustainable timber harvest from concessions and a selective harvest from natural forests will be necessary for a sustainable timber industry in Zambia. The diversity of people engaged in the timber sector, however, calls for tailoring the sustainable management approaches for forests and woodlands to suit their social, economic, and technological capacities across the country (Kokwe, 2004).

Both current and projected future land clearing is expected to increase pressure on forests. Deforestation can be reduced through a combination of demographic, energy, agriculture, and development policies (Wunder, 2000). Agriculture intensification, alternative energy sources and livelihood diversification strategies are all mentioned in the literature.

SI likely plays little role in land abandonment or expansion. A descriptive meta-analysis of independently published studies highlighted the fact that the abandonment of agricultural land is a phenomenon mostly driven by socio-economic factors such as immigration into areas where new economic opportunities are offered to rural people. Ecological drivers such as elevation and land mismanagement leading to soil erosion are of secondary importance (Benayas et al., 2007). This is also likely true of land expansion. Expansion appears to be more related to population growth and non-agricultural income generation opportunities than it is to unsustainable agricultural practices.

There appears to be a spatial incompatibility of promoting agriculture and biodiversity conservation within the same areas, particularly when conservation involves large mammals. Conservation organizations tend to promote low-input, improved agricultural practices (specifically the practices associated with conservation agriculture) and combined with the focus on conservation in protected areas (like GMAs in Zambia), this may exacerbate rather than mitigate conflicts between biodiversity conservation and agricultural production (Baudron and Giller, 2014).

3. Methods

In this discussion of the methods employed in the SILL project, we begin by describing the system dynamics-modeling tool that was used to conceptualize the linkages between agricultural development, forest protection, and wildlife conservation. We then review the participatory modeling approach that was used to examine causal loops, identify sources of data, and validate the structure of the models. We conclude this discussion of methods with a brief description of the inventory of datasets that were identified in the development of the models.

A. System Dynamics Modeling

System Dynamics Modeling (SDM) is a quantitative modeling tool that uses systems thinking to analyze the impact of feedback loops in complex and dynamic systems. SDM has been applied to the adoption and diffusion of new agricultural technologies (Dea Fitri et al., 2014), development (Saysel et al., 2002), global climate change (Sterman, 2011), forest fire management (Collins et al., 2013), and water resource management (Simonovic, 2002; Stave, 2003), among other topics.

SDM captures the essential temporal self-regulating feature of many systems. This means that feedbacks among the system components incrementally adjust the state of the system. A change in one part of the system affects another that then affects others with some delay. Some of these changes will eventually feedback to amplify or dampen the effect of the original change. SD modeling recognizes that changes do not occur in isolation and many systems do not respond instantaneously to these changes.

Interactions between the system components are represented with causal loops, which indicate the direction and polarity of the relationships. Two types of feedback exist: positive or reinforcing feedback (e.g., population growth is exponential because more people make more people), and negative or balancing (e.g., an increase in deforestation leads to less wood availability thus potentially decreasing deforestation). The result is often one of two outcomes: when a balancing feedback dominates there is an oscillating or static equilibrium. When a positive feedback occurs there is exponential change. To measure the dynamics of changing levels of stocks and rates of flows SD models solve a system of simultaneous differential equations.

The modeling reported in this document uses system dynamics because it is a flexible quantitative simulation tool based on logical flow where causation drives the model. We use quantitative data to parameterize the model collected from existing studies, gray literature, and in some cases rely on anecdotal evidence or best guesses when empirical data is not available. We allow patterns to emerge from the relationships between the stocks and flows and use these patterns to identify key levers of change that drive the system. We validate model assumptions through sensitivity analysis, participation of experts and stakeholders, comparison with other data, or a combination of these.

B. Participatory Modeling

Participatory modeling has grown in popularity in recent years as a tool to investigate environmental systems and generate consensus among stakeholders around environmental problems. System dynamics has been consistently used to incorporate stakeholder input into the model-building process and is easily adaptable to a participatory format (Van den Belt 2004). We use participatory system dynamics modeling as a framework to integrate stakeholders' knowledge with scientific analysis and develop a discussion among them about model dynamics. This approach is particularly useful for analyzing coupled human-environment systems with dynamic connections between social and ecological systems.

Participatory modeling is a broad field with a diverse set of methods, but generally the goal of constructing a model in collaboration with stakeholders is to elicit their views of a system and how it operates, and to use that information to inform the model. Interviews, textual analysis, focus group discussions, and group diagramming exercises are all techniques that are commonly used (Andersen and Richardson, 1998). We chose to use group diagramming and scenario-building exercises in combination with key stakeholder interviews to build the participatory system dynamics model for this project. We did this to garner a diverse set of views and knowledge on the system, while drawing on the specialized knowledge of stakeholders who were unable to attend the group meetings. The model-building process was iterative, meaning that it involved the modeling team returning to the stakeholders repeatedly for further model development and feedback on what had been done so far. Throughout the duration of the project, there were four interactions with stakeholders that informed subsequent model building (described below). The modelers used participant input to frame the model structure, the algorithms used in the model, and the scenarios run by the model.

In May 2014, two workshops were held, one in Mfuwe in Eastern Province and the other one in Lusaka with various partners in Zambia. Participants in the workshop included representatives from USAID Washington and Zambia Mission, CIFOR, ICRAF (Zambia/Malawi), WWF, Total Landcare, the Forestry Department, IAPRI, and MSU. In the Mfuwe workshop participants were grouped in their respective sectors (agriculture, the environment, and livelihoods) and asked to identify the drivers of the problems that their organizations work on related to agriculture and the environment. In Lusaka the participants were integrated across sectors but asked to do the same activity. The groups were all asked to look at the system and identify causal relationships. They then generated different causal loop diagrams, which were later integrated into one diagram. This diagram became the conceptual map used to start building the system dynamics model.

A third workshop was held in August 2014 in Chisamba, Zambia and focused on validating the assumptions and parameters used in the landscape-level model. A similar set of participants attended this workshop. Small group discussions were centered on the initial model behavior and dynamics and interventions, which would disrupt the model behavior. Discussion also focused on the assumptions and data underlying the model and scenarios or policies that might change the operation of the model. We usually attempted to validate stakeholder feedback with other data and literature before we incorporated it into the model. For example, during the workshop, participants expressed skepticism that rural fuelwood gathering was contributing significantly to deforestation (as the model then depicted), since most believed that rural Zambians collect deadwood to burn rather than cutting live trees. We consulted the literature and found that this was likely the case, and changed the model algorithm to reduce the amount of fuelwood required for the rural population.

The feedback from these workshops was used to develop two provincial level models for Eastern Province and Lusaka Provice. These models were validated in a fourth interaction with stakeholders via a conference on improving integration among agriculture, forestry, and land tenure, held in Lusaka in June 2015. This conference was attended by many of the same stakeholders who participated in the earlier workshops, and included presentations on the contributions of the stakeholder organizations to integrated programming to address agricultural-environmental linkages.

Throughout the participatory modeling process, the interventions discussed by participants focused on three broad categories: more efficient energy sources, energy governance (such as legalization and enforcement of charcoal production and marketing), and social programs targeting fuelwood consumption such as community-based natural resources management. Scenarios identified that needed further exploration ranged from human-environment conflict (climate change and carbon sequestration) to exogenous market scenarios (food price spikes, increase in value of tourism) to development led scenarios (land grabbing by foreign entities and road building). Some additional landscape level topics that arose were the prevalence of fire in grassland areas, illegal logging in Eastern Province, gender dimensions of fuelwood harvesting, and the contribution of migration and land access to extensification.

C. Inventory of datasets

We inventoried various datasets to quantitatively parameterize both provincial and national models. We empirically collected the data from existing studies, gray literature, and national statistics in Zambia whenever available and accessible. In some cases we, however, used proxies from either miombo countries or from other countries in sub-Saharan Africa. We summarized the parameters of each of the three models in a supplemental spreadsheet that is submitted separately and is not included in this report. For each model, we classified the parameters by module: landscape; population; and agriculture and drought. We further grouped these parameters into different categories per module. For example, the landscape module has 15 categories, whereas the population module has only three categories. These categories include land use and land cover, wood recovery from land clearing, miombo clearing and regrowth, deadwood availability in forests and woodlands, charcoal production and use, agroforestry and fuelwood, population in GMAs, maize production and export, and drought and conservation agriculture among others. Key national studies include the ILUA study on forest cover and land use (Kalinga et al., 2008), Zambia 2010 census of population and housing (CSO, 2011), rural agricultural livelihoods survey (RALS) datasets by IAPRI (Tembo and Sitko, 2012), and estimates of adoption of agricultural technologies in Zambia (Ngoma et al., 2014b). FAOStat datasets, as well as CIFOR, ICRAF, GRZ, REDD programme, and World Bank studies as well. We derived specific parameters from publications such as rate of miombo regrowth by Chidumayo (1988), density of agroforestry plantation by Sileshi et al. (2007), and delay in agroforesty growth by Kwesiga et al. (1999). We used most of these rates to calculate either the parameters or their initial values in the model.

4. Models

In this section, we describe the system dynamics models that were developed at the national and provincial scales and the scenarios that were simulated to consider various policy interventions and hypothetical situations. We developed three separate models. The first is a national-level model that assesses the impact of the main drivers of deforestation at the landscape level. The second model focuses on Eastern province and encroachment into GMAs. The third model is of Lusaka province and focuses on urban energy consumption needs. Each of these models is discussed briefly and basic simulation results are presented.

A. National-Level Model of Deforestation

We first constructed a dynamic simulation model of land use change in Zambia to explore the drivers of deforestation. Model parameters are drawn from an exhaustive search of the literature and existing datasets. Consultations and a participatory modeling workshop with experts in the field were used to generate assumptions where they were required. The model is then used to examine the current and future role of the drivers of deforestation given population and urbanization trends in Zambia and to simulate possible interventions to curb deforestation.

The National Level Deforestation Model focuses on the interactions between biophysical aspects of forests and drivers of deforestation over time in Zambia. The model consists of three subsystems: landscape, population, agriculture and drought. Each of these modules represents stocks (boxed variables) and flows (arrows). At the core of the landscape module is a chain of different types of forests stocks disaggregated into three categories: miombo woodland, deciduous and evergreen forests, and grass and shrub land (see Figure 2a). The disaggregation follows the classification of forest types used in the recent Zambian Integrated Land Use Assessment (ILUA) with field data collection in 2005-2007 (ZFD and FAO, 2008a; 2008b). The flows represent rates of change in forest cover types at the following stages: (1) natural regrowth; (2) conversion into agricultural land; (3) deforestation for charcoal and firewood; (4) land/forest degradation, and (5) land abandonment. Each of the three forest cover types can be converted into agricultural land and can lead to degraded land over time with unsustainable farming practices. Degraded land is restored through fallowing to any of the three forest types. Miombo woodland and deciduous forests are degraded through selective cutting for charcoal production or industrial round wood. Deadwood and trimmings of branches are harvested for fuelwood, with a small proportion of households that cut down live tree in degraded natural forests (Gumbo et al., 2013).

In addition to the stock-flow structure, other exogenous causal relationships are represented with blue arrows. Such variables include urban charcoal demand, rural demand for firewood, efficiency of the charcoal kiln, deadwood biomass, number of people cutting live trees, biomass density, relative price of electricity, and electrification rate among others. The structure described above forms the core of the landscape module for all three models.

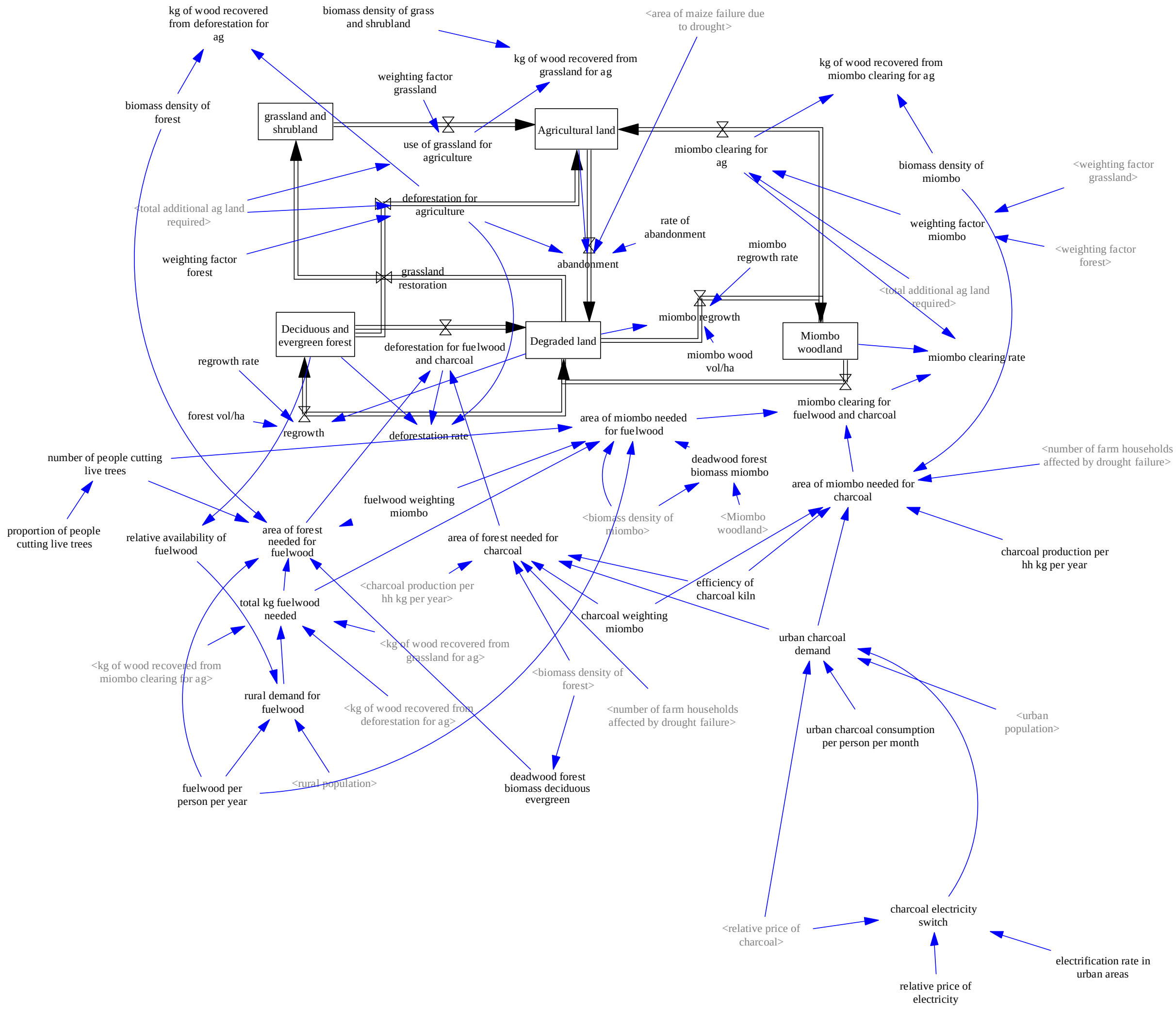

Figure 2a. National landscape module of deforestation in Zambia

National-level model parameterization

In the baseline run, total area for miombo woodland was set at 34.1 million hectares, that of the deciduous and evergreen forest at 15.8 million hectares, and that of grasslands and shrubs at 12.1 million hectares (Kalinda et al., 2008). We also considered two other land use types in the model: agricultural land and degraded land. The former had an initial value of 7.78 million ha and that for the latter was set at 7,000 ha (ZFD and FAO, 2008a; 2008b). Various flows (rates) link the four main stocks, including regrowth rates, land clearing and fallow rates, and deforestation rates. If undisturbed, miombo forest can regrow at 1.93 m^3per ha per year (Chidumayo, 1988).

Based on discussions with experts, we assumed that agricultural expansion prioritizes grasslands and shrublands, reflecting agricultural conversion preferences reported in the ILUA forest inventories. Similarly, charcoal production targets miombo woodlands (Gumbo et al., 2013). We used the mean per capita charcoal consumption in urban areas estimated by Nyembe (2011), which is 7.1 kg per month per person (Nyembe, 2011), though there is a range of estimates for this parameter. The initial per capita firewood consumption in rural areas was set at 1,025 kg per annum (Kalumiana, 1996). Finally, we set the initial value for the proportion of people who cut live wood in rural areas at 10 percent across the country, presumably because they live in areas with little or degraded natural forest (Gumbo et al., 2013). This assumption is supported by the fact that 2 percent of the total biomass stock is accessible deadwood, estimated at about 434 million tons in the ILUA study by Kalinda et al. (2008). Finally, we set the initial value for charcoal kiln efficiency at 25 percent (Schenkel et al., 1998) and that of electrification rates at 22 percent in urban areas versus 2.5 percent in rural areas (CSO, 2013). These key assumptions provided the basis for the baseline run.

B. Provincial-Level Models of Deforestation

We used insights gained from the national landscape model to develop provincial level models for Eastern and Lusaka Provinces. Each of these models is made up of the three core modules: landscape, population, and agriculture. Each landscape module incorporates an agroforestry component (Figure 2b). The agroforestry component mainly represents the effects of the adoption of agroforestry systems on household fuelwood demand in rural areas of Zambia. Variables considered in this component include average area under any agroforestry system, density of agroforestry plantation, average growth rates of agroforestry tree species, fuelwood yield, and growth delay of agroforestry species. Specific datasets were used to separately parameterize each of the modules of the provincial modules. The model dynamics are simulated over a 50-year period from 2010 to 2060. Main datasets are summarized in the section below.

Provincial-level model parameterization

Similar to the national model, we used data from the ILUA inventory by Kalinda et al. (2008) to parameterize the variables on land cover and land use types in Eastern and Lusaka Provinces. The extent of the land cover and use types differed between the two Provinces, reflecting the difference in the urbanization rate between them. For example, the total area for the three forest cover types considered in the landscape module for Eastern Province is over 5 million ha, or five-fold that for Lusaka Province. The initial values were set at 2.89 million ha for the total area for miombo woodland, 1.55 million ha for deciduous and evergreen forest, and 687,000 ha for grass and shrub land in Eastern Province. Similarly, parameters were set at 556,251 ha for miombo woodland, at 296,850 ha for deciduous and evergreen forest, and at 132,159 ha for grass and shrub land in Lusaka Province (Kalinda et al., 2008). The total agricultural land was also set at 1.37 million ha for Eastern Province and 659,800 ha for Lusaka Province. It is important to note that we modeled the stock of deadwood in forest and woodlands using an estimated ratio of deadwood and live biomass of 2 to 6 percent by Kamelarczyk (2009). We also included the export of charcoal produced in Eastern Province to other urban areas in Zambia and to neighboring countries like Malawi (Gumbo et al., 2013). This is one of the main differences between the Eastern and Lusaka Province models.

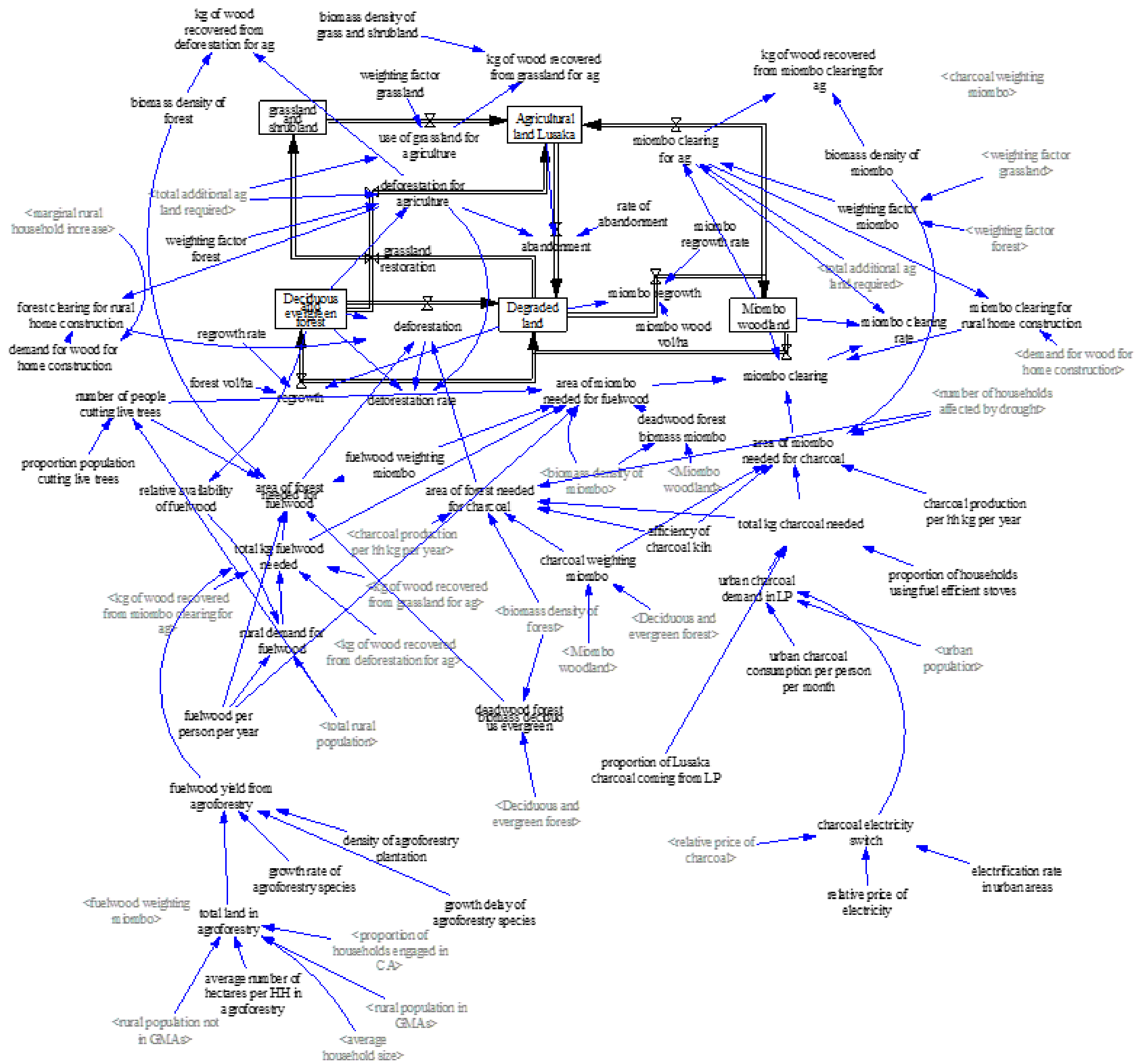

Figure 2b. Provincial landscape module of deforestation in Zambia

Population module

The population module is presented in Figure 3. Zambian population estimates were derived from the 2010 Zambian census (CSO, 2010). The population stock is further divided into urban and rural population for both the national and provincial level modules. Eastern is still rural, with an initial rural population set at 87.4 percent (CSO, 2010), versus only 15.5 percent for Lusaka Province. In addition, the population is growing faster in Lusaka, with an average growth of 4.6 percent per annum, than in Eastern Province (2.6 percent per year) (CSO, 2010). Each of them has dynamic significance because they drive the demand of charcoal in urban and firewood in rural settings, respectively (World Bank, 2015). The flow-rates are birth and death. The flow-rates drivers include marginal increases in both household and population and average household size as well. We further differentiated the rural populations living in GMAs versus those living in non-GMAs because the location drives both encroachment and the adoption of agroforestry practices by rural households. The total population in GMAs was estimated at 54,146, with an approximate growth rate of 0.17 percent per annum in 2010 (CSO, 2010) in Lusaka. The same estimates were set at 193,879 total people in GMAs and at 3.3 percent per annum for the growth rate in Eastern Province (CSO, 2010). We finally factored in the effects of immigration in the marginal increases in populations. For simplicity, the model does not explicitly depict the additional effects of the population aging, which affects death rates as well as farm sizes. Birth rates are modeled as declining over time, which reflects the available data from the Zambian census.

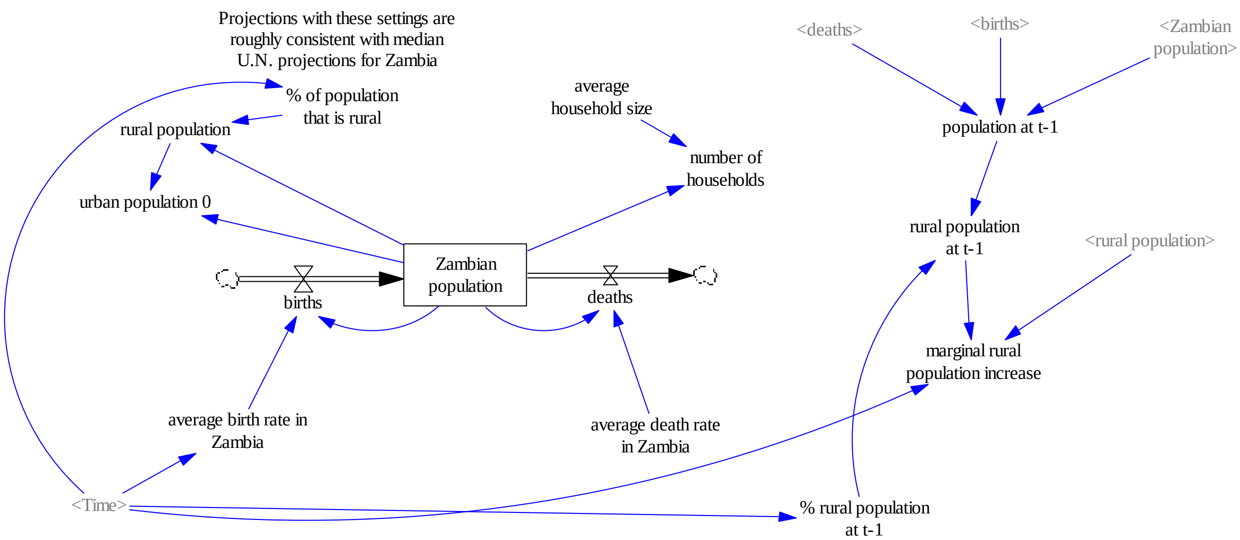

Figure 3. Population module

Agriculture and drought module

Figure 4 depicts the agriculture and drought effects module. This module is made up of variables allocating agricultural land to the main crops grown in Zambia and in each province. Maize, cassava, and groundnuts were considered in all three models. For simplicity, we modeled maize as the quantity of maize required for feeding the increasing population and for current export levels. The choice was guided by the importance of maize to the caloric intake of Zambians, which was estimated at 57 percent of Zambians' daily caloric consumption by FAOStat in 2014. The main drivers include the caloric intake of maize set at 1,000 Kcal per capita per day and the total calories contained in 1 Kg of maize set at 3,600 Kcal (FAO). Other drivers are the total population, the average land cleared by a new farmer, maize yield and an annual increase in maize yield (assuming average technological improvements), post-harvest losses, drought severity and the number of farming households affected by drought, and the proportion of households engaged in conservation agriculture as well. Each of these variables drive the total maize demand for food, the total area required to produce maize, as well as the total production of maize. Agricultural parameters were derived from the agricultural census data, compiled by Tembo and Sitko (2012). The annual average increase of maize yield is estimated at 93 kg per ha (Tembo and Sitko, 2012) and that of the average post-harvest loss for maize was initially set at 18 percent for Zambia (Rembold et al., 2011).

We assumed that a moderate drought occurs every 8 years and affects 40 percent of the total agricultural land, and a severe drought occurs every 40 years and affects 70 percent of the area under cultivation. Drought leads to yield loss of 30 to 70 percent (Thurlow et al., 2009). This loss is estimated at 30 percent on CA plots with drought year (Grabowski et al., 2015). Assuming an average area of 0.25 ha under CA per household in Eastern Province, this suggests 315 kg of maize produced from CA plot by adopters. It further implies that farmers who practice CA are less affected by drought than their fellow non-adopters. The total area under CA, therefore, influences the total maize production during drought years. A key assumption is the reliance of farmers on charcoal production as a coping strategy for the rural poor during drought years (Zulu and Richardson, 2013). This assumption provided an insight to simulate the effect of drought on forest cover types using the landscape model. For the sake of simplicity, we did not consider the price of maize and the effects of inputs on the productivity.

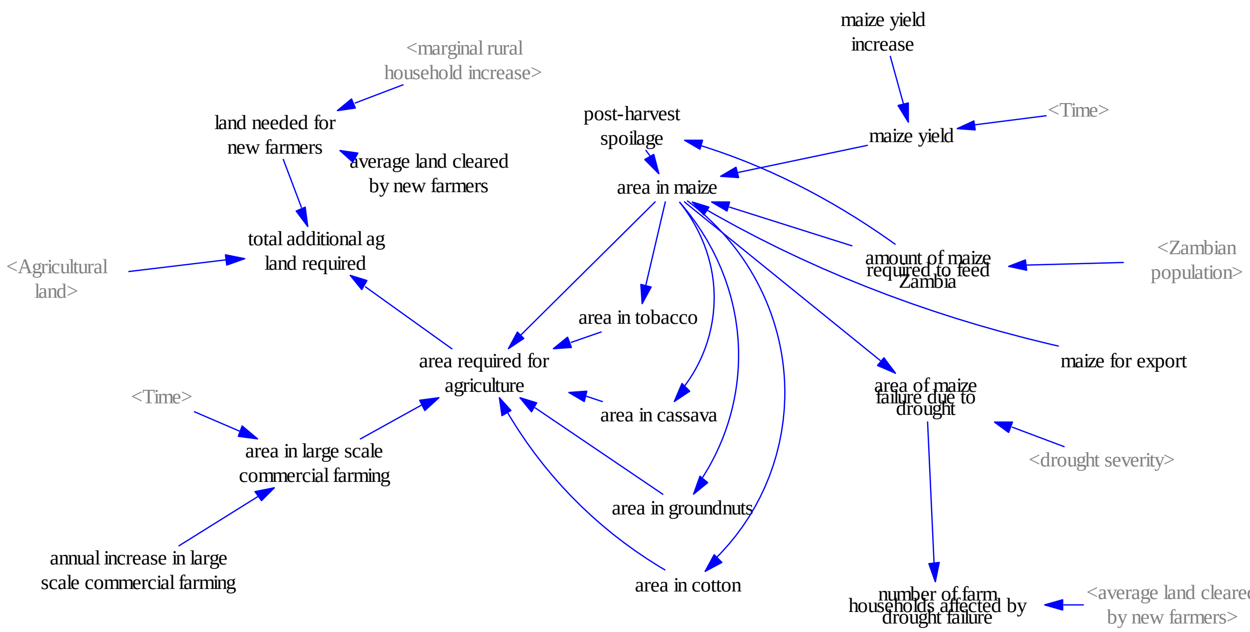

Figure 4. Agricultural and drought effect module

C. Results and Simulation Scenarios

In this discussion of results and simulated scenarios, we first present the baseline scenario for the national-level model, which represents how the system is likely to perform with no interventions. In the baseline scenario, we simulated deforestation by each of the main national drivers, including charcoal production and forest clearing for fuelwood harvesting from cutting down live trees, extraction of wood for construction and timber production, as well as agriculture expansion. Then we ran four additional scenarios, whereby we simulated the effect of i) drought, ii) full electrification, iii) maize yield increase, and iv) the effect of fuel-efficient stoves on forest cover in Zambia and in each province.

Results and discussion of national-level model

The results of the baseline scenario on forest loss and the contribution of each driver to deforestation by forest type over a 50-year period (from 2010 to 2060) are presented in Figure 5 for the national model.

The national model simulated an area loss of 156,364 ha of all forest types in 2010. The simulated deforestation rate is lower than those reported by FAO (2010) at 167,000 ha in 2010 or even the range of 250,000 to 300,000 ha reported in the ILUA study by Kalinda et al. (2008) and Mukosha and Siampale (2009). The slightly lower rate may be attributed to the exclusion of mining and infrastructure development, as well as the additional demand of wood by agriculture and industry in Zambia (Vinya et al., 2011). CSO (2013) estimated an additional 1.2 million tons of firewood and 48,000 tons of charcoal used in agriculture, industry, and mining in 2010. In term of relative decline in the total forest cover, the model reported a simulated value of 0.33 percent in 2010, which is consistent with that reported in the World Bank document (World Bank, 2012).

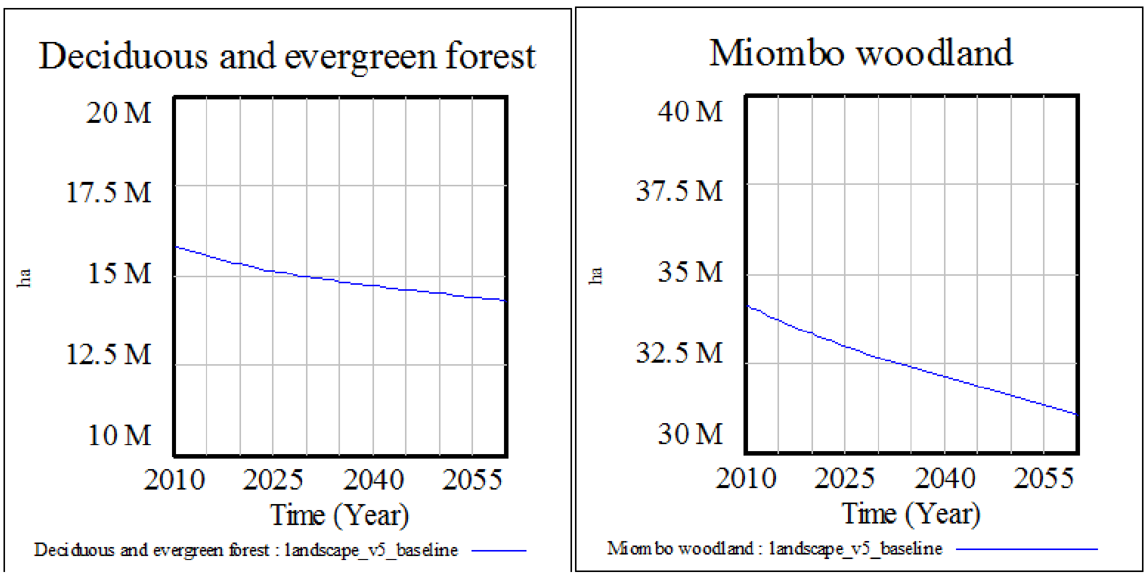

Figure 5. National deforestation rates for deciduous and evergreen and miombo woodlands from 2010 to 2060 in Zambia.

The simulated relative decline in forest cover differed between forest cover types over time. The deciduous and evergreen forest type would decline by 2.84 million ha or 18 percent in 2060. This suggests a deforestation rate of approximately 56,880 ha per annum. The miombo woodland areas would reduce by 14 percent, or 4.77 million ha over a 50-year period. This rate is translated to a loss of 95,000 ha of miombo area per year. If we apply the estimated rate of 79.37 tons of wood is produced per ha in intact forests by Kalinda et al. (2008), the total wood of over 12 million tons per year would be produced from the simulated total area of 153,350 ha in both forest cover types. Assuming an average density of 750 kg per m^3 (UNEP, 2015), this suggests a minimum extraction of over 16 million m^3 of wood per year across the country. If we consider the additional wood demand in agriculture and mining sector, the annual wood extraction would rise over 17.6 million m^3, representing the lower bound of wood extraction. This estimate is still greater than the MAC, estimated at 17.5 million m^3 per annum by Kalinda et al. (2008). The simulated forest loss, therefore, supports the proposition that Zambia's forests are being lost at an unsustainable rate (Kalinda et al., 2008; Gumbo et al., 2013).

Figures 6a and 6b depict the deforestation by each driver at the national level. Agricultural expansion accounted for the largest proportion of deforestation in 2010. The model simulated approximately 28,000 ha cleared for agriculture compared to about 7,000 ha of forest lost to charcoal production. The model simulated an increase of about 40 percent in agricultural expansion in 2060. Charcoal production would induce a steady increase in forest loss of 385 percent in 2060 Figure 6a). The rate of deforestation induced by charcoal production would override that of agriculture expansion beyond 2070. Our results support the prevalence of both charcoal production and agriculture expansion as national drivers of deforestation reported in various studies in Zambia (Kalinda et al., 2008; Vinya et al., 2011; Gumbo et al., 2013; Mulenga et al., 2014). The other important drivers are firewood harvesting from cutting down live trees in rural areas and wood harvesting for construction in rural areas. They showed similar patterns, with marginal increases of about 2 percent for wood harvesting and 18 percent for fuelwood harvesting. At the national level, the simulations showed similar patterns for most drivers in the miombo woodland, with the exception of commercial timber harvesting (Figure 6b). Key findings suggest that affordable alternative sources of energy are urgently needed to reduce the increasing demand of charcoal by the fast growing urban population. Similarly, widespread adoption of agricultural intensification practices will be required to produce more food from existing agriculture land to feed to increasing population in Zambia.

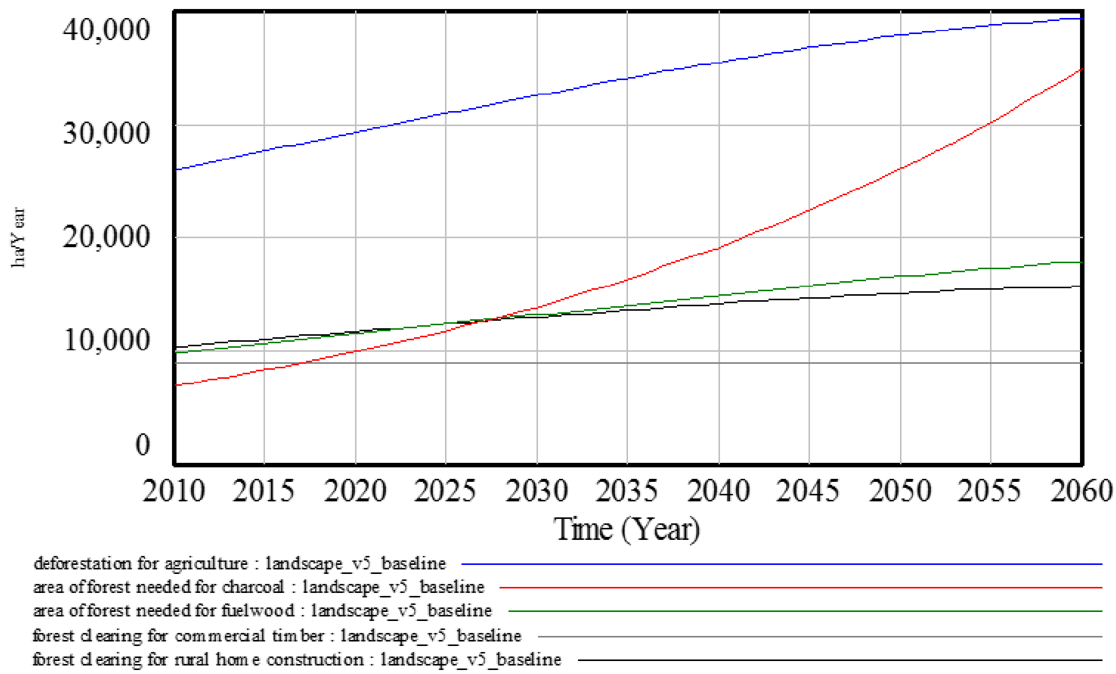

Figure 6a: National deforestation rates by driver for all forest cover types in the baseline scenario in Zambia

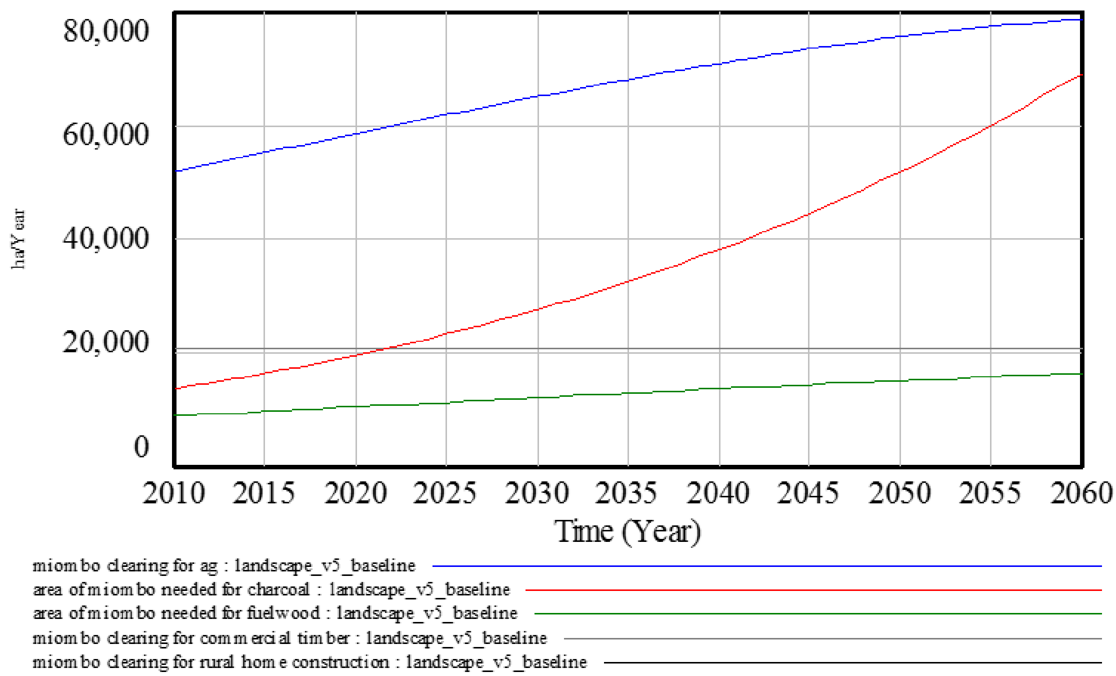

Figure 6b: National deforestation rates by driver for miombo woodland in the baseline scenario in Zambia.

Results and discussion of Eastern Province model

The results of the baseline run for Eastern Province are summarized in Figure 7. The Eastern Province model simulated a loss of over 11,000 ha of deciduous and evergreen forest since 2010 or a rate of 0.74 percent per year. Similarly, the simulation suggested a lower relative decline in miombo woodland at 0.54 percent, but a higher deforested area of over 15,000 ha in 2010. This total loss represents approximately 9 percent of the national deforestation rate, estimated at 300,000 ha per year by UN-REDD and cited in many reports (e.g., World Bank, 2012). If we apply the estimated rate of 79.37 tons of wood is produced per ha in intact forests by Kalinda et al. (2008), the simulation results suggest the extraction of at least 2.1 million tons of wood in all forest types in 2010. Assuming an average density of 750 kg per m^3 (UNEP, 2015), this would translate into approximately 2.75 million m^3 of wood biomass harvested in 2010. If we consider the total wood biomass, estimated at 264 million m^3 by Kalinda et al. (2008), the total wood extracted in 2010 represents only 1 percent the total stand stock in Eastern Province. This suggests a sustainable harvesting of wood in the Province. The estimate of the maximum allowable cut at 1.6 million m^3 per annum by Kalinda et al. (2008), however suggests over-exploitation in some areas.

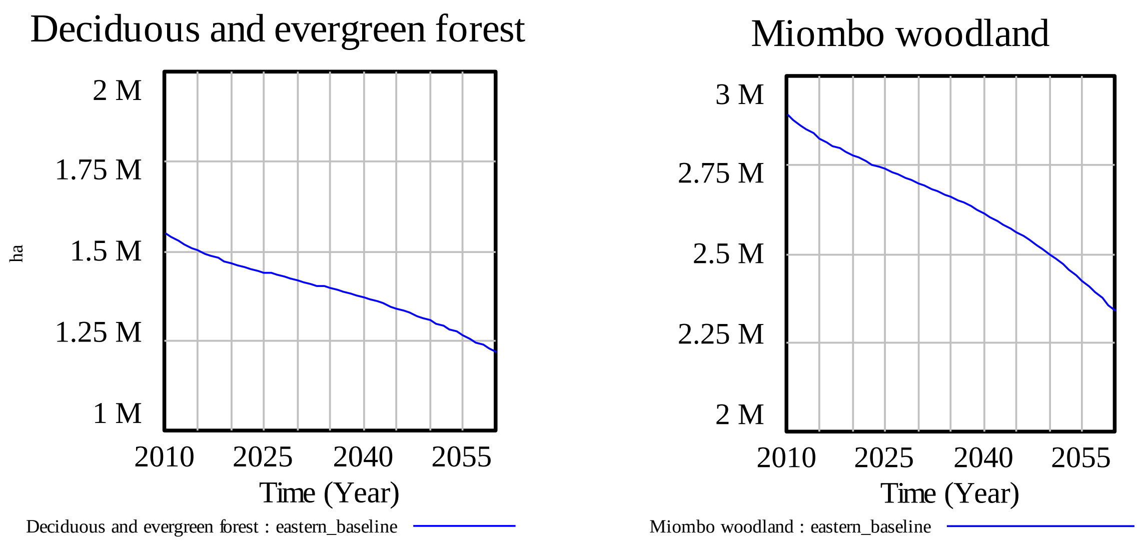

Figure 7: Simulated deforestation rates for deciduous forest and miombo woodland in the baseline scenario in Eastern Province

The model further simulated a total forest loss of 350,000 to 490,000 ha in Eastern Province over the next 50 years. Such loss represents an estimated annual deforestation rate between 7,000 ha and 9,800 ha per annum in deciduous and miombo woodland, respectively. As in the national model, forest loss in Eastern Province is mainly driven by agriculture expansion and charcoal production (Figure 8a). The model simulated about 5,000 ha of all forest type cleared for agriculture in 2010. The area required for charcoal production in 2010 was 0.6-fold that of agriculture at about 3,000 ha in the same year. The simulation showed a significant increase in forest loss for charcoal production by 400 percent over time, rising up to 15,000 ha in 2060. Agriculture expansion would increase by 120 percent over the 50 years, averaging approximately 11,000 ha in 2050. The simulation suggested that more land would be cleared for agriculture than for charcoal production in the next 30 years. In about 2045, charcoal production would induce more deforestation than agriculture expansion. The rates for wood harvesting for construction and fuelwood harvesting showed linear and marginal increases of about 50 percent over the simulation period.

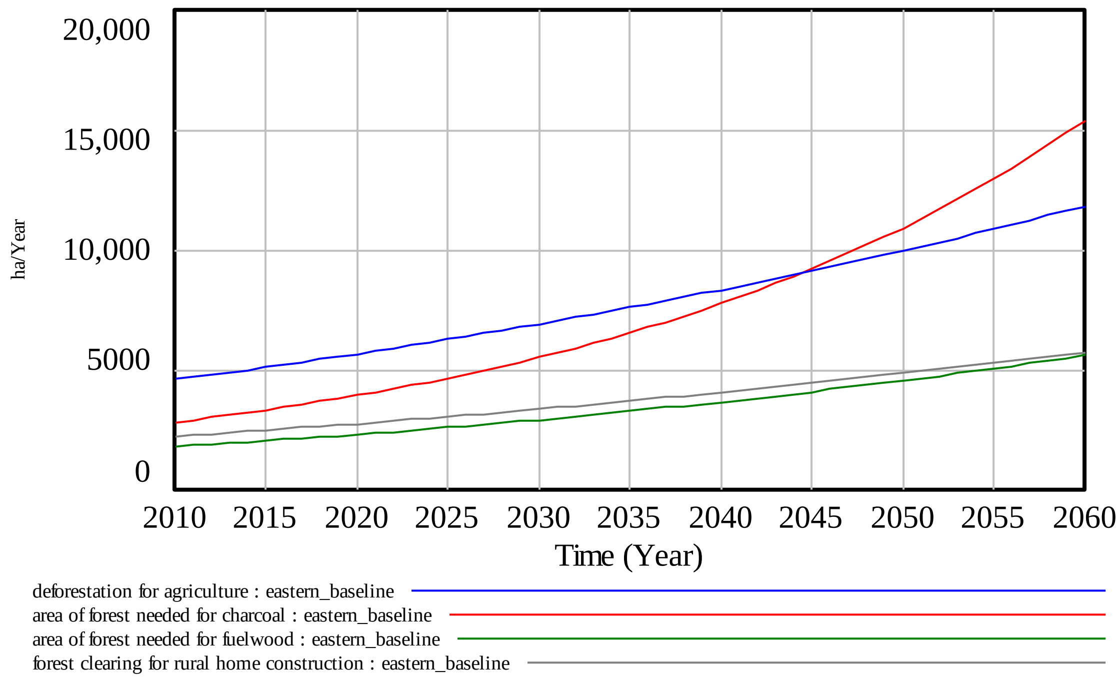

Figure 8a: Simulated deforestation rates by driver for all forest cover types in the baseline scenario in Eastern Province

The simulation further showed similar patterns for the drivers of deforestation of the miombo woodland (Figure 8b). But the model reported greater land clearing for charcoal production and agriculture in the miombo than in other forest cover types. The area cleared for charcoal in woodland was about 1.2 times that in all forest types in 2010, with a higher multiplier of 1.8 in 2060. The simulation suggests that charcoal production would require the clearing of over 26,000 ha of miombo in Eastern Province in 2060. Similarly, land clearing for agriculture in all forest types increased by 48 percent in miombo woodland alone in 2010, doubling in 2060. The greater demand for charcoal in the middle of the 21st century may be led by charcoal demand of total population, which is projected to more than double in the next generation, reaching 44 million in 2050 (United Nations, 2012). The large clearing of miombo woodland for charcoal production supports the preference of tree species of the genera of miombo ecosystem *Brachystegia*, *Julbernadia* and *Isoberlinia* for charcoal production reported by many studies like (Chidumayo, 1993; Dewees et al., 2011). The province is, however, poor in valuable timber tree species, with only 0.1 million m^3 of MAC per year. This timber stock may not sustain the supply of industrial wood for both local consumption and export. The findings raise the need to enforce the laws and regulations to combat illegal logging and to control charcoal production in Eastern Province.

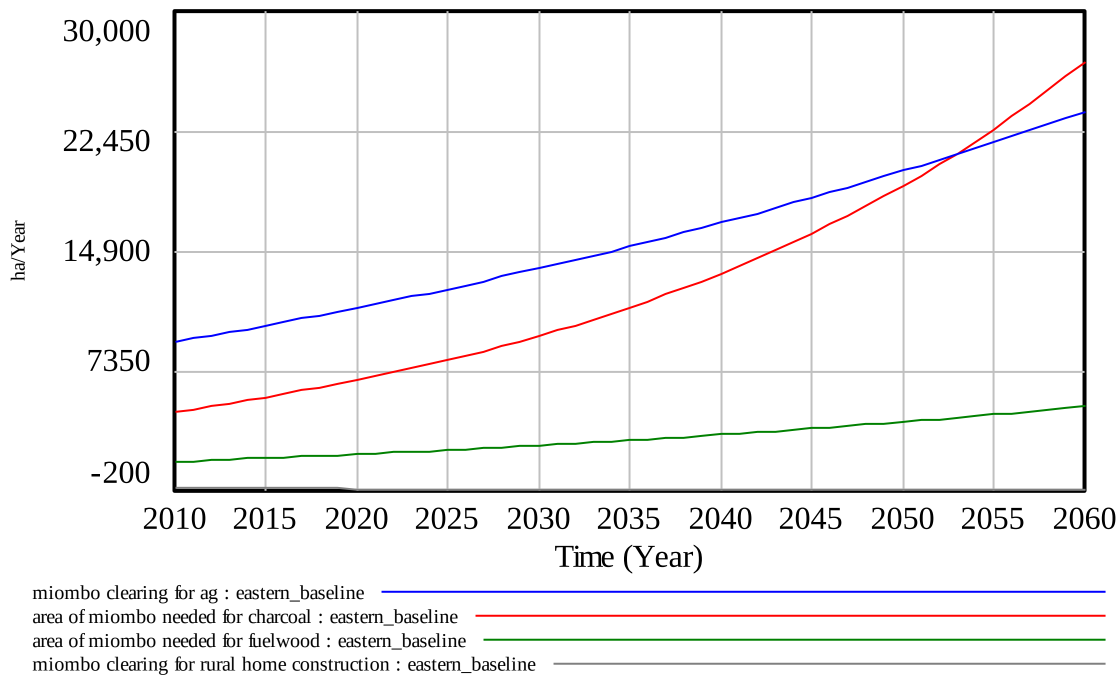

Figure 8b: Simulated deforestation rates by driver for miombo woodland in the baseline scenario in Eastern Province

Scenario analysis for Eastern Province

The first scenario simulated for Eastern Province is the effects of drought on forest cover change. The model reported negligible effect of both moderate and severe droughts on forest cover loss if all maize farmers practice conservation agriculture (Figure 9a). Grabowski et al. (2015) reported an average maize yield of 1,260 kg per ha for poor farmers in rural areas. Given an average cultivated area of 1.8 ha per household in Eastern Province, farmers who adopt CA would harvest about 2,268 kg of maize per household in bad years. Assuming the estimated daily requirement of 0.28 kg of maize per capita and the average household size of 5 members, the total maize production would satisfy food requirement throughout the year. The model assumes production loss on 40 percent or 70 percent of the maize area, which is translated into a total loss of production for the portion of farmers who use drought-prone land. The main impact of drought on forest loss is through farmers with reduced production using charcoaling as a coping strategy. But their charcoal harvest is dwarfed by the regular charcoal demand, and has relatively little long-term impact on forest dynamics.

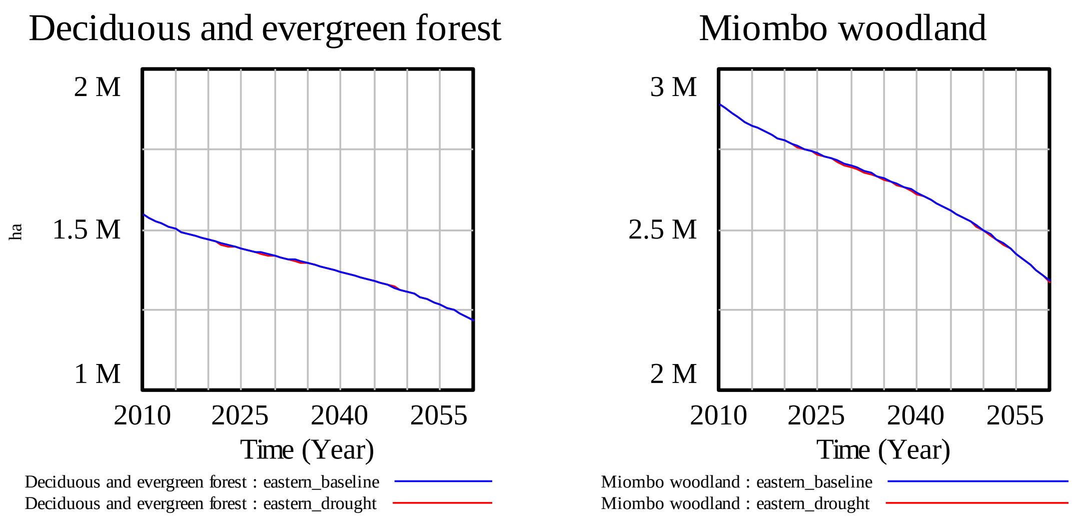

Figure 9a. Simulated results of the effects of drought on land cover change in Eastern Province

The next scenario we simulated was the impact of full electrification coverage for Zambia on forest cover change. Full electrification would have marginal effects on land cover loss in Eastern Province (Figure 9b). The model reported a gain in the cover of deciduous and evergreen forest of about 250,000 ha over the 50-year period compared with the baseline scenario. This gain increased by 40 percent, averaging 350,000 ha in the miombo woodland over the same time period. Full electrification would preserve about 6,000 ha of the forest cover per annum in Eastern Province. This low gain is explained by the greater proportion of rural population in Eastern Province, estimated at 87 percent in 2010 (CSO, 2010), for whom electrification would not replace woodfuels, and the high use of fuelwood among them.

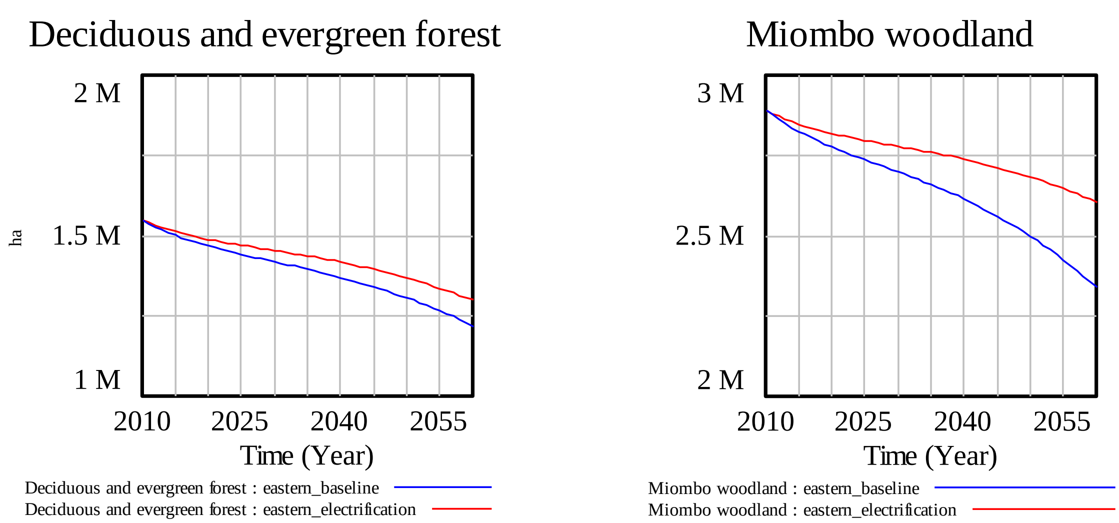

Figure 9b. Simulated results of the effects of full electrification on land cover change in Eastern Province

We also simulated an increase in maize yield on forest cover change. The model reported an increase in current average maize yield by three-fold would not change the current rate of land clearing in both deciduous and evergreen forests as well as in miombo woodlands (Figure 9c). Assuming an average maize yield of 1,696 kg per ha for rural smallholder, the yield increase simulation suggests a smallholder maize yield of over 5,000 kg per ha. Attaining such level of productivity would require a tailoring of agricultural technologies that suit specific locations across the country. These may include nutrient use efficiency (NUE), no-till, crop protection, integrated soil fertility management, drought tolerant varieties, water harvesting and/or the combination of the promising technologies recommended by Rosegrant et al. (2014) for maize production in Southern Africa.

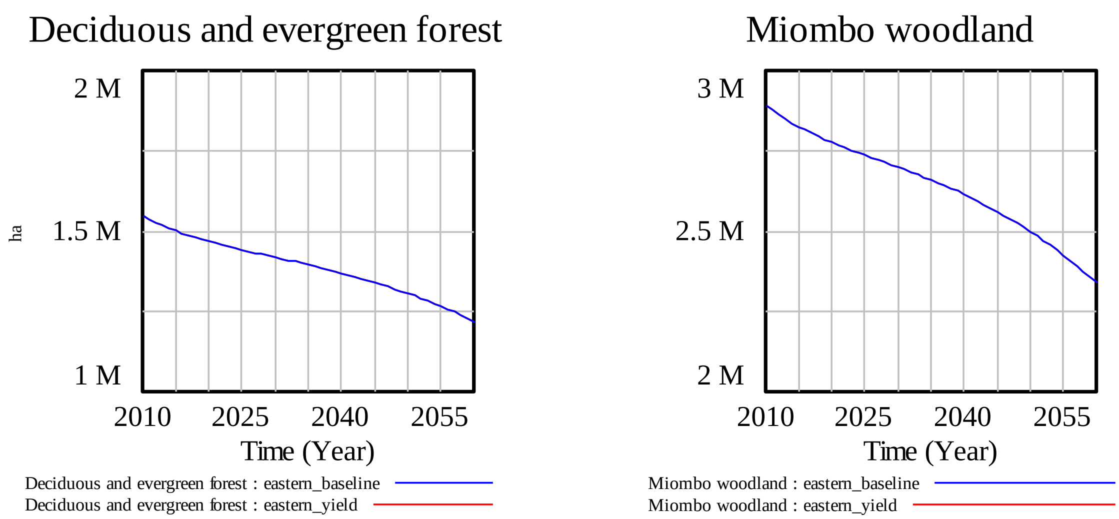

Figure 9c. Simulated results of the effects of increased maize yield on land cover change in Eastern Province

The final scenario we simulated for the eastern province model was widespread use of more fuel-efficient stoves on forest cover change. The use of efficient stoves would induce marginal reductions in forest loss in Eastern Province. The model reported increased gains in forest cover over time for both forest cover types compared with the baseline scenario (Figure 9d). The reduction in forest loss was about 62,500 ha in deciduous and evergreen forest type over the simulation time period. This suggests an average gain of approximately 1,250 ha per annum in the same forest type. Gains in miombo woodland are 4-fold that in the deciduous forest, simulated at 250,000 ha. The use of efficient stoves would preserve about 5,000 ha of miombo woodland area per annum.

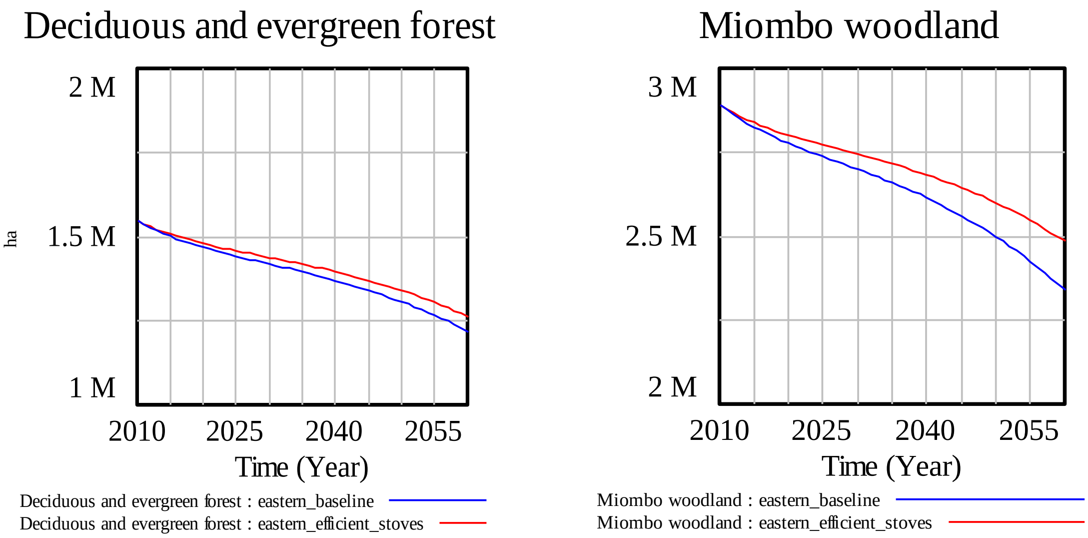

Figure 9d. Simulated results of the effects of efficient cook stoves on land cover change in Eastern Province

Results and discussion of Lusaka Province model

In the baseline run there was a total loss of about 3,600 ha, or 1.23 percent of deciduous and evergreen forest in Lusaka province in 2010. Higher losses were simulated in the miombo woodland, reaching 5,170 ha, or 0.93 percent of forest cover in the same year (Figure 10). This total loss represents approximately 3 percent of the national deforestation rate, estimated at 300,000 ha per year by UN-REDD and cited in many reports (e.g., World Bank, 2012). If we apply the estimated rate of 79.37 tons of wood produced per ha in intact forests by Kalinda et al. (2008), the simulation results suggest a partial extraction of over 700,000 tons of wood in both the deciduous forests and miombo woodland in 2010. Assuming an average density of 750 kg per m^3 (UNEP, 2015), this would represent approximately 930,000 m^3 of wood biomass harvested in 2010. The suggested wood extraction in 2010 exceeds by far, almost doubling the MAC, estimated at 500,000 m^3 by Kalinda et al. (2008). The simulation results suggest the over-exploitation of forests and woodlands in Lusaka province (e.g., Chidumayo, 2001), despite the seemingly low loss rates. Lusaka, therefore, relies heavily on import from forest-rich provinces to supply both charcoal and timber needs.

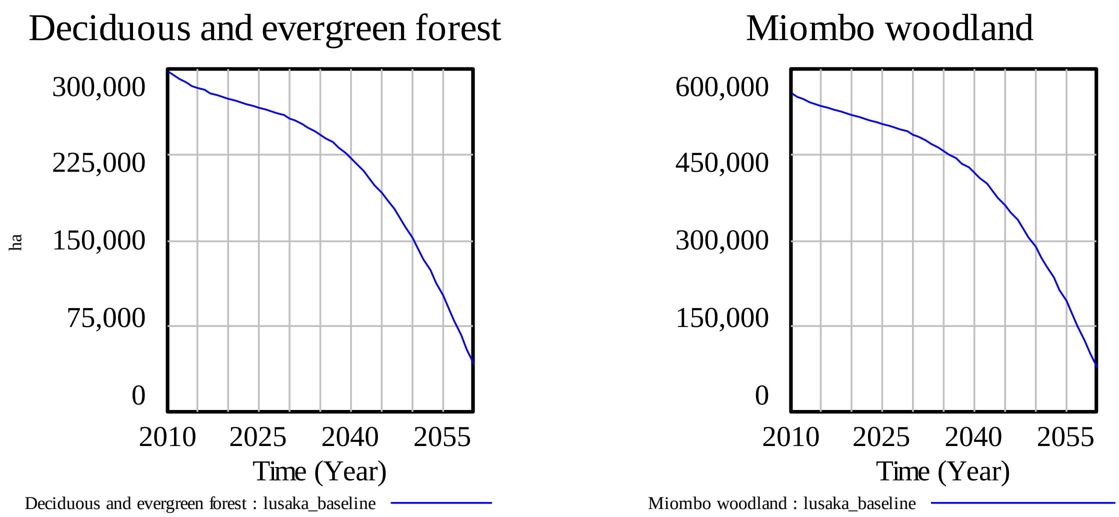

Figure 10: Simulated deforestation rates for deciduous forest and miombo woodland in the baseline scenario in Lusaka Province

The model further showed a sharp decline in both forest and woodland cover types over time. Each of these forest cover types is projected to decline by at least 87 percent by 2060. This suggests an average loss of about 260,000 ha in deciduous and evergreen forests over 50 years, or about 5,170 ha per annum. Loss in miombo woodland would almost double that in deciduous forest, simulated at 487, 000 ha over 50 years, or over 9,700 ha per annum. The results indicated a sharp increase in the rate of deforestation by 44 to 88 percent in simulated forest and woodland cover types over time. The model further suggested a complete destruction of the forest and woodland in the next 60 to 70 years. The total land clearing of 14,870 ha by 2060 would represent over 1.5 million m^3 of wood biomass, which is over three-times that of the estimated MAC in Lusaka province. This suggests that meeting the charcoal and timber demand in Lusaka province would require a clearing of over 10,000 ha of land in other provinces.

The simulation showed that charcoal production is the most important driver of deforestation in all forest types, increasing exponentially over time. Land clearing for charcoal production would increase by over 11-fold, or 1025 percent, from 2,000 ha in 2010 to 22,500 ha by 2060 (Figure 11a). Charcoal production alone would require over 2.38 million m^3 of biomass wood in 2060. Assuming the MAC of 500,000 m^3 per year, the province will record a deficit of over 1.8 million m^3 for charcoal production, suggesting a minimum land clearing of 17,000 ha in neighboring areas. The second important driver was agricultural expansion that showed a linear increase over the 50-year time period. Agriculture expansion would increase by 3.5-fold, from about 2,000 to over 7,000 ha of land clearing by 2060. Fuelwood and wood harvesting for construction showed similar patterns of marginal increases over the simulation time.

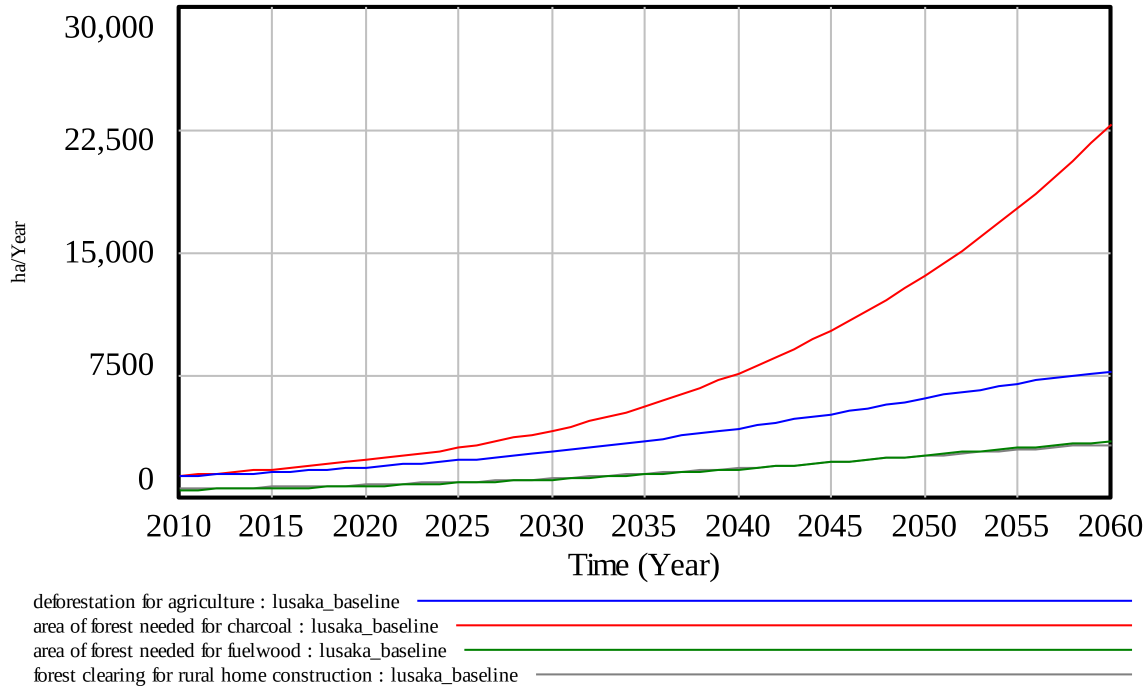

Figure 11a: Simulated deforestation rates by driver for all forest cover types in the baseline scenario in Lusaka Province

The model showed the same pattern in the miombo woodland, with higher levels of land clearing than in all forest types (Figure 11b). Charcoal production would increase land clearing by around 19-fold, rising up to 38,000 ha by 2060 in the miombo woodland. Similarly, land clearing for agriculture rose up to 10,000 ha, increasing by 5-fold over the 50-year time period. The simulation showed an increase in land clearing in the miombo woodland by 1.4- to 1.7-fold that in all forest types. The simulation results support the findings that charcoal production is the most important driver of deforestation in Lusaka province (Vinya et al., 2011). It further emphasizes the preference of miombo woodland for charcoal production as well as its vulnerability to over-exploitation of wood extraction (e.g., Dewees et al., 2011). Moreover, the exponential increase in land clearing for charcoal production over time reflects the exponential growth in energy needs of the urban population. Our findings are consistent with studies in many other countries in SSA, where charcoal demand is expected to increase for several decades with the fast growing urbanization (Arnold et al., 2006; Hofstad et al., 2009; Zulu and Richardson, 2013; World Bank, 2014).

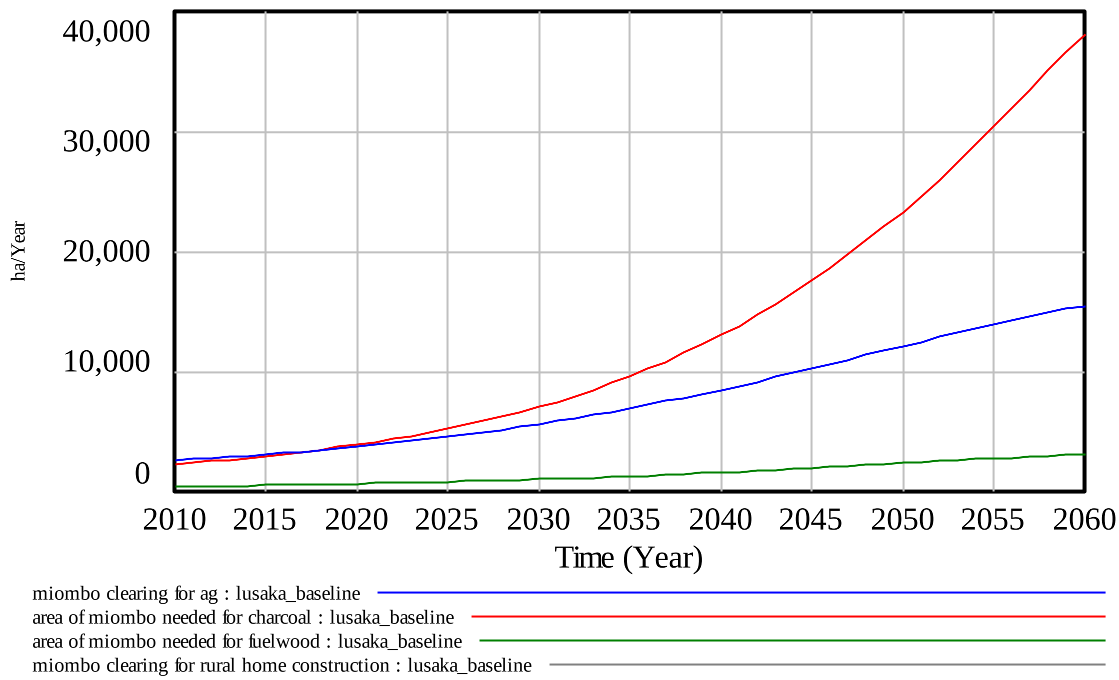

Figure 11b: Simulated deforestation rates by driver for miombo woodland in the baseline scenario in Lusaka Province

Scenario analysis for Lusaka Province model

The first scenario simulated was the effects of drought on forest cover change. Similar to Eastern Province, the model reported negligible effects of both moderate and severe droughts on forest cover loss if all maize farmers practice conservation agriculture in Lusaka Province (Figure 12a). Grabowski et al. (2015) reported an average maize yield of 1,260 kg per ha for poor farmers in rural areas. Given an average cultivated area of 0.98 ha per household in small and medium scale production in Lusaka Province, farmers who adopt CA would harvest about 1,234 kg of maize per household in bad years. Assuming the estimated daily requirement of 0.28 kg of maize per capita and the average household size of 4.8 members, the total maize production would satisfy the food requirement of rural people throughout the year. But given that access to market, the lack of alternative livelihood options, and household income strongly influence rural household dependence on natural resources (Bwalya, 2011), some rural households are likely to produce charcoal to supply the charcoal demand by urban populations as a coping strategy during drought years. Based on these insights, we assume that, when 40 percent or 70 percent of the land area loses maize production due to drought, some farmers turn to charcoaling. But their charcoal harvest is dwarfed by the "regular" charcoal demand, and does not impact forest dynamics greatly over the long-term.

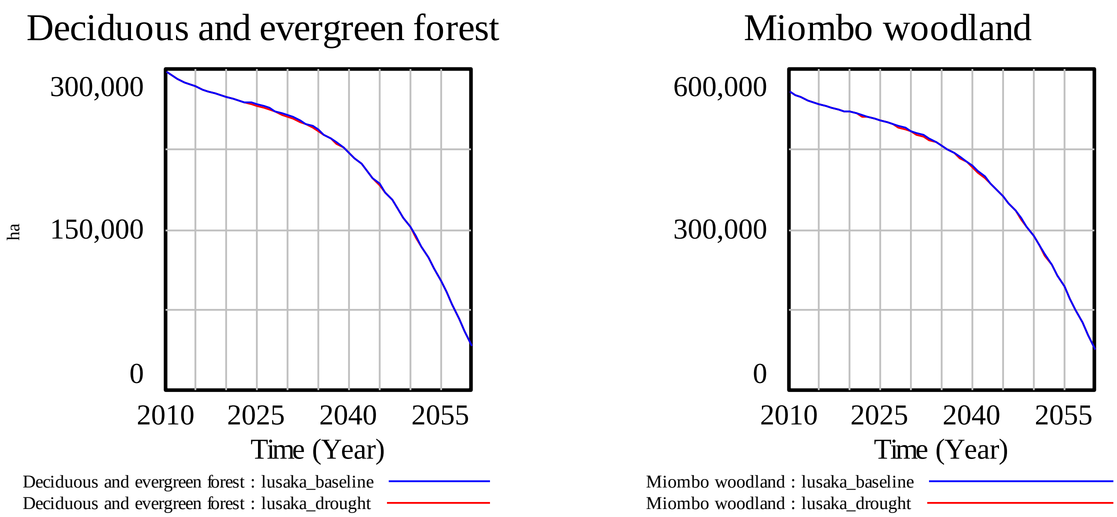

Figure 12a. Simulated results of the effects of drought on land cover change in Lusaka Province

The next scenario we simulated was the impact of full electrification coverage in Zambia on forest cover change. Full electrification showed significant effects on land cover loss in Eastern Province (Figure 12b). The model reported a total gain in the cover of both deciduous forests and miombo woodland of about 487,500 ha over five decades compared with the baseline scenario. This suggests that full electrification would preserve about 9,750 ha of the total forest cover per annum in Lusaka Province. In addition, over 75 percent of this total gain would occur in miombo woodland over the same time period. The simulation suggests that full electrification is likely to reduce more than half the current rate of deforestation in simulated forest types over the next 50 years in Lusaka Province. The gain is higher in the miombo woodland, estimated at 68 percent of the area being currently cleared in the Province. These gains are attributable to the large population of urban dwellers in Lusaka province, estimated at 84 percent in 2010 (CSO, 2010) and the high use of charcoal among them.

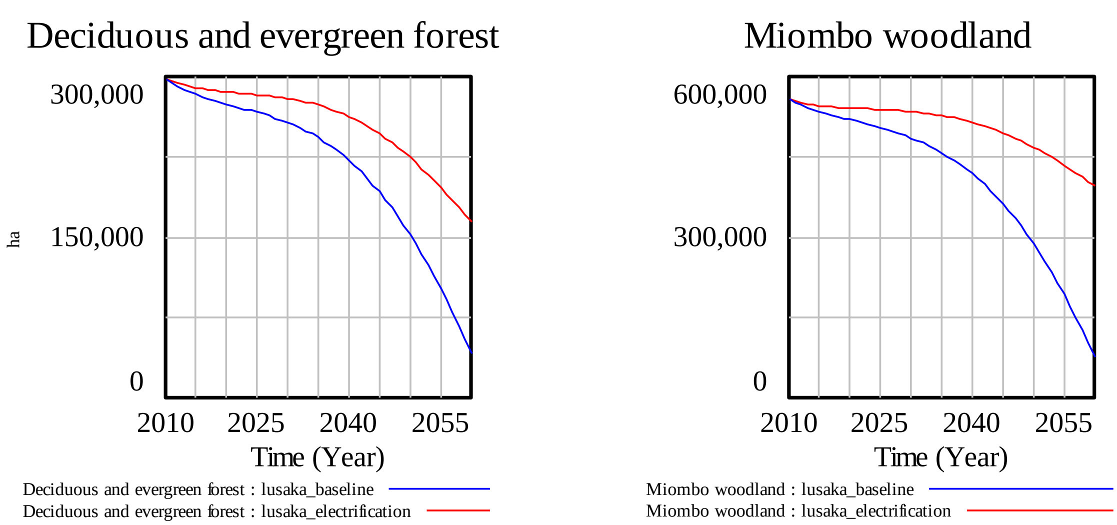

Figure 12b. Simulated results of the effects of full electrification on land cover change in Lusaka Province

Next we simulated an increase in maize yield on forest cover change. The model reported an increase in current average maize yield by three-fold to maintain the current rate of land clearing in both deciduous and evergreen forests as well as in miombo woodlands (Figure 12c). Assuming an average maize yield of 1,725 kg per ha for small- and medium-scale farmers, the simulation suggests a maize yield of over 5,175 kg per ha to prevent encroachment into forests and woodlands in Lusaka Province. Attaining such level of productivity would require a tailoring of agricultural technologies that suit specific locations across the country. Rosegrant et al. (2014) provides such specific promising technologies for maize production for Southern Africa. Large-scale commercial farmers, however, produce approximately 4,850 kg of maize per ha in Lusaka Province (Tembo and Sitko, 2012). Their productivity represents 94 percent of the desired maize yield, suggesting that a few farmers could increase their productivity with better financial support.

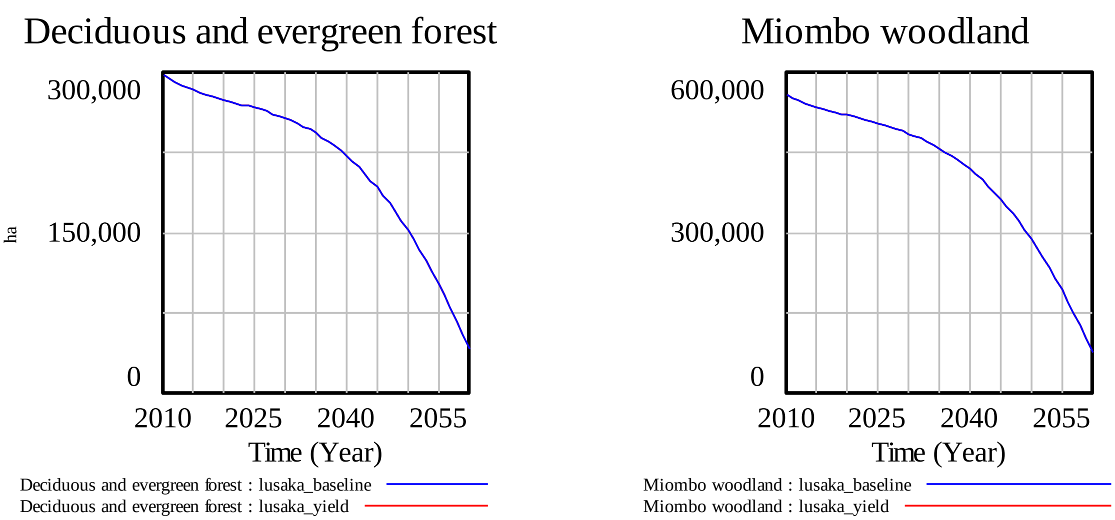

Figure 12c. Simulated results of the effects of increased maize yield on land cover change in Lusaka Province

The final scenario we simulated was the impact of widespread use of fuel-efficient stoves on forest cover change. The use of efficient stoves would induce significant reductions in forest loss in Lusaka Province. The model reported increased gains in forest cover over time compared with the baseline scenario, totaling 237,500 ha by 2060 (Figure 12d). Over two-third of the projected gain would occur in miombo woodland alone, with a simulated area of 150,000 ha over the five decades, or 3,000 ha per annum. About 87,500 ha would be preserved in the deciduous forest, representing an average annual gain of 1,750 ha over the same time period compared with the baseline. The simulation suggested that efficient stoves would reduce the current rate of deforestation by 69 percent in all forest types, rising up to 71 percent in miombo woodland alone. Our results support on-going initiatives of fuel-efficient cook stoves by various NGOs like SVA and COMACO in urban areas. It is, however, important to note that such initiatives should be spatially targeted in order to yield desirable outcomes. For example, the on-going promotion of cook stoves in the rural areas would have little, if not, zero effect on the level of urban charcoal consumption. Concerted efforts should, therefore, be made to promote their widespread adoption and sustained used among charcoal users in Lusaka Province.

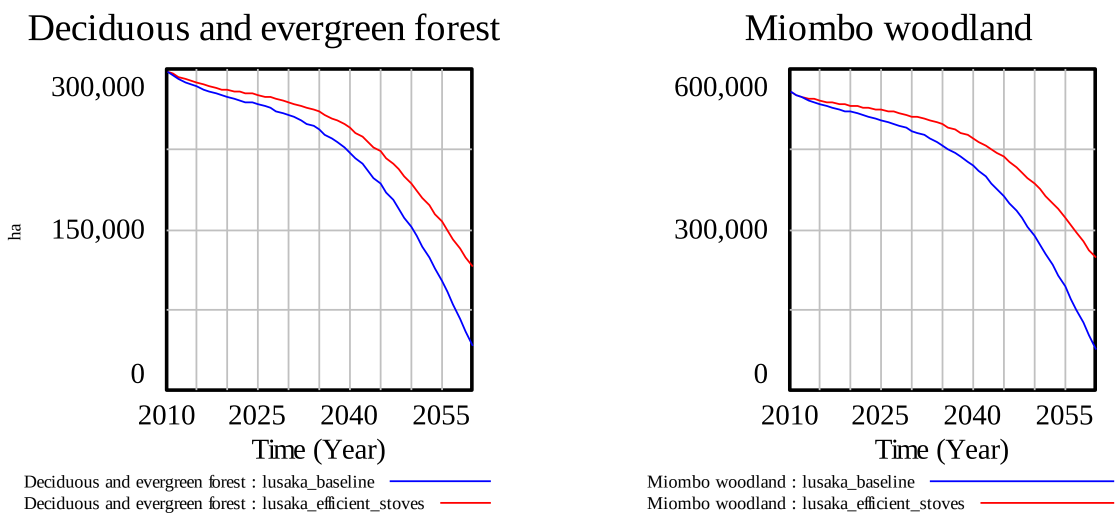

Figure 12d. Simulated results of the effects of efficient cook stoves on land cover change in Lusaka Province

5. Conclusions and Recommendations

In this discussion of conclusions and recommendations, we consider the implications of the results, in light of limited evidence of linkages between farm-level SI practices and the objectives of forest protection and wildlife conservation. We then offer concluding recommendations for the development of integrated programming that addresses the dual objectives of agricultural development and environmental conservation.

A. Conclusions

In the context of this integrated project, the current study examined the agricultural-environmental linkages in Eastern and Lusaka Provinces in Zambia with a focus on the implications of farm-level decisions for landscape-level impacts. A comprehensive literature review reveals that these agricultural-environmental linkages in the context of Zambia are complex and dynamic, especially in light of rapid population growth and urbanization. The results of the participatory systems dynamic model demonstrate that agricultural expansion is currently the largest contributor to forest loss, but in the future, charcoal production will soon outstrip expansion. At the provincial level there are different patterns. Charcoal production dominates in Lusaka Province throughout the 50-year simulation time, primarily because of urban demand. By contrast, agricultural expansion dominates in Eastern Province until about 2045, at which point charcoal production becomes the dominant driver.

Charcoal production is driven by urban population growth and energy demand. Agricultural expansion is driven by rural population growth, as opposed to low yields and/or land abandonment. Currently, wood is extracted in an unsustainable way from forests and woodlands, posing a threat to the remaining forest and woodland resources across the country. Moreover, miombo woodlands are the most vulnerable to deforestation of all forest cover types in Zambia despite their ability to regenerate. The results support other projections of the expected increase in charcoal demand as a result of rapid urbanization (e.g., Zulu and Richardson, 2013; World Bank, 2014). Reducing charcoal consumption would likely involve coordinated policies that address rural household dependence on charcoal as an income source in addition to policies that seek to reduce urban charcoal demand.

This SILL project had two main objectives. The first objective was to provide an evidence base for the linkages between SI practices and climate change mitigation and biodiversity conservation. For the reasons described in the summary of agricultural-environmental linkages in Section 2, the overall conclusion is that there is no evidence of linkages between adoption of sustainable intensification practices (such as CA and AF) and environmental objectives in terms of climate change mitigation, forest protection, and wildlife conservation. There is theoretical and empirical evidence that agriculture can have negative implications on wildlife when promoted in areas where human-wildlife conflict is likely. However, the cause is related to encouraging agricultural activities in sensitive areas more generally, not the promotion of SI or CA specifically. Agroforestry appears to have a net positive effect on biodiversity (for some species), and on-farm fuelwood provision from agroforestry is of particular benefit to women and children in rural Zambia by reducing their burden of walking long distances to collect firewood due to local scarcity. However, given the minimal contribution of firewood collection in rural areas to overall deforestation, the results suggest that agroforestry will contribute little to reduce the largest driver of deforestation, which is the rapidly rising demand for charcoal consumption in urban areas. Therefore, we conclude that in the context of agricultural development in Zambia, there is little evidence of linkages between adoption of SI practices and either forest protection objectives associated with climate change mitigation or wildlife protection activities associated with biodiversity conservation.

B. Integrated Programming and Further Research

The second objective of the SILL project was to provide recommendations to inform the design of integrated programs involving agricultural development and environmental conservation in Zambia. The connection between livelihoods, food security and biodiversity is fundamental to sustainable development in Zambia. This link, however, continues to pose challenges for both the conservation and agricultural development sectors. The rural population heavily depends on smallholder agriculture, which employs the vast majority of the labor force and contributes about 20 percent of the national GDP (Tembo and Sitko, 2012; World Bank, 2012). Encroachment and degradation remain significant threats to forested areas in Zambia. However, in the future, charcoal production from increasing demand for cooking fuel in urban areas poses the greatest threat to the remaining forests in Zambia (Kalinda et al., 2008; Gumbo et al., 2013).

Human encroachment also threatens protected areas in Zambia, more than in most other African countries (Pfeifer et al., 2012), and poses a serious threat to biodiversity, habitat degradation, and species loss in Zambia. The most effective integrated programming would recognize that agricultural development and biodiversity conservation activities may be incompatible in some regions, particularly in the context of large mammal conservation, and so promoting these activities in the same location is not fruitful. Evidence suggests that population growth in GMAs has been associated with human-wildlife conflicts in the form of crop losses (Richardson et al., 2012), and such conflicts are counter to the objectives of sustainable intensification and biodiversity conservation. Conservation initiatives may be more effective by focusing on local governance and promoting community-based natural resource management programs such as community forestry, community wildlife conservation, and sustainable management and use of forests. At the same time, programs designed to enhance rural food security in biodiversity-rich areas should consider the challenges of agricultural development in biodiversity-rich areas, particularly with respect to the conservation of large mammals. The conclusions of this project suggest that such programs should not include the promotion of livelihood activities that might incentivize expansion of agricultural land.

Integrated programs that effectively address agricultural and environmental objectives should consider the landscape-level implications of agricultural-environmental linkages described in the results of this report. Charcoal production is most likely to be addressed effectively through a rapid reduction in the demand for charcoal, which is likely to require widespread adoption of more efficient and sustainable approaches for producing and consuming charcoal in the near term, and a sweeping shift to affordable alternative energy sources for urban households in the long term. Given the importance of charcoal production to deforestation, this linkage may also be effectively addressed through a better understanding of the dynamics of household behavior and decision-making in sub-Saharan Africa regarding both urban household energy choices and rural household participation in charcoal production and trade. The so-called "energy ladder" theory postulates that as household incomes increase and individuals and countries develop economically, people's energy preferences will transition up an energy ladder from "inferior" biomass fuels to modern, cleaner alternatives. Although this theory largely holds globally, recent evidence shows that for Africa, the energy transition has proceeded much more slowly than anticipated, and it ultimately may not fully shift, since households continue to use charcoal for energy even as incomes rise (Zulu and Richardson, 2013). Community-based woodlots and community-based forest management arrangements may offer considerable opportunities for reducing deforestation and enhancing charcoal-based poverty reduction because of their potential to more deeply reach into communities, and to be more locally relevant, pro-poor, equitable, and more just than top-down government approaches.

The problem of encroachment and degradation is most likely to be addressed through better land allocation policies, improved monitoring of forest cover, and better enforcement of forest laws and land use regulations. In consultations with stakeholders, land tenure emerged as an important issue regarding rural land use. These consultations highlighted several issues, including (i) the existence of a dual tenure system of state and customary land with conflicting incentives that are often poorly understood, and (ii) the fact that all forest and wildlife resources were vested in the State, leaving rural communities with little authority or agency regarding natural resource management.

From consultations in Eastern Province we concluded that in general, people felt relatively secure that the land that they farm will not be taken away from them, so their perception of land security is high. At the same time, there is a perception that there is no more land available to allocate to youth for agricultural expansion as the population grows. In terms of gender implications, there were perceptions that trends associated with characteristics of matrilineal cultures were transitioning to those associated with characteristics of patrilineal cultures.

Other examples of integrated programs to address the problem of encroachment and degradation would include working with communities to draft community resource management and conservation plans for communities around GMAs to ensure that benefits of the GMAs are spread across the community. Integrated programs to address the linkages examined in this study may also include community-based woodlots and reforestation initiatives.

Integrated programs should consider the spatially extensive nature of these drivers, since the charcoal sector meets energy demands for urban households and provides livelihoods and income for both rural and urban households through the production, transport, and marketing processes. For example, an integrated program of this nature could involve the development of alternative livelihoods for rural households, such as the production of briquettes made from agricultural residues or sawdust from construction to meet the energy demands of urban households.

Other recommendations for integrated programming would include the promotion of alternative livelihood activities and off-farm income generation activities in GMAs rather than food security activities. Certainly, the promotion of agricultural technologies that address both the biophysical and socio-economic needs of farm households such as fertilizer tree species, intercropping, mulching, and minimum tillage is likely to contribute to the increase in rural household resilience and the conservation of agricultural landscapes while also contributing to climate change mitigation and protection of biodiversity. In particular, climate-resilient sustainable intensification, which includes the diversification of smallholder farming systems, holds a great potential to build and/or increase the resilience of smallholders to climate change. The high vulnerability of maize to projected weather shocks and its estimated yield loss of 18 percent by 2050 in Southern Africa (Zinyerere et al., 2012) call for an integrated approach of sustainable land use with both adaptation and mitigation effects to climate change. Such approach could combine agroforestry systems, such as fertilizer trees, nitrogen-fixing legumes, and climate-resilient maize varieties to improve food security while conserving biodiversity (e.g., Verchot et al., 2007; Syampungani et al., 2010). All these are components of the climate-smart agriculture approach being promoted in several countries, including Zambia. The results of this integrated project also suggest the need to raise awareness among all stakeholders on the linkages between environmental conservation objectives and charcoal production and use, including among policy makers, rural and urban households, schools, and other institutions.

It is worth mentioning that the objectives of this integrated project included focusing on two provinces in Zambia as pilot sites. Effective integrated programming must be spatially explicit and targeted to the needs of the community at a finer scale. For example, deforestation tends to be localized and closer to urban areas for easier access to markets, but the examination of agricultural-environmental linkages in this pilot study at the province level was limited in its ability to explore the more concentrated drivers of deforestation and may mask localized forest pressures. Implications for further research may include an examination of the localized drivers of deforestation, perhaps in the context of GMAs and the associated demographic changes that appear to be occurring there. Spatial analysis could be effective in such an examination of localized impacts at a finer scale.

A final recommendation for further research would include an in-depth analysis of household behavior and decision-making in sub-Saharan Africa regarding urban household energy choices generally, and rural household participation in charcoal production and trade. Such an analysis could help enhance understanding of the drivers of charcoal production and consumption, as well as the pathways to promote behavioral changes required to shift urban households away from charcoal consumption towards more sustainable energy sources.

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Appendices

Appendix 1: Species used in charcoal production

Faidherbia Albida (Apple-ring acacia or Ana-tree, Musangu)
Acacia gerrardii (Grey-haired acacia, Mtete)
Afzelia quanzensis (Pod mahogany, Mbangozi)
Anisophyllea boehmii (Mufungu, Mutobo, Nfungo)
Brachystegia boehmii (Prince of Wales feathers, Mfundaluzi)
Brachystegia bussei (Smooth bark)
Brachystegia longifolia (Mubombo, Musamba)
Brachystegia spiciformis (zebrawood, bean-pod tree, Muputu, Kampela)
Cassia abbreviata (Long-tail cassia, Musamafwa, Munsokansoka, Mseza)
Combretum collinum (Bicoloured bush willow, Nkalalama)
Diplorhynchus condylocarpon (Horn-pod tree, Mtowa)
Julbernardia globiflora, *Julbernardia paniculata* (Mutondo)
Maprounea Africana (Mpasa), *Pericopsi*s *angolensis* (Bloodwood, Mubanda)
Piliostigma thonningii (wild bauhinia, Rhodesian bauhinia, monkey bread, Musekesi), *Pseudolachnostylis maprouneifolia* (Duiker-berry, Kudu-berry, Msolo, Ngalati)
Stereospermum spp. (Kasokosoko)
Terminalia sericea (Silver cluster leaf, Gonondo)
Terminalia stenostachya (Rosette cluster leaf, Muweti)
Terminalia stuhlmanni (Gonondo)

Appendix 2: High-value timber tree species

About 19 tree species are widely harvested from the Zambian forests for timber and other industrial round wood production. Most recently, GRZ (2013) through its Statutory Instrument No. 52 of 2013 under the Forest Act (Amendment Regulations, 2013) identified 13 species of high value for timber production in Zambia. These are *Afzelia quanzensis* (Mupapa, Mwande, or Pod Mahogany), *Albizia* species (Musase, Mutanga), *Baikiaea* species (*Baikiaea plurijuga* also known as Mukusi or Zambezi teak), *Brachystegia* species (Miombo), *Daniella alsteeniana* (Mkukulabushiku), *Entandrophragma* species (Mofu, Mofwe, or Mipumema), *Erythrophleum africanum* (Kayimbi, Mukoso, Mubako), *Faurea Saligna* (Saninga, Mushokoso), *Guibourtia coleosperma* (rosewood or, Muzauli, Mushibi), *Khaya anthotheca* (Mululu, Mbewa), *Mitragyna stipulosa* (Mupa), *Pericopsis angolensis* (African Ebony or Mubanga, Mukwa), and *Pterocarpus angolensis* (Mukwa, Muombwa, Mulombe, Mukula).

The most preferred and targeted species are the Zambezi teak (*Baikiaea plurijuga*), the Kiaat (*Pterocarpus angolensis*) and the African rosewood (*Guibourtia coleosperma*) (Kalinda et al., 2008; Gumbo et al., 2013). These 13 high value timber tree species represent 12.4 percent of the national timber gross value or 364.7 million m^3. These species have an estimated average commercial value of 8.8 m^3 per hectare (Kalinda et al., 2008).

Appendix 3: Population growth in Game Management Areas

Name	Type	Location description	Area (ha)	Census area (ha)	GMA Coverage (%)	Population within GMA 2010 [a]	Population growth, 2000-2010 (%)
Kanakantapa South	Forest reserve	W of Lower Zambezi, not contiguous	5,409	33,536	16	unknown	unknown
Chiawa	GMA	West of Lower Zambezi	234,400	286,332	82	5,892	207
Luano	GMA	Portion of Lusaka & Central provinces	893,000	936,170	95	16,911	154
Rufunsa	GMA	East and NE of Lower Zambezi	317,900	807,139	39	31,343	135
Lower Zambezi	National Park		409,200	673,104	61	6,814	156
West Petauke	GMA	Western portion of Nyimba	414,000	643,394	64	3,815	123
North Luangwa	National Park		463,600	[b]			
Musalungu	GMA	N and NE of North Luangwa	1,735,000	1,969,992	88	84,571	147
Munyamadzi	GMA	Between N and S Luangwa	330,000	981,041	34	8,738	141
Luambe	National Park	Between N and S Luangwa	25,400	[c]			
Lumimba	GMA	N of Lupande	450,000	986,508	46	29,313	143
Lukusuzi	National Park	E of the northern part of S Luangwa	272,000	[c]			
South Luangwa	National Park		905,000	[d]			
Lupande	GMA	SE of South Luangwa	484,000	633,315	76	61,108	161
Sandwe	GMA	S of South Luangwa	153,000	311,812	49	6,334	117
		Total	**7,091,909**		**Even distribution**	**254,839**	**145%**
		Minimum			**Minimum**	**200,000**	**133%**

Notes: [a] Unfortunately GMA boundaries do not line up perfectly with ward boundaries, requiring assumptions about the distribution of the population across the ward. For simplicity, an even distribution is assumed, although this likely overestimates populations in the GMAs, which tend to be farther from main roads and towns. The minimum population and growth rates are therefore also presented. The population growth rates are estimated as the change in population between 2000 and 2010 from the 2010 census. Earlier census data are not available at the ward level. Many wards changed boundaries between the two census years so they were grouped into areas with the same boundaries for both years in order to estimate growth rates for those areas. Census area was left blank in instances where there was overlap between wards. [b] not calculated separately; see Chifungwe ward. [c] not calculated separately - see Lumimba GMA. [d] not calculated because largely not in Eastern province.