Ecosystem services trade-offs in a *Miombo* landscape

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A Research Report submitted to the faculty of Science, University of the Witwatersrand, Johannesburg in partial fulfilment of the requirements for the degree of Master of Science

Faculty of Science

University of the Witwatersrand

Johannesburg

South Africa

January 2019

Contents

DECLARATION

I declare that this Research Report is my own, unaided work. It is being submitted for the degree of Master of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other University.

(Signature of Candidate)

31 Day of January 2019 in Braamfontein (Wits)

ABSTRACT

Ecosystem services are the benefits people obtain from the ecosystems. The ability of the ecosystems to continue to deliver these services sustainably depends on humanity ability to ensure that the extraction rate for each service does not exceed the ability of the ecosystem to replenish the service. This was a desktop study based on an actual 10×10 km landscape near Kilombera in southern Tanzania, mapped and quantified by the Vital Sign Project, was explored using scenarios of future development. Three land use scenarios were mapped in addition to the current (baseline) landscape which were transformed to commercial agricultural landscape, transformed to smallholder agricultural landscape and the untransformed (protected) landscape. The current landscape was used a baseline study with the various identified land covered inferred to various land uses. The flows of the following ecosystem services were water yield, water quality, fuelwood production and biodiversity habitat provision quantified under each land use scenario. The flows of these services were quantified using -Excel^{TM} models. The monetary valuation of each ecosystem service was calculated using the Natural Capital Project where the value of each resource was calculated as the Net Present Value of future flows at a 3% discount rate. This valuation was conducted over a 25 years period as this was the assumed asset life. The analysis revealed trade-offs between water quality and crop production; crop production and biodiversity; crop production and fuelwood production; biodiversity and fuelwood production.

DEDICATION

It is said in an African proverb that ‘it takes a village to raise a child’ I would like to share my appreciation for the people in my life who were always cheering me on through this journey. To my mother Ms EN Rikhotso no words can express enough how grateful I am for your continuous love and support. To my siblings Ms YA Shingange, Mr C Shingange and Mr M Shingange you have always been there encouraging and allowing me to lean on you when it all became too much to handle. To Mr SJ Radebe your love and support has seen me through some tough times and for that I can’t thank you enough. To Mbalenhle Rivoningo I wish my experience will inspire you to always follow your dreams and to never give up,

ACKNOWLEDGEMENT

I would like to express my sincere gratitude to my supervisor, Prof R.J. (Bob) Scholes for his insights, time, knowledge, patience and encouragement. Your contribution to this project and my journey as a scholar can never be emphasised enough. I would like to thank the Global Change Institute for providing funding for this project under the NRF grant Delivering Food Security on Limited Land (DEVIL), which made it possible to pursue this qualification.

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Chapter one – Introduction

1.1 Rationale

"Miombo" is a vernacular name used to describe a woodland dominated by species of the genera *Brachystegia*, *Julbernardia*, and *Isoberlinia* of the subfamily *Caesalpiniodae* (Lupala, et al., 2015; Syampungani, et al., 2014; Sawe, et al., 2014; Ryan and Williams, 2011). Miombo is one of the largest extant savanna ecosystems in in the Sub-Saharan Africa: it covers over 2.7 million km^2 spanning- several countries in south central Africa: Angola, Burundi, the Democratic Republic of the Congo, Malawi, Mozambique, Namibia, Tanzania, Zambia and Zimbabwe (Kachamba, et al., 2016; Ryan, et al., 2010; Mwakwalukwa, et al., 2014, Soka and Nzunda, 2014).

Miombo woodlands play a critical role in the livelihoods of more than l00 million people (Ryan, et al., 2010). They provide a range of ecosystem services, including but not limited to food, fuel, medicine, timber, carbon storage, and water regulation (Kachamba, et al., 2016; Ryan, et al., 2010; Mwakwalukwa, et al., 2014, Soka and Nzunda, 2014). Up to half of the rural income in miombo regions is derived from the woodlands component of the landscapes alone (Ryan, et al., 2010). In addition, they are an important habitat for a variety of fauna and avifaunal species (White, 1980).

In Tanzania, the miombo covers about two thirds of the total 'forest land' and supports 87% of the combined urban and rural livelihoods (Lupala, et al., 2015; Sawe, et al., 2014; Soka and Nzunda, 2014). Despite the essential ecosystem services, they provide, there is a steady conversion of the woodlands to other cover types, such as farmlands, pastures and industrial development (Lupala, et al., 2014). Excessive wood extraction for uses such as timber, fuel or conversion to charcoal, for the generation of private economic returns, leads to the degradation of the woodlands and often paves way to conversion to croplands (Vira et al, 2015; Garedew et al., 2009; Pimentel et al., 1997). Deforestation for cultivation has transformed natural land cover in order to improve food security and provide employment opportunities for the rural poor (Lupala, et al., 2015; Soka and Nzunda, 2014). Degradation of the untransformed woodlands could result in social and economic crisis, because a large percentage of the rural poor depend on these woodlands for their livelihoods (Soka and Nzunda, 2014). Tanzania has a rate of deforestation and degradation (largely in miombo) estimated at 403 000 ha/year (Lupala, et al., 2014).

When intact miombo is transformed, several different land covers and uses can result. The key contrast in miombo-derived landscapes at present is that between smallholder farmlands and large commercial farmlands. The former can be traditional slash-and-burn agriculture (technically called swidden agriculture, known as *chitemene* in the region), or more settled, permanent but still small farms (<5ha) which largely practice a multispecies agroforestry cropping system with maize, cassava and beans as key crops under a canopy of mango trees and other fruits. The large-scale farms grow commercial commodity crops such as maize and soybeans in rotational monocultures. These different land uses have different consequences for the ecosystem services that can be delivered by the landscape, and thus represent different trade-offs relative to the intact miombo and each other. This research quantifies ecosystem services (biodiversity, crop production, fuelwood production, water quality and water yield) trade-offs in a single, typical miombo woodland landscape, by modelling four key ecosystem services delivered from three hypothetical land use patterns applied to the same landscape; in other words, assuming a constant geomorphology and climate. These trade-offs with be based on the economic value of each ecosystem service.

1.2 Literature Review

1.2.1 The ecology of miombo woodlands

Climate in the miombo ecoregion is characterised by alternating wet and dry seasons, with an average annual rainfall between 600 and 1500 mm (Abbot, et al., 1999; Chidumayo, 1997). There are two major forms of miombo woodland: "dry miombo" has a mean annual precipitation (MAP) below 1000 mm and "wet miombo" has MAP above 1000 mm (Chidumayo, 1997). The trees are temperature sensitive, with frost intolerance at minimum temperatures of below -4 ^{o}C (Chidumayo, 1997). The miombo region has soils that are generally eluvial, acidic and nutrient poor in nature on interfluves (Chidumayo, 1997). Wet miombo are slightly more acidic than dry miombo, and its acidity increases with depth (Campbell, et al., 1998; Chidumayo, 1997). The miombo vegetation is absent on alkaline soils underlain by certain geologies, and on the water-logged soils which occur in the fluves (Campbell, et al., 1998; Chidumayo, 1997).

Several major African rivers have their headwaters in miombo woodlands: the Zambezi, Okavango, Congo and Ruaha are examples. The miombo landscape is characterised by a gently rolling topography with sandy interfluves covered by woodland and clayey, seasonally-waterlogged valleys (called *dambos*) covered by species-rich grasslands. The combination provides a steady flow of high quality water into the rivers.

The miombo woodland is appreciated worldwide for its biodiversity and is the location of several of the iconic national parks of Africa, including the Selous Nature Reserve in southern Tanzania, the largest park in Africa (WWF Report, 2012). Timber tree harvesting in miombo is currently unsustainable (Geldenhuys, et al., 1996); this is exacerbated by the fact that only a few, high-value species are used for timber, thus leading to their over-utilization. An example is *Dalbergia melanoxylon*, the African Blackwood used for woodwind instruments (Jenkins et al., 2002). The main non-timber wood use is for fuelwood, principally in the form of charcoal for consumption in the regional towns and cities (Schure et al., 2011).

1.2.2 Land use and change process in Sub-Saharan Africa

Deforestation and degradation in the miombo woodlands is a great challenge. The increased demand for woodland-derived products has resulted in a significant decline in woodland cover and the ecosystem services provided by the woodlands (Bond, 2010). Woodland conversion to other land covers and uses is widespread (Bond, 2010). Masanja (2013) and Mitchard and Flintron (2006) cited population growth as the underlying driving force. Masanja (2013) observed that woodland degradation is caused by land clearing (for both small- and large-scale farming), domestic and industrial use, planting of tobacco, and the procurement of building material. The demand for forest products way surpasses the supply, there by resulting in a decline in natural forest cover (Masanja, 2013; Abbot, et al., 1999). Woodland degradation due to over exploitation of both timber and non-timber products is on the rise (Masanja, 2013; Abbot, et al., 1999; Geldenhuys, et al., 1996). The miombo woodland is contracting due to internal and external socio-economic issues such as population growth, domestic fuel wood, and agricultural expansion, charcoal production and an increased demand for tobacco curing (Masanja, 2013). Proper management of the woodland must be specifically targeted based on particular land use and cover changes and the important services that are needed from the ecosystem. This research will consider the maximization of human well-being to be the guiding principle for choosing different patterns of use in miombo landscapes.

There are a lot of factors that impact on woodland degradation. The reasons for the woodland degradation are since conversion offers higher short-term benefits to communities (Bond, 2010), or because the traditional controls on overexploitation have broken down. But these rates and types of use are unsustainable, as the exploitation of these woodlands will not lead to the alleviation of poverty in the region (Bond, 2010), since degradation decreases the capacity of these ecosystems to provide these essential services. Development for road infrastructure, and mining, tends to attract migrants into areas previous unsettled or sparsely settled (Bond, 2010). Campbell et al., (2007) found that drivers of land use change vary between countries due to economic growth rates, location and land tenure, but three common drivers were identified i.e. agriculture and settlement, fuel wood extraction, and timber extraction.

1.2.3 The concept of ecosystem services

The Millennium Ecosystem Assessment (MEA) created a unifying framework for the effects of environmental change on human wellbeing and ecosystem (MEA, 2003; Carpenter, et al., 2006; Montana, 2018). Ecosystem services are defined as the benefits humans derives from their environment. These are divided into the following top-level categories: provisioning, regulating, cultural, and supporting services (MEA, 2003). This classification and the terms used to describe it have subsequently been changed by Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Diaz et al 2018), but since this work was done under the old framework, those terms will be used. The following changes were made; the provisioning services (now called material contributions) are the physical goods that we derive directly from ecosystems (including agroecosystems) such as food, fresh water, wood, fibre and fuel (MEA, 2003; Khan, et al., 2014). Ecosystems provide a variety of regulating services (now called regulating contributions) such as water purification, climate, disease, and flood regulation (MEA, 2003; Khan, et al., 2014). Cultural services (now nonmaterial contributions) are less tangible, but still often have a market: they include aesthetic, spiritual, educational, and recreational services (MEA, 2003; Khan, et al., 2014). The category of supporting services has been discontinued, since they are now considered to be the ecosystem functions, rather than services. They are those processes that enable the other services to be delivered: they include nutrient cycling, soil formation, and primary production, and are not themselves directly consumed (MEA, 2003; Khan, et al., 2014). There is a postulate positive link between ecosystem services and human wellbeing: an ecosystem that is functioning in its full capacity will be able to provide the resources required by people to sustain their wellbeing. The framework offered by Diaz et al. 2018 came out too recently to have been the basis for the system applied in this study, and changes only the nomenclature rather than the findings. Although there are varying schools of thoughts with regards to ecosystem services and nature's contribution to biodiversity, these frameworks are mutually reinforcing (Strassburg, 2018). Much as there is varying frameworks for explaining the interaction between humans and the environment there are consensus that the loss of biodiversity and degradation of ecosystem services must be prevented (Montana, 2018).

Land use changes (for farming, industrial development, mining, and residential use) results in a trade-off between the ecosystem services supplied by that landscape. There are trade-offs in ecosystem services if the enhancement of one service negatively impacts the capacity of the ecosystem to deliver another service (MEA, 2003). For instance, agricultural food production usually comes at the expense of water quality, increased emission of greenhouse gases (ie. a reduction in climate regulation service), loss of biodiversity and reduction in timber and fuelwood supply (Bennet, et al., 2009; Khan, et al., 2014, Raudesepp-Hearne, et al., 2010). It is also possible for there to be a positive relationship between certain ecosystem services, sometimes called a “synergy”. An example is the restoration of wetland riparian zones, which may both enhance flood protection, and indirectly increase crop production (Bennet, et al., 2009).

There are often contradictory challenges posed by the need to increase food production to feed the growing global population and the need to reduce pressure on the natural environment and the ecosystem services it supplies. This tension led to the concept of “sustainable intensification” (Schulte, et al., 2014). It is defined by Schulte et al. (20014) as “increasing total food production from current global agricultural land area, thus negating increased competition for land with ecological habitats, while reducing or at least decoupling the environmental impacts associated with agricultural production”. This is to limit the trade-offs between the provision of food and other affected ecosystem services. As a result, there is a need to develop a framework that will quantify the supply of and demand of ecosystem services at local, regional and global levels (Schulte, et al., 2014). The increase in agricultural production does not guarantee increased food security (Fischer et al., 2017). As food security means there must be sufficient availability, access to, and utilization of nutritious food (Fischer et al., 2017). As a result, the aim should not only be an increase in production, but it should be ensuring global food security.

The ability to determine the multidimensional relationships that exist between ecosystem services is the key to sustainable management of these ecosystems (Raudesepp-Hearne, et al., 2010). Ecosystem services bundles have been defined as “a set of ecosystem services that appear together across space or time” (Raudesepp-Hearne, et al., 2010). For instance, crop yield (from smallholder agriculture), water quality (yield), erosion control, and soil fertility can be bundled together as they will appear together across space and time (Bennet, et al., 2009; Raudesepp-Hearne, et al., 2010).

1.2.4 Agricultural yields

Poverty and hunger in rural southern Africa have been negatively impacted by recurrent droughts, decrease in output prices, and the removal of subsidies for agricultural inputs (Kuntashula, et al., 2006). An increased market demand for food, fodder and fibre tends to favour intensive agricultural practices. This results in an increased need to explore sustainable agricultural practices with respect to the impacts agro-chemical pollution will have on water quality, biodiversity, and soil (Kuntashula, et al., 2006). In contrast, the increased demand for livelihoods and food among rural poor and the frequently-inadequate transport infrastructure favours smallholder agriculture for local consumption over large-scale commercial agriculture for export. National policies are somewhat schizophrenic on which to favour: export-led economic growth favours the latter, while domestic politics favours the former. The two contrasting types of agricultural systems (smallholder and commercial) will be analysed in this study.

The smallholder agricultural practices in the region are usually family based and used for sustaining the livelihoods of rural communities (Campbell, 1996). This type of agriculture comprises of a mixture of maize, cassava, cowpea, and several other short-duration crops, and tropical fruit trees (typically mango) all planted in a small patch of land, usually less than 5 ha (Fischer et al., 2017; Mwila, et al., 2008). Maize is a staple carbohydrate in most southern and east African countries. The multiple crops and the low level of nutrient and water input lead to low yields for any individual crop. The multicropping strategy (resulting in high agro-diversity) is motivated by risk reduction: the variety is to ensure that the family have food even if one crop produces poor yield. It provides a more balanced diet than a single crop would, or a reliance on store bought food on a restricted budget. It also helps to provide food throughout the year. The poor soil quality typical over much of tropical Africa (and especially in miombo landscapes) constrains the yields achievable by traditional crop production methods (Rattan, 1987).

Commercialisation is determined by the degree of participation in crop and livestock markets, with the exclusion of distress sales by poor farmers (Smalley, 2013; Rattan, 1987). Large scale (typically >100 ha) commercial farms are usually privately owned, either by a family or faming enterprise. The individual farm size, is far above that of the national average, and agronomic decisions are profit driven rather than family nutrition-driven, for instance. Because of high inputs of fertiliser, and use of mechanisation, improved germplasm, pesticides and sometimes irrigation, commercial farms produce relatively high yields per hectare, often ten-fold larger than is typical for small-scale, low input farmers (Smalley, 2013; FAO, 2012). Poor infrastructural development (e.g. roads, rail, electricity, water supplies, and processing plants) means higher start-up capital is needed to ensure smooth running of the operation (Smalley, 2013). This type of farming model will not necessarily decrease food insecurity, despite the higher yields. That is because the harvest is export driven, and the locals cannot afford to buy at the same prices as the international markets (Fischer et al., 2017). It also offers limited work opportunities for local people, because of the substitution of hand labour by machinery, and the requirement for higher skills levels than are often locally available. Sometimes commercial farming uses hired seasonal workers, with a small number of permanent employees, depending on the type of crop or livestock produced (Smalley, 2013). The commercial farms are usually mixed in the sense that each produce a variety of crops and livestock (for instance maize and wheat cultivation, and cattle ranching), with a portion of uncultivated land, but the crops take the form of sequential monocultures of relatively large extent and are spatially separate from grazing areas.

African agriculture is the new buzz word with an increase in foreign direct investment (Smalley, 2013). The miombo region offers an opportunity for commercial farming as it is input intensive, which can compensate for the nutrient poor soils in the region by the use of fertilizers (du Plessis, 2003; FAO, 2012). Commercialising agriculture will increase yields, but it also comes with potentially adverse environmental impacts which can be minimised using sustainable intensification (FAO, 2012). The diversity of land tenure systems on the continent means that there is no one-size-fits-all model for commercial agriculture in the region (Fischer et al, 2017): it may be indigenously owned and operated or foreign owned and or operated; on leased or owned land; be performed by a collective of small farmers or a few individual large farmers; funded by private capital, development aid or state capital.

Tropical sub-Saharan Africa includes 200 million hectares of wetlands ranging from coastal wetlands, river flood plains, lakes, and small inland valleys (Owen, et al., 1995). The dominant form in miombo landscapes is valley bottom wetland called "*dambo*" in Tanzania, "vleis or *mapani*" in Zimbabwe. It is similar to the "*fadamas*" in Nigeria and Sierra Leone (Owen, et al., 1995). In this research these seasonally flooded grasslands will be referred to as *dambos*. Breen et al. (1977) in Kuntashula et al (2006) defined *dambos* as 'a wide and low-lying, gently sloping, treeless, grass-covered depression, which has a water table that for most of the years in the upper 200- 1000 cm of the soil profile from which it drains into streams". They are usually partly used for crop production in smallholder agricultural practices (Owen, et al., 1995), while Kuntashula et al. (2006) advices that serious consideration must be given to more intensified farming method in *dambos*.

There are negative impacts to the increased use of *dambos* for cropping, which will happen because of recurrent droughts and the poor performance of rain fed crops in the region (Kuntashula, et al., 2006). Their location at the head of the drainage system (Kuntashula, et al., 2006; Owen, et al., 1995) means that application of pesticides can result in the contamination of water sources in the catchment area (Kuntashula, et al., 2006). Excessive fertilizer application would also increase nutrient loading by nitrogen and phosphorus in the rivers, potentially leading to eutrophication, and tillage close to the river increases sediment yield, which reduces water quality. The drainage of *dambos* for dryland crop production interferes with their flood regulation function.

1.2.5 Water yields and quality

Water yield is defined as "the total outflow from all or part of the drainage basin through either surface channels or subsurface aquifers within a given time period" (Chiew, et al., 2003). The usable water yield must consider the pattern of flow over time, but also water quality, in other words the contamination from sediments and fertilizers in the catchment area (Campbell, et al., 1998; Chiew, et al., 2003). Sediment loss from the catchment area is because of poor erosion control, which is caused by land use changes such as deforestation or inappropriate agricultural practices such as large bare fields, on steep slopes without proper contouring. When fertilizers (including manures) are applied to crops, not all the nutrient supplied is taken up by the crops. Some is lost to the atmosphere and some leaches into the water. The fraction which ends up in the river system depends on the amount applied in relation to the crop demand (Chiew, et al., 2003, Chidumayo, 1997).

Water yield and water quality are modelled using different approaches. The annual flow from a large catchment can be modelled using very simple approaches such as the Budyko curve, which represent the relationship between runoff to the ratio between precipitation and evaporation (the aridity index P/Ep) in a supply-demand framework (Gerrits, et al., 2009). These models are appropriate for moisture and/or energy constrained locations, such as sub-humid landscapes (Gerrits, et al., 2009), and for large space and time scales, (annual and whole landscapes, as considered here). Analytical derivations of the Budyko curve are based on a conceptual model that uses measurable parameters. The curve can be generalised (Fu, 1981), by introducing a parameter which can be adjusted to represent different land use options (Teng, et al., 2013).

Water quality in agricultural landscapes can be modelled based on sediment yield due to soil erosion under various land uses (Mulemgera, 2008). Soil erosion emanating from heavy grazing, unsustainable cropping practices, and declining woodland cover results in increased sediment load in streams and rivers (Mulemgera, 2008). The water resource infrastructure then experiences rapid siltation (Mulemgera, 2008). Mulemgera (2008) used empirical soil erosion models to predict sediment yields from watersheds following two approaches. The first used a sediment delivery ratio based on plot-scale measurements, upscaled to the catchment. The second approach divided the watershed into morphological areas where all elementary factors of selected models can be evaluated (Mulemgera, 2008). The use of nitrogen and phosphorus-based fertilizers in agriculture also negatively impacts water quality (Campbell, et al., 1998). Leakage to the river can be represented as a fraction more than crop demand.

1.2.6 Fuelwood and charcoal production

Woodfuel production in the miombo region plays an important role as it is used as a domestic and industrial energy source (Abbott, et al., 1999). Fuelwood and charcoal production for both domestic and industrial use are the main drivers of deforestation in the miombo ecoregion (Campbell, 1993). The production of charcoal is largely to satisfy the domestic energy needs of a growing urban population, even though it is not as energy efficient as the direct use of firewood on a 'tree-to-table' basis (Kutsch, et al., 2011). The commercialisation of charcoal (Kutsch, et al., 2011) combined with the ease with which it can be transported (Campbell, 1993; Kutsch, et al., 2011) has changed the spatial patterns of deforestation in the woodland. The difference between fuelwood production and charcoal production is that the former is mostly harvested in smaller quantities as dead wood, while the latter is produced in large quantities by felling live trees (Kutsch, et al., 2011). Currently, charcoal production is an unsustainable use of the woodland with the cleared fraction being greater than the fraction able to re-grow (Kutsch, et al., 2011). This causes an impairment in the capacity of the ecosystem to supply all the other services. The migrant charcoal producers are not invested in these other ecosystem services provided by the woodlands, hence their disregard for sustainable use of the woodland (Kutsch, et al., 2011). There is a need for better regulation on the harvesting of trees for charcoal in order to encourage sustainable harvesting and reduce the rate of deforestation (Campbell, 1993; Kutsch, et al., 2011). This research calculates fuelwood production in the woodlands, acknowledging that some of the fuelwood may be converted to charcoal.

Fuelwood production models are based on the annual rainfall, the woody basal area and the stem size distribution (Vital Signs thread document, 2014). Scholes and Shackleton (in review) developed a framework for a tree production model using annual basal area growth rate of trees in the South African savanna biome. This model was created by measuring the change in circumference of each tree stem which was subsequently converted to stem cross sectional area. According to the Vital Signs Wood Fuel Thread Document (2014) the tree production model in Scholes and Shackleton (in review) is developed from woody biomass using the Colgan et al., (2012) algorithm based on cover and height and wood density.

1.2.7 Climate regulation

The increasing level of greenhouse gases (principally carbon dioxide, methane, and nitrogen oxide) in the atmosphere is the cause of global climate change (Bond, 2010); an important source of these emissions is agriculture and land use change. Human activities have seen increased levels of deforestation and forest degradation, including in the miombo woodlands. The clearing of forests has impaired the ability of the forest to regulate climate, since it removes a carbon sink and creates a carbon source (Bond, 2010; Shelukido, et al., 2014). Management of these woodlands to preserve or enhance their carbon sink, and avoid carbon losses, would contribute to the mitigation of climate change. This is the intent of the Reduced Emissions from Deforestation and Degradation (REDD+) initiatives (Lupala, et al., 20014). The intact woodlands stores tons of carbon per hectare (Lupala, et al., 2014). The carbon is stored principally in the soil, with a lesser contribution from tree biomass. The millions of diverse hectares covered by the miombo woodlands make it difficult to give an accurate estimate of the total tree biomass or soil carbon stock (Tamene, et al., 2016). Initiatives to address the problem of deforestation and forest degradation in miombo have over the past two decades been based on Community Based Forest Management (CBFM) and Participatory Forest Management (PFM) (Lupala, et al., 2014; Lupala, et al., 2015). The REDD+ initiative offers cash payment per tonne of carbon equivalent that is stored in the natural forest (Bond et.al, 2010), thus encouraging forest conservation. For the REDD+ initiative to be a success it needs to offer benefits that will be equal to or exceeds those that are offered by the benefits derived from deforestation (Bond, 2010)

1.2.8 Biodiversity habitat in the miombo woodlands

The miombo ecoregion forms part of the Zambezian phytochorological region (White, 1980; Campbell, 1996; Chidumayo, 1997). It home to a spectrum of savannas, woodlands and wetlands. The woodland component typically comprises of 10-20 m high, closed canopy trees that are broad leafed (Campbell, 1996; Chidumayo, 1997). These tree species are mostly of the dominant genera *Brachystegia*, *Julbernardia*, and *Isoberlinia* (Campbell, 1996; Chidumayo, 1997; Lupala et al., 2005; Syampungani et al., 2014; Sawe et al., 2014; Ryan and Williams, 2011). It also includes *mopane* (*Colophospermum mopane*), thickets and woodlands and *Munga*- open dry savannas dominated by *Acacia* (now *Vachellia* or *Senengalia*) and *Terminalia* species (Chidumayo, 1997). The region has an estimated 8500 higher plant species, of which over 54% are endemic (Campbell, 1996; Chidumayo, 1997). These different species are distributed across the region, with Zambia having the highest tree diversity (Campbell, 1996; Chidumayo, 1997).

The miombo region is home to some important large mammal populations, including elephants, rhinos, lions, buffalos and leopards, Lichtnestein's hartebeest and staple antelope (Campbell, 1996; WWF Report, 2012). It also has a diverse and endemic avifauna including Grey Tit, Shelley's Sunbird, Rock Thrush, and Stierling's Woodpecker (Campbell, 1996; WWF Report, 2012). Most animal diversity has been concerned with larger mammals (Campbell, 1996; WWF Report, 2012; Chidumayo, 1997). Habitats on the river terraces have more palatable grasses, and a greater animal biomass, while the upland landscapes are nutrient poor and support smaller but more diverse animal populations (Campbell, 1996).

1.2.9 Bundles of ecosystem services

It is crucial to determine a method of managing multiple ecosystem services sustainably across a landscape (Raudsepp-Hearner et al., 2010). This is a key challenge with regard to ecosystem management. Society is often faced with the dilemma of having to enhance the supply of some ecosystem services which are often provisioning services (such as food and fibre), which often leads to a decline in the ability of the ecosystem to supply other services such as the regulating (such as nutrient cycling and flood regulation) and cultural services (such as recreation) (Raudsepp-Hearner et al., 2010; Bennet et al., 2009). This key challenge can be addressed by identifying synergies and trade-offs that exists among ecosystem services at different scales (Raudsepp-Hearner et al., 2010).

There is a complex relationship between ecosystem services across space and time in a landscape (Bennet et al., 2009; Bennet et al., 2009; Martinez-Harmz and Balvanera, 2012). This existing relationship or interaction among the services results in ecosystem services bundles (Raudsepp-Hearner et al., 2010), which are defined as a set of ecosystem services appear together across space and time repeatedly (Raudsepp-Hearner et al., 2010; Bennet et al., 2009). These bundles of ecosystem services can be used to identify synergies and trade-offs, to ensure effective management by putting strategies in place to limit the trade-offs and enhance the synergies (Bennet et al., 2010). Raudsepp-Hearner et al. (2010) and Bennet et al. (2009) identified two causes of the interaction that exists between ecosystem services which are because of; multiple services responding to the same driver of change or when interactions among services themselves causes changes in one service which then alters the provision of another.

Land use/ land cover are often used to determine these complex interrelationships among ecosystem services, which tends to simplify human influence on a landscape (Bennet et al., 2010; Raudsepp-Hearner et al., 2010). Bennet et.al (2010) proposed that the area where ecosystem services of interest are saturated be identified with respect to other services when deciding on whether to invest or not. For instance, nutrient runoff emanating from agricultural production can be reduced by reducing the quantity of fertilizer used, by substituting it with conservation tillage or maintaining the riparian zones, all which will not result in undue losses of food production (Raudsepp-Hearner et al., 2010). Being cognisance of the fact that there are both positive and negative interactions among ecosystem services, this paper only identifies trade-offs that exists in ecosystem services in a landscape. These trade-offs are determined by quantification of the services supplied by each landscape. Determination of the 'optimum' level of such trade-offs requires that the competing services be expressed in a common unit. This is typically monetary value, and this step is called ecosystem services valuation.

1.2.10 Valuation of ecosystem services

Valuation of ecosystem services was developed as a vehicle to integrate ecological understanding and economic considerations, to redress the neglect of ecosystem services in policy decision (Chee, 2004). Valuation is about trade-offs towards achieving a goal (Costanza et al., 2014). The value of ecosystem services can be thought of as the relative contribution of ecosystems to that goal, this contribution can be assessed in a variety of ways. (Costanza et al., 2014). Ecosystem services valuation attempts to attach monetary value for the benefits that nature provides humanity (Chee, 2004; Khan et al., 2014; Farber et al., 2002). Four types of capitals (or in other schemes, three or five or six, being various combinations and splits of the same basic idea) are recognised by economists, financial, human, manufactured and natural capital (Chee, 2004). The interaction between these capitals is what results in human wellbeing

(Constanza et al., 2014). Ecosystem services are mostly related to natural capital (Chee, 2004; Khan et al., 2014). Ecosystem services represent a flux (an annually-renewed supply) of some good or benefit which has a value. In other words, ecosystem services have units of benefit delivered per unit area per unit time, which can be converted to monetary value if the price per unit can be established. If the net present value of future benefit flow is calculated for a given a social discount rate, and summed over all ecosystem services, it is a measure of Natural Capital (UNU-UNEP, 2014). Development economics has seen the transformation of natural capital into consumer products and services (Chee, 2004). Ecosystem services tend to be included in categories of open access and pure public services, meaning no one owns or has rights to these services and others cannot be excluded from using them (Chee, 2004). They frequently fall outside the commercial markets, which impedes the establishment of price (Chee, 2004; Khan et al., 2014; Hein et al., 2006; Robinson et al., 2013).

The growing concern about the pressure that humanity is exerting on the earth's resources has increased the focus on environmental sustainability (Robinson et al., 2013), thereby creating a need for the inclusion of ecosystem services in decision making tools such as the Gross Domestic Product (Chee, 2004; Khan et al., 2014; Robinson et al., 2013). The difficulty in the valuation of ecosystem services has resulted in the only value being bestowed on the benefits that have a direct and usually market-related impact on human welfare (Robinson et al., 2013). Note that when the value of ecosystem services flows is used to help guide land use decisions, it is the *change* in ecosystem services or natural capital that is considered, rather than the absolute value, and this reduces some of the concerns regarding assumptions in the valuation process. There are a variety of methods used for the valuation of ecosystem services. The World Bank uses the Net Present Value approach, and the Inclusive Wealth Report (IWR) uses estimates that are based on quantity price (Khan et al., 2014; Hein et al., 2006). The former approach is the one used in this research.

The valuation technique applied in this research report will include provisioning services, cultural services and regulating services only, excluding the supporting services so as to avoid double counting (Khan et al., 2014; Hein et al., 2006), over the full extent of a spatially-defined landscape of 10×10 km, for a hypothetical year.

To value ecosystem services, the service flows must first be quantified. For the production services this involves quantifying the flow of goods harvested in an ecosystem using physical units (Khan et al., 2014; Hein et al., 2006). The non-market cultural services are dependent upon the value humans place on an ecosystem, thus making them highly subjective (Khan et al., 2014; Hein et al., 2006; Constanza et al., 2014). But many cultural services are marketed, such as nature-based tourism, and these can be more easily and robustly valued. Regulating services require a spatially explicit analysis of the bio-physical impacts of a service on the environment in the ecosystem or its surrounding areas (Khan et al., 2014; Hein et al., 2006). The value attributed to these ecosystem services is dependent upon the stakeholders benefiting from these services (Hein et al., 2006). It has been proposed that valuation of ecosystem services will improve the understanding of trade-offs in ecosystem services (Chee, 2004; Constanza et al., 2014; Farber, 2002). Specifically, it offers the possibility that services of different types can be directly compared in terms of common units. The optimal landscape design, in this ‘utilitarian’ view, is then the one which delivers the most value to society.

1.3 Aim

- The aim of this study is to quantify the yields of four ecosystem services from a typical miombo landscape, to inform trade-off analysis, under three land use scenarios: untransformed (protected), transformed to commercial agriculture, and transformed to smallholder agriculture.

1.4 Objectives

- Model the delivery of four ecosystem services under the four-land use scenario.
- Quantify the monetary values of those services.
- Identify the trade-offs that exists between the services across all scenarios.

Chapter two – Methods and Materials

2.1 Methodological approach

This study is based on data collected by the Vital Signs Project (www.vitalsigns.org). This data provided the baseline information the land cover within the chosen landscape, from which various land-uses can be inferred. The chosen landscape (Vital signs L19) is a 10×10 km^2 area, in the Kilombero district in the Morogoro region in Tanzania. This landscape forms part of the Southern Agricultural Growth Corridor of Tanzania (SAGCOT), which is a multi-stakeholder initiative launched in 2010, aimed at poverty alleviation, promotion of food security, stimulation of economic growth and developing Tanzania as a major player in food export. This corridor covers approximately 300 000 km^2 stretching between the Indian Ocean and the borders of Zambia, Malawi and the Democratic Republic of the Congo.

The various land use scenarios were created from the baseline (current landscape) land use covers sourced from the Vital Signs Projects. The approach used was the creation of shape files from the JPEG pictures of the current landscape enorder to enhance the quality of the maps. The type of scenario planning used was the one highlighted in Peterson et al. (2003) where investigate the outcomes of different ecological management regimes. The scenarios were based on the premise that it a structured account of a possible future, in essence it is futures that could rather than futures that will be (Peterson et al., 2003).

The data only offers information on land cover. Land uses in the landscapes were compiled by the Vital Signs Project from surveys they conducted. The land uses were inferred based on the land cover maps taken at <1m resolution from the Vital Signs project. All suitable agricultural land was assumed to be used for maize- soy rotation agricultural production in the commercial virtual landscape, and for agroforestry production (mangoes, cow peas, maize and cassava) for the smallholder virtual landscape. In all landscapes, the area covered by roofs was assumed to represent the areas of human settlements. For the protected virtual landscape, the residential areas assumed to be indicated by roofs on the maps were fenced out to reduce uncontrolled access to the protected area.

2.2 Study area

The study area was in the Kilombero Cluster where landscape 19 is found has a broad floodplain extending southeast to northeast between the Udzungwa Mountains and Mahenge Highlands. This floodplain is watered by Ruhudji, Mnyera and Pitu Rivers. It is one of the largest freshwater floodplains in East Africa. As a result, it has ecological value because it provides habitats to many wetland plant and animal species and cleans and regulates water flow for downstream users. It has a high proportion of the world's puku antelope populations and has several bird species that are endemic to the valley. In the 1980s thousands of wildlife species would move into the area during the dry season, but these numbers have since declined. The area is characterised by soils that are suited to agricultural production of a variety of crops, as a result the area already has many smallholder and commercial farmers. The landscape has the following land cover classes: agricultural land, woodlands, shrubs, bare soil (inferred to be fallow or abandoned agricultural land), roads, water and bare rocks (including gravel). These diverse land covers mean there are diverse ecosystem services supplied by the landscape, in varying proportions. This study hypothesises that existing land cover and land use in the landscape will change over the next decades in response to the planned changes in road infrastructure under the SAGGOT, which will facilitate transportation of agricultural inputs and products for commercial farming, or alternately may attract new smallholder farmers.

Figure 2.2.1: Map of the study area of Kilombera cluster in Morogoro (Vital Signs atlas, 2014)

2.3 Data Collection

2.3.1 Land cover

This was a desktop study with the initial landscape identified from the open-access data collected by the Vital Signs Project, which has operated in Tanzania since 2012. This landscape was chosen for its diverse land covers as part of the Vital Signs Project. At <1 m resolution over the 10x10 km landscape, the following eight land covers have been mapped: roads, roofs, trees, shrubs, agricultural land, soil, rock/gravel and water. The 8 land covers on the baseline or current landscape were mapped from the Vital Signs Project. The roofs represent residential areas. From the Vital Signs Project remotely sensed land cover maps were sourced to identify the distribution of the different land uses across the landscape. Since the available images were not clear as they were in a JPEG format. Shape files were created from ArcMap10.4.1. These shapefiles were then used to estimate areas of the different land covers using ArcMap as polygons to create the commercial, smallholder and protected land use scenarios. The various land use scenarios per landscape in hectares are given on table 2.3.1 below.

Table 2.3.1 Land areas occupied by various cover types under current land use in the Kilombera landscape, and the eventual area occupied under three hypothesised future scenarios. Note that a 10×10 km landscape is equivalent to 10 000 ha.

	Croplands	Road	Roof	Trees	Shrubs	Bare soil	Rock/Gravel	Water	Total Area (ha)
Current	5262	79	9	1578	3004	50	66	1	10 000
Commercial	8100	121	9	629	1050	50	40	1	10 000
Smallholder	8100	81	9	669	1050	50	40	1	10 000
Protected	__	90	9	4076	5714	50	60	1	10 000

Since the mapping was conducted using an automated algorithm, there was a small category (1%) called 'shadows'. It was redistributed to the 'tree' and 'shrub' classes.

2.3.2 Crop yield

Crop yields in hectogram (a metric unit of mass which is equal to one hundred grams) per hectare (Hg/ha) for maize, soybeans, cassava, mangoes, and water melons were sourced from the FAOStat database for Tanzania, whereas that for the commercial landscape South African data was used. This was done because South Africa because of data scarcity for Tanzania for all crops. The area planted was estimated using the ArcMaps shapefiles developed using the current landscape the Vital Signs Project as a baseline.

2.3.3 Water yield and water quality

Water yield data was mainly obtained from the FAO Aquastats databank (www.fao.org/nr/water/aquastat/data/query/index..html?lang=en, 2017), since there is an issue of water data scarcity for the area. Tanzania is a data scarce country with respect to mean annual actual evaporation and mean annual potential evaporation. A variety of data was provided by scholars such as Nyezi et al., (1981), Zemadim et al., (2011), Trambauer et al., (2014), and Dag and Woodhead (1970). Mean annual precipitation was sourced from FAO Aquastats. Water yield was then estimated using the Budyko equation with a landcover-dependant yield value, w. This parameter depended on vegetation cover, soil permeability, soil depth and soil permeability (Zhang et al., 2004), but soil depth and soil permeability will be the same across each scenario. The equation is given in subsequent section. Sediment yield was calculated using the Modified Universal Soil Loss Equation (MUSLE), with the input data estimated from Gemechu (2016), and Nitrogen flow were sourced from FAO Aquastat.

2.3.4 Fuelwood

Fuelwood data was sourced from the Vital Signs Project measurements of tree circumferences within plots in the landscape. The tree circumference was then converted to diameter at breast height, and then to tree biomass. The number of tree stems of each diameter was also sourced from the Vital Signs Project. Mean annual precipitation was sourced from FAO. Together with the tree biomass, these variables allowed the annual wood production to be estimated using the equations documented by Vital Signs.

2.3.5 Biodiversity Intactness

Species richness data was collected from a variety of sources with bird diversity sourced from Birdlife Africa, mammal diversity was sourced from Barnes and Douglas-Hamilton (1982), Institute of Applied Ecology (2017) and WWF Ecoregion and plant diversity was sourced from African Plant Richness Databases (NEES-Uni.Bonn, 2017).

2.4 Data Analysis

Essentially the same simple predictive models were used in each real and hypothetical landscape to predict the average flow of ecosystem services once a new configuration had been reached. What varied between the landscapes was the proportions of different activities, and in some instances, the parameters used in the models (Table 2.4.1).

Table 2.4.1 Table of parameters and variables used in the four models for the current, commercial smallholder and protected landscape. The parameters are defined in subsequent text.

Parameters in models	**Current Landscape**	**Commercial landscape**	**Smallholder landscape**	**Protected landscape**
Crop yield				
Maize yield (y_i) (kg/ha)	1267	2403	1267	—
Soy beans yield (y_i) (kg/ha)	—	1207	—	—
Cassava yield (y_i) (kg/ha)	7754	—	7754	—
Mango yield (y_i) (kg/ha)	13150	—	13150	—
Cow peas yield (y_i) (kg/ha)	604	—	604	—
Area planted to crops (A_i) (ha)	5262	8100	8100	—
Water yield				
Mean Annual Precipitation (mm)	1071	1071	1071	1071
Mean annual potential evaporation (mm)	1938	1938	1938	1938
Catchment characteristics function (w)	3.00	2.85	3.05	3.15
Water quality				
q_p (Peak flow rate) (m^3/sec)	1.0	1.0	1.0	1.0
K (Erodibility)	1.0	1.0	1.0	1.0
C (Crop Management/ land cover)	0.9	1.0	0.85	0.8
S (Slope steepness)	1.0	1.0	1.0	1.0
L (Slope length)	1.0	1.0	1.0	1.0
P (Erosion control practice)	0.9	1.00	0.85	0.35
Area planted to crops (ha)	5262	8100	8100	1000
Fuelwood Production				
Area used for fuelwood production	1772	631	717	—
Bmax (potential basal area)	16.5	16.5	16.5	—
c is the dimensionless sensitivity to annual variations in rainfall	2.46	2.46	2.46	—
a is a term accounting for the allometry of the trees	3.2	3.2	3.2	—
Species richness used in calculating BII				
Mammals	11	11	11	11
Birds	28	28	28	28
Reptiles	14	14	14	14
Amphibia	8	8	8	8
Plants	30	30	30	30
Area reserved for biodiversity (ha)	2658	945	970	9687

Table 2.4.2 Variables used in the estimation of ecosystem services valuation

Landscapes	Commodities	Output sale price	Input Costs	subsidies	Taxes
Current / Smallholder Landscape	Maize	R 1 500 per ton	60% of output sale price	0	0
	Cassava	R 1 300 per ton	60%	0	0
	Mangoes	R 1 200 per ton	60%	0	0
	Cow Peas	R 631 per ton	60%	0	0
	Water Services	R 24.04 per Kl	75% (Smallholder) and 70%(Current)	20%	28%
	Fuelwood	R1 200 per tonne	75% input price	15%	28%
	Biodiversity	R 65 per person	R 834 per ha	20%	28%
Commercial Landscape	Maize	R 2 600 per ton	75% of output sale price	10%	28%
	Soy Beans	R 3 200 per ton	75% of output sale price	10%	28%
	Water Services	R 24.03 per Kl	80%	20%	28%
	Fuelwood	R1200 per tonne	R80% of input price	15%	28%
	Biodiversity	R 50 per person	R 834 per ha	20%	28%

Protected Landscape	Water Services	R24.03 per kL	40%	R20	28%
	Biodiversity	R 100 per person	R834 per ha	20%	28%

The values given above on table 2.4.2 for input costs are the total costs of the following inputs; labour, marketing, transportation, packaging, fertilizers (commercial agricultural production), manure (smallholder agricultural production), pesticides (commercial agricultural production), irrigation systems, insurance, electricity, maintenance and services of vehicles and machinery. It was assumed that the bulk infrastructure necessary to ensure that all services are rendered are already in place. This is due to the high costs associated with investments in bulk infrastructure projects which will make it difficult to estimate the values of each services. For the smallholder agricultural production, it was assumed that the market is informal and as a result does not have subsidies and tax incentives. The commercial agricultural production system was assumed to be input intensive and driven by export prices. The value of the water resource was estimated based on the monetary valuation of water abstracted for public water supply (Khan, et al. 2014). This means the estimate will be lower than the actual value. Resource rent for abstraction had to take into account collection, treatment, supply and labour. It was assumed that 50 million litres of water were abstracted from each landscape per annum. The input price sale for fuelwood was computed at the 13% efficiency for conversion to charcoal. The input costs associated with biodiversity services was calculated by dividing the profits for Tanzania's Tourism Sector (which is largely tourism based) with the total protected area per hectare for Tanzania. It was assumed that 38% of the total terrestrial area in Tanzania was protected (The United Republic of Tanzania, 2015).

2.5 Modelling

All the models used to quantify the supply of four ecosystem services in a virtual landscape are executed in a spreadsheet (Excel™). Crop yields in tonnes for all landscapes were computed as the product of crop yield in kg/ha and the area planted to crop $Y = y_i \times A_i$ where Y is crop yield and y_i is crop yields per hectare and A_i is the area planted to crop. The value of each crop was computed using the Natural Capital Project (Khan et al., 2014) and the value of each type of crop (maize, soy etc) was calculated and add these together to get the total per landscape

The actual evaporation for Tanzania (Ea) was calculated using $\frac{E_a}{P} = 1 + \frac{E_p}{P} - [1 + (\frac{E_p}{P})^w]^{1/w}$. Where w is a unitless parameter that depends on catchment characteristics such as vegetation cover, soil depth and soil permeability. The values for w estimated for each landscape are given in table 2.3.1. estimated from Zhang et al. (2004). Water yield (runoff) was then computed from water balance equation $P = E_a + Q$, where Q is the total runoff from the catchment. Water quality was computed using the Modified Universal Soil Loss Equation (MUSLE) $Y = 11.8 \times (Q \times q_p)^{0.56} \times K \times LS \times C \times P$, where the modifier terms K, L and S were assumed to be equal to 1 as these parameters will not vary in each landscape. Sediment yield in tonnes was then converted to total suspended solids (TSS) in mg/L by dividing by the water yield computed from the Budyko equation above.

The fuelwood data was analysed by first computing the circumference of trees to diameter at breast height in meters. The number of stems of each diameter were counted. The mean annual precipitation was used to estimate the potential basal area using the following; $B_{max} = 0.03(P_{average}-100)$ for $P_{average}$ <650 mm, 16.5 for $P_{average}$>650 mm, and 0 for $P_{average}$<100 (Scholes and Shackleton, in review). The woodfuel model was computed using the Scholes and Shackleton model

($\Delta M = a \bullet B \bullet c\Delta P \bullet (1-B/B_{max}) \bullet (1.0+42.0e^{-0.29d})/100$). The mean basal increment for the landscape was calculated to be 1.73% per annum using this equation. Using the assumption that above ground biomass to be 59.5 tonnes per hectare for Tanzania (Malimbi et al. 2016), this value was multiplied by 0.0173 to give an annual wood production per hectare of about 1 tonne.

Biodiversity Intactness Index was calculated using the richness of various groups of species (mammals, birds, amphibians and plants), and the impact on their abundance under different land covers. The data was analysed using the product sum of species richness (R_{jk}), Area of land use k in ecosystem j (A_{jk}) and population of group i under land use activity k in ecosystem j (I_{ijk}) divided by the product sum of R_{jk} and A_{ij} as articulated in the Scholes and Biggs (2005) paper. It was assumed to be 1 in intact woodland or wetland, 0.4 in small-scale multispecies agroforestry, and 0.1 under commercial cropland and settled area.

2.6 Ecosystem Services Valuation

The valuation of the ecosystem services delivered per landscape was calculated using the natural capital method by the means of a net present value of future flow of ecosystem services.

$$Value\ of\ component = \sum_{t=0}^{n} \frac{resource\ rent\ in\ year\ t}{(1+r)^t}$$

Resource rent was computed as follows:$resource\ rent = (output\ sale\ price) - input\ costs - subsidies + taxes$. The value of each component was computed as over a period of 25 years to ensure simplicity and comparability with other international studies. The discount rate (r) was assumed to be 3% per annum (i.e. 0.03) to account (Khan et al., 2014). This discount rate converts the expected stream of resources into the current period estimate of the overall value (Khan et al., 2014) The inputs for commercial farming considered were fertilizer, pesticides, transportation, labour (workforce), seeds, irrigation and mechanised equipment. The values given are acknowledged to be very approximate estimates, performed to show that the calculations can in principle be done, but with no confidence expressed in their reality in the actual situation analysed. Due to the lack of data for Tanzania the prices used were based on the South African market commodity prices as the prices for Tanzania and other East African Countries were not available for all commodities valuated in this paper. A tax rate of 28% was applied to all ecosystem services (South African Revenue Services, 2018). For smallholder agricultural production it is assumed that the market is not formalised and as result there are no subsidies and tax incentives.

The valuation of biodiversity only considered the prices entrance fees to access such space. It did not consider the 'leverage effects' such as money for transportation or accommodation that people are willing to pay access such spaces. This was due to the difficulty in accessing such values.

Chapter three- Results

3.1 Results

Table 3.1.1 Table of quantifying flows or delivery of ecosystem services per landscape.

Land use scenario	Ecosystem Services						
	Water quality (mg/L)		Water yield (m^3)	Fuelwood production (tyr^{-1})	Food Production (tonnes)		Biodiversity Intactness Index (BII) (%)
	TSS (Total Suspended Solid)	Nitrogen					
Current	13.68	8.32	9 800 000	182 401	Maize	3 333	50.00
					Cassava	6 920	
					Mangoes	1 224	
					Cow peas	318	
Commercial	28.87	27.81	11 800 000	64 931	Maize	19 465	27.30
					Soy Beans	9 783	

Smallholder	19.33	17.26	10 300 000	73 779	Maize	5 132	42.05
					Cassava	10 651	
					Mangoes	1 884	
					Cow pea	4 48	
Protected	0.86	0	9 000 000	0	0		98

Table 3.1.1 above illustrated the flow or delivery of each ecosystem services per landscape. The food production in tonnes was calculated for each crop type because each crop will have different yields depending on the method of production employed and the area planted. Commercial agricultural production had higher yields as it is input intensive whereas smallholder agroforestry had lower yields as they are not input intensive and relies on natural rainfall for irrigation. The total suspended solids (TTS) was higher under the commercial landscape and lower in the protected landscape. This was because of low erosion control in the commercial landscape as compared to the higher flood control in the protected landscape offered by the woodlands, nitrogen yield also exhibited similar characteristics. Water yield was higher in the commercial landscape and lower in the protected landscape. Fuelwood production was higher in the current landscape and a larger area was reserved for this purpose in this landscape. It was the lowest at zero (0) for the protected landscape as this was prohibited in this landscape.

Table 3.1.2: Tables of the total monetary values of each ecosystem service under the four land-use scenarios. The valuation was computed over a 25-year period accounting for the discount rate.

Ecosystem Services	Value of Each Service in Rand per landscape			
	Current	Commercial	Smallholder	Protected
Food Production	R120 897 956	R675 494 236	R185 890 426	—
Fuelwood Production	R172 899 520	R61 580 061	R69 935 766	—
Water Services	R9 513 061	R3 724 329	R7 300721	R15 043 909
Biodiversity	R14 580 890	R7 915 727	R7 350 318	R40 159 848

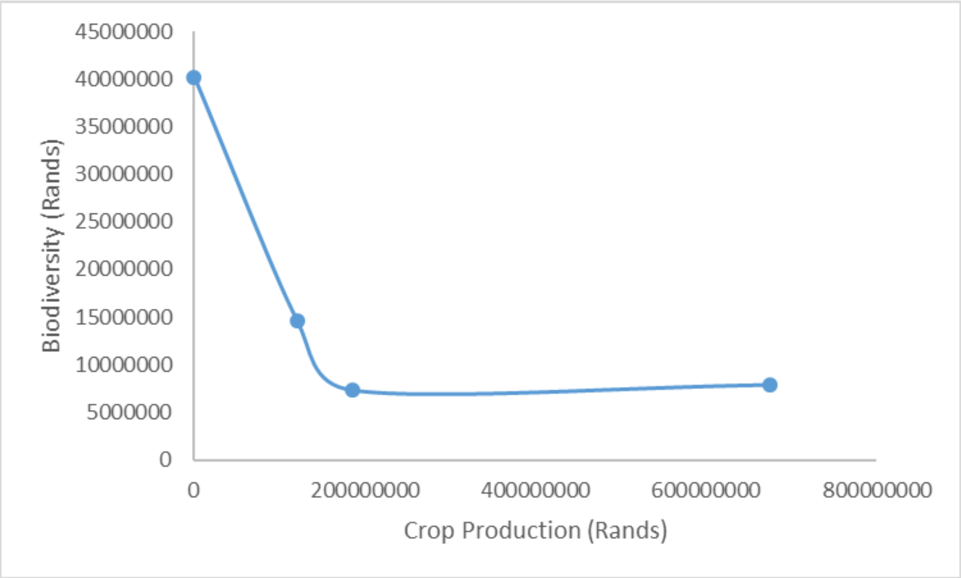

Figure 3.1.1: Graph of economic value of biodiversity versus crop production in Rands under the four land-use scenarios.

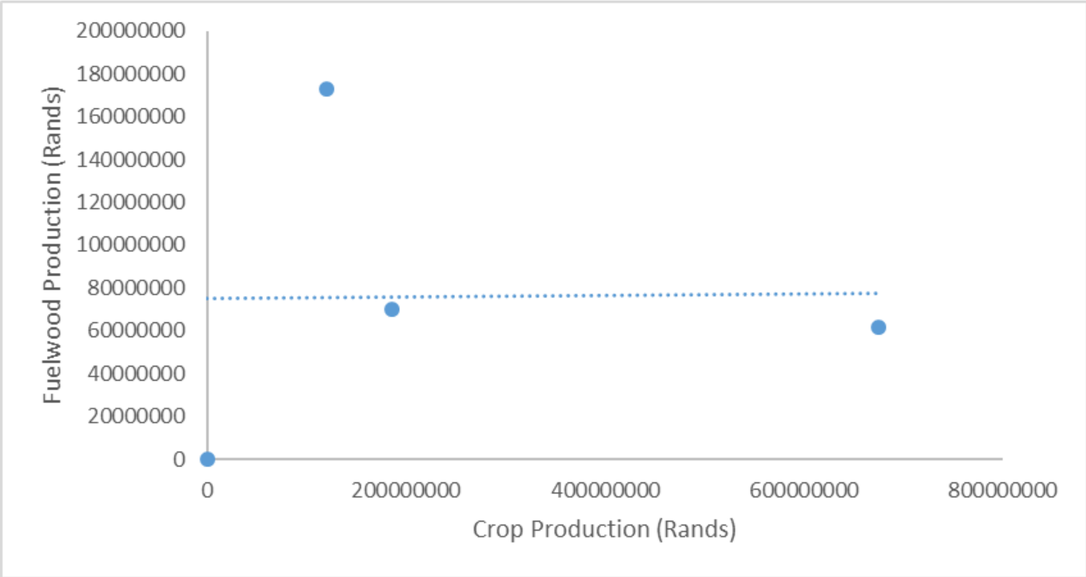

Figure 3.1.2: Graph of economic value of fuelwood production versus crop production in Rands under the four land-use scenarios.

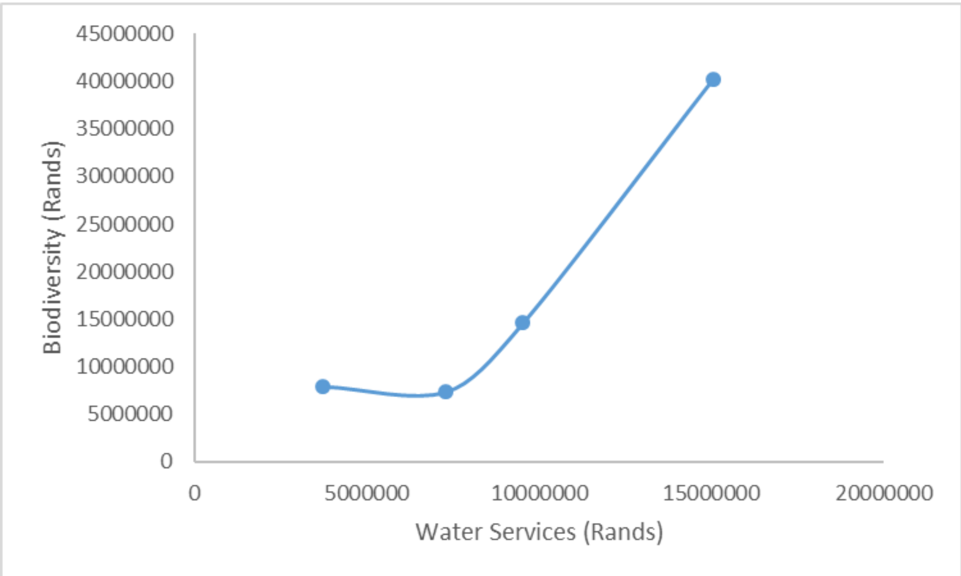

Figure 3.1.3: Graph of economic value of biodiversity versus water services in Rands under the four land-use scenarios.

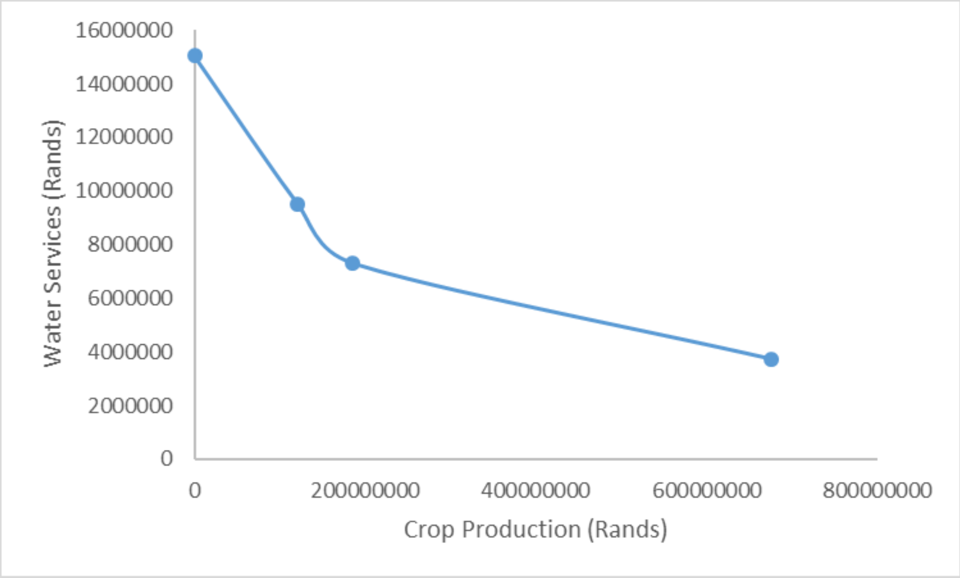

Figure 4.1.4: Graph of economic value of water services versus crop production in Rands under the four land-use scenarios.

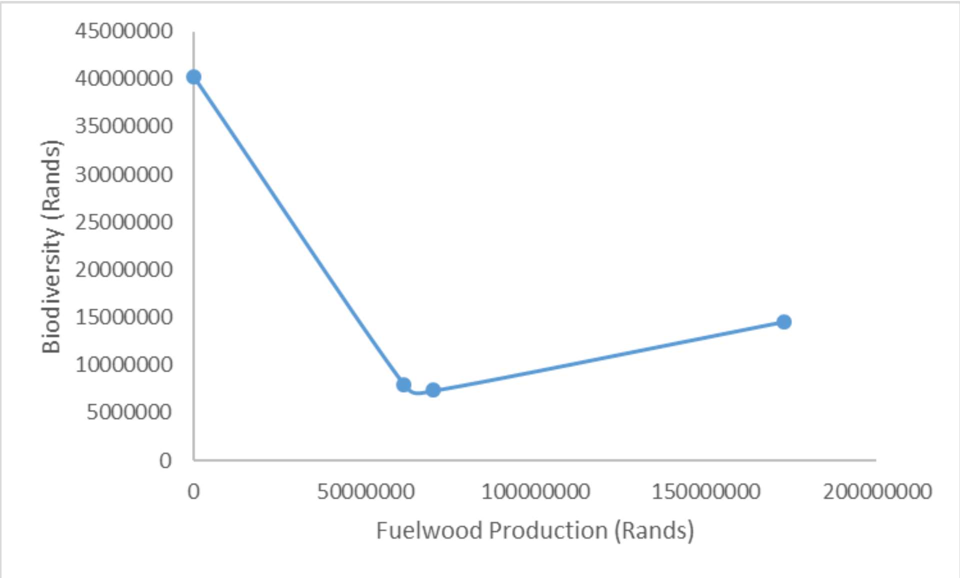

Figure 3.1.5: Graph of economic value of biodiversity versus fuelwood production in Rands under the four land-use scenarios.

3.1.1 Fuelwood - crop yield relationship

Fuelwood is often used to as a source of energy (domestic fuel) in both rural and urban areas in Sub-Saharan Africa (Amous, 1999; Scholes and Biggs, 2004). The energy content of fuelwood used as a domestic energy source is lower than that derived from charcoal (Zidago and Wang, 2016; Kaale et al., 2000), which is why charcoal is preferred as a fuel to be transported for sale in distant towns. However, a large fraction (over 75%) of the energy present in the fuelwood is lost in the process of converting it to charcoal, and a large amount of greenhouse gas is generated (a negative impact not calculated here). The extraction of fuelwood if not managed sustainably, in a manner where the rate of extraction is equal to the rate of tree growth will lead to vegetation loss. Loss of vegetation will result in some negative consequences such as loss of habitat for fauna, loss of flood control, ect.

Crop yield is one of the most valued ecosystem services as it directly affects food security, which is a key player in human wellbeing (Garnett et al., 2013). Two types of crop production were considered in this paper. The first was commercial crop production where by 8100 ha of land were converted from trees and shrubs to commercial maize and soy rotation, the second is a smallholder agroforestry where the same hectares as above of land were converted, but to mixed crops in smaller farms with lower yields. The smallholder agroforestry (multi-cropping family farms) was assumed to be applied in both smallholder scenario and current landscape.

The highest crop yield was estimated for the commercial landscape and the lowest fuelwood production was estimated in this landscape. The smallholder landscape had a larger area allocated for fuelwood production in comparison to the commercial agricultural production landscape. There is a trade-off between crop production, which requires vegetation clearance and fuelwood production. Figure 3.1.2 illustrate the negatively correlated relationship that exists between the two services.

3.1.2 Fuelwood - biodiversity relationship

Fuelwood production and biodiversity intactness are negatively correlated as illustrated in Figure 3.1.5. The extraction of fuelwood has the capacity to threaten habitat for species if it is not managed sustainably. The protected landscape has higher species diversity (with a Biodiversity Intact Index of (BII)=98%) because a larger area of land was protected and the fuelwood production of 0 t/ha/yr in this landscape as the extraction of fuelwood was forbidden (this is not a necessary rule but is the one usually applied). To have an area of high biodiversity, there must be habitat for the species to live on, which is provided for in the protected landscape in larger hectare compared to the other three landscapes. These two services (fuelwood production and Biodiversity Intactness) are competing as the enhancement of one service will negatively impact the capacity of the landscape to provide the other service.

3.1.3 Water - biodiversity relationship.

All landscapes had an area that was allocated to the protection of biodiversity. The landscape with the largest area for biodiversity protection was the protected landscape, followed by the current, smallholder and then the commercial landscape. Water yield increases mainly due to land degradation from anthropogenic activities (Geng et al, 2015). Water quality decreases with an increase in land degradation. The protected landscape had higher water quality and lower water yield because trees use more water than shorter vegetation because they offer higher interception and evaporation losses (Lang et al., 2017). Areas of higher biodiversity have a higher capacity for erosion control. There was an obvious difference in the quantity of water that will be yielded onto the catchment area under different land use types. The relationship between biodiversity intactness and the water services is synergistic or positively correlated. An increase in biodiversity results in increased water quality and an increased in water yield. The cost of supplying potable water was higher in the commercial landscape and lower in the protected landscape as illustrated in Figure 3.1.3, thus making the value of the service over a

25-year period lower in the commercial landscape. Thus, meaning that there is a positive correlation between water services and biodiversity.

3.1.4 Crop yield - biodiversity relationship

The clearing of vegetation cover for crop production means that the habit required for biodiversity is removed (Nelson et al., 2009). When there is no habitat then animal diversity will decline as well. From the above results illustrated in table 3.1.1 and figure 3.1.1, biodiversity intactness which assess species distribution and population sizes in a landscape decreases as the area allocated for crop production increases. Hence this negative correlation implies a trade-off must be made between the two services.

Species diversity was higher in the protected landscape as this environment has suitable habitat for these species. The commercial landscape had the lowest BII at 27%. This was due to fact that this landscape was the most transformed of all landscape. The protected landscape had no area allocated for crop production hence it had the highest habitat to support biodiversity.

3.1.5 Crop production – water relationship

The production of crops will always require clearing of vegetation. The scale or magnitude of this vegetation clearance depends on the type of agricultural production practiced. In this paper the smallholder agricultural management practiced is multi-cropping agroforestry. In the commercial landscape it is maize-soy rotation that is resource intensive. As a result, there is a larger rate of sediment yield and nitrogen yield onto the catchment areas in the commercial landscape. Thus, meaning water quality will decline while water yield will increase because the annual sum of transpiring leaf area is reduced under dryland cropping. Note that irrigation cropping was not considered here, it would lead to a reduction in river flow. This increase in water yield (surface water) has the potential to result in flooding if they are inadequate measures in place to prevent flooding.

There exists a trade-off between crop production and water services as the costs of providing portable water for domestic use will be higher for the commercial landscape as illustrated on figure 4.1.4. The poorest water quality was observed for the commercial landscape and the best quality was observed for the protected landscape. Water yield was much lower for the protected landscape where crop production was very low.

4.2 Bundles of positively associated services

Bundles that are positively associated are referred to as synergies. A synergy occurs in an event where the enhancement of one ecosystem service results in a simultaneous increase in another ecosystem service (Tomcha and Gergel, 2016; Elmqvist et al., 2011; and Turkelboom et al., 2016). This positive correlation should be encouraged as it ensures that the ecosystem is able to supply two or more services without much negative impacts. That’s if a linear relationship between two or more services is applied. Given the variety of services derived from a landscape the enhancement of one service can result in a simultaneous increase in another service. Such a relationship is demonstrated between water services and biodiversity intactness illustrated in the protected landscape as illustrated in figure 3.1.3. Since an area of high biodiversity can only exist in an environment that can provide habitat for the animal species. This relationship is positively correlated as the higher vegetation cover increases the capacity of the landscape to control erosion which reduces sediment yield rate there by improving or increasing the water quality at catchment area (both down and upstream).

Chapter Four- Discussion

Trade-offs occur in instances where an increase in the delivery of one ecosystem service results in a direct or indirect decrease in the delivery of another ecosystem service (Castro et al., 2014; Deng et al., 2016; Elmqvist et al., 2011; Nelson et al., 2009; Tomscha and Gergel, 2016).

Provisioning ecosystems services (food, fibre, fuelwood, freshwater, and fibre) are often enhanced at the expense of supporting (soil formation, nutrient cycling and primary production), regulating (climate regulation, water purification and pollination) and cultural (aesthetic, educational and recreation) services. Provisioning services are often prioritised as they have an immediate and direct impact on human wellbeing, often for a subset of privileged individuals, and they are easier to valuate as there is already an existing market for these products (Deng et al, 2016; Nelson et al., 2009). However, the capacity of a landscape to continue a sustainable delivery of these provisioning services is dependent on the environments capacity to continue the delivery of supporting and regulating services (Turkelboom et al., 2016; Elmqvist et al., 2011). This paper modelled the delivery of four ecosystem services bundles and mapped the trade-offs existing between the services given, for a given set of scenarios in the same landscape. The relationship between these services was analysed with trade-offs exhibited between the following; water services and crop production, biodiversity and crop production, fuelwood and crop production, and fuelwood and biodiversity. A synergetic relationship was observed between biodiversity and water serves. This paper set out to identify the trade-offs in ecosystem services using ecosystem services valuations through the Natural Capital Project under different land use scenarios. The outcomes or results were as expected from studies such as Raudesepp-Hearne et al. (2010) and Woollen (2016).

This study was based on secondary data analysis. Furthermore, many assumptions had to be made to allow the ecosystem services yields and values to be estimated. Scenario planning is by its nature full of uncertainties (the future is unknown), to which the above sources of potential error are added. It follows that if there are substantive errors in any of the above sources of information, the errors will be propagated to the results and the conclusions drawn from them. However, qualitative inspection of the results obtained suggests that they are not unreasonable or unexpected, and thus that the errors are within the bounds of what is acceptable in scenario-based exercises.

Chapter Five- Conclusion

This project has shown that trade-offs exist between four key ecosystem services in a *miombo* landscape. These trade-offs result in the reduction in supply of certain services where a landscape is transformed to maximise the provision of one particular service. Specifically, maximising for commercial food production decreases the supply of clean water, woodfuel and biodiversity habitat. An attractive land-use scenario would be one where there is an attempt to ensure a variety of land-uses are allowed in a landscape, such that there is an adequate supply of all desired services, for all people, now and in the future. Where there is a strong trade-off the consequences of prioritisation of one service over others must be carefully considered by policy makers and land managers, lest it adversely affect the wellbeing of some groups in the landscape or increase social costs while providing benefits to just a few.

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