AN ECONOMIC AND INSTITUTIONAL ANALYSIS OF COMMUNITY WILDLIFE CONSERVATION IN ZIMBABWE

By

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Declaration

I, Herbert Ntuli, do hereby declare that the work presented in this thesis, is my own, except where acknowledged and that this thesis or any part of it, has not been previously submitted for the award of a degree at any university.

Signed

Candidate ……Signed by candidate……… Date ……07-12-2015……

Abstract

This thesis focuses on the economics and institutional aspects of community wildlife conservation in the context of local communities living adjacent to the Gonarezhou National Park in Zimbabwe. A significant proportion of wildlife in Zimbabwe, and in Southern Africa in general, is managed as a common pool resource by communities under community-based natural resource management. Several challenges threaten conservation efforts at both local and higher levels, thus hindering its ability to bring about development that might improve the welfare of poor rural communities participating in wildlife conservation.

The most pressing issues in the wildlife sector include: i. inability to extract resource rents from wildlife conservation that in turn affect household welfare in terms of total household income and reduction in poverty and inequality, ii. lack of capacity by local communities to solve collective action problems or lack of incentives to self-organise, and iii. lack of comparable successful outcomes in CBNRM communities such as the wildlife conservancy communities. Learning from other successful communities that use 'community-based' models, such as wildlife conservancies, might provide important insights for policy makers and development practitioners. These issues are explored in three substantial papers included in this thesis. The thesis consists of five chapters starting with an introduction, followed by three papers and finally conclusions and policy implications. The study makes use of purpose-collected primary data from local communities living adjacent to Gonarezhou National Park in Zimbabwe.

The first paper investigates the effects of wildlife resources on community welfare. Specifically, the paper examines the contribution of environmental income to i) total household income, ii) poverty reduction, and iii) reduction in income inequality. Furthermore, it investigates the impact of environmental income on households in different income categories, the role of wildlife in the portfolio of environmental income and the determinants of environmental income generated by different households. To achieve the objectives above, the paper makes use of income quintile analysis, the Foster-Greer-Thorbecke poverty measure, Gini coefficient analysis, Gini decomposition analysis, ordered logit regression model and instrumental variables estimation using heteroskedasticity-based instruments.

The results show that the relative contribution of environmental income towards total household income is more pronounced in poor households, while the relative contribution of agricultural income is noticeable in wealthier households. In particular, wealthier households consumed more wildlife products in total than relatively poor households. However, poorer households derive greater benefit from the consumption of wildlife resources than wealthier households. Excluding wildlife compromised the relative contribution of environmental income and, at the same time, increased the relative contribution of farm and wage income. Environmental income has more impact in terms of poverty and inequality reduction in the lower income quintiles than in the upper quintiles. Wildlife income alone accounted for about 5.5% reduction in the proportion of households living below the poverty line. Furthermore, wildlife income had an equalizing effect, bringing about a 5.4% reduction in measured inequality.

The results of the ordered logit model suggest that the likelihood of belonging to a wealthier category of income increases with an increase in environmental income. However, the marginal effect of environmental income is very small, suggesting that only households that are positioned on the boundary will be able to move to the next income quintile. Its impact may be less pronounced for households that are located further away from the boundary. As expected, household wealth significantly and positively affect environmental income generated by households. The results reveal some evidence of the relationship between benefits and the quality of the resource system. Households generated more environmental income in areas with good biodiversity than in areas where there is an unhealthy population of wild animals.

The second paper examines the role of local institutions in community wildlife conservation in Zimbabwe. Using ordinary least squares and instrumental variables estimation with heteroskedasticity-based instruments, the results confirm that sound institutions are an important ingredient for cooperation in the respective communities. The paper finds evidence that cooperation positively and significantly affects biodiversity conservation. Taken together, these two main results imply that institutions directly affect the ability of a community to self-organise, which in turn affects the success of biodiversity conservation in a community. Group size, community trust, number of stakeholders and existence of punishment were the other important variables explaining cooperation. Besides cooperation, variables such as training, benefits, distance from the nearest urban canter, distance from the fence, social capital average

age of household head, fence and information sharing were also found to be very important in explaining the success of biodiversity conservation *ceteris paribus*.

The third paper develops a bio-economic model to analyse wildlife conservation in two habitats adjacent to a national park by two types of communities in the context of Gonarezhou National Park in Zimbabwe. One community is made up of peasant farmers operating under a benefit-sharing scheme (CAMPFIRE), while the other is made up of commercial farmers practising game farming in a conservancy (the Save Valley Conservancy). Both communities exploit wildlife by selling hunting licenses to foreign hunters but with different levels of success. The park agency plays a central role by authorizing the harvest quota for each community. The paper formulates a bio-economic model for the three agents, optimises the market problem for each agent and compares the outcomes of the different communities to the social planner's solution.

The results show that the size of wildlife stock recommended by social planner could be higher than the market solution for both the CAMPFIRE community and the conservancy. The level of anti-poaching enforcement employed by the park agency is suboptimal. The level of anti-poaching enforcement exerted by the conservancy community achieves social optimality. This could drive the stock in the conservancy towards the social planner's solution, starting from a lower level. CAMPFIRE communities exert more poaching effort than what the social planner would recommend. As a result, the size of the shared stock roaming inside the national park and on communal land diverges from the social planner's prescription over time.

The thesis sets out to investigate how people are benefiting from wildlife conservation and what drives conservation in CAMPFIRE communities. Its main contribution to the resource economics literature is to examine the effects of wildlife income (in the context of environmental income) on household welfare, the application of the general framework for analysing complex social-ecological systems developed by Ostrom (2007a; 2007b) in the wildlife sector in the context of a developing country and use a bioeconomic model to evaluate the behaviour of various actors in order to propose institutional changes that might move individual decisions closer to the social optimum. Furthermore, Ostrom's framework has been used extensively in areas such as forestry, fisheries, rangelands and water resources management, while little has been done in the wildlife sector, particularly in Southern Africa.

While other studies may have considered the role of wildlife in the income stream of CAMPFIRE, they limited themselves to the analysis of revenues of CAMPFIRE activities. Our study takes a step further by examining the subsistence values from wildlife. In the case of Zimbabwe, the success of CBNRM hinges on the interaction between local communities and wildlife. This is why disaggregation can help to make an appropriate assessment of the extent to which wildlife income as a component of environmental income is a catalyst of the CAMPFIRE programme. From a policy perspective, this thesis sheds light on the processes governing the human-environment systems and provides results comparable to other studies. Furthermore, the findings of this thesis will allow policymakers to interrogate their policies and strategies while at the same time identifying areas that need to be improved.

Several other important policy implications can be drawn from the analysis in this chapter. To begin with, wildlife conservation has an important role in mitigating poverty and income inequality in CAMPFIRE communities. There is therefore a need to design policies that increase access to wildlife income because this could have an impact on their welfare. In line with our goal of using wildlife resources to bring about community welfare, we believe that policy experiments of increasing wildlife income could highlight the importance of wildlife resources for redistributive policy targeting. Wildlife income can greatly reduce poverty, but not inequality because it is less responsive to policy induces increments in wildlife income. Reducing wildlife income slightly more 15% could reduce poverty in the whole population to zero.

There is need for a policy shift away from the resource itself to encompass the broader local economy in order to enhance livelihoods. Quick policy interventions such employment creation are required to increase household earnings from wildlife conservation in the short to medium term. In the long-run, household welfare in CAMPFIRE communities could benefit through policies and programs that stimulate an increase in wildlife earnings from activities such as non-consumptive tourism, tourism business ventures and investment in infrastructure. However, wildlife-based land reform also needs to empower poor households in the area of capital accumulation while imposing restraint on harvesting by well-off households.

The wildlife sector should be re-configured to facilitate the implementation of collective strategies that are endogenous to the community aimed at providing public goods such as wildlife through conservation. Government policy should recognize CAMPFIRE communities as important stakeholders with an important role to play, but not as beneficiaries of wildlife conservation. Future policy reforms should also consider further devolution of natural resource management function and full benefits from the Rural District Council to sub-district producer communities or increased autonomy. This will allow innovation among communities, while the government create an enabling environment for CAMPFIRE communities to operate. The capacity of CAMPFIRE to develop own rules, rather than just following externally-imposed rules, should not be undermined.

Since wildlife is a fugitive resource that roams freely on communal land, there is therefore a need for coordination among CAMPFIRE communities in order to supply the public good, the required habitat and to benefit from collective. The government, NGOs and private sector should act as catalyst of collective action to ensure that there are incentives for CAMPFIRE communities to self-organize into multiple layers of nested organizations through policies and programmes that allow resource users to participate in decision making process. Development programmes or policy interventions with both a welfare and a conservation component should not be designed as 'one size fits all' but should recognize and understand the differences in community characteristics and incentives to self-organize in order to promote conservation and safeguard the livelihood interests of pro-poor communities.

Government programmes can target capacity building and skills development in order to have a positive impact on biodiversity conservation. There is need to increase spending on training relevant to natural resource management, targeting not only community leadership or project committees, but also ordinary community members as a path way to conservation. Therefore, the capacity building efforts of government agencies, NGOs and other stakeholders should complement each other to ensure that the necessary resources are mobilised and all communities receive the necessary training and resources.

Resolving the problem faced by CAMPFIRE communities will necessarily not require market based instrument alone, but also institutional reforms. This study calls for three main strategies to allow campfire communities to move from a seemingly inferior outcome to one that is optimal.

Firstly, this result calls for policy instruments that can facilitate the development of sound institutions that are tailored to suit local conditions and endogenous to the community. For sustainability of the programme, there is also need to investment in institutional building blocks such as governance structures at the local level, democracy, monitoring and enforcement, and community level trust, and to source additional funding in order to equip CAMPFIRE communities with much-needed resources. The government should therefore allow CAMPFIRE communities to learn through past experiences and mistakes, and to be innovative so that they can develop and experiment with new conservation models.

Secondly, CAMPFIRE communities suffer double taxation from the premium charged by the safari operators and a significant proportion of wildlife income that remains in the hands of the Rural District Council. From a policy standpoint, CAMPFIRE communities would benefit immensely if they could operate with the same autonomy and self-sufficiency as the conservancy because both taxes could be avoided. This could be achieved by hiring a manager or building internal capacity to match that in the conservancy community. This implies a different model or an improvement of the current benefit-sharing scheme, instead of receiving cash transfers. Innovative policy instruments are required to assist CAMPFIRE communities in various activities of wildlife conservation so that they can integrate into the main stream economy and fully commercialize their operations.

Thirdly, designing policy instruments that increases the risk premium could decrease the effective price of the illegal off-take. Reducing the effective price of the illegal off-take discourages the community from poaching by eroding the incentives. It is possible to integrate the risk premium into local institutions by carefully designing policy instruments that are adapted to local conditions. We recommend that these three strategies should be deployed simultaneously for an effective and fast solution to the challenges faced by CAMPFIRE projects. This implies that policy makers should use a combination of both market-based instruments and institutional reforms since they seem to complement each other.

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Dedication

I dedicate this work to my mother Gladys Ntuli, late father Muxabango Ntuli, my wife Jean Felistas Ntuli and the rest of my family.

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Acronyms and Abbreviations

CA	Conservation Area
CAMPFIRE	Communal Areas Management Programme for Indigenous Resources
CBNRM	Community-Based Natural Resource Management
CPRs	Common Pool Resources
FGT	Foster-Greer-Thorbecke
FTLRP	Fast-Track Land Reform Programme
GNP	Gonarezhou National Park
ICDPs	Integrated Conservation and Development Projects
IV	Instrumental Variables
NGOs	Non-Governmental Organizations
NRM	Natural Resource Management
OLS	Ordinary Least Squares
RDC	Rural District Council
SES	Social-Ecological Systems
SVC	Save Valley Conservancy
VIF	Variance Inflation Factor

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Chapter 1

1. Introduction

The management of common pool resources (CPR) such as rangelands, fisheries, forests, water and wildlife is a critical element in the promotion of rural livelihoods and environmental sustainability. Throughout the developing world, a significant proportion of natural resources are managed by poor rural households as CPRs. They provide income, food, fuel, clothing, furniture, medicines and construction material among other things. In the process of harvesting these resources, local people pose a serious threat to the environment when the resource extraction is not controlled (Hardin, 1968; Johnson, 1972; Baland & Platteau, 1996; Dietz et al., 2003; Ostrom, 2007). Consequently, if the natural capital base is utterly decimated, then it implies that the means of livelihood for many poor rural households is at risk. This, in turn, exacerbates rural poverty and inequality. Two unique characteristics of CPRs, namely non-excludability and rivalryness, make them vulnerable to resource overexploitation and eventually to total collapse of the resource system (Ostrom, 2003).

Despite over 30 years of notable effort, human ingenuity has failed to generate a plausible solution to the environmental sustainability problem (Harich, 2010). The sustainability problem suggests that human beings should extract or harvest from the environment in such a way that the ability of the resource system to provide the same goods and services in future is not compromised. As a result, there has been tremendous growth in research to understand the complex relationships between people and the environment. The human-environment relationship is multifaceted and includes both human and ecological dimensions which together forms what Ostrom (2007a) refers to as the 'social-ecological system' (SES). Just like any system, these two dimensions of a SES cannot be studied in isolation.

When we consider Southern Africa and the different types of natural resources used by rural communities, we observe that wildlife is more threatened, despite its importance to the rural economies of Africa. Wildlife is very important in for most governments in Southen Africa because it generates revenues through both consumptive and non-consumptive tourism. In

Zimbabwe, tourism used to be the fourth foreign currents earner after tobacco, cotton and gold during the 1990s generating income in excess of US$857 million per annum and contributed more than 11.6% towards Gross Domestic Product. Wildlife conservation also employs thousands of people (over 98 500 jobs) in Zimbabwe and generates income for communities living adjacent to national parks. Since independence, over US$20 million has been distributed to local communities.

The link between human beings and the wildlife system is a topical issue worth investigating from both a developmental and conservation objective. This thesis is motivated by the fact that very few studies done in the region have actually attempted to operationalise the theory developed by Ostrom (2007a) and considers the following: i. the effects of wildlife income in the portfolio of environmental income on household welfare for local communities living adjacent to protected areas, ii. the role of local institutions in managing common pool wildlife in the context of Southern Africa, and iii. the lack of comparable successful outcomes in CBNRM communities such as the wildlife conservancy communities to inform conservation and development policy. These issues are so pressing that they deserve to be looked at in greater detail in three separate papers.

The first issue seeks to establish the extent to which the benefits generated by environmental resources (including wildlife) have made environmental income a part of the rural economy of local households. In particular, the study seeks to investigate the effects of wildlife resources on community welfare – total household income, poverty and income inequality. It is indisputable that environmental resources contribute significantly to rural economies in Southern Africa (Cavendish, 2000; Fisher, 2004; Shackleton & Shackleton, 2006; Thondhlana et al., 2012; Thondhlana & Muchapondwa, 2014) and in other developing countries elsewhere in world (Lopez-Feldman et al., 2007; Uberhuaga et al., 2012).

As part of the global research agenda addressing rural poverty and the environment in the past three decades, particular emphasis has been placed on improving the livelihoods of poor rural communities in most developing countries and conserving natural resources. As a result, previous studies have looked at environmental dependence (Uberhuaga et al., 2012; Thondhlana et al., 2012; Thondhlana & Muchapondwa, 2014) the contribution of environmental resources to total household income (Cavendish, 1999; Cavendish, 2000; Shackleton & Shackleton, 2006), the role of environmental income in reducing rural poverty (Cavendish, 1999; Lopez-Feldman et al., 2007; Thondhlana et al., 2012) and its effects on income inequality (Cavendish, 1999; Fisher, 2004; Lopez-Feldman et al., 2007; Fonta & Ayuk, 2013).

Harvesting, consuming and selling of environmental resources form part of rural livelihood diversification strategies in Southern Africa. Ellis (1998) defines livelihood diversification as the process by which rural households construct a diverse portfolio of activities and social support capabilities in order to survive and to improve their standards of living. Viewed from a different angle, environmental income also acts as a safety net against income shocks associated with extreme weather conditions, crop failure, market shocks, health shocks and idiosyncratic shocks (Hegde & Bull, 2008; Shackleton & Shackleton, 2006). Use of valuable natural resources such as wildlife should be incorporated into poverty and income inequality reduction strategies.

In the process of mining the environmental or harvesting natural resources, there is also a threat posed by local communities to the environment (Hardin, 1967; Johnson, 1972; Baland & Platteau, 1996; Dietz et al., 2003). Scholars argue that one way of dealing with this challenge is through devolution and decentralization of natural resource management (NRM) functions and decision-making to the community's grassroots level (Agrawal 2001; Ostrom, 2007; Berkes, 2008). When we consider different types of environmental resources used by rural households in Southern Africa, we observe that wildlife is more threatened, despite its importance to local communities living adjacent to protected areas. This study argues that the utilization of wildlife resources has resulted in different contributions to livelihoods (Cavendish, 1999; Shackleton & Shackleton, 2006) and incentives to conserve resources. The different channels through which wildlife contributes to livelihoods need to be investigated in order to inform rural development policy.

It is essentially a good outcome if poor households derive significant benefits from managing their own natural resources. However, when the community fails to adequately manage the resources, this always produces bad conservation outcomes. Therefore, the second issue address the linkage between local institutions[1] and biodiversity outcomes. When we consider the CAMPFIRE communities in Zimbabwe, we observe that conservation is not doing as it should. This leads us to a very important question, what drives community wildlife conservation? In theory, conservation is forged by cooperation and cooperation by institutions (Ostrom, 2007). The results in the literature are not quite conclusive. This is why this study seeks to investigate whether institutions affect biodiversity through cooperation.

There is substantial evidence of the role played by community level institutions in forestry, fisheries and water resources management (Pomeroy, 1995; Agrawal, 2001; Sokile et al., 2003; Dungumaro & Madulu, 2003; Mehringa et al., 2011). Institutions are required to safeguard the long term integrity of a SES. Since institutions act as a constraint to human behaviour, it therefore makes sense that institutions directly affect ability to self-organise or cooperation. The literature distinguishes between triggering institutions (i.e. institutions that are required to prompt cooperation) and those institutions that are required to sustain cooperation over time. However, both triggering and sustaining institutions are needed during the different phases of collective action.

Furthermore, the CPR literature seems to suggest that the success of environmental outcomes depends directly on collective action (Wade, 1987; Ostrom, 1990; Agrawal, 2001; Harris, 2003; Ostrom, 2010). This implies that institutions do not affect biodiversity outcomes directly, but instead work through the community's ability to self-organise. This has a more direct impact on biodiversity. Integrated Conservation and Development Projects (ICDP), also referred to as community-based natural resource management (CBNRM) in the literature, assume that it is possible to harmonise both conservation and development goals in such a way that local communities are able to self-organise in order to conserve resources, while at the same time deriving benefits from conservation (Alpert, 2015). The critical assumption behind the so called people oriented approaches is that local communities are interested and able to self-organise for a common purpose (Sorongorwa, 1999).

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[1] Institutions define the set of operational rules that are used to determine who is eligible to make decisions in some arena, what actions are allowed or constrained (Ostrom, 1990).

Ostrom (2003) argues that the property rights system might be responsible for generating incentives to self-organise and, hence, to conserve resources. From the CPR literature, the idea of protected areas is viewed as the only way to protect biodiversity in third world countries. High on the list of policy options is private ownership of natural resources, which is believed to generate adequate incentives to conserve resources (Bromely, 1991). Given this common assumption, the tenure system operating in communal areas has been the subject of considerable debate (Murombedzi, 1999). The common property system was viewed by some scholars as inferior, largely normative and based on ideological construct (e.g., Hardin, 1968; Johnson, 1972); while others view it as an alternative to both state control and privatization of CPRs (Schlager & Ostrom, 1992; Ostrom et al., 1999; Agrawal, 2001). Due to other undesirable consequences associated with private and state ownership of CPRs, common property systems have been found to deliver superior results under certain conditions (Agrawal & Ostrom, 2001).

Finally, the third issue looks at different communities with different outcomes in order to establish the conditions under which the outcomes of the two communities could be the same. For us to be able to deal effectively with institutions there is a need to talk about a different type of community other than the CAMPFIRE community. For the purposes of this study, we chose the Save Valley Conservancy community. Both communities are conserving wildlife in a community-based fashion, but their level of success is different. The performance of the CAMPFIRE community is much worse than the conservancy community. Perhaps there could be room for institutional reform in the community that is not performing well. There is need to investigate why conservancies perform better than CAMPFIRE communities and what policies need to be instituted in order for CAMPFIRE communities to achieve comparable results.

In particular, the paper develops a bioeconomic model in order to compare wildlife conservation under private versus communal property systems in terms of management (stock size) and utilization (harvesting effort). Many studies have used bioeconomic modelling in the past to demonstrate the importance of local communities in wildlife and biodiversity conservation (Schulz & Skonhoft, 1996; Skonhoft, 1998; Johannesen & Skonhoft, 2004; Skonhoft & Schulz, 2005; Bulte & Rondeau, 2007; Fischer et al., 2011; Johannesen & Skonhoft, 2014).

Fischer et al. (2011) formulated a bio-economic model to analyse community incentives for wildlife management under the benefit-sharing programs under the CAMPFIRE programme in Zimbabwe. They established that resource sharing does not necessarily improve community welfare or incentives for wildlife conservation. They argued that the results depend on the exact design of the benefit shares, the size of the benefits compared with agricultural losses, and the way in which the parks agency manages hunting quotas. Johannesen and Skonhoft (2004) developed a model for wildlife species migrating seasonally between a conservation area and a neighbouring area. Contrary to what is argued in the literature, their study demonstrated that handling the property rights over to the local people does not automatically translate into more wildlife and sustainable resource utilisation. They suggested that the reason might lie in the nuisance motive for harvesting.

Skonhoft (1998) identified several ways through which the park agency and local communities can share wildlife benefits. It can be revenue sharing from tourism, safari hunting, establishing user rights through hunting quotas, or through local job creation in tourism. Another way of providing local people with park revenues is by compensating farmers for wildlife induced agricultural damages. However, damage compensation may typically work as an agricultural subsidy and possibly trigger agricultural land expansion, thereby reducing the size of wildlife habitat (Johannesen & Skonhoft, 2014). Bulte & Horan (2007) established that damage compensation stimulates habitat conversion (with negative effects on wildlife stock), but it also reallocates labour from wildlife hunting to agriculture (the effect works in the opposite direction). They also found that compensation is less likely to work as intended when agricultural land is a poor substitute for natural habitat for wildlife.

The ICDPs literature suggests that benefit-sharing initiatives have been poorly linked to the conservation objective and, hence, such initiatives rarely work (Wells et al., 1999; Fischer et al., 2011; Johannesen & Skonhoft, 2014). Common pool resource scholars stress the need to change incentives from indirect measures such as compensating farmers for wildlife induced agricultural damages to direct measures such as transfers conditional on the conservation target (Ferraro, 2001; Ferraro & Kiss, 2002). Zabel et al. (2011) analysed the impact of conditional revenue transfers conditional upon the size of the wildlife stock. Using a bio-economic model of wildlife harvesting and livestock herding, they found that conditional transfers based on the size of the wildlife stock may reduce wildlife harvest and thus encourage wildlife conservation. Additionally, it may give incentives for the herders to better protect their livestock from the predation pressure.

This work differs from previous studies in that it uses bio-economic modelling to compare wildlife management under two distinct communities where one produces superior results. There is overwhelming evidence showing that conservancies perform much better in terms of wildlife management and utilization compared to the CAMPFIRE projects in Zimbabwe where local communities have frequently been accused of destroying wildlife (Murombedzi, 1999). Both the land under conservation and wildlife population has increased tremendously in the conservancies (Kreuter et al., 2010). Furthermore, local communities have failed to adapt in such a way that they are able to extract resource rents through conservation while the private game farms have evolved into vehicles of extraction (Emerton, 2001). These observed outcomes could be a result of the discrepancies in community level institutions. As such, appropriate institutional reforms might be necessary for CAMPFIRE communities to benefit from and conserve wildlife resources.

This thesis contributes to the resource economics literature by using purpose collected primary data to examine the effects of wildlife income (in the portfolio of environmental income) on household welfare, by operationalizing the framework for analysing complex SESs developed by Ostrom (2007a; 2007b) to investigate the role of institutions on community wildlife conservation, and by using a bioeconomic model to evaluate the behaviour of various actors in order to propose institutional changes that might move individual decisions closer to the social optimum. Furthermore, Ostrom's framework has been used extensively in areas such as forestry, fisheries, rangelands and water resources management, while little has been done in the wildlife sector, particularly in Southern Africa.

While other studies may have considered the role of wildlife in the income stream of CAMPFIRE, they limited themselves to the analysis of revenues of CAMPFIRE activities. Our study takes a step further by examining the subsistence values from wildlife. In the case of Zimbabwe, the success of CBNRM hinges on the interaction between local communities and wildlife. This is why disaggregation can help to make an appropriate assessment of the extent to which wildlife income as a component of environmental income is a catalyst of the CAMPFIRE programme. From a policy perspective, this thesis sheds light on the processes governing the human-environment systems and provides results comparable to other studies. Furthermore, the findings of this thesis will allow policymakers to interrogate their policies and strategies while at the same time identifying areas that need to be improved.

1.2 Objective of the thesis

Given the research issues articulated above, this thesis aims to achieve the following broader objectives:

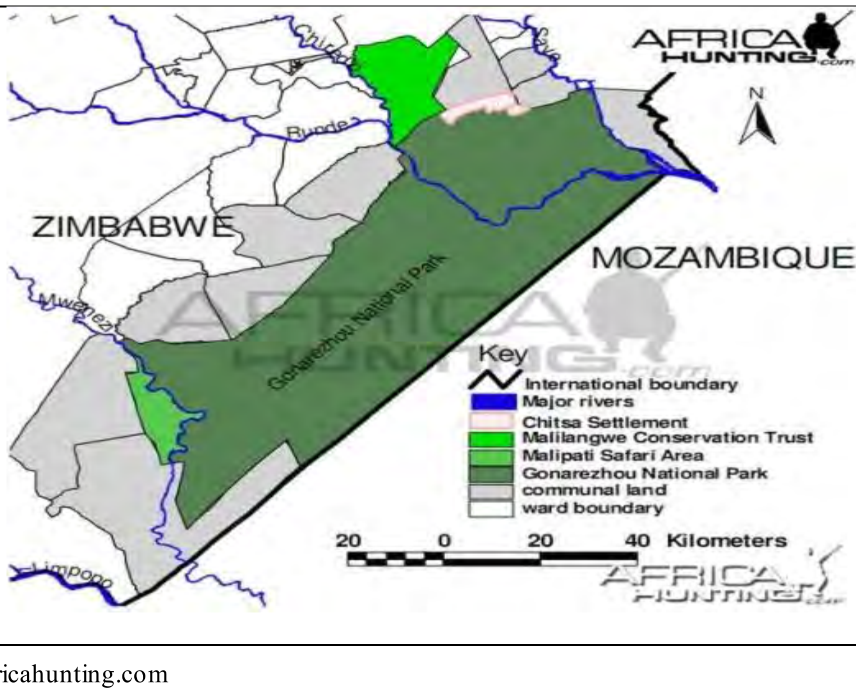
1. to examine the effects of wildlife resources on community welfare – household income, poverty and inequality;
2. to investigate the role of institutions in community wildlife conservation using local communities living adjacent to Gonarezhou National Park in Zimbabwe as a case study; and
3. to compare wildlife management on common property systems operating under the banner of the Communal Areas Management Programme for Indigenous Resources and the private game farming community in the Save Valley Conservancy.

1.3. Description of study area and sampling

This study collected primary data from 30 local communities in 13 wards around Gonarezhou National Park (GNP) in Zimbabwe that are participating in wildlife conservation. GNP is the second-largest game reserve in the country after Hwange National Park. It is located in south-eastern Zimbabwe (coordinates 21^{0} 40′ S 31^{0} 40′ E) and covers about 5 053 km^2. It forms part of the Great Limpopo Trans-frontier Park which links Gonarezhou with Kruger National Park in South Africa and Limpopo National Park in Mozambique. Owing to its vast size, rugged terrain and location away from the main tourist routes, large tracks of Gonarezhou still remain pristine wilderness. The national park is located in natural region five, which is very dry with very low agricultural potential. The mean annual rainfall for the area is about 499 mm, with a standard deviation of about 195 mm. The average maximum monthly temperature ranges from less than 25.9^{o}C in winter to over 36^{o}C in summer, while the average monthly minimum ranges from 9^{o}C to 24^{o}C in winter and summer respectively (Gandiwa, 2011). The vegetation of the Gonarezhou ecosystem is typical semi-arid savanna, dominated by *Colophospermum mopane* woodlands (Gandiwa & Kativhu, 2009).

The area under study is located approximately 100km away from the nearest Chiredzi town, relatively sparsely populated and predominantly occupied by the Shangani people; other ethnic groups such as the Venda, Ndau, Shona and Ndebele people are also found in the area. Apart from the CAMPFIRE programme (which accounts for over 95% of the local communities involved in wildlife conservation), there are also peasant farmers operating under resettlement schemes (slightly less than 3%) and about four conservancies. The mode of production of peasant farmers is primarily subsistence in nature, while commercial farms are heavily capitalised. Although peasant farmers engage in a diversified portfolio of economic activities including wildlife conservation, the most dominant livelihood activities are livestock production and crop cultivation, e.g., maize, sorghum, millet, groundnuts and cotton. The most important activity in the conservancy is wildlife conservation with little agriculture, particularly, livestock ranching. Figure 1.1 below shows the map of GNP (in dark green) and the communal areas (in grey) bordering the national park.

Figure 1.1: Map of Gonarezhou National Park and the local communities



Source: www.africahunting.com

The data for the analysis was drawn from a household survey conducted in June/July 2013. Local communities participating in wildlife conservation were identified with the help of community leadership, park authorities and the Rural District Council (RDC). From each community, the chairperson of the wildlife management committee was identified and interviewed as a key informant[2]. The survey employed a simple random sampling procedure using the list of beneficiaries from each community as the sampling frame. Initially, sampling was done using information gathered at district level from the RDC. There was a huge disparity between the information supplied at the district level and what was on the ground in terms of the number of community groups (projects) and number of households participating in community wildlife conservation as the RDC did not update its records regularly. As a result, the sampling procedure changed ex-post, i.e., information gathered from the chairperson of the respective community groups was used to update the initial calculation of the sample size, and this made some wards to be over-sampled. This complicated the exercise because the information needed to keep the same sample size among the areas was not readily available. Local communities around the Gonarezhou National Park share similar ecological conditions and perhaps similar culture, traditions or languages, which makes the results comparable across communities.

Table 1.1 below shows the number of households interviewed by ward against the total number of households participating in wildlife conservation in each ward, and ward population expressed in terms of households. The sample size was further compromised by the fact that some enumerators did not meet the required target, particularly in areas that were sparsely populated. In addition, some questionnaires were not usable due to non-response or lack of critical information. Table 1.1 confirms that some Wards, particularly Ward 8 and Ward 15, were heavily sampled (i.e., heavily sampled more than others). Another explanation is that these two wards had the highest number of CAMPFIRE communities than any other ward, i.e., six and five groups respectively (refer to Table A2 in the appendices). Since the study purposefully targeted CAMPFIRE communities, a complete enumeration of all CAMPFIRE projects in each ward (full sample) was carried out.

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[2] Local communities are required to organise and form wildlife management committees by the government in order to participate and to benefit from wildlife conservation. All the communities visited during the field work had committees in place. The only difference between them was the extent to which these committees are functional and involved in wildlife management.

Table 1.1: Number of households by ward

Ward	Interviews		Household involved in CWM		Ward Population	
	Number	Percent	Number	Percent	Number	Percent
Ward 10	20	5.95	820	7,82	1300	7,46
Ward 12	7	2.08	43	0,41	43	0,25
Ward 13	15	4.46	861	8,21	1000	5,74
Ward 14	9	2.68	492	4,69	1750	10,04
Ward 15	64	19.05	955	9,11	2000	11,47
Ward 22	10	2.98	81	0,77	81	0,46
Ward 23	13	3.87	181	1,73	181	1,04
Ward 29	19	5.65	2000	19,08	2000	11,47
Ward 30	29	8.63	960	9,16	1700	9,75
Ward 4	10	2.98	350	3,34	1850	10,61
Ward 5	32	9.52	1589	15,16	2100	12,04
Ward 8	84	25.00	1235	11,78	1470	8,43
Ward 9	24	7.14	917	8,79	1960	11,24
Total	336	100.00	10484	100.0	17435	100,00

Source: Survey data August 2013

It is not likely that some of the household covariates used in the study and our results could be biased because the primary target for the study were CAMPFIRE projects, and the goal was to collected data from household in these communities in order to understand their resource extraction patterns in the first and second paper and in the third paper their conservation pattern. The idea was to sample households in CAMPFIRE communities – over and under-sampling is therefore taking place at project level and not community level. Given that we are controlling from community effects in order to capture resource heterogeneity and other spatial differences that influence incomes, we don't believe there is going to be bias on account of over-sampling at project level.

Out of the 30 communities visited, 25 are CAMPFIRE projects while the remainder, namely, Nyangambe, Chizvirizvi and Gonakudzingwa, are resettlement schemes (see Table 1.4 below). Although Gonakudzingwa has the least number of households (about 43), it had about three CPR groups, each with its own wildlife management committee, thereby bringing the total number of communities under resettlement schemes to five. Resettlement schemes differ from communal areas in that the land tenure system was based on written permits which allowed settlers to reside and use the land on which they settled in a setting overseen by resettlement officers rather than by established local authorities while the latter provides communal tenure.

Table 1.4: Number of communities involved in wildlife conservation by tenure

Type of tenure	Freq.	Percent	Cum.
Communal	25	83.33	83.33
Resettlement	5	16.67	100.00
Total	30	100.0	

Source: survey data August 2013

By design, all CAMPFIRE projects were supposed to operate at ward level. However, due to conflict and unequal distribution of resources within communities, they had to split in order to spread the benefits to every community member. About three communities were operating at ward level, namely, Mahenye, Mutandahwe and Nyangambe. The rest of the wards were divided into several community groups ranging from two to six different sub-groups in a single ward. In the case of communal areas, these different community groups were referred to as CAMPFIRE villages[3].

1.4. Organisation of the thesis

This chapter introduced three primary issues of the study, gave an outline of the broader objectives of the thesis and a description of the study area. These research issues and objectives are closely intertwined but each deserves to be looked at in greater depth, and as such they shall be dealt with in three separate papers each constituting a thesis chapter. The first paper is presented in chapter 2, the second paper in chapter 3 and the third paper in chapter 4. Chapter 2 examines the effects of environmental resources (including wildlife) on community welfare using income quintile analysis, the Foster-Greer-Thorbecke poverty measure, Gini coefficient analysis, Gini decomposition analysis, ordered logit regression model and instrumental variables estimation using heteroskedasticity-based instruments. Chapter 3 relies on Ostrom's framework for analysing complex social-ecological systems to examine the role of local institutions in community wildlife conservation in Zimbabwe. Chapter 4 develops a bio-economic model with numerical illustrations to compare wildlife management under common property systems and the private game farms. It seeks to establish the possibility of wildlife tenure reforms or conditions under which local communities can behave like the private game farming community. Finally, chapter 5 presents conclusions and policy implications.

[3] By definition, a CAMPFIRE village comprises several political administrative villages, each with its own headman. Its primary objective is conservation.

Chapter 2

Effects of Wildlife Resources on Community Welfare: Income, Poverty and Inequality[4]

2.1. Introduction

There is increasing consciousness among policymakers, development practitioners and academic scholars about the importance and value of environmental resources[5] in the livelihoods of poor rural communities, including the Southern Africa region (e.g., Cavendish, 2000; Fisher, 2004; Shackleton & Shackleton, 2004; Lopez-Feldman et al., 2007; Thondhlana et al., 2012; Uberhuaga et al., 2012; Fonta & Ayuk, 2013; Thondhlana & Muchapondwa, 2014). All these studies demonstrated considerable economic contribution made by environmental resources to rural livelihoods. However, based on the studies done in the Southern Africa region so far, it is still not clear how the analysis of environmental income and welfare[6] is altered when wildlife and poor rural households living adjacent to national parks are considered. Furthermore, previous studies were conducted in areas where wildlife conservation is not an important activity in the community. This chapter argues that wildlife conservation is an important component of environmental income generation for communities living adjacent to the Gonarezhou National Park in Zimbabwe that are participating in the Communal Areas Management Programme for

[4] A version of this chapter has been desseminated as an EfD Discussion Paper No. DP 15 – 21 and ERSA Working Paper 554.

[5] We define environmental resources in this paper as goods that are freely provided by nature or "nature's bounty" (Cavendish, 1999), accessible to everyone in the community, and which community members can collect without incurring any other cost except their own time. Ownership of assets such as carts and dogs help in the collection process of environmental resources.

[6] Household welfare is measured in terms of three dimensions: total household income, poverty and income inequality.

Indigenous Resources (CAMPFIRE), a program that seeks to balance both conservation and development goals by including local communities in wildlife management[7].

Therefore, the purpose of this study is to examine the economic contribution of wildlife resources (as part of the portfolio of environmental income) to household welfare and incentives to conserve resources. This study seeks to fill an important gap in the literature. Little is known about the effect of wildlife income on household welfare, i.e., total income, poverty and inequality. It is undeniable that unequal utilization of wildlife resources results in different contributions to livelihoods among the users of such resources. The different channels through which wildlife contributes to livelihoods need to be investigated in order to inform rural development and conservation policy in Southern Africa.

In addition, little is known about the relative contribution of environmental income (with and without wildlife) when compared with other income sources. From a policy standpoint, it has become imperative to recognise the relative importance of different income sources in deriving inter-household poverty and inequality (Leibbrandt et al., 2000). Using Gini decomposition, Leibbrandt et al. (2000) considered six income sources that include wage income, remittances, agriculture, capital income, transfers and self-employment for rural households in rural South Africa. They found that wage income is both the most important income component and the most important source of inequality. This chapter applies such a technique and includes environmental income in the analysis in addition to the six components of standard household income considered above. In doing so, the chapter extends existing knowledge about the human-environmental resources nexus in the context of developing countries. This study is relevant given the occurrence of a major institutional reform in 2000 which affected both the land tenure system and wildlife policy in Zimbabwe.

Analysis of the human-environment relationship is constrained by inadequate data encompassing both environmental and economic activities (Dasgupta, 1993; Deaton, 1997; Cavendish, 1999;

[7] The CAMPFIRE programme was established during the mid-1980s to accommodate peasant farmers in communal areas that are located in the vicinity of national parks (Balint & Mashinya, 2006). CAMPFIRE allows local communities to manage wildlife through their respective Rural District Councils and to get income from wildlife conservation. It is expected that villagers carryout anti-poaching enforcement in their communal areas because they now benefit from conservation.

Cavendish, 2000; Luckert et al., 2000). Cavendish (2000) argued that traditional studies miscalculated rural incomes and welfare measures simply because environmental income is ignored in the analyses. He argued further that measures of poverty and inequality are overstated in conventional studies. As a result, such measures do not reflect the true picture on the ground and policies based on these measures achieve limited success. The lack of appropriate and comprehensive household data-sets encompassing both economic and environmental aspects presented further stumbling blocks for past researchers to undertake such rigorous quantitative analysis (Cavendish & Campbell, 2002; and Cavendish, 2000). To overcome this challenge, this study made use of purposely-collected survey data capturing both economic and environmental aspects from local communities around the Gonarezhou National Park in Zimbabwe.

In light of the policy issues discussed above, a number of questions arise. How does the utilization of environmental resources affect welfare (total income, poverty and inequality) and incentives to conserve resources when wildlife is considered? Specifically, we ask: i. Does environmental income (including wildlife) contribute significantly toward total household income, reduction in rural poverty and reduction in income inequality? ii. How does environmental income compare with other sources of income? iii. What determines the different amounts of environmental income that households generate?

This chapter is organised as follows. Section 2.2 provides the background and reviews the literature, while Section 2.3 discusses data issues, defines key variables and gives an outline of the research methods, analytical framework and empirical model specifications. We then proceed to discuss the results in the Section 2.4 and wind up with conclusions and policy implications in Section 2.5.

2.2. Background and Literature Review

Poor rural households in Southern Africa depend heavily on the natural capital base to sustain their welfare through the provision of both consumptive and non-consumptive goods (Cavendish, 2000; Fisher, 2004; Shackleton & Shackleton, 2004; Thondhlana et al., 2012; Thondhlana & Muchapondwa, 2014). Apart from land restitution programmes or any other justice objectives, the realization that the livelihoods of poor rural households in the region depend heavily on environmental resources has led to devolution and decentralization of natural resource management, particularly wildlife resources, into the hands of local communities. Such a policy is believed to provide appropriate incentives to the communities in question to conserve natural resources, while at the same time making sure that they also benefit from managing their own resources (Balint & Mashinya, 2006).

Demonstrating the complementarity between development and conservation goals, which were previously thought to be incompatible, has been on the agenda of regional and international policy since the mid-1980s, when devolution started in Southern Africa (Dubois, 2003; Shackleton & Shackleton, 2006). Zimbabwe was among the first countries in the region to implement the so-called 'people-oriented approaches' to natural resource management. CAMPFIRE is one good example of local communities that are managing natural resources to their own benefit (Murombedzi, 1999; Balint & Mashinya, 2006). The livelihood of local communities living adjacent to national parks is heavily dependent on natural resources, including wildlife. As a result, enhancing the utilization of environmental resources in the domestic and wider markets will not only increase livelihood security and reduce rural poverty and inequality, but will also provide incentives for conservation and sustainable utilization of resources (Wunder, 2001; Thondlana et al., 2012).

Considering the management side within the CAMPFIRE projects, some local communities have managed to put in place a wildlife management committee, develop sound common pool resource institutions in order to control resource extraction (i.e., the technology to use, when and where to harvest certain types of resources), and set aside a portion of their land for conservation or as wilderness area. These communities have a constitution in place, which spell out the rules and regulations that govern the utilization of resources in their area in addition to traditional institutions such as the village headmen and chiefs, and bylaws enacted by the Rural District Council and park authorities. For instance, some CAMPFIRE communities restrict unnecessary harvesting of trees either for firewood or as timber, thatch grass and other valuable non-wildlife resources. Although poaching is still persistent in the study area, it is illegal to harvest wildlife in all communities.

When we consider how individuals think about bylaws and household behaviour in terms of illegal harvesting there is great variations in the timing, harvesting place and technology used by CAMPFIRE communities. Most households prefer to hunt at night with dogs, while others hunt during the day with bows are arrows. Some households use wire snares and poison, but these technologies are not selective, i.e., both wildlife and domestic animals are killed. Some households prefer to hunt in the periphery of the community where they harvest small plains game, while others prefer the wilderness area far away from the community but immediately before the boundary of the national park where they harvest bigger game.

Cavendish (2000) identified a number of channels through which environmental resources contribute to rural livelihoods. To begin with, a household can harvest natural resources such as wild vegetables, wild fruits, timber or firewood and consume them directly as part of its own consumption activities. This is in line with the notion of a standard rural household as both a production and consumption unit. It is also possible that a household might use environmental resources as inputs in another production activity, such as the use of firewood in beer brewing or brick making, in which case they are referred to as input goods. Environmental resources can also be used by rural households as output goods for sale, for example, households gather natural resources which they do not consume themselves and sell them in order to supplement total household income. Finally, rural households harvest resources from the environment to produce household durables such as furniture or keep stocks of environmental resources for future use. Timber and firewood are examples of resources that can be stored for future use.

Closely related to the discussion above is the literature on forest income, which explores the three roles of environmental resources, that is, preventing poverty by acting as insurance or safety nets (Shackleton et al., 2008; Thondhlana & Muchapondwa, 2014), reducing poverty via increased earnings (Fisher, 2004; Vedeld et al., 2007), and, finally, playing a role in equalizing income (Cavendish, 1999; Fonta & Ayuk, 2013). Shackleton et al. (2008) argued that income derived from the environment acts as a safety net for poor rural households by mitigating agricultural risk through direct and indirect provisioning. Poor rural households use environmental income[8] as a method of diversification to cushion themselves against shocks

[8] Environmental income is defined here as the sum of direct use values and cash income derived from environmental resources (Thondhlana et al., 2012).

associated with illness, crop failure, loss of employment, changes in food prices or extreme weather conditions.

From the literature, a number of issues stand out with regard to the study of the human-environment relationship. These issues relate to unequal utilization and differentiation in the types of environmental resources used in communal areas, the contribution of environmental resources to total household income, wealth differentiation and resource utilization, and whether environmental income reduces poverty and rural inequality. Environmental resources offer goods to rural households that have considerable differentiation in terms of their economic characteristics and utilization (Cavendish, 1999; 2000). Shackleton and Shackleton (2006) and Kar and Jacobson (2012) argued that, within any given community, there is significant socio-economic differentiation and it is important to acknowledge such differentiation when considering policy formulation and management interventions in order to support rural livelihoods and promote sustainable utilization of natural resources. Thondhlana et al. (2012) and McGregor (1995) emphasised the role of contextual factors such as culture, social institutions, ecological conditions and infrastructure in influencing access and the ultimate utilization of resources.

In Southern Africa, poorer households depend heavily on environmental resources, which contribute about 40% to their incomes, although richer households use greater quantities of these resources in total (Cavendish, 2000). Through a detailed examination of use and value of four non-timber forest products (NTFPs), Shackleton and Shackleton (2006) found evidence supporting Cavendish's claim that poorer households benefit more than wealthier classes from environmental resource utilization in proportional terms. In addition, wealthier households purchase more NTFPs, while a greater proportion of poor households were actually involved in selling NTFPs (McGregor, 1995; Shackleton & Shackleton, 2006). Richer households generated more environmental income in total than poorer households because they have more man-made assets for collecting resources (Cavendish, 1999; Uberhuaga et al., 2012; Ambrose-Oji, 2003).

There are mixed results with regard to the effect of environmental income on poverty reduction. It is still not clear whether environmental income can actually move poor households across the poverty line, but there is agreement that such resources can mitigate poverty and, at least, make some households less poor. For example, Cavendish (1999) reported that environmental income is important in mitigating poverty, but might not be responsible for lifting poor households out of poverty. In contrast, in a study of forest income and resource dependence in lowland Bolivia, Uberhuaga et al. (2012) reported that forest income has the potential to move households out of poverty, provided that the environmental resources are fully commercialised and the rural households are integrated into the mainstream economy. Fonta et al. (2011) and Lopez-Feldman et al. (2007) also found evidence that forest income reduces rural poverty in Nigeria and Mexico respectively. Using meta-analysis of 51 case studies from 17 countries, Vedeld et al. (2007) established that forest environmental income represents on average 22% of the total income in the sampled population.

There is general consensus about the role of environmental income in reducing rural inequality (Cavendish, 2000; Cavendish & Campbell, 2002; Fisher, 2004; Vedeld et al. 2007; Fonta et al. 2011). For example, using a sample of 213 households from rural Zimbabwe, Cavendish and Campbell (2002) found that environmental income is strongly and significantly equalizing[9], bringing about a 30% reduction in inequality. A study by Fonta et al. (2011) found that forest income reduces income inequality in rural Nigeria. Using Gini decomposition, Fisher (2004) showed that access to forest income reduced measured income inequality in Malawi. Vedeld et al. (2007) found that forest environmental income has a strong equalizing effect on local income distribution. Most studies conducted so far are concerned with the broader environmental income category. This chapter contributes to the literature by defining two specific categories of environmental income (with and without wildlife benefits).

2.3. Research Methods

2.3.1 Empirical strategy

The study is motivated by three specific questions. i) Does environmental income (including wildlife) contribute significantly toward total household income and reduction in rural poverty and income inequality? ii) How does environmental income (and specifically income from

[9] Environmental income has an equalizing effect on rural income distribution if it can result in a reduction in inequality. Therefore, access to environmental income improves social welfare through its role in both increasing and equalizing incomes.

wildlife) compare with other sources of income? iii) What determines environmental income generation among poor households living adjacent to protected areas? To address these questions, this chapter uses income quintile analysis, the Foster-Greer-Thorbecke poverty measure, Gini coefficient analysis, Gini decomposition analysis and regression analysis. A brief description of these analytical techniques is given in the next section.

To answer these questions, we first define three different income measures used in this study. Standard household income (Y_0) is defined as income derived when standard household budget surveys are implemented (Cavendish, 1999). This comprises all household economic activities (i.e., wage income, remittances, gifts and transfers, farm income, capital income and self-employment), but excludes environmental income. We expect the results of the standard household income not to differ significantly from previous studies. Standard household income is used as the baseline against which we measure the relative contribution of environmental income. Environmental income includes both direct use values and cash income (i.e., direct consumption of resources by households, environmental-based labour income and sales of environmental resources).

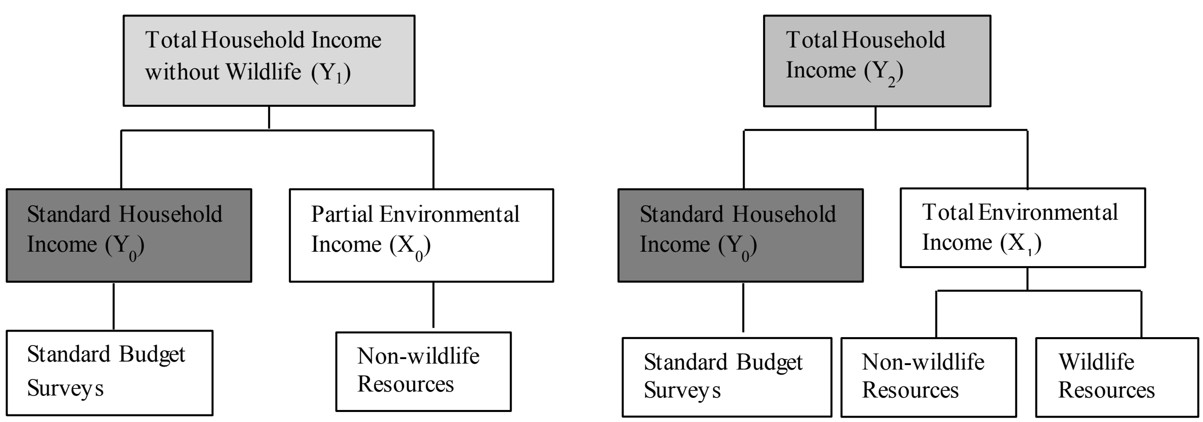
Because we are dealing with local communities living adjacent to a game park and whose livelihoods depend on wildlife conservation, the main goal of this chapter is therefore to examine the effects of wildlife resources on welfare. We are concerned with two scenarios, a scenario with wildlife benefits and a scenario without. As a result, a distinction is made between non-wildlife and wildlife income. Wildlife income captures household consumption and sales of bush meat, small animals, fish and birds (illegal harvest), in addition to income generated from legal activities which includes the money and meat from wild animals killed through trophy hunting[10] in the community's conservation area. It is important to note that the term 'income' as used in this chapter refers to income per adjusted adult equivalent unit and not per capita. Per capita is an extreme form of the adult equivalence scale that adjusts household income on a per person basis. The adjusted adult equivalent unit accounts for the different contributions between household members as per capita measurements underestimate individual contributions since they overlook differences in household composition (Cavendish, 1999; 2000). In computing income per

[10] Local communities get income from legal hunting activities by engaging a Safari operator or local professional hunter who utilises the quota on their behalf.

adjusted adult equivalent, we divide household income by the number of adults in that household and where there are children, we add some weights. The different incomes defined above were measured for the 12 month period spanning the 2012/2013 agricultural season. As a result, all the calculations and analysis in this chapter were done with reference to this period of time.

We also differentiate partial environmental income or environmental income without wildlife resources (X_0) from total environmental income (X_1) derived from both non-wildlife and wildlife resources. As suggested earlier, a distinction is made between these two categories of environmental income because other studies concerned themselves with the broader environmental income category. We define total household income excluding wildlife (Y_1) as the sum of standard household income (Y_0) and environmental income without wildlife resources (X_0). Finally, we define total household income (Y_2) in terms of standard household income (Y_0) and total environmental income (X_1). Thus, the main difference between the two measures of total household income, i.e., Y_1 and Y_2, is that the former does not include wildlife income, while the latter includes wildlife income. Figure 2.1 below summarises the different measures of income computed in this chapter.

Figure 2.1: Summary of income measures



Cavendish (1999) highlighted a number of controversies in poverty research when it comes to measuring poverty and the choice of a measure of welfare. This chapter follows standard practice in poverty research, considers these issues and chooses income as a measure of welfare. There is also disagreement in the literature about whether the poverty line should be absolute (fixed), relative or subjective (e.g., Ravallion, 1992; Deaton, 1997). Cavendish (1999) argued that deciding the exact position of the poverty line is not as important as comparing results for poverty measures under different assumptions about the location of the poverty line. For the purposes of this analysis, the robustness of poverty measures to different poverty lines is therefore more important than a point estimate of poverty. Following Cavendish (1999), this study uses poverty lines, fixed with reference to the standard income distribution, that span a wide range of incomes in our sample. The poverty lines correspond to the uppermost incomes of the income quintiles of the standard income distribution and are consistent with other studies.

a) Income quintile analysis: To analyse the contribution of environmental income to total household income, the study used income quintile[11] analysis. Based on this technique, the sample was divided into five income groups in such a way that 20% of the population lies in each group. This made it easier to examine the contribution of environmental income with and without wildlife to total household income by income category. The household questionnaires included quantitative questions about a wide range of environmental goods and their uses and values as part of household's income, consumption and expenditure. As a result, household values were calculated on the basis of environmental use rather than resource availability. Mostly, economic transactions were valued at local market prices[12] and value addition calculated for subsistence agricultural output. In cases where market prices could not be determined, household reported values were used to allow for a comparison of environmental income against a full accounting of the household's other economic activities.

Using standard principles for agricultural households involved in both market and non-market activities, the environmental resource use and non-environmental economic data were valued and aggregated to produce household income accounts (see Grootaert, 1982; Cavendish, 1999; Cavendish, 2000; Thondhlana et al. 2012). Therefore, the method employed for valuing environmental resource utilization was similar to all other economic transactions, i.e., household's own reports of quantity, total value of resource utilization (consumption) or sales. However, there is a limitation to this method in that most environmental goods are not traded on the market. This is the main reason why environmental resources have been excluded from

[11] Quintiles are points taken at regular intervals from the cumulative distribution function of a random variable such as income, where the sample is divided into fifths according to the neighbourhood socioeconomic status.

[12] Market prices are ideal because they represent clearing prices.

standard household income analysis in the past (Cavendish, 2000). To minimise this problem, the quantitative data was supplemented by qualitative data collected through key informant interviews.

b) The Foster-Greer-Thorbecke (FGT) poverty measure: The Foster-Greer-Thorbecke (FGT) metric is used to examine whether environmental income (including wildlife) can reduce rural poverty. It is a generalised measure of poverty within an economy. We ask, is environmental income capable of lifting people out of poverty or, at least, making people less poor than they were without it? Hence, the FGT metric measures the income shortfall expressed as a share of the poverty line and is weighted by a "sensitivity parameter" α (Donaldson & Weymark, 1986). Algebraically, we have

$$FGT^{(\alpha)} = \frac{1}{N}\sum_{i=1}^{H}\left(\frac{z-y_i}{z}\right)^{\alpha} \text{.....................}(2.1)$$

where z is an agreed upon poverty line (e.g., the most common poverty line used for Africa by the World Bank is US$450.00 per person per annum, which corresponds to about US$ 1.25 per day adjusted for purchasing power parity), N is the number of people in an economy, H is the number of poor households (i.e., those with income at z or lower), and y_i are individual incomes. The interpretation of the sensitivity parameter draws its inspiration from Atkinson (1970). A low value of the sensitivity parameter (α) implies that the FGT measure weights all the individuals with incomes less than z roughly the same, while a high value puts more weight on those individuals with the lowest incomes (i.e. furthest below z). A very high FGT statistic implies more poverty in the economy.

For specific values of α, the FGT statistic corresponds to other measures of poverty. For example, if $\alpha = 0$, the formula reduces to the Headcount ratio (H), which is the fraction of the population living below the poverty line. Mathematically, this is written as:

$$FGT^{(0)} = \frac{H}{N} \text{.....................}(2.2)$$

If $\alpha = 1$, then the FGT metric reduces to the average poverty gap[13], that is, the average amount of income required to move those in poverty up the poverty line. Thus, we have

$$FGT^{(1)} = \frac{1}{N}\left(\frac{z - y_i}{z}\right) \text{...}(2.3)$$

While a great deal of the literature on poverty uses these two versions of the FGT, other studies make use of the FGT statistic where $\alpha = 2$, so that the index reduces to the poverty severity measure.

$$FGT^{(2)} = \frac{1}{N}\sum_{i=1}^{H}\left(\frac{z - y_i}{z}\right)^2 \text{...}(2.4)$$

Using such a tractable form, the statistic combines information on both poverty and income inequality[14]. Rewriting the FGT statistic, we obtain

$$FGT^{(2)} = H\mu^2 + (1 - \mu^2)C_v \text{...}(2.5)$$

where H is the number of poor households as defined above, C_v is the coefficient of variation among those with income such that $y_i < z$, and $\mu = \frac{1}{N}\sum_{i=1}^{H}\left(\frac{z - y_i}{z}\right)$

The FGT class of decomposable poverty measures discussed above was first introduced by Foster, Greer, and Thorbecke (1984).

c) Gini coefficient analysis: To establish whether environmental income reduces rural inequality, the Gini coefficient method is employed. The Gini index is a summary statistic that measures how equitably or inequitably a resource, e.g., income, is distributed in a society. The advantage of using the Gini coefficient is that the statistic is a self-contained summary of economic data which is easy to compute and interpret (Farris, 2010). It can be defined mathematically based on the Lorenz curve[15]. Therefore, the Gini coefficient can be thought of as

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13 This is equivalent to the amount an average person in the economy would have to contribute in order for poverty to be barely eliminated.

14 The FGT also considers inequality among the poor, but as the proper amount for α is not defined (i.e. it is a normative question).We are thus not able to say that the Gini is part of the FGT.

15 By definition, the Lorenz curve shows the distribution of a quantity in a population. For a resource Q, the Lorenz curve is the curve $y = L(p)$, where the Q -poorest fraction p of the population has a fraction $L(p)$ of the whole and the value of p is called the percentile variable. If everyone in the economy had exactly the same amount of Q,

the ratio of the area that lies between the line of equality (i.e., the 45^0 line) and the Lorenz curve over the total area under the line of equality. The Gini index is thus defined as an integral that shows how much the Lorenz curve in question deviates from perfect equity, i.e., the 45^0 line. Mathematically we have:

$$G = 2\int_0^1 [P - L(p)]dp \qquad (2.6)$$

where the factor 2 scales the area in such a way that the Gini index varies between 0, perfect equity where everyone in the economy has the same share of the good, and 1, where one person has everything. This is an indirect method of calculating the Gini coefficient through the construction of the Lorenz curve. However, the index can be computed through a direct method as follows:

$$G = \frac{1}{\mu N(N-1)} \sum_{i>j} \sum_{j} |y_i - y_j| \qquad (2.7)$$

where μ is mean income, N is the total number of observations, y_i and y_j are the dollar values of income for individuals i and j (Thomas et al., 2000).

***d) Gini decomposition analysis**:* This chapter uses the Gini decomposition approach to investigate the relative contribution of income components to income inequality for local communities living adjacent to Gonarezhou National Park in Zimbabwe. In a South African study, Leibbrandt et al. (2000) considered income from six different sources i.e. wage income, remittances, agricultural income, capital income, state transfers and self-employment. In this study, we expand these variables and include environmental income in the analysis. By going deeper into each and every component, we ask whether there are any policy interventions that can either increase or decrease income generation.

Following Leibbrandt et al. (2000) and Shorrocks (1993), we assume n households deriving income from K different sources. Let y_i represent the total household income $i = \{1,...,n\}$ and y_{ik} represent total income for household i and source $k = \{1,...,K\}$; hence, $y_i = \sum_{k=1}^{K} y_{ik}$. Assuming

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then the order of our imaginary line-up would be completely arbitrary and $L(p) = p$, the curve of perfect equilibrium.

that the distribution of household income and the income components are represented by $\gamma = \{y_1,...,y_n\}$ and $\gamma_k = \{y_{1k},...,y_{nk}\}$ respectively, the Gini coefficient for the distribution of total income within the group can be defined as:

$$G = \frac{2Cov[\gamma, F(\gamma)]}{\mu} \qquad (2.8)$$

where μ represent mean household income, $F(\gamma)$ is the cumulative distribution of total household income i.e. $F(\gamma) = f(y_i),........, f(y_n)$, and $f(y_i)$ is the rank of y_i divided by the number of observations n. The key aspects of the Gini decomposition technique can then be summarised as in Leibbrandt et al. (2000) and Stark et al. (1986) as follows:

$$G = \sum_{k=1}^{K} R_k G_k S_k \qquad (2.9)$$

Where S_k denotes the share of income source k in total group income, i.e., $S_k = \frac{\mu_k}{\mu}$, G_k represents the Gini coefficient measuring inequality in the distribution of income component k within the group and R_k is the Gini correlation of income from source k with total household income, defined as follows:

$$R_k = \frac{Cov[\gamma_k, F(\gamma)]}{Cov[\gamma_k, F(\gamma_k)]} \qquad (2.10)$$

To check for robustness of our results, we consider results from the Gini decomposition using the Rao's (1969) analytical approach, the Shapley (1953) decomposition approach and the FGT decomposition approach.

e) Econometric modelling of the relationship between poverty and environmental income: This section is concerned with modelling the relationship between environmental income and relative poverty using regression analysis techniques. The human-environment relationship raises two important questions that need attention. First, does environmental income affect households in various income quintiles in the same way? Second, what are the determinants of environmental income? Given these two questions, it is hypothesised that environmental income is capable of moving households across income categories, and that household wealth and the status of biodiversity in an area are key determinants of environmental income generation. If these assertions are correct, then it is conceivable through appropriate policy designs to improve household welfare by increasing access to wildlife income by relatively poor households, while imposing restraints on wealthier households.

To answer the first question, it is possible to capture the impact of environmental income (EI) on households in different income quintiles using an ordered logistic regression model. These quintiles were calculated based on standard household income and therefore exclude environmental income. Suppose the sampled population can be divided into five categories according to the level of income by calculating the first, second, third, fourth and fifth income quintiles; then, for $i = 1, 2, 3, 4, 5$ the dependent variable Y_i is ordered and increasing. We can therefore rewrite the dependent variable as follows: 1=low income, 2= lower middle income, 3=middle income, 4=upper middle income, and 5=high income. Thinking in terms of relative poverty, we can say that households in a lower quintile are relatively poorer than households in a higher quintile. Algebraically, we can write the following model.

$$Y_i = \beta_0 + \beta_1 Age + \beta_2 Areacult + \beta_3 EI + \beta_4 Hhsize + \beta_5 Employ + \beta_6 Lhead + \beta_7 Educ ... + \xi_i ....(2.11)$$

where the explanatory variables in equation 2.11 are defined as follows: age of household head (Age), area under crop cultivation (Areacult), environmental income (EI), household size (Hsize), household head employed (Employ), number of years living in the area (Lhead), and number of years in school of the household head (Educ). Theoretically, it is plausible to believe that most households tend to move from lower income quintiles to higher quintiles as the age of the household head, area under cultivation, household size, number of years living in the area and the level of education increase and as their employment status changes from unemployed to employed. This is so because opportunities to get employment and acquire more wealth (including farm land) and other assets come with age and education. As the number of years living in the area increase, people accumulate knowledge of their environment and become more away of the opportunities it offers through learning and experience. We use community level dummies to capture resource heterogeneity and other spatial differences that influence incomes.

In a study of factors affecting poverty dynamics in rural Zambia, Chapoto et al. (2011) demonstrated that accumulation of assets, such as farm land, offer a pathway to sustained high levels of income. The results of Chapoto et al. (2011) underscore the importance of education and employment, both for avoiding chronic poverty and for ensuring that livelihood earnings remain consistently above the poverty line. In a study of rural Nigeria, Apata et al. (2010) found a negative relationship between poverty and educational attainment of the household head. The study of Bogale et al. (2014) for rural Ethiopia established that the probability of a household being poor tends to diminish as household size and age of the household head increases using per capita household calorie consumption.

In the second part, we find the determinants of the different amounts of environmental income that people generate, i.e., $EI = f(S_i, W_i, ...)$. So we propose a regression model of the form:

$$EI_i = \beta_0 + \beta_1 S_i + \beta_2 W_i \cdots + \mu_i \text{..}(2.12)$$

where EI_i represents environmental income; S_i denotes household characteristics such as the age of the household head, education, employment status, household size e.tc; W_i indicate household wealth; and μ_i is the error term. As is common in the economics literature, a proxy for the wealth variable is recovered from household assets, livestock ownership, agricultural implements (Shackleton & Shackleton, 2006) and farm size[16] using factors analysis.

From empirical studies done in the developing countries, there is substantial evidence suggesting that households generate more environmental income as the age of the household head and household size increase (Godoy et al., 1997; Escobal & Aldana, 2003; Mamo et al., 2007; Fonta et al. 2011). On the other hand, there is also evidence showing that the household dependence on environmental income diminishes with the level of educational attainment and as the employment status of the head of the household changes from unemployed to being employed (Angelsen & Kaimowitz, 1999; Godoy & Contreras, 2001; Fisher, 2004; Mamo et al., 2007; Kamanga et al., 2009; Fonta et al. 2011). Shackleton and Shackleton (2006) found evidence of the association between household wealth status and environmental resource use in the Kat River Valley in South Africa. Contrary to the finding of Shackleton and Shackleton (2006), Vedeld et al. (2007) and Fisher (2004) reported a negative relationship between wealth and forest income.

[16] Farm size is included in this computation because it captures agricultural income from crops thereby marking our index a better proxy of the wealth index.

As before, we control for resource heterogeneity and other spatial differences that influence incomes using community level dummies.

In this study, it is assumed that, as household wealth increases, people invest in technology, such as carts and draught power, to harvest more environmental resources. Conversely, as households obtain more environmental income, they use the excess income to accumulate more assets, thereby increasing their wealth status. Because of potential reverse causality between environmental income and wealth, we suspect that t endogeneity is an issue in this relationship. In such a case, the most appropriate way forward is using the standard instrumental variables regression model as our method of choice if instruments were available. However, this study could not find suitable external instruments for use in traditional instrumental variables estimation from the primary data collected. Because of this obstacle, this study makes use of instrumental variables estimation with heteroskedasticity-based instruments that methodologically deal with the endogeneity problem (Lewbel, 2012; Baum et al., 2013).

Based on Lewbel (2012), this method estimates an instrumental variables regression model that provides the option to generate instruments and allows the identification of structural parameters in regression models with endogeneity or mismeasured regressors in the absence of traditional identification information such as external instruments. Identification is achieved in this context by having explanatory variables that are uncorrelated with the product of heteroskedastic errors, which are a key feature of models where the correlations in the error terms are due to an unobserved common factor (Baum et al., 2013). Because instruments are constructed as simple functions of the model's data, the approach may be applied in cases where no external instruments are available, or may be used to supplement weak external instruments in order to improve the efficiency of the instrumental variables estimator (Lewbel, 2012).

Thus the instrumental variables estimation with heteroskedasticity-based instruments is a good substitute of the standard IV approach in terms of addressing the problem of endogeneity. The choice one uses depends on the availability of sound external instruments. If good external instruments are available, then the standard IV approach is superior. If external instruments are either weak or not available, then the method of heteroskedasticity-based instruments is superior to the conventional IV approach. Accordingly, the standard IV did not add value in our case as it is inferior to heteroskedasticity-based instruments. This method is gaining popularity and it is being used widely in many studies (e.g., Mishra & Smyth, 2015; Banerjee et al. 2013; Emran et al., 2012). Mishra & Smyth (2015) used two data sets from China to compare the identification strategy suggested by Lewbel (2012), which utilizes a heteroskedastic covariance restriction to construct an internal IV, and the standard IV. They found that Lewbel's approach provides plausible estimates in datasets in which conventional IVs are not available.

For a detailed description of the method for constructing instruments as simple functions of the model's data, please refer to Appendix A. There are potential shortcomings of this approach. One of the major drawbacks of Lewbel's approach is that identification relies upon higher moments, and is likely to be less reliable than identification based on coefficient zero restrictions. However, in the absence of plausible identifying restrictions like in our case, this approach may be the only reasonable strategy.

2.3.2 Data sources

The data for the analysis was drawn from a household survey conducted in June/July 2013 with local communities living adjacent to the Gonarezhou National Park in Zimbabwe. Both formal and informal methods of primary data collection were employed e.g., structured interviews using questionnaires, semi-structured interviews such as key informant interviews diaries to record environmental resource utilization and income sources. The data was collected from 336 randomly selected households, in 13 Wards and 31 communities involved in wildlife conservation. The household questionnaire collected information about a wide range of environmental resources utilization and economic characteristics (including wildlife resources) in addition to the usual household socio-economic characteristics and agricultural activities. Comparing households in communal areas and those in resettlement schemes seemed to be problematic because the sample for the latter tenure regime is quite small relative to the former[17].

[17] We will not therefore compare the two tenure systems in most of the analysis, except in regression models where an attempt is made to control for the effects of tenure. This implies that the poverty analysis will be based only on the pooled sample.

2.4. Results and discussion

2.4.1 Household characteristics

A total of 336 interviews were conducted. Since over 90% of peasant farmers involved in wildlife conservation are from the communal areas and less than 10% are found in the resettlement schemes, the interviews consisted of 306 households from the former and 30 households from the latter. Table 2.1 shows some characteristics of the sampled households around the Gonarezhou National Park in Zimbabwe. The mean age of the household heads, number of years in school, number of years living in the area, household size, area under crop cultivation, distance to the nearest town (i.e. some measure of market integration) and mean number of dogs[18] owned for the sampled population are 48.9years, 5.52years, 36.6years, 6.4, 2.7ha, 65.5km and 1.1 respectively. The proportion of household heads born in the area is about 70.8%, while the proportion of household heads who are Christians is 59.2%. Using a scale from zero to one hundred, the mean wealth index (31.8) shows that the average household in the sampled population is generally poor, i.e., the index lies below half (50.0).

Table 2.1: Household characteristics by tenure category

Variables	***N_i***	***Mean***
age of household head	336	48.88
number of years in school	336	5.524
number of years living in the area	336	36.64
household head born in the area [0,1]	336	0.708
religion of household head [0,1]	336	0.592
household size	336	6.423
area under crop cultivation	336	2.656
distance to the nearest town	336	65.45
wealth index	336	33.37
number of dog	336	1.080

Source: survey results 2013

Households were categorised in terms of income quintiles. Five categories of income corresponding to the 1st, 2nd, 3rd, 4th and 5th income quintiles were defined as follows: low income, lower middle income, middle income, upper middle income and high income households. Table 2.2 presents the uppermost incomes of the income quintiles of the standard income distribution. 20% of the households lie below US$208.35 in the first income quintile,

[18] The number of dogs owned by a household matters in this analysis because dogs are used by local people for hunting purposes.

40% of the households lie below US$280.63 in the second quintile, and so on. The analysis makes use of two poverty lines widely used in the literature. Zimbabwe's official poverty line of US$1 per day or US$360.00 per annum (adjusted for purchasing power parity) corresponds to the threshold of the third income quintile (Zimbabwe National Statistics Agency, 2014). The lowest poverty line used by the World Bank (1990) of US$1.25 per day or US$450.00 per annum (average line of the poorest 15 countries) corresponds to the threshold of the fourth income quintile. The poverty line for the World Bank was chosen for comparison purposes, to check for robustness of the results and the idea of basing the analysis on a relative poverty line rather than a fixed line. However, the interpretation of the results is based on Zimbabwe's official poverty line.

Table 2.2: Income quintiles

Income quintiles	Uppermost income
1st quintile	208.35
2nd quintile	280.63
3rd quintile	376.23
4th quintile	495.25
5th quintile	668.63

Source: survey results 2013

2.4.2 Utilization of environmental resources

There are a number of resources from which communities living adjacent to the Gonarezhou National Park derive their livelihoods; these include rangelands, woodlands, watering points, rivers and dams. These resources sometimes form part of what the community refers to as its conservation area or wilderness. By law, communities living adjacent to national parks are required to set aside a piece of land or keep the land for conservation purpose, if they are to participate in wildlife conservation. The land usually lies between the community and the game park, but within the vicinity of the game park. This land traditionally belonged to the communities in question but, with the establishment of the game park in 1975, communities lost part of their land to the state. Households are not allowed to harvest wildlife directly from their conservation area, but they can do so as a community through trophy hunting and tourism activities done by the Safari operators, in which case the community receives some income.

Table 2.3 below shows that households from the study site harvested and used an enormous range of environmental resources, which in turn provided a wide range of economic characteristics. For example, wood can be used as firewood, fencing material, furniture or for construction purposes. Cavendish (2000) categorised these resources as consumption goods, input goods, output goods and durables or stocks, according to their economic functions. The resources can also be categorised as food and non-food items. As stated earlier, these resources come from a wide range of different ecological niches and are either owned communally or individually, depending on whether the resources are found on communal land or on an individual plot, such as in resettlement schemes. It is therefore common for access or use rules to exist for common pool resources, such as wildlife, in order to guide their utilization over time. If such institutions are not in place, then the resource system is subject to an open access regime.

Table 2.3: Classification of Environmental Resources by Economic Function

Consumption goods	Inputs	Output goods	Durables & stocks
Wild vegetables	Firewood-brick making	Wild vegetable sales	Furniture
Mushroom	Firewood-beer brewing	Mushroom sales	Timber
Wild fruits	Leaf litter	Wild fruit sales	Firewood store
Bush meat (large animals)	Thatching grass	Bush meat sales	
Small animals	Livestock fodder	Wine sales	
Fish	Termitaria*	Firewood sales	
Wild medicines	River sand	Insect sales	
Insects	Watering points	Construction wood sales	
Wine	Pastures	Thatching grass sales	
Firewood (cooking & heating)		Carpentry/furniture sales	
Agricultural implements		Woven goods sales	
Household utensils		Pottery sales	
Woven goods - Baskets		Gold sales	
Pottery		Broom grass	
		Carving	
		Bricks	

Source: Adapted from Cavendish (1999)
* A nest built by a colony of termites underground or above ground (usually as a mound). Poor rural households in Southern Africa use the soil from this nest to improve the soil nutrients in their fields.

Considering major food items and non-food items consumed by local communities around the Gonarezhou National Park, the results show that richer households consume more environmental resources (both food and non-food items) in total than poorer households (see Table 2.4 below). This is consistent with the results of Cavendish (1999; 2000), Shackleton and Shackleton (2006), and Thondhlana and Muchapondwa (2014). In line with the study of Twine et al. (2003), richer households consume more of valuable resources such bush meat, fish, timber, firewood and livestock fodder, while poorer households consume more of less valuable resources such as wild vegetables, wild fruits, insects and thatch grass. As one of the main motivations of this chapter, the results reveal that relatively wealthier households consume more wildlife products in the aggregate than do relatively poor households. Table A1 in the appendix shows the gender composition of the data, i.e., how gender influences resource extraction. The results illustrate that male headed households consume more valuable resources in aggregate such as wildlife products and other major non-food items than female headed households, while female headed households consume more of less valuable resources such as wild foods and thatch grass.

Table 2.4: Quantity consumed by income quintile (kgs)

Variables	**Income quintiles**					**Total**
	1st quintile	*2nd quintile*	*3rd quintile*	*4th quintile*	*5th quintile*	
Quantity of major wild foods items consumed						
vegetables	9.14	7.843	7.455	6.306	4.537	**7.056**
mushroom	0.485	0.56	0.179	0.761	0.946	**0.586**
insects	11.75	10.6	8.881	7.34	7.981	**9.310**
fruits	2.787	2.582	1.928	1.067	0.791	**1.831**
honey	0.326	0.328	0.459	0.994	1.545	**0.730**
Sub-total	**24.49**	**21.91**	**18.90**	**16.46**	**15.80**	**19.51**
Wildlife products only						
bush meat	3.397	6.269	8.075	10.537	15.104	**8.676**
small animals	4.787	4.299	4.903	3.903	3.881	**4.355**
fish	1.809	4.515	5.254	8.313	10.851	**6.148**
birds	1.934	1.687	1.858	1.082	0.724	**1.457**
Sub-total	**11.93**	**16.77**	**20.09**	**23.84**	**30.56**	**20.64**
Quantity of major non-food items consumed						
timber	19.55	19.85	24.94	26.25	46.04	**27.33**
firewood	664.2	679.9	688.3	754.9	825.9	**722.64**
thatch grass	61.49	59.49	52.84	44.1	39.78	**51.54**
basket	1.897	2.045	2.612	3.343	3.851	**2.750**
livestock fodder	3.235	9.552	9.848	12.16	20.3	**11.02**
Sub-total	**750.374**	**770.840**	**778.514**	**840.756**	**935.873**	**815.271**
Total consumed						
Total	**786.794**	**809.522**	**817.535**	**881.062**	**982.235**	**855.437**

Source: survey results 2013

The analysis in the table above may not be robust since per capita household income is a flow variable which is based on cross-sectional data to classify households and understand their natural resource consumption pattern. Household income may change over time due to shocks such as loss of employment or remittances. Thus we need to check for robustness of our results using a stock variable that is constant over time such as livestock or farmland since farmers need more time to change the stock of land or livestock. Table A2 in the appendix uses farmland to check for robustness of our results.

The results in Table 2.5 show that 53.1% of the households purchased environmental resources during the 2012/2013 agricultural season, while 59.7% sold environmental resources. Consistent with the study of Shackleton and Shackleton (2006), wealthier households purchased more environmental resources than did poorer households, while poorer households sold more environmental resources. Please note Table 2.5 is showing the number of households involved in selling environmental resources, but it does not say anything about the actual values. Table A.3 shows that the most sold environmental resources are food items, which are less valuable and less bulky in nature compared to non-food items. Poorer households harvest and sell more food items than richer households. On the other hand, wealthier households retain most of the valuable resources they extract from the environment such as timber, firewood and livestock fodder. Consequently, wealthier households extract and consume more environmental resources in aggregate and in value.

Table 2.5: Percent household purchased or sold environmental resources by income quintile

Quintile	% Household purchased or sold environmental resources	
	Purchased	*Sold*
1st quintile	0.429	0.762
2nd quintile	0.507	0.657
3rd quintile	0.463	0.616
4th quintile	0.582	0.582
5th quintile	0.672	0.367
Total	0.531	0.597

Source: survey results 2013

2.4.3 Contribution of environmental income (including wildlife) to total household income

The household income accounts in Table 2.6 show that wage income, farm income and environmental income are the three most important sources of household income for communities around the Gonarezhou National Park. Although agricultural income (35.4%) dominates all the sources of income, the contribution of environmental income (28.7%) is also quite substantial. The environmental income is made up of environmental-based labour income[19], wildlife income and other environmental resources, each contributing 7.8%, 6.2% and 14.7% respectively, to total household income.

Table 2.6: Household income accounts per adjusted adult equivalent unit per annum

Variable	Obs	Mean	Std. Dev.	Income share
Full time employment	336	60.20	198.4	9.5
Casual labour	336	34.48	43.45	5.4
Self -employment	336	44.65	101.9	7.0
Total wage income	**336**	**139.3**	**228.8**	**21.9**
Crop sales	336	108.9	120.6	17.2
Livestock sales	336	100.3	135.4	15.8
Animal product sales	336	15.26	19.33	2.4
Manure sales	336	0.0818	1.046	0.01
Total farm income	**336**	**224.5**	**216.0**	**35.4**
Income from land rented out	336	0.470	2.460	0.07
Income from draught power hired out	336	41.20	41.66	6.5
Capital income	**336**	**41.67**	**42.02**	**6.6**
State transfers	336	3.237	16.47	0.5
Community projects	336	2.755	14.02	0.4
Food relief	336	6.929	6.902	1.1
Net gifts	336	1.978	7.867	0.3
Total transfers	**336**	**14.90**	**30.18**	**2.3**
Remittances	**336**	**32.59**	**44.32**	**5.1**
Environmental based labour income	336	49.33	83.25	7.8
Wildlife income	336	39.19	46.29	6.2
Environmental income (without wildlife)	336	93.19	91.47	14.7
Environmental income	**336**	**181.7**	**174.8**	**28.7**
Total household income	**336**	**634.7**	**381.6**	**100.0**

Source: survey results 2013

To examine the contribution of environmental income to total household income, the chapter used income quintile analysis. Three definitions of income are used to accomplish this objective: standard household income, total household income without wildlife (including standard household income and non-wildlife income), and total household income (including standard household income, non-wildlife income and wildlife income). Standard household income is used as the baseline in this analysis. Wildlife income is made up of income (including the actual

[19] Environmental-based labour income includes labour income derived from harvesting and processing environmental resources such as digging termitaria, thatching, brick moulding, etc.

consumption of game meat) from hunting and tourism activities done legally by the community through engaging Safari operators and illegal hunting activities (game meat consumed by the household out of its own production and income realized from selling game meat).

In the aggregate, non-wildlife environmental resources contributed about 31.5% to total household income, while the total contribution of environmental resources including wildlife is 40.1% implying a net effect of 6.6% from wildlife alone. Disaggregating the three measures of income by income quintiles, the results show that adding both non-wildlife and wildlife income to standard income increases total household income across income quintiles (see Table 2.7 below). The increase in total household income resulting from the inclusion of non-wildlife and wildlife resources is much higher for households in lower income quintiles than it is for households in higher quintiles. This standard result confirms the findings of Cavendish (2000), Shackleton and Shackleton (2006) and Thondhlana and Muchapondwa (2014) that poor households derive greater relative benefits than richer households from utilizing environmental resources. The contribution is to show that, in particular, poorer households derive greater benefit from the consumption of wildlife resources than wealthier households.

Table 2.7: Income measures by quintile

Quintile	Standard household income	Total household income without wildlife		Total household income		Effect of wildlife
		Mean	*% change*	*Mean*	*% change*	*% change*
1st quintile	204.6	279.9	36.9	300.1	45.7	7.2
2nd quintile	301.1	405.6	34.7	432.6	45.0	6.6
3rd quintile	367.8	502.4	34.6	542.8	43.6	6.5
4th quintile	527.2	668.3	29.8	710.1	35.7	6.2
5th quintile	867.9	1126.0	26.7	1194.0	32.6	6.0
Total	**453.0**	**595.5**	**31.5**	**634.7**	**40.1**	**6.6**

Source: survey results 2013

2.4.4 Environmental income and poverty

Table 2.8 below presents the results of the FGT poverty statistics. Three measures of poverty are used: the headcount ratio, the poverty gap, and the poverty severity measure. Overall, the results illustrate that the proportion of people in the full sample living below the poverty line is greatly reduced when we account for non-wildlife income initially and wildlife income later (i.e., from about 47.6% to 24.7% and then to 22.1%). The inclusion of non-wildlife resources accounts for approximately 48.1% reduction in poverty, while the inclusion of wildlife resources accounts for 53.6% reduction in poverty (refer to Table A.4 in appendix A). The net effect of wildlife income alone is about a 5.5% reduction in the proportion of people living below the poverty line. Comparing the results of the headcount ratio with the poverty gap and poverty severity indices, the reduction is massive. Poverty depth in the full sample is reduced from 16.6% to 5.5% with non-wildlife income and then 4.3% with wildlife income.

Table 2.8: Comparison of FGT indices assuming a poverty line of US$360.00 per capita

Units of analysis	**Mean income**	**Headcount ratio (%)**	**Poverty gap**	**Poverty severity**
Standard household income				
All households (N=336)	452.97	47.6	16.6	7.4
1st quintile	204.62	100.0	42.5	20.1
2nd quintile	301.06	100.0	17.1	6.2
3rd quintile	367.75	25.9	9.2	4.2
4th quintile	527.22	0	0	0
5th quintile	867.93	0	0	0
Total household income without wildlife resources				
All households (N=336)	595.50	24.7	5.5	1.7
1st quintile	279.95	100.0	22.7	6.9
2nd quintile	405.59	9.6	0.5	0
3rd quintile	502.44	0	0	0
4th quintile	668.29	0	0	0
5th quintile	1125.96	0	0	0
Total household income with wildlife resources				
All households (N=336)	634.69	22.1	4.3	1.2
1st quintile	300.12	100.0	17.9	4.8
2nd quintile	432.59	0	0	0
3rd quintile	542.76	0	0	0
4th quintile	710.13	0	0	0
5th quintile	1194.00	0	0	0

Source: survey results 2013

Analysis by income quintiles when the poverty line is US$360.00 revealed that only the first income quintile had 100% poverty counts with or without environmental income and with or without wildlife. The second quintile reduces from 100% poverty counts with standard household income to 9.6% poverty counts with non-wildlife income. With the inclusion of wildlife income, it further reduces to zero. The third quintile reduces from 25.9% with standard income to zero with or without wildlife income. No poverty is recorded for the fourth and fifth quintiles for all the three scenarios, suggesting that wealth in the area might be tied to environmental income. Using a different poverty line (e.g., the average line of the poorest 15 countries computed by the World Bank), there are still dramatic differences in measured poverty between standard income and total household income with and without wildlife. Table A.5 in appendix A shows that, when the poverty line is changed to US$450.00, the first two quintiles had 100% poverty counts throughout the scenarios. The third quintile reduces from 100% poverty counts in the baseline to 87.2% with non-wildlife, while the inclusion of wildlife reduces the headcounts to zero. The results are robust to different poverty measures and poverty lines.

Table A.6 in the appendix shows the results of policy experiments by considering, for example, a 5%, 10% and 15% increase in wildlife income from the base case on poverty measurements. The results show that increasing wildlife income reduces the headcounts to zero for households in the 2^{nd}, 3^{rd}, 4^{th}, and 5^{th} income quintiles under all policy scenarios while the headcount ratios for households in the 1^{st} income quintile reduced from 100% to 85.6%, 61.4% and 39.5% with a 5%, 10% and 15% increase in wildlife income respectively. This is a significant improvement from the original results, where 100% of the households in the first quintile remained poor with and without wildlife income. Considering the full sample, increasing wildlife income by 5%, 10% and 15% reduces the headcounts from 47.6% to about 8.8%, 2.1% and 0.7% respectively. This result helps to highlight the importance of wildlife resources for redistributive policy targeting.

2.4.5 Environmental income and inequality

This section discusses the sample estimates of measured inequality for Rao's (1969) approach, the Shapley (1953) decomposition and FGT decomposition approaches. It is intuitive to start by comparing inequality in the sample data against measured inequality in other studies. Comparing the Gini indices computed from the survey data with studies done in other countries, the results show similarities in terms of measured inequality to the figures reported by Cavendish (1999) for Zimbabwe and Morocco, but differ significantly from South Africa, Lesotho, Guinea and Zambia (see Table 2.9 below).

Table 2.9: Gini indices from other studies

Country/study	Indices		% reduction in inequality
	Standard income	*Total household income*	
Zimbabwe (Cavendish, 1999)	0.36	0.30	18.6
Morocco	0.33	-	-
Guinea	0.47	-	-
Lesotho	0.56	-	-
South Africa	0.58	-	-
Zambia	0.46	-	-

Table 2.10 shows a significant reduction in measured inequality when environmental income is considered both with wildlife income (16.1%) and without wildlife income (11.3%). Thus, environmental income (with and without wildlife) appears to have a strong and significant equalizing effect on income. Surprisingly, wildlife income on its own also has an equalizing effect, bringing about a 5.4% reduction in measured inequality. As a result, policies that seek to increase access to wildlife income by poor rural communities through ICDP might help reduce income inequality in rural areas. These results are also supported in Figure A1 in appendix A.

Table 2.10: Gini indices for standard income and total income with & without wildlife

Standard income		**Total income (without wildlife)**		**Total household income**	
Estimate	*Std. Error*	*Estimate*	*Std. Error*	*Estimate*	*Std. Error*
0.333176	0.016490	0.295609	0.015028	0.279633	0.013661
Effect of including non-wildlife resources on total household income					
Standard income		**Total income (without wildlife)**		**% reduction**	
0.333176		0.295609		11.28	
Effect of including wildlife resources on total household income					
Standard income		**Total household income**		**% reduction**	
0.333176		0.279633		16.07	
Contribution of wildlife resources					
Total income (without wildlife)		**Total household income**		**% reduction**	
0.295609		0.279633		5.4	

Source: survey results 2013

Table A.8 in the appendix shows the results of policy experiments by considering a 5%, 10% and 15% increase in wildlife income from the base case on measured inequality. The results show a massive reduction in inequality as we move from standard household income to total household income of the magnitude of 21.3%, 25.4% and 32.6% for the respective policy scenarios compared to 11.3% in the original results when wildlife was factored in. The effect of wildlife income alone increased from 5.4% for the original case with wildlife income included to 11.3%, 15.9% and 24.7% when wildlife income increase by 5%, 10% and 15% respectively. Again, this result also supports the importance of wildlife resources for redistributive policy targeting. The results show that measured inequality is not as responsive as poverty to policy induced increments in wildlife income. Thus policies that aim to reduce poverty in rural areas should consider increasing access to wildlife income.

Disaggregating the Gini indices for our three measures of income by income quintiles, we observe a tremendous reduction in inequality when we consider environmental income with and without wildlife, particularly for poorer households (refer to Table 2.11 below). The reduction in measured inequality when we consider total household income (accounting for wildlife resources) is approximately 73.0%, 61.6%, 51.2%, 33.4% and 27.1% for households in the 1st, 2nd, 3rd, 4th and 5th income quintile, against a reduction in inequality of 62.1%, 56.3%, 44.9%, 27.4% and 21.8% respectively with non-wildlife resources included. Wildlife alone accounts for about 28.9%, 12.6%, 11.3%, 8.2% and 6.8% reduction in measured inequality. These findings seem to suggest that environmental income and, in particular, wildlife income has a stronger equalizing effect for relatively poor households than for wealthier households. This might be true because the ratio of environmental income to total household income is very high for relatively poor households compared to richer households, implying that there is heavy dependence on environmental resources by the former households. Moreover, poorer households do not have many alternative sources of income compared to their wealthier counterparts.

Table 2.11: Comparison of the Gini indices by income quintile

Group	**Standard income**	**Total income (without wildlife)**		**Total household income**		**Effect of wildlife**
	Index	*Index*	*% reduction*	*Index*	*% reduction*	*% reduction*
1st quintile	0.156	0.059	62.1	0.042	73.0	28.9
2nd quintile	0.137	0.060	56.3	0.053	61.6	12.6
3rd quintile	0.144	0.079	44.9	0.070	51.2	11.3
4th quintile	0.132	0.096	27.4	0.088	33.4	8.2
5th quintile	0.275	0.176	21.8	0.164	27.1	6.8
Population	0.333	0.296	11.1	0.280	15.9	5.4

Source: survey results 2013

2.4.6 Contribution of individual income sources to total household income

Six income sources were considered: employment, agricultural income, capital, transfers, remittances and environmental income. As noted earlier, three different approaches were employed to check for robustness of our results, i.e., Rao's (1969) method, Shapely's (1953) approach and FGT decomposition. Overall, the results in Table 2.12 below show that agriculture is not only the biggest contributor to total household income, but it is also the most important source of rural inequality. This finding diverges from the study of Leibbrandt et al. (2000) done in rural South Africa, which established that wage income is both the most important income component and also the most important source of inequality. This might be true because employment opportunities are scarcer in rural Zimbabwe than in South Africa, and because most households depend more heavily on agriculture (crop cultivation and livestock rearing) than on any other livelihood activities. Although employment is the second most important source of inequality in this study, the relative contribution of environmental income to total household income surpasses that of wage income.

Table 2.12: Decomposition of the Gini Index by Income Sources - Rao's 1969 Approach

Sources	**With wildlife**				**Without wildlife**			
	Income Share	**Concentration Index**	**Absolute Contribution**	**Relative Contribution**	**Income Share**	**Concentration Index**	**Absolute Contribution**	**Relative Contribution**
employment	0.194	0.286	0.0554	0.145312	0.2581	0.328	0.0632	0.213219
	(0.0169)	(0.0625)	(0.0159)	(0.0249)	(0.0162)	(0.0580)	(0.0155)	(0.0449)
agriculture	0.401	0.385	0.155	0.552284	0.4093	0.415	0.166	0.579487
	(0.0153)	(0.0251)	(0.0131)	(0.0429)	(0.0146)	(0.0235)	(0.0124)	(0.0409)
capital	0.0732	0.232	0.0170	0.062878	0.0728	0.263	0.0192	0.06460
	(0.0034)	(0.0255)	(0.0019)	(0.0079)	(0.0034)	(0.0255)	(0.0019)	(0.0077)
transfers	0.0194	0.00289	0.0000056	0.000208	0.0193	0.0450	0.00087	0.002935
	(0.0020)	(0.0892)	(0.0017)	(0.0064)	(0.0020)	(0.0920)	(0.0017)	(0.0059)
remittances	0.0580	0.0675	0.00392	0.014503	0.0577	0.0152	0.00088	0.002965
	(0.0044)	(0.0404)	(0.0024)	(0.0088)	(0.0046)	(0.0439)	(0.0025)	(0.0086)
environmental	0.204	0.0392	0.254	0.205232	0.1827	0.194	0.0500	0.168589
	(0.0242)	(0.0068)	(0.0111)	(0.0520)	(0.0068)	(0.0184)	(0.0047)	(0.0125)
Total	**1.000**	**---**	**0.270**	**1.000000**	**1.0000**	**---**	**0.296**	**1.000000**
	(0.0000)	**---**	**(0.0191)**	**(0.0000)**	**(0.0000)**	**---**	**(0.0193)**	**(0.0000)**

Source: survey results 2013
NB: Standard errors are shown in brackets

If we compare the two scenarios, with and without wildlife resources, we find that agriculture remains both the most important income component and also the most important source of rural inequality, though wage income quickly catches up as another important source of inequality if wildlife is excluded from the analysis. The without-wildlife scenario severely compromises the relative contribution of environmental income, which is now completely overshadowed by employment or wage income. The results cast doubt on the credibility of capital income, transfers and remittances as important sources of income in the study area. Furthermore, the relative contribution of these income sources is not affected by the inclusion or removal of wildlife resources. The same conclusion is quickly arrived at if we consider the results of the Shapely decomposition technique in Table A10 in the appendices.

The use of the FGT decomposition approach brings a different view to the analysis by making use of the idea of the relative poverty line, which is missing in the Rao decomposition approach. Table 2.13 presents the FTG decomposition results based on the relative poverty line with and without wildlife resources. The first column shows the income share of each income source. The results confirm that employment, agriculture and environmental income are the most important sources of income in the study area. Considering the headcount ratio and accounting for wildlife income, the results show striking similarities between the relative contribution of environmental income and agricultural income to total household income when the poverty line is pegged at US$360.00 per capita. However, the poverty gap and poverty severity measures indicate that the relative contribution of environmental income clearly surpasses that of agricultural income and employment. If we change the poverty line from US$360.00 to US$450.00 while holding other things constant, the relative contribution of agricultural income dominates that of environmental income only for the headcount ratio. The results, however, remain the same under the other two measures of poverty.

With the removal of wildlife, the relative contribution of agricultural income completely overshadows that of environmental income for all three measures of poverty. At the same time, the relative contribution of farm income increases tremendously, while that of environmental income worsens to lower levels. The contribution of wage income is also increased, but only slightly when compared to farm income, implying that, in the absence of wildlife income, employment also becomes an important contributor to total household income. These results seem to suggest that the exclusion of wildlife severely compromises the relative contribution of environmental resources to the livelihoods of poor rural communities living adjacent to the national park in the study area. The fact that the relative contribution of farm income completely dominates that of environmental income as we move from a lower to a higher poverty line suggests that the relative contribution of environmental income to total household income might be more pronounced in poor households, while the relative contribution of agricultural income is noticeable in wealthy households.

Table 2.13: Decomposition of the FGT index by income components

Income Source	**Income share**	**Relative contribution**			
		With wildlife		**Without wildlife**	
		Poverty line (360.00)	*Poverty line (450.00)*	*Poverty line (360.00)*	*Poverty line (450.00)*
Headcount (α=0)					
employment	0.194	0.178931	0.193688	0.192810	0.197343
agricultural income	0.401	0.330893	0.401268	0.354966	0.383689
capital income	0.073	0.072899	0.073194	0.085627	0.068350
transfers	0.019	0.018698	0.019432	0.019648	0.019594
remittances	0.058	0.069015	0.058009	0.075116	0.068026
environmental income	0.254	0.339564	0.254409	0.271834	0.262998
Poverty gap (α=1)					
employment	0.194	0.174060	0.175109	0.181909	0.184369
agricultural income	0.401	0.266978	0.281742	0.278442	0.296615
capital income	0.073	0.132913	0.118989	0.140351	0.127007
transfers	0.019	0.035194	0.031898	0.036525	0.033463
remittances	0.058	0.098059	0.091576	0.101748	0.095726
environmental income	0.254	0.292795	0.300686	0.261024	0.262819
Poverty severity (α=2)					
employment	0.194	0.170979	0.172269	0.176046	0.178615
agricultural income	0.401	0.247143	0.257112	0.253930	0.266195
capital income	0.073	0.150435	0.141412	0.154699	0.146971
transfers	0.019	0.046950	0.042072	0.048110	0.043337
remittances	0.058	0.110622	0.104853	0.11311	0.107878
environmental income	0.254	0.273870	0.282281	0.254104	0.257004

Source: survey results 2013

2.4.7 Econometric modelling of relative poverty and environmental income

To examine the nature of the relationship between poverty (measured in relative terms) and environmental income, we used regression analysis. Firstly, the ordered logit model is used to establish whether environmental income has a differential impact on households in different income quintiles. To derive the dependent variable, sampled households were grouped into five categories of income: low income, lower middle income, middle income, upper middle income, and higher income[20]. We can also think of these income categories in terms of relative poverty, that is, households in the first quintiles are relatively poorer than households in the second, third, fourth and fifth income quintiles, while households in the second quintile are relatively less poor than households in the first quintile but relatively poorer than those in upper quintiles, and so on.

Secondly, the study use both the ordinary least squares (OLS) technique and instrumental variables estimation with heteroskedasticity-based instruments to model the determinants of environmental income generated by these poor households. We suspect that an endogeneity problem exists between environmental income and household wealth. The test for endogeneity revealed that the instrumental variables estimation with heteroskedasticity-based instruments could be better than the OLS results. However, for purposes of comparison, we present the results of both models, but do not interpret the OLS results. The VIF tests for the two models whose results are interpreted and discussed below rule out the possibility of multicollinearity among the explanatory variables (see Table A.11 in the appendices). Table A.13 and Table A.14 in Appendix A show the full results with community level dummies to control for resource heterogeneity and other spatial differences that influence incomes.

The Brant test was used to test for the parallel regression assumption. We expect that the results of the Brant test to be non-significant as a rule of thumb suggest a significant test statistic provides evidence that the parallel regression assumption has been violated. The results in Table A.12 of the Brant test indicate that we have not violated the proportional odds assumption under the null hypothesis that there is no difference in the coefficients of the models. If we had, we would want to run our model as a generalized ordered logistic or probit model. We calculated the marginal effects for each category and the results are reported in Appendix A. Standard tests for under-identification, weak identification, over-identification and heteroskedasticity show that it is safe to proceed with the instrumental variables estimation with heteroskedasticity-based

[20] In calculating the dependent variable, we excluded environmental income from total household income so that it does not appear on the RHS. We then used partial household income (i.e., total household income without environmental income) to group these households into relative poverty categories. We think that environmental income is more related to full income and not necessary to the standard income. Endogeneity is quite clear when we have environmental income versus full income. However, we can't completely rule out the possibility of endogeneity between environmental income and income categories and to deal with this problem we need good external instruments or away of generating internal instruments. Unfortunately, we do not have good external instruments. Given that we are running a logit model, it is not possible to generate internal instruments. Accordingly, we acknowledge that there might be some bias in these results and hence we exercise caution in their interpretation.

instruments in both models. Because we are considering 30 communities (which imply about 30 observation), there is a worry that we might be dealing with too few observations. One way to deal with that is bootstrapping. We applied the bootstrapping procedure to check for robustness of the standard errors and found that the results are consistent (see Table A.16 and Table A.17).

Table 2.15 below presents the results of the ordered logistic model of the relationship between poverty (measured in relative terms) and environmental income, plus other household characteristics. The model is highly significant (at the 1% level) and tells us that the amount of environmental income generated by households, educational level of the household head, whether or not the household head is employed, household size, religion of the household head and tenure have an effect on households in the different income quintiles or relative poverty. There is no evidence of the effects on the dependent variable of the age of the household head and whether or not he or she was born in the area on the dependent variable.

As anticipated, the coefficient of environmental income is positive and highly significant. The results suggest that the likelihood of households moving from lower income quintiles to higher quintiles increases with an increase in environmental income generated by households. In other words, the chance of belonging to a wealthier category of income increases with an increase in environmental income. Considering Table A.15 in the appendices, the marginal effect of environmental income is very small, suggesting that only households that are positioned on the boundary might be able to move to the next income quintile because of the increase in environmental income, while its impact may be less pronounced for households that are located farther away from the boundary. Although environmental income might not be able to push households farther away from the boundary into the next income class, we argue that such households are better off in that they are less poor with environmental income than without it.

Table 2.15: Ordered logit regression model

Ordered logit estimates	Number of obs	= 336
	LR chi2(8)	= 99.41
	Prob>chi2	= 0.000
Log likelihood = -491.2	Pseudo R2	= 0.0922

Quintile	**Coef.**	**Std. Err.**
environmental income	0.00217	0.0026***
age of household head	0.00729	0.0328
education of household head	0.582	0.2652***
household head employed [0, 1]	0.466	0.1474*
household size	-0.324	0.0253***
household head born in area [0, 1]	-0.143	0.0636
religion of household head [0, 1]	0.0580	0.2203**
gender [0,1]	0.0042	0.0436
Tenure [0, 1]	-0.695	0.4577**
cut1	-2.236	0.8283
cut2	-0.878	0.7505
cut3	0.0567	0.8564
cut4	1.485	0.8402

Source: survey results 2013

The coefficients of education, employment status and religion are also positive and significant, while household size and tenure have negative and significant coefficients. These results suggest that the likelihood of belonging to a wealthier category of income increases with educational attainment and employment. This might be true if wage income is substantial enough to positively impact on total household income. An educated household head finds it easier to secure employment than their uneducated counterparts. Being a Christian increases the likelihood of belonging to a wealthier class. The likelihood of belonging to a wealthier category of income diminishes with household size. This might be true because a larger household implies many mouths to feed. This is a problem especially when the majority of the household members are unemployed. The likelihood of belonging to a wealthier category also diminishes as we move from resettlement schemes to communal areas. Evidence gathered through qualitative interviews established that households in the former tenure regime are relatively richer than are households in the latter.

The results of the determinants of environmental income generated by households living adjacent to the Gonarezhou National Park are presented in Table 2.16. The instrumental variables estimation model explains about 77.4% of the variation in our dependent variable. The endogeneity test suggests that instrumental variables estimation yields better results than OLS.

Both the Kleibergen-Paap LM test for underidentification and the Hansen J statistic for overidentification show that it is safe to proceed with instrumental variables estimation. Moreover, the number of explanatory variables that are significant increases, and the value of R-squared and the significance level also improve. All variables were significant except for whether or not the household head lives on the farm and whether or not the household head was born in the area.

As expected, household wealth significantly and positively affects environmental income generated by households. The relationship between environmental income and wealth has some interesting policy implications given that wealthier households accumulate more assets that can be used to harvest more environmental resources. This implies that wildlife-based land reform also needs to empower poor households in the area of capital accumulation while imposing restraints on well-off households' use of capital investments to harvest resources. We also expect the age of the household head and household size to have a positive relationship with environmental income. Thus, as the household head grows older, household size increases and the amount of environmental income generated by the household also increases because the number of people required to harvest environmental resources has increased. Interestingly enough, the results show a negative relationship between the educational attainment of the household head and environmental income. As the number of years in school increases, dependence on environmental income diminishes because the head of the household has more opportunities at his or her disposal due to this educational attainment. Employment reduces environmental income generated by the household *ceteris paribus*.

Table 2.16: Determinants of environmental income generation

	Ordinary Least Squares		IV Estimation	
Environmental income	*Coef.*	*Std. Err.*	*Coef.*	*Std. Err.*
wealth index	8.649***	3.472	7.26***	3.813
age of household head	3.265**	7.516	3.678***	7.354
education of the household head	-62.82**	87.84	-73.70**	77.58
household head employed [0, 1]	-287.4**	165.0	-256.2***	179.0
household head live on the farm [0, 1]	7.053	52.56	4.435	44.02
household head born in this area [0, 1]	89.0	43.02	20.18	51.76
household head is a Christian [0, 1]	-117.3*	60.35	-125.2**	72.34
household size	54.34**	20.57	47.63***	33.21
distance to the market	4.381**	2.139	5.565**	3.411
number of dogs	31.36	49.18	53.07*	48.42
biodiversity	125.6*	77.57	87.3**	98.73
tenure [0, 1]	-531.7***	225.4	-535.3***	242.6
constant	346.1	541.6	324.0	653.5
Observations	336		336	
R-squared	0.655		0.774	
Underidentification test (Kleibergen-Paap rk LM statistic):			Chi-square	27.827
			P-value	0.0065
Weak identification test (Cragg-Donald Wald F statistic):			F-statistic	9.1053
Hansen J statistic (overidentification test of all instruments):			Chi-square	9.2462
			P-value	0.6170

*** p<0.01, ** p<0.05, * p<0.1

Source: survey results 2013

Again, being a Christian reduces dependence on environmental resources. Blakemore (1975) established that Christian households in Ghana have better education and hence better employment opportunities than non-Christian households. The author further posits that allegiance to Christianity significantly reflects a shift toward the acceptance of formal education, suggesting that Christian households represent 'family environments' generally oriented towards success norms and educational achievement. As discussed above, better education and more employment opportunities will in turn lead to less dependence on the natural capital base. In addition, by virtue of belonging to an organised group, there is more social cohesion among group members. Also, when children leave the villages to find greener pastures elsewhere, they keep ties with their relatives and send back money in times of need.

The results show that, as the distance to the market increases, environmental income generated by households also increases. This might be the case because rural households face fewer opportunities as we move farther away from the urban areas. If there are fewer opportunities for households, especially employment opportunities, then the natural capital base becomes the most important source of livelihoods. Environmental income also increases as the number of dogs owned by household increases. Dogs are very important and form an integral part of the livelihoods of poor rural households in the study area because of their role in the provision of security at home (sometimes including protecting field crops) and hunting activities.

The results reveal some evidence of the relationship between benefits and the quality of the resource system. As anticipated, environmental income increases as biodiversity increases[21]. This means that households generate more environmental income in areas with good biodiversity than in areas where there is an unhealthy population of wild animals (too few). Finally, the results also show that households in communal areas collect fewer environmental resources than do households in resettlement schemes. This might be true because households in resettlement schemes are relatively wealthier compared to those in communal areas. As a result, they possess better technology and more assets (e.g., carts, draught power and guns) that are useful in harvesting environmental resources. The policy implication of this result is that wildlife-based land reform also needs to empower poor households in communal areas while imposing restraint on harvesting well-off households in resettlement schemes.

2.5. Conclusion and policy implications

Awareness of the importance and value of environmental resources in the livelihoods of poor rural households in developing countries has increased tremendously. As a result, there has been a growing body of literature attempting to quantify the value of environmental resources and their impact on poverty and inequality in the rural economies of Southern Africa. In particular, wildlife has become popular with policymakers and development practitioners as a vehicle for rural development. However, unequal utilisation of wildlife has resulted in different contributions to livelihoods. Therefore, the main objective of this study was to investigate the effects of environmental resources and, in particular, wildlife on household welfare. Specifically, we asked the following questions: i. Does environmental income (including wildlife) contribute significantly toward total household income and reduction in rural poverty and income inequality? ii. How does environmental income (specifically, wildlife income) compare with

[21] The Shannon index was used as a measure of the health of biodiversity in the study area. The index provides information about rarity and commonness of wildlife species in the area.

other sources of income? iii. What determines the different amounts of environmental income that households generate?

To address these questions, the chapter made use of income quintile analysis, the Foster-Greer-Thorbecke poverty measure, Gini coefficient analysis, Gini decomposition analysis and instrumental variables estimation, using heteroskedasticity-based instruments on purposefully-collected household data from local communities living adjacent to the Gonarezhou National Park in Zimbabwe, whose livelihoods depend on wildlife conservation. By so doing, the study expands the existing knowledge concerning the nexus between environmental income and poverty and inequality. From a policy standpoint, it has also become imperative for policymakers and development practitioners to understand the relative importance of wildlife income in driving rural poverty and inequality as they formulate strategies for operationalising wildlife-based land reform.

The households sampled in this study harvested and used an enormous range of environmental resources, which in turn provided a wide range of economic benefits. Considering major food items and non-food items consumed by local communities around the Gonarezhou National Park, the results show that richer households consume more environmental resources (both food and non-food items) in total than poorer households. Richer households consume more of valuable resources such bush meat, fish, timber, firewood and livestock fodder, while poorer households consume more of the less-valuable resources, such as wild vegetables, wild fruits, insects and thatch grass. As one of the main motivations of this study, the results suggest that relatively wealthier households consume more wildlife products in total than relatively poor households. However, poorer households derive proportionally greater benefit from the consumption of wildlife resources than wealthier households. Furthermore, wealthier households purchase more environmental resources than poorer households, while poorer households sold more environmental resources.

Wage income, farm income and environmental income are the three most important sources of household income for the communities in question. Although agricultural income (35.4%) dominates all the sources of income, the contribution of environmental income (28.7%) is also quite substantial. The increase in total household income resulting from the inclusion of non-wildlife and wildlife resources is much higher for households in lower income quintiles than it is for households in higher quintiles.

Overall, the results illustrate that the proportion of people in the full sample living below the poverty line is greatly reduced when we account for non-wildlife environmental resource income initially and wildlife income later. The inclusion of non-wildlife resources accounts for approximately a 48.1% reduction in poverty, while the inclusion of wildlife resources accounts for a 53.6% reduction in poverty. The net effect of wildlife income alone is about a 5.5% reduction in the proportion of people living below the poverty line. The separate consideration of wildlife and non-wildlife income is a key contribution of this chapter.

The results show a significant reduction in measured inequality when environmental income is considered both with wildlife income (16.1%) and without wildlife (11.3%). Thus, environmental income (with and without wildlife) appears to have a strong and significant equalizing effect on income. In particular, wildlife income has an equalizing effect, bringing about a 5.4% reduction in measured inequality. Disaggregating the Gini indices for our three measures of income by income quintiles, we observe a tremendous reduction in inequality across income quintiles when we consider environmental income with and without wildlife. The reduction in inequality is greater for poorer households than for relatively wealthier households. Wildlife alone accounts for about 28.9%, 12.6%, 11.3%, 8.2% and 6.8% of the reduction in measured inequality. These findings seem to suggest that environmental income and, in particular, wildlife income has a stronger equalizing effect for relatively poor households than for wealthier households. As a result, policies that seek to increase access to wildlife income by poor rural communities through IDCP might reduce poverty and income inequality in rural areas.

Agriculture is both the most important income component and also the most important source of rural inequality in the area studied. Considering the headcount ratio and accounting for wildlife income, the results show striking similarities between the relative contribution of environmental income and agricultural income to total household income when the poverty line is pegged at US$360.00 per capita. The without-wildlife scenario severely compromises the relative contribution of environmental income, which becomes completely overshadowed by both farm and wage income. At the same time, the relative contribution of farm income increases tremendously in the without-wildlife scenario, while that of environmental income plunges to lower levels.

In line with our goal of using wildlife resources to bring about community welfare, we believe that policy experiments of increasing wildlife income could highlight the importance of wildlife resource for redistributive policy targeting. So accordingly, we considered 5%, 10% and 15% policy induced increments in wildlife income from the baseline case. The result show that increases in wildlife income can greatly impact poverty, but not inequality because inequality is less responsive to policy induces increments in wildlife income. The results also show that we need an increase in wildlife income slightly greater 15% in order to complete reduce poverty in the whole sample to zero. However, it should be noted that the GINI decomposition method might not be suitable for changes as large as 10% to 15% as it is designed for marginal changes.

The results of the ordered logit model suggest that the likelihood of belonging to a wealthier category of income increases with an increase in environmental income. The marginal effect of environmental income is very small, suggesting that only households that are positioned on the boundary will be able to move to the next income quintile because of an increase in environmental income, while the impact of environmental income in general and wildlife in particular may be less pronounced for households that are located farther away from the boundary. As expected, household wealth significantly and positively affects environmental income generated by households. Finally, the results reveal some evidence of the relationship between benefits and the quality of the resource system. Households generated more environmental income in areas with good biodiversity than in areas where there is an unhealthy population of wild animals.

In this chapter, we conducted different kinds of analysis to examine the effects of wildlife resources on community welfare in the portfolio of environmental income. All the results in this analysis speak to each other since they address key policy issues pertaining to the effects of environmental income on household welfare and its role in alleviating rural poverty and income inequality through its contribution to livelihoods or total household income. The intended beneficiaries of the study are local communities, wildlife agencies, development practitioners and policy makers who will acquire relevant and empirically grounded evidence. Our research findings will allow them to interrogate their wildlife management strategies and policies while at the same time identifying areas that need to be improved.

The main policy message of the findings of this study to take home is that, wildlife conservation has an important role in mitigating poverty and income inequality in CAMPFIRE communities. Several other policy implications can be drawn from the analysis in this chapter. Reduced access to wildlife income could have substantial impact on welfare in CAMPFIRE communities and might potentially increase poverty and inequality in the study area. There is therefore a need to design policies that increase access to wildlife income by poor rural households living adjacent to national parks because this could have an impact on their welfare. To avoid further marginalization of the poor, attention to equity in resource management and access to resources should be a prime consideration, particularly with valuable resources such as wildlife. Increased devolution of wildlife management function from the Rural District Council into the hands of local communities could allow more access to wildlife income and potentially contribute towards reducing poverty and inequality. The relationship between environmental income and wealth has some interesting policy implications given that wealthier households accumulate more assets with which to harvest more environmental resources. This implies that wildlife-based land reform also needs to empower poor households in the area of capital accumulation while imposing restraint on harvesting by well-off households.

In line with our goal of using wildlife resources to bring about community welfare, we believe that policy experiments of increasing wildlife income could highlight the importance of wildlife resources for redistributive policy targeting. The results illustrate that increases in wildlife income can greatly impact poverty, but not inequality because inequality is less responsive to policy induces increments in wildlife income. Thus policies which seek to reduce poverty in rural communities should consider increasing access to wildlife income by these communities. The results demonstrate that a policy induced increment in wildlife income of slightly more than 15% is needed to reduce poverty in the whole population to zero.

There is need for a policy shift away from the resource itself to encompass the broader local economy in order to enhance livelihoods. Quick policy interventions such as formal and informal employment in the craft industry are required to increase household earnings from wildlife conservation in the short to medium term. In the long-run, household welfare in CAMPFIRE communities could benefit through policies and programs that stimulate an increase in wildlife earnings from activities such as non-consumptive tourism, tourism business ventures such as hotels or accommodation, investment in infrastructure, filming and live animal sales.

Chapter 3

The Role Institutions in Community Wildlife Conservation in Zimbabwe: A Social-Ecological Approach[22]

3.1 Introduction

Wildlife conservation has become popular with policymakers and development practitioners alike as a vehicle for rural development because of abundant tourism opportunities in Southern Africa. A significant proportion of wildlife is managed as a common pool resource (CPR) under various forms of community-based natural resource management (CBNRM) arrangements involving both local communities and private game farms.[23] Wildlife shares some characteristics with other CPRs, such as water, forests, rangelands and fisheries, thus making its management and utilization under joint-use arrangements a daunting task. For instance, conservation efforts are affected by global market trends, past and prevailing governance and institutional arrangements (Gibson & Marks, 1995). The situation is made even worse by the fugitive character of the resource, which makes it difficult to assign property rights to wildlife (Muir-Leresche & Nelson, 2000).

The challenges associated with the management and utilization of CPRs in most developing countries has resulted in a search for policy options in an effort to make social-ecological systems (SESs) sustainable[24] over time. Following the publication of Hardin (1968), both state control and private ownership were embraced by colonial governments in the region as panaceas for all environmental problems. While both policy instruments have benefited minority groups and persisted even after independence, the colonial legacy created tension between wildlife authorities and local communities living adjacent to national parks (Songorwa, 1999). At the rate

[22] A version of this chapter has been desseminated as an EfD Discussion Paper No. DP 15 – 23.

[23] Private game farms are normally organised into conservancies, which have dissolved their internal boundaries in order to manage wildlife as a common pool resource. Private game farms are referred to as landowner communities because each one usually comprises multiple private landholdings (Kreuter et al., 2010).

[24] Sustainability is defined as the ability of an ecosystem to provide goods and services today without compromising its ability to provide the same goods and services in the future (Brundtland et al., 1987).

at which wildlife is being decimated by local communities, the region could lose some of its prime wildlife species sooner than 2050 (Thuiller et al., 2006). This realization has led policymakers to shift attention to people-oriented approaches rather than the conventional top-down governance systems.

This chapter argues that the ability of a community to manage resources sustainably depends on the capacity of local communities to self-organise and that self-organisation depends on the nature of the institutions[25] that are in place. Existing institutions have failed to protect wildlife and biodiversity, and hence there is a need for more ideas to feed into future policy or institutional reforms. The study was conducted in local communities around the Gonarezhou National Park in Zimbabwe that are participating in community wildlife conservation. The overall objective of this study is, therefore, to enhance our understanding of the role of local institutions in promoting sustainable management of CPRs and biodiversity conservation using Zimbabwe's CAMPFIRE nature conservation programme[26] as a case study. Following the discussion above, three important questions arise. Are there any practical differences in terms of resource units, resource users, and quality of institutions in common pool wildlife systems within and across communities? Under what conditions will the users of common pool wildlife self-organise? What attributes of resource units, resource users and local institutions are consistent with sound biodiversity outcomes?

This chapter contributes to the literature on common pool resources by applying Ostrom's framework and collecting primary data on a little-studied topic. The framework has been used extensively in areas such as forestry, fisheries, rangelands and water resources management, while little has been done in the wildlife sector, particularly in Southern Africa. From a policy perspective, this chapter sheds light on the processes governing human-environment systems and provides results comparable to other studies or ongoing projects. Furthermore, our research findings allow both policymakers and development practitioners to question their wildlife

[25] North (1991) defined institutions broadly as humanly devised constraints that structure political, economic and social interactions. In this thesis, we define institutions as systems of established and prevalent social rules that structure social interactions (Hodgson, 2006), including community-level organisations.

[26] The Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) is a benefit-sharing scheme that involves local communities that live in the vicinity of national game parks and suffer wildlife intrusions. The programme was instituted by the government of Zimbabwe during the mid-1980s in order to create incentives to conserve wildlife by directly transferring benefits from conservation to the local communities.

management strategies and policies while at the same time identifying areas that need to be improved.

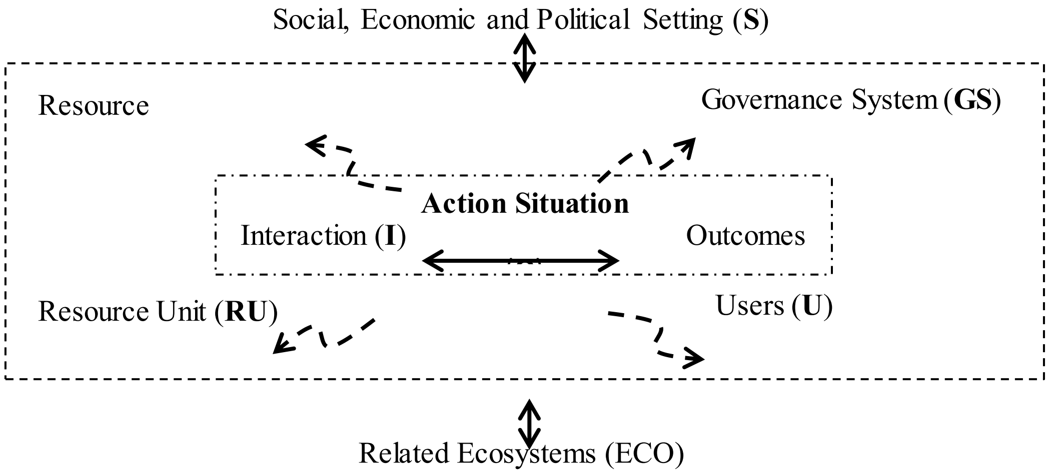
The rest of the chapter is organized as follows. Section 3.2 presents a review of the theory of collective action and the conceptual framework. Section 3.3 gives an outline of the research methods, i.e., the analytical framework, empirical model specifications and data issues. We then proceed to present and discuss the results in Section 3.4 and end with conclusions and policy implications in Section 3.5.

3.2 Review of Theory

3.2.1 Conceptual Framework

This study employs the general framework for analysing complex social-ecological systems (SESs) as developed by Ostrom (2007). Figure 3.1 shows eight core subsystems that have been observed to affect both the ability of a community to self-organise and the sustainability of a resource system. In empirical work, many variables have been observed to affect the patterns of interaction and outcomes. In an earlier study, Agrawal (2001) identified more than 30 variables that had been posited in major theoretical work to affect incentives, actors, and outcomes related to sustainable governance of a resource system. By unpacking the eight core subsystems and expanding on Agrawal's work, Ostrom et al. (2007) went further to provide an even longer list of variables under each subsystem (see Table B.1 in appendix B). This framework helped to identify the variables that are relevant in studying community wildlife conservation, designing the research instrument and analysing findings about the sustainability of a SES.

Figure 3.1: The core subsystems in a framework for analysing social-ecological systems



Source: Extracted from Ostrom (2007: 15182)

By applying Ostrom's framework, it is possible to analyse different environmental problems under different scenarios and predict the associated outcomes. Table 3.1 below shows how we can adapt this framework in the context of the wildlife sector in Southern Africa. For example, this study deals with a fugitive and finite renewable resource which is either harvested legally through trophy hunting activities by engaging a safari operator (in which case the community gets wildlife income, thereby maximizing societal welfare), or illegally by poachers, who maximise their individual short-term gains. Local communities are not allowed to hunt wildlife for their own purposes both inside and outside the national game park (i.e. in their own conservation area).

The community does not benefit from tourism, research and wildlife photography or filming activities. This means that trophy hunting is their only means of generating revenue from conservation. The community apply for a hunting quota from the Zimbabwe National Parks and Wildlife Authority, and generate income by selling the quota to safari operators who will, in turn, source clients at organised international events and sell the quota at a premium. The quota is eventually utilised by the client. The most important institutions involved in community wildlife conservation are the wildlife management committee, park authorities, Rural District Councils (RDCs, the main administrative organ and entry point into the community) and non-governmental organisations (NGOs).

Table 3.1: Second tier variables used in this paper

Resource System	Governance System
RS1: Sector – wildlife sector RS2: Resource size – finite RS3: Renewable Resource	GS1: Wildlife Management Committee *- Expected to continue with conservation outside park* GS2: Rural District Council *- enact bylaws and sometimes monitoring & enforcement* *- has the appropriation rights* *- collect and distribute revenues* GS3: National Parks *- custodian of wildlife* *- set hunting quotas* *- monitoring & enforcement inside protected area*
Resource Units	**Users**
RU1: Fugitive Resource *- Wildlife destroy crops and livestock* RU2: Legal harvesting by the PH *- generate income to the community* RU3: Illegal harvesting by poachers	U1: Large number of users U2: Conflict of Interest *- Maximize community welfare (altruistic motive)* *- Maximize short-term gain (self-interest)* *- Nuisance motive for harvesting wildlife*
Interaction	**Outcome**
I1: Maximum harvesting levels by poachers I2: PH guided by quota	O1: Resource overexploitation O2: Destruction of the ecological system

Source: adapted from Ostrom (2007)

The interaction between the community and the resource system produces undesirable outcomes. For instance, harvesting levels by poachers exceed the maximum sustainable yield because they maximise personal gain. Therefore, if the communities in question do not have the means of controlling extraction, then the end result is resource overexploitation and, eventually, total collapse of the social-ecological system. The prediction of resource collapse usually comes true in a very large and highly valuable resource system under open-access conditions when users are diverse, do not communicate and have failed to develop institutions for managing resources (Berkes et al., 2006).

3.2.2 The Collective Action Problem

The theory of collective action[27] has matured tremendously since the publication of Olson (1965), entitled 'The Logic of Collective Action.' It relates to the group or an individual's lack of capacity (except under certain conditions) to solve what is referred to as the 'collective action problem.' Resource economists and theorists ask questions about the

[27] Collective action is defined as any action taken together by a group of people whose goal is to enhance or achieve a common objective (Olson, 1965).

conditions under which those who face the 'tragedy of the commons' are able to organise a system of rules by which the tragedy is averted (Wade, 1987). The analytical framework in contemporary analysis is that of a rational, self-interested individual who maximises short-term gain while the society maximises community welfare (Ostrom, 2003). Conflict of interest exists between the community's objective function and that of an individual. Of particular interest to this study is the literature that focuses on creating incentives for collective action through designing sound CPR institutions. What causes communities to develop institutions is the scarcity of natural resources and the need to avoid tragedy in the commons.

There is a great deal of literature focusing on the role of punishment and social sanctioning or ostracism in promoting cooperation among resource users. Much of this literature comes from the field of experimental economics (Fehr & Gachter, 2000; Cardenas et al*.*, 2000; Murphy & Cardenas, 2004; Akpalu & Martinsson, 2011). Agrawal (2001) emphasises the differences between a self-organised community and externally imposed collective action in terms of rule enforcement and sanctioning. Studies reveal that communities benefit when institutions are endogenised by the community, compared to the case when rules and regulations are externally enforced by the government (Murphy & Cardenas, 2004; Akpalu & Martinsson, 2011). Community-based wildlife conservation in Zimbabwe, for example, relies to a great extent on state authorities to monitor and enforce rules and regulations, but the capacity to do so is limited by the budget. Knowledge of local collective action and informal institutions in natural resource management provide crucial information for designing policy instruments or interventions aimed at simultaneously addressing both conservation and poverty issues.

It is certainly inadequate to study collective action without recognizing the important concept of participatory development and good governance and the contributions made by Gandhi (1934), Chambers (1983) and Cernea, (1985). Chambers (1983) define participatory development as a process through which stakeholders can influence, share control over development initiatives, and make important decisions over resources that affect their livelihoods. The goal of Integrated Conservation and Development Projects is to engage local populations in development projects in such a way that development and conservation objectives are met, while at the same time setting in motion a process of self-reliant and sustainable development[28]. Participatory development has taken a variety of forms, since its emergence in the 1970s, when it was introduced as an important part of the 'basic needs approach' to development and later formalized into various forms of Participatory Rural Appraisal (PRA) approaches[29]. Since then, broader participation and engagement of key stakeholders, public transparency, and institutional accountability have gained greater importance in participatory development.

Local communities have, for a long time, been marginalized when it comes to the management and utilization of valuable natural resources in their jurisdiction such as wildlife. Blind policies that neglect the needs of local communities have exacerbated environmental crime or poaching and conflict in the past between local communities and the park authority or wildlife. Chambers (1983), argued that 'putting the last first' was the only way to halt the decimation of natural resources and to achieve rural development. The participatory development movement was important in applying people oriented approaches to small-scale development in ways that would allow the poor to be informed participants in development, with external agents acting mainly as facilitators and sources of funds. It is therefore imperative for policymakers to design policies and programmes that engage local communities in resource management, utilization and decision making at both local and higher levels, since they are the ones threatening the existence of wildlife in the Southern African region.

3.3 Research Methods

3.3.1 Analytical Framework

The chapter utilises both household survey data and key informant interviews. We provide a detailed characterization (mapping) of local CPR institutions of 25 CAMPFIRE projects and

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28 Self-reliant development means building the endogenous mechanisms of society that will enable local communities ultimately to manage their natural resources to their own benefit. Sustainable development means continuing a stable growth pattern in such a way that development is in harmony with the environment.

29 The term Participatory Rural Appraisal describes a growing body of approaches and methods to enable local communites to share, enhance and analyze information and knowledge of life circumstances in order to plan and act in an informed manner (Chambers, 1992).

5 communities from three resettlement schemes[30]. From a qualitative perspective, the analysis will articulate how institutions in different communities differ and why they differ given similar ecological conditions. This is followed by empirical model estimations to investigate the link between institutions and cooperation, and the relationship between cooperation and success of biodiversity outcomes.

3.3.2 Empirical Methods

Model 1 - Participation in Community-Based Wildlife Management Programs*:* A variable measuring ability to self-organise or a signal of cooperation is used as a dependent variable in our first model. This could be participation of individuals in community wildlife projects or activities at a community level. Following McCarthy et al. (2002) and Pennings and Leuthold (2000), ability to self-organise (cooperation) is assumed to be a latent variable. Factor analysis of variables thought to be associated with cooperative capacity is employed in order to recover the latent variable. Indicators of cooperation are drawn from two main categories, namely, networks and organisational performance variables. Network indicators include the density of organisations and density of household participation, while organisational performance indicators include number of rules, regulation, activities and meetings. By definition, density is a measure of the size of a network (e.g., how many organizations are found in a particular community) while participation is a measure of the quality or strength of a network (e.g., membership of these organizations, how frequent people attend meetings or interact with each other, and whether they share information important for the organization). Mathematically, we write:

$$C_i = \beta_0 + \beta_1 Institutions + \beta_2 Groupsize + \beta_3 Trust + \beta_4 Ethinicity + \beta_5 Wealth + \beta_7 \Pr ojectyear + \beta_8 Punishment + \beta_9 \operatorname{Re} sourcesize + \beta_{10} Stakeholders ...... + \varepsilon_i \qquad (3.1)$$

where C_i is a variable measuring the level of cooperation in the community and the other variables in the model stand for community institutions, group size, the level of trust in the community, number of ethnic groups, an asset index used as a proxy for wealth, the year the project was established, whether punishment is endogenized by the community, the size of

[30] Peasant farmers around the Gonarezhou National Park are found under two different types of land tenure systems, namely, communal areas and resettlement schemes. The resettlement scheme is a product of the government of Zimbabwe's land reform program created after independence in 1980 when the country embarked on a land redistribution exercise in which some of the land that belonged to large-scale commercial farmers was transferred to poor households from the overcrowded communal areas (Mushunje et al., 2003).

the resources system and number of stakeholders working with the community to conserve wildlife resources respectively.

To measure the quality of institutions in a community, we made use of the extended version of Ostrom's (2010) design principles for stable local common pool resource institutions (refer to Table 3.2 below). A complete enumeration of CPR institutions in each community provided vital information for calculating an index of the quality of institutions. We then used this index plus other explanatory variables to explain cooperation as described in Model 1 above.

Table 3.2: Design principles for CPR institutions

Variable	Description
P1:	Clearly defined boundaries (effective exclusion of external non-entitled parties)
P2	Rules regarding the appropriation and provision of common resources that are adapted to local conditions
P3	Collective choice arrangements that allow most resource appropriators to participate in the decision making process
P4	Effective monitoring by monitors who are part of or accountable to the appropriators
P5	A scale of graduated sanctions for resource appropriators who violate community rules
P6	Mechanisms of conflict resolution that are cheap and easy to access
P7	Self-determination of the community recognised by higher-level authorities
P8	In the case of larger common-pool resources, organisations in the form of multiple layers of nested enterprises, with small local CPRs at the base level

Source: Ostrom (2010) - Analysing Collective Action

Model 2 - Success of Biodiversity Outcomes*:* To measure the success of biodiversity outcomes across communities, a measure of relative abundance or diversity was used. The relative abundance of a species in a community is defined as the proportion of individual organisms in the community that belongs to that species. Let P_j for $j = 1,2,.....,N$ be the relative abundance of species j and N the number of species. We can define the Shannon index as:

$$S_j = \sum_{j=1}^{N} P_j \left|\log P_j\right| \quad\text{..........}3.2$$

The Shannon index (S) provides important information about rarity and commonness of species in a community. Mouillot and Lepretre (1999) and Nagendra (2002) suggest that a good measure of species diversity should be able to capture two important dimensions of biodiversity, namely, species richness and species evenness. Thus, the Shannon index is a quantitative measure that reflects how many different groups, types or species there are in a data set. The value of S rarely exceeds 4 in most ecological studies and increases when the number of types (species richness) and evenness increases. Although the index does not tell us anything about the endangeredness of a species, it is sufficient for the purposes of this analysis since it incorporates both components of biodiversity. Moreover, endangered species such as rhinos are not found in the communities in question.

To calculate the Shannon index, we used information about commercial count of animal species[31] done by the RDC and the respective communities. Each year, communities keep information by type and species about the number of wild animals traversing their conservation area[32]. The records are kept at the RDC offices and this information is then used by the community as justification when applying for a quota. Wild animal counting is done at the community level by a team of people that includes members from local communities, the RDC and sometimes park authorities. So, algebraically, we have the following equation linking biodiversity and cooperation:

$$S_j = \beta_0 + \beta_1 Cooperation + \beta_2 Benefits + \beta_3 Market + \beta_4 Distfence + \beta_5 Socialcapital + \beta_6 Fence$$
$$\beta_7 Age + \beta_8 School + \beta_9 \mathrm{Re}\, sidence + \beta_{10} Yearsliving + \beta_{11} Training + \beta_{11} Info .... + \varepsilon_i .... 3.3$$

where S_i is the Shannon index representing a measure of biodiversity in the community, while the other variables represent the level of cooperation in the community, benefits from wildlife conservation, distance to the market, distance from the fence of the national park, an index for social capital, whether the community has been affected by the electric fence separating the community from the national park, average age of the household head in the community, mean number of years in school for the household head, whether the head of the household is resident on the farm or not and number of years living in the area respectively.

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[31] In calculating the Shannon index, we included the major commercial aninal species found in the area (both prey and predators) and these are: elephants, buffalos, giraffe, zebra, wilddogs, eland, kudu, antelope, impala, nyala, waterbuck, bushbuck, springboks, warthogs, lion, leopard, hyena, cheater, hippos, crocodile, duiker, and baboons.

[32] The community's conservation area (CA) is an area set aside by the community for the purpose of conservation. This is a requirement by the state for the community to participate in the CAMPFIRE programme. In addition, local communities are tasked to continue with conservation work in order to keep their land bordering the National Park. Failing to comply with this requirement means that the community could risk losing part of their land.

The models presented in this chapter are based on the idea or theory which suggests that institutions act as a constraint on human behavior and this will in turn have an influence on biodiversity outcomes (Ostrom, 2007a; Ostrom, 2007b). This theory puts emphasis on the development of sound local institutions to govern people's behavior so that people can manage their resources sustainably. In this chapter, human behaviour is measured in terms of cooperation or rule compliance. We can think of institutions as a treatment and cooperation as an outcome variable in the first model, while in the second model, cooperation becomes a treatment and biodiversity conservation the outcome variable so that the relationships between our key variables is unidirectional and the variables maintain a certain order (i.e., $Institutions \rightarrow cooperation \rightarrow biodiversity\ outcome$). This order is very important in this chapter because it supports the two models that we have developed based on the theory stated above, and for this reason, we will maintain this kind or reasoning throughout the chapter. However, we will relax the assumption that the relationship between the variables is unidirectional to make the model more realistic.

Based on this theory, we can deduce a structural model, which explains wildlife or biodiversity conservation and that structural model consist of two equations where in the first round institutions determine cooperation and in the second round cooperation determine biodiversity. Thus it is appropriate to assume that institutions indirectly affect biodiversity based on this structural model. So from our view, it is misleading to assume that the institutional variable enters the biodiversity equation as it ignores the nature of things in a natural system such as in wildlife conservation.

Table 3.3 below describe the explanatory variables that are used in the two models. The last column shows the expected signs of the variables. Factors that have been studied in the collective action literature include group size, group heterogeneity and inequality, ethnicity, trust, resource size and punishment (Wade, 1978; Runge; 1986; Ostrom, 1990; Baland & Platteau, 1996; Agrawal, 2001). These variables were analysed in theoretical, empirical and experimental studies in third world counties. From an empirical and theoretical perspective, we expect our measure of institutions, trust and number of stakeholders in the community, in the first model, and our measure of cooperation, benefits from wildlife conservation, distance to the market, distance from the fence of the park, educational level of the head of the household, number of years living in the area, whether community members received training, whether community members share information, in the second model, to carry a positive sign. We expected group size, ethnicity and punishment, in the first model, to carry a negative sign (e.g., Olson, 1965; Ostrom, 1990; Baland & Platteau, 1996; Ostrom, 2007a). However, the expected signs of some variables could not be determined theoretically or from the literature review. These variables included in the model were identified in a framework to analyse social-ecological systems developed by Ostrom (2007a; 2007b) as important for collective action and sustainable management of common pool resources.

Table 3.3: Nature of variables and expected signs

Variable	Nature of variable	Expected sign
Variables for the first model		
Cooperation index	Dependent variable measuring ability to self-organise or cooperation	
Institutions	Institutional index (continuous)	+
Clarity	Clarity of institutions	+
Fairness	Fairness of institutions	+
Enforcement	Monitoring & enforcement	+
Governance	Governance & Democracy	+
Groupsize	Group size (continuous)	±
Trust	Measured on a scale from 0 – 10 (continuous)	+
Ethnicity	Number of ethnic groups (continuous)	-
Wealth	Wealth index (continuous)	±
Projectyear	Year project was established (continuous)	+
Punishment	Punishment [0=punishment exogenous 1=endogenised]	+
Resource size	Resource size in ha (continuous)	±
Stakeholders	Number of stakeholders (continuous)	undetermined
Variables for the second model		
Biodiversity index	Dependent variable measuring success of biodiversity outcomes	
Benefits	Benefits from wildlife conservation (continuous)	+
Market	Distance to the nearest urban centre in km (continuous)	+
Distfence	Distance to the fence (continuous)	undetermined
Social capital	Social capital index (continuous)	±
Fence	New electric fence	undetermined
Age	Average age of household head (continuous)	±
School	Average number of years in school (continuous)	±
Training	Received training (number of training courses)	+
Yearsliving	Average number of years living in the area (continuous)	+
Information	Information sharing index	+

Cooperation is used in this study as a signal for self-organization. Social capital is a measure of the relationships, networks or linkages (bond and bridging linkages) between people, i.e., how relatives, neighbours, colleagues and friends work together in a community. Defined this way, social capital can be viewed as an asset. The institutional variable is disaggregated further into four components: clarity, fairness, enforcement and enforcement, and governance and democracy. Clarity of institutions and fairness are attributes that measure how people perceive community institutions in terms of whether rules are clear, easy to understand and fair to everybody. The information index tries to measure the flow of information in the community and at meeting and the type of information that is conveyed and whether the information is relevant in managing their natural resources. Training is measured in terms of number of training workshops or courses received, participation and type of training, i.e., the relevance of the training received in natural resource management.

Like cooperation, institutions and biodiversity, all other indices (i.e., trust, clarity, fairness, governance and democracy, monitoring and enforcement, information and social capital) were also recovered using factor analysis. A battery of questions related to each variable was asked to respondents and this information was then used to compute the index. The most appropriate variables for factor analysis are binary in nature since it is easy to interpretation the index (e.g., the yes or no type of variables). Continuous type variables (including a scale from 0 to 10 to indicate the presence or absence of an attribute) were also collected depending on the type of question being asked and level of difficulty in soliciting a response. The main idea is not to interpret the index, but to use the variable in a regression model. The institutional index was disaggregated into five different components and each component was measured separately. The overall institutional index was then recovered using factor analysis from the five components. For the specific type of questions that were asked, we refer the reader to the questionnaire in the appendix section.

We suspect the problem of endogeneity in both models, particularly in the relationship between cooperation and biodiversity. This is because the theory posits that there is reverse causality between biodiversity outcomes and cooperation. In other words, less biodiversity (scarcity) translates into more cooperation in order to avoid the tragedy of the commons, and vice versa. As a result, some scholars argue that the incentive to self-organise does not always hold, especially when resources occur in abundance (Ostrom, 1990). However, this is not the case in Zimbabwe because wildlife resources have declined tremendously since the turn of the 21st century. To put this into perspective, the wildlife management policy in the country speaks to the scarcity of resources. This is also the reason that the government institutionalised the CAMPFIRE programme in order to enhance the stock of wildlife.

Ostrom (2007a; 2007b) argue that institutions come first and cooperation later, since people need to conform to some behavioural standards (i.e., rules, values or norms) commonly referred to as institutions in the common pool resources literature. In the absence of institutions (rules, values, and norms) it is difficult to measure cooperation since there is no reference point (or standards) against which we can compare people's behaviour. So we need these standards for us to be able to talk about either compliance or deviation from the rules, norms or cultural values set by the community. Based on this idea, we might assume the relationship between institutions and cooperation is unidirectional, particularly for communities in which relevant institutions are not yet capable of addressing their current needs for resource management.

In a different paper, Ostrom et al. (1994) distinguished between triggering institutions, which trigger cooperation among community members, and sustaining institutions, which are responsible for sustaining cooperation over time, thereby leading to sustainability of a SES. In this case, the problem of endogeneity between cooperation on institutions comes into play when community members demand better institutions in a later phase of development in order to sustain cooperation after realizing the benefits of accruing to the group as a whole.

The literature on conditional cooperation seems to support the idea that institutions come first, followed by cooperation and finally biodiversity outcomes. This literature suggests that users of a common pool resource cooperate conditional on punishment (e.g., Fehr & Gachter, 2000; Masclet et al., 2003; Nikiforakis et al., 2007; Herrmann et al., 2008), and on cooperation of others (Rustagi et al., 2011; Fischbacher & Gachter 2010; Fischbacher et al., 2001). This might imply that punishment triggers and sustains cooperation in order to achieve some desired objectives. Punishment is part of a community's institutions. It is administered to offenders so that people will cooperate with the rules, values or norms of the group.

We acknowledged that both cooperation and institutions might not be the only endogenous variables in our models. Because of the endogeneity issues discussed above, we first estimate the models using ordinary least squares (OLS), ignoring any issues of endogeneity. We then employ instrumental variables estimation with heteroskedasticity-based instruments, which methodologically deal with the problem of endogeneity, and compare the results. Following Lewbel (2012), this method estimates an instrumental variables regression model providing the option to generate instruments and allows the identification of structural parameters in regression models with endogeneity or mismeasured regressors in the absence of traditional identification information such as external instruments. For a detailed description of Lewbel's approach and the mathematics of constructing instruments as simple functions of the data, please refer to Section 2.3 and Appendix A.

Lewbel (2012) proposed a novel identification strategy, which utilizes a heteroskedastic covariance restriction to construct an internal IV. Identification is achieved in this context by having explanatory variables that are uncorrelated with the product of heteroskedastic errors.[33] Correlation in the error terms due to an unobserved common factor is a key feature in many models (Baum et al., 2013). According to Lewbel (2012), instruments may be constructed as simple functions of the model's data. As a result, the approach may be applied in cases where no external instruments are available or used to supplement weak external instruments in order to improve the efficiency of the instrumental variables estimator.

We acknowledge that the benefits from wildlife conservation might cause the problem of endogeneity in the biodiversity model and, therefore, should not be used directly. We use the predicted values of the variable to isolate the independent effects of benefits on our measure of biodiversity. The new variable benefits-hat is obtained by running a linear regression model with benefits or environmental income as a function of other covariates that are not part of the biodiversity equation and then proceed to generated predict values. These predicted values are then used in the biodiversity model. We hope that the problem of endogeneity would be reduced by performing this procedure.

3.3.3 Data sources

Through household surveys and key informant interviews, the study collected data from a sample of 336 households and 30 key informants. The household questionnaire collected

[33] The greater the degree of scale heterosckedasticity in the error process, the higher will be the correlation of the generated instruments with included endogenous variables, which are the regressands in the auxiliary (first stage) regression (Lewbel, 2012).

information about the household's socio-economic characteristics, such as demographics, agricultural activities, assets, income, expenditure and involvement in community wildlife activities, while the key informant questionnaire collected information at the community level about the community's involvement in wildlife activities. Secondary data was also collected from the respective RDCs to complement the survey data.

3.4.Results and Discussion

3.4.1 Characterization of the Community and its Institutions

The data shows great variability in terms of household, community and institutional characteristics. Table 3.4 shows that the average group size is about 451.6 and ranges from 6 to about 2000 households, depending on whether the community is operating at ward level and whether we are talking about CAMPFIRE projects or resettlement schemes[34]. The average number of ethnic groups for the communities in question is 2.6. The data shows that the average age of the head of the household is 48.9 and ranges from about 22 to 89 years. About 50.0% of the communities have managed to endogenise punishment; they have systems in place for monitoring and enforcing rules and regulations, but the degree to which punishment is internalised varies as we move from one community to another. Whether monitoring and enforcement are externally done by a third party or internalised by local communities has implications for cooperation, which in turn affects the success of biodiversity outcomes.

The benefits from wildlife conservation range from 0 to US$68 880.00 during the survey period, with a mean of US$20 047.00, while the average size of the conservation area is 14 186 hectares (ha), ranging from about 7 614 ha to 26 000 ha. In some communities, the conservation area has been greatly reduced because the fence of the national park has recently expanded its boundaries, partly due to pressure from the state to increase land under conservation and the availability of donor funding. This affected the benefits flow because some communities are no longer entitled to benefits. About 62.5% of the committee members in the respective communities have received some form of training related to wildlife management. A number of stakeholders were involved in administering the training,

[34] As part of the land reform programme during the 1990s, peasant farmers were allocated plots around the Gonarezhou National Park which was previously reserved for white commercial farmers.

including the RDC, Zimbabwe National Parks and Wildlife Authority and several NGOs[35]. The average number of stakeholders was 4.2 per community, which is quite substantial given the size of the area under consideration (refer to Table B.3 in appendix B).

Table 3.4: Descriptive statistics of the variables used in the econometric model

Variable	Obs	Mean	Std. Dev.	Min	Max
Group size	336	451.6	461.4	6	2000
Trust [scale from 0 - 10]	336	4.896	2.192	1	10
Number of ethnic groups	336	2.631	0.901	1	5
Year project was established	336	12.99	7.157	4	31
Nature of punishment [0=Endo, 1=Exo]	336	0.506	0.501	0	1
Resource size (ha)	336	14186	7614	0	26000
Training (0=no 1=yes)	336	0.628	0.484	0	1
Benefits	336	20047	17290	0	68880
Benefits-hat	336	0.528	0.335	0	0.852
Distance to the market (km)	336	65.45	25.88	33	133
Distance to the fence (km)	336	9.843	16.77	0.100	80
Number of poaching incidents	336	7.955	6.192	0	22
Average age of household head	336	48.88	13.62	22	89
Average number of years in school	336	5.524	4.257	0	15
Residence status for the head	336	0.848	0.359	0	1
Head born in this area	336	0.708	0.455	0	1
Average number of years living in the area	336	36.64	13.57	6	73

Source: survey data Aug 2013

Table 3.5 below shows that the performance of most communities in the study area is well below the desirable level in terms of many characteristics that matter for conservation. On a scale from 0 to 100, where 0 signifies the complete absence and 100 the complete presence of an attribute, all community attributes in Table 3.5 fall below half (50.0), except for information sharing, which has a mean of about 60.56. Most communities share vital information such as financial matters, past actions and knowledge of the SES, mainly through village meetings. The mean level of cooperation is about 39.26, while the mean for the overall institutional index is 34.06. Disaggregating the institutional index into four attributes, namely, clarity of institutions, fairness, governance (including participation and democracy) and monitoring and enforcement (including formal punishment and social sanctioning), we observe that the mean of each attribute is still worrisome, especially for the governance, participation and democracy index. This seems to suggest that, in most communities, the

[35] Training offered by these organisations ranges from basic courses, such as bookkeeping or record keeping, to specialised courses, such as constitution development, leadership courses, veldt fire management, training for armed game guards (including the use of firearms) and general wildlife management (including animal counting, provision of watering points, trophy quality, live animal cropping, etc.).

quality of local institutions is very poor. This could have implications for cooperation in a community and, by extension, to the whole CAMPFIRE programme.

Table 3.5: Summary of indices

Variable	Obs	Mean	Std. Dev	Min	Max
Cooperation index	336	39.26	28.47	0	100
institutional index	336	34.06	22.05	0	100
clarity index*	336	40.97	23.62	0	100
fairness index*	336	40.81	20.85	0	100
governance index*	336	34.96	20.82	0	100
monitoring index*	336	37.67	24.40	0	100
wealth index	336	31.84	24.18	0	100
social capital index	336	18.18	14.49	0	100
information index	336	60.56	22.95	0	100
biodiversity index**	336	1.520	0.950	0.06	3.14

Source: Survey data Aug 2013
* Indices for institutional characteristics are combined using factor analysis to give an overall index of 34.06
** The Shannon index for biodiversity

In general, qualitative interviews revealed that communities that joined the CAMPFIRE programme earlier had better institutions in place compared to communities that either joined later or are operating outside the programme, except for Nyangambe.[36] The discrepancies in institutional characteristics across communities can be attributed to the fact that communities that started earlier, such as the Mahenye CAMPFIRE project, enjoyed a lot of donor funding and privileges from the state, which led to their success. This also meant that the relationship between earlier communities and state authorities improved over time because of the attention they got from the international community. Furthermore, these communities had ample time to learn from their own mistakes, experiences and past achievements, and hence adapted accordingly. In addition to this, such communities also received adequate training to build their own institutional capacity from the CAMPFIRE programme itself, state apparatus and various other local and international non-governmental organisations that were involved in wildlife conservation at that time.

Table B.4 in the appendices shows that community involvement in wildlife conservation in the study area varied significantly across communities. Most respondents (96.7%) indicated that the community uses awareness campaigns as a vehicle to fight illegal harvesting of

[36] Its success came from the fact that the community was part of the Save Valley Conservancy from its establishment and was involved in wildlife conservation from 1990 until 2004, when Nyangambe temporarily pulled out due to political interference in the conservancy.

wildlife, to educate the community and to foster cooperation, while 59.5% indicated that they actually carry out anti-poaching activities. About 42.3% indicated that they have a veldt fire management committee in place, while 36.3% use whistle-blowers to alert the authorities to any illegal activities happening in their community. Fewer respondents indicated that the community participates in quota setting (32.4%) and game cropping[37] (12.5%).

About 38.7% of the respondents stated that they enjoyed use rights, while 33.6% indicated decision making rights (refer to Table B.5 in the appendices). When asked about the extent to which the communities enjoyed both use and decision making rights on a scale from 0 to 10, there was great variation in the observations across projects, with means of 1.97 for use rights and 1.55 for decision making. About 91.3% of the respondents indicated that the option to enter or exit is not available for community members, while 66.4% indicated that the community is not able to exclude external or untitled parties. Many respondents (about 86.0%) were aware of the existence of a constitution for the community, while fewer respondents (25.6%) were aware of the existence of a wildlife management plan. It is essential at this stage to highlight that wildlife management committees exist for all communities (100.0%), since this is a requirement for local communities to be recognised by the state as a conservation group and for them to benefit from wildlife conservation.

Table B.6 in the appendices shows the number of rules, meetings and activities, and the participation rates for each. On average, using a scale from 0 – 10, the extent to which community rules are recognised by community members and higher level authorities is 3.51 and 6.10, respectively. This indicates that a number of communities still have problems in terms of rule compliance. This is also supported by the high number of poaching incidents (about 7.9 per year) in the study area.

3.4.2 Results of the Regression Models

This chapter argues that institutions directly affect cooperation (defined as the ability to self-organise) and indirectly influence the success of biodiversity outcomes through cooperation.

[37] Eltringam (1994) defines game cropping as the taking of a sustainable yield from a completely wild population. This definition implies regular harvest from a wild population. Cropping would have the objective of either wild animal population control or harvesting to provide bushmeat and other wild animal products for local consumption and/or for income generation.

Using regression analysis, we analyse the association between institutions and cooperation and the relationship between cooperation and success of biodiversity outcomes. As stated earlier, we are aware of the endogeneity problem; although we did not expect our data to suffer from it, it has to be corrected. First, we ignore any endogeneity issues and use ordinary least squares regression analysis, and then we use instrumental variables estimation with heteroskedasticity-based instruments to methodologically deal with the problem. For robustness check, an experiment of introducing the institutional variable directly in the biodiversity model, i.e., equation 3.3 is done. As expected, the variable was insignificant see results in Table B.13 Appendix B. This is consistent with the structural model deduced by the researcher.

The Durbin-Wu-Hausman tests for endogeneity (refer to Table B.7 in the appendices) seem to suggest that OLS yields better results in the first model (relationship between cooperation and institutions), while the instrumental variables estimation with heteroskedasticity-based instruments yields superior results in the second model (relationship between biodiversity and cooperation). The VIF tests show that multicollinearity is not a severe problem for both models (please refer to Table B.8 in the appendices). The rule of thumb suggests that the VIF should be less than 10; otherwise, we have multicollinearity issues (Menard 1995; Neter et al., 1989; O'Brien 2007). Furthermore, the correlation results in Table B.14 and Table B.15 in the appendices also support the results of the VIF test above.

The results of the under-identification, weak identification, over-identification and heteroskedasticity tests obtained from the instrumental variables estimation models are presented in Table 3.6 and Table 3.7 below. Under the null hypothesis that the equation is under-identified, the Kleibergen-Paap test shows that it is safe to proceed with the instrumental variables estimation with heteroskedasticity-based instruments in both models. Usually, the p-value should be very small in order to reject the null hypothesis in favour of the alternative hypothesis that the equation is not under-identified. The rule of thumb for the weak identification test using the Cragg-Donald Wald F-statistic suggests that we reject the null hypothesis of weak identification if the F-statistic is large. Weak identification arises when the excluded instruments are weakly correlated with the endogenous regressors. Estimators can perform poorly when instruments are weak.

The Hansen J statistics for testing over-identification of all instruments also reveals that it is safe to use instrumental variables estimation with heteroskedasticity-based instruments in both models under the joint null hypothesis that the instruments are valid instruments, i.e., uncorrelated with the error term, and that the excluded instruments are correctly excluded from the estimated equation. A rejection casts doubt on the validity of the instruments. Therefore, the p-value must be very large in order not to reject our null hypothesis.

Because we are considering 30 communities (which imply about 30 observations), there is a worry that we might be dealing with too few observations. One way to deal with that is bootstrapping. We applied the bootstrapping procedure to check for robustness of the standard errors and found that the results are consistent with each other. The results in Table B.10 through Table B.12 in the appendices do not vary significantly from the original results. We take this to be a validation of the results and, henceforth, we will interpret them. In the section below, we present both OLS and instrumental variables estimation results for both models.

Regression Model 1 - Relationship between Cooperation and Institutions: The results in Table 3.6 below show that both OLS regression model and instrumental variables estimation are highly significant and explain over 80.0% of the variation in the dependent variable. We consider two models, a model with the overall institutional index and another with disaggregated institutional indices.[38] Replacing the overall institutional index with disaggregated institutional indices does not affect the signs of the explanatory variables in the model and so the interpretation remains the same. Comparing OLS and the IV results, we observe that most variables are highly significant, except for group size, ethnicity, resource size and wealth. The variable ethnicity measured heterogeneity in a community or group, but the variable is insignificant in both models, suggesting that ethnicity is not an important variable explaining cooperation in the area. However, with disaggregated institutional indices, group size, resource size and wealth become significant under both OLS and IV estimation, implying that cooperation is explained better with the latter model than with the former.

[38] As highlighted earlier, the overall institutional index can be disaggregated into four institutional characteristics: clarity of institutions, fairness, governance and monitoring, and enforcement. The objective of disaggregating the overall institutional index into these four attributes is to establish those characteristics of institutions that matter most for cooperation.

The results show that the institutional variable is positive and significant at the 1% level of significance. This suggests that an improvement in the quality of institutions increases cooperation in the respective communities. This result is consistent with theory and confirms our hypothesis that institutions matter for self-organisation, i.e., good institutions translate into higher levels of cooperation and vice versa. It is important to note at this point that all communities in the study area have some form of institutions, but these institutions differ in terms of their characteristics as we move from one community to another. Hence, we can rule out the possibility of an open access regime where members of the community can access resources at any time without restraint, i.e., a system where there are no rules governing access to and utilization of resources.

When we consider the model with the disaggregated institutional index, we observe that both the governance index and the monitoring and enforcement index are highly significant and positive, implying that an improvement in these variables might increase cooperation. The results suggest that governance, monitoring and enforcement are more important for cooperation than fairness and clarity of institutions. The results underscore the need for institutional arrangements that allow local communities to fully participate in wildlife conservation, govern their resources in a democratic way, monitor each other, and enforce rules and regulations internally, since this is more likely to encourage higher levels of cooperation with possible implications for biodiversity outcomes.

However, the fairness index is highly significant and carries a negative sign. This result seems counter-intuitive, but there is an explanation for this. Because we collected qualitative data, a possible explanation for this anomaly could be that beneficiaries closer to the park fence frequently suffer from wildlife intrusion and hence feel that they should be treated differently from those farther away from the fence. Pooling beneficiaries in the same ward together and treating them as equal is problematic because of the difference in their experience with wildlife since some of them are located near the boundary and hence suffer more from intrusion while other live far away from the border and hence experience less disturbances from wildlife. This also explains why group size is very unstable at ward level and why communities frequently divide into smaller groups in order to ensure that communities that suffer more from wildlife intrusion benefit more than those farther away.

Table 3.6: Relationship between cooperation and institutions

Cooperation	Model with overall institutional variable		Model with institutional attributes	
	OLS	*IV*	*OLS*	*IV*
Number of obs	336	336	336	336
Prob>F	0.0000	0.0000	0.0000	0.0000
R-squared	0.8542	-	0.8854	-
F (10, 325)	-	223.22	-	320.2
Centered R2	-	0.8191	-	0.844
Uncentered R2	-	0.9378	-	0.946
overall institutional index	0.129***	0.557***		
	(0.0487)	(0.177)		
clarity index			0.117*	0.195
			(0.0696)	(0.137)
fairness index			-0.160***	-0.515**
			(0.0537)	(0.150)
governance index			0.192***	0.603***
			(0.0732)	(0.168)
monitoring and enforcement index			0.360***	0.245***
			(0.0512)	(0.103)
group size	0.00121	0.00472*	0.00309*	0.00205**
	(0.00181)	(0.00197)	(0.00161)	(0.00152)
Trust	3.120***	1.151***	1.800***	1.496***
	(0.574)	(0.893)	(0.554)	(0.756)
Ethnicity	-1.172	0.231	-0.626	-0.594
	(0.767)	(1.135)	(0.714)	(0.960)
wealth index	0.0673**	0.0254	0.0555**	0.0642**
	(0.0268)	(0.0342)	(0.0239)	(0.0278)
year of establishment	1.672***	1.181***	1.255***	1.166***
	(0.202)	(0.328)	(0.187)	(0.267)
punishment [0 = Exo, 1 = Endo]	12.99***	11.23***	7.195***	9.815***
	(1.843)	(2.170)	(1.831)	(2.528)
resource size	-0.000150	-0.000287***	-0.000265**	-0.000366***
	(9.22e-05)	(9.84e-05)	(0.000111)	(0.000157)
number of stakeholders	1.552***	1.741**	0.912*	0.557***
	(0.564)	(0.702)	(0.533)	(0.596)
Tenure	-8.657***	-7.465***	-6.617**	-12.17***
	(2.883)	(2.153)	(3.271)	(4.855)
Cons	-12.04***	-12.62**	-9.085***	-3.016
	(3.635)	(5.121)	(3.360)	(4.072)

Underidentification test (Kleibergen-Paap rk LM statistic):			72.208		54.450
	Chi-sq(10) P-val	=	0.0000	=	0.0000
Weak identification test (Cragg-Donald Wald F statistic):			19.373		16.281
Hansen J statistic (overidentification test of all instruments):			10.574		9.5841
	Chi-sq(9) P-val	=	0.2308	=	0.2623
Breusch-Pagan test for heteroskedasticity	chi2(1)	=	18.543	=	29.172
	Prob>chi2	=	0.0000	=	0.0000

Source: survey data Aug 2013
NB: Standard errors shown in brackets
* Significant at 10% ** Significant at 5% *** Significant at 1%

Community level trust, year of establishment, punishment and number of stakeholders have positive and significant impacts on cooperation. This seems to suggest that cooperation is better in communities where members trust each other than in communities where trust is lacking. As suggested in the previous section, cooperation is higher in communities that joined wildlife conservation earlier than in communities that joined later. This makes sense because the longer a community is involved in wildlife conservation, the more likely it is for that community to develop robust institutions that are adapted to local conditions. Cooperation is also higher in communities that have endogenised punishment as opposed to communities that still rely on external force in order to enforce adherence to rules and regulations.

Policymakers and development practitioners should seriously consider institutional reforms that convey greater control of natural resources through devolution and decentralization of managerial functions, decision making and authority into the hands of local communities, while the state maintains regulatory functions. Ostrom et al. (2007) argue that it is cheaper for local communities to engage in monitoring and enforcement activities than it is for state apparatus to do so, due to budgetary and information constraints. The results also reveal that cooperation increases with an increase in the number of stakeholders. This is true in the study area because there are a number of NGOs working with local communities in wildlife conservation, particularly in providing training or capacity building.

The tenure variable is negative and highly significant (at 1%) under both models, implying a negative relationship between tenure and cooperation. This suggests that cooperation declines as we move from communal areas into the resettlement schemes. This is true for two main reasons. Firstly, the households in resettlement schemes are permit holders operating on individual plots, who are in most cases maximizing individual objectives at the farm level but not as a community as the scope of joint activities is small. Given the nature of resettlement schemes, it is difficult for park authorities to monitor and enforce rules unless farm owners come together and act as a community in order to achieve a common objective of conservation. Secondly, the areas of collaboration tend to be fewer in resettlement areas to the extent that farmers act individually and no one knows what the other farmer is doing in most instances; hence, their institutions are not as well developed as for communal areas.

Furthermore, with the introduction of disaggregated institutional indices, group size, the size of the resource system and wealth become significant under both models, although the interpretation is not immediately intuitive. The results show that there is a significant and positive relationship between group size and cooperation. In general, the theory posits that the effect of group size on self-organisation tends to be negative, given the higher transaction costs associated with getting people together and agreeing on important issues (Ostrom, 2009; Wade, 1994). However, if the tasks of managing a resource system are very costly (e.g., monitoring an extensive resource system), larger groups are in a better position to mobilise the necessary resources required for such undertakings. Wildlife systems are typically extensive and impose a huge burden on the communities because they require more human capital in addition to other resources. Hence, the size of the group is always relevant, but its influence on the ability of a community to self-organise is contingent on other variables of the SES (Ostrom, 2009).

The results also show that the level of cooperation in a community declines as the size of a resource system increases. Chhatre and Agrawal (2008) argue that self-organisation is more unlikely in very large and very small resource systems. The reason could be that very large resource systems are associated with high costs of defining boundaries, monitoring and enforcement, and gaining ecological knowledge, while very small resource systems do not generate substantial flows of valuable products. Hence, a moderate-sized resource system is most conducive to self-organisation. We therefore argue that, for most communities considered in the study area, the resource systems are big enough to generate tangible benefits, and communities with reasonable conservation areas are better off in terms of fostering cooperation than communities with larger conservation area.

The wealth index becomes significant and carries a positive sign under both models when the disaggregated institutional attributes are introduced, suggesting more cooperation in wealthier communities than in poor communities. The conventional wisdom from field experiments suggests that, at the group level, both average group wealth and variance in the distribution of wealth decrease the level of cooperation or social efficiency achieved by the group (Ostrom et al., 1994; Cardenas, 2003). However, in this case, the burden imposed by an extensive resource system (e.g., monitoring) can better be handled by communities with greater mean than relatively poor ones.

Regression Model 2 - Relationship between Biodiversity and Cooperation*:* Both OLS and IV estimation models are significant at the 1% level and explain over 72.2% of the variation in our dependent variable. Although the Durbin-Wu-Hausman tests suggest that IV estimation with heteroskedasticity-based instruments is superior to its OLS counterpart, all variables in both models are significant and do not vary much in terms of their coefficients, standard errors and the level of significance for the two models.

Table 3.7 below shows that cooperation is positive and highly significant. This implies that cooperation is an important variable explaining biodiversity outcomes, as suggested in the CPR literature. Hence, we expect to find more sound ecological outcomes in those communities with high levels of cooperation and strong institutions than in communities with less cooperation and weak institutions. We maintain that institutions affect biodiversity outcomes indirectly through their ability to self-organise or cooperation. Alternatively, we might think of cooperation as refraining from illegal harvesting of wildlife resources, so that, in areas where the level of cooperation is very low and poaching activities are rife, wildlife is either quickly decimated due to overharvesting, or may respond to higher levels of poaching by retreating back into the park, thereby leaving few wild animals in the community's conservation area. As a result, a community that is less poached has more animals in its vicinity than a community that is heavily poached.

As expected, training is positive and highly significant, implying that this variable is an important factor explaining the success of biodiversity outcomes. This suggests that communities that received training are better off in terms of managing and conserving wildlife than communities where training has not been administered. However, not all communities have received training relevant to wildlife management. The number of households and communities involved in wildlife conservation is growing, signalling the need for more training in the study area. Qualitative interviews also revealed that committee membership changes quite frequently and, at times, there is a total overhaul of the entire management committee which may severely affect operations. There is, therefore, a need for continuous training so that the institutional memory, entrepreneurial and leadership skills acquired through training are not lost when such a dramatic change occurs. From this perspective, government programmes should target capacity building in terms of institutional capacity and skills development in order to have a positive and significant impact on biodiversity.

Table 3.7: Model 2 - Relationship between biodiversity and cooperation

Biodiversity	**OLS**		**IV**
Number of obs	336		336
Prob>F	0.0000		0.0000
R-squared	0.7223		-
F (10, 325)	-		283.38
Centered R2	-		0.807
Uncentered R2	-		0.9224
Cooperation	0.00595***		0.00758***
	(0.00246)		0.00215
training [0=yes, 1=no]	0.215***		0.208***
	(0.0745)		(0.07620)
Benefits-hat	0.5165***		0.2651***
	(3.56e-06)		(3.37e-06)
distance to nearest urban centre	0.0122***		0.0126***
	(0.00151)		(0.00130)
distance to the fence	0.0148***		0.0131***
	(0.00212)		(0.00205)
social capital index	0.00704***		0.00710***
	(0.00200)		(0.00202)
average age of household head	-0.357***		-0.381***
	(0.12374)		(0.13522)
average number of years in school	-0.0926***		-0.0892***
	(0.02271)		(0.02345)
average number of years living in the area	-0.00538**		-0.00597**
	(0.00218)		(0.00247)
Fence	-0.00345***		-0.00490***
	(0.00129)		(0.00182)
Information sharing index	0.242***		0.210***
	(0.04296)		(0.04558)
Cons	9.631***		10.34***
	(3.02523)		(3.34761)
Underidentification test (Kleibergen-Paap rk LM statistic):			139.319
	Chi-sq(10) P-val	=	0.0000
Weak identification test (Cragg-Donald Wald F statistic):			25.420
Hansen J statistic (overidentification test of all instruments):			0.0430
	Chi-sq(9) P-val	=	0.8352
Breusch-Pagan test for heteroskedasticity	chi2(1)	=	21.94
	Prob > chi2	=	0.0000

Source: survey data August 2013
NB: Standard errors shown in brackets
* Significant at 10% ** Significant at 5% *** Significant at 1%

As expected, the benefits from wildlife conservation significantly and positively affect biodiversity outcomes in a community. If the resource system is very important in the eyes of the users and generates a substantial flow of benefits, then users attach high value to sustainability of the resource (Berkes & Folke, 1998; Chhatre & Agrawal, 2008); otherwise, the cost of organising and maintaining a self-organised system may not be worth the effort (National Research Council, 2002). Communities from the study area have came to realise that using the proceeds from wildlife conservation to invest in public goods, such as schools, clinics, water, grinding mills, and electricity, is much more beneficial than getting dividends at the household level. Viewed from this angle, the benefits from wildlife conservation are tangible in the eyes of the community and thus community wildlife conservation, from this perspective, is successful. This line of reasoning differs from previous studies which considered benefits in terms of income flowing directly into the household.

Market integration and global market trends are viewed worldwide as potential threats to wildlife conservation in developing countries. In this chapter, we used distance to the nearest urban centre as a proxy for market integration. The variable distance to the nearest urban centre is positive and highly significant, suggesting that biodiversity outcomes improve as the distance to the market increases. We argue that the incentives for poaching are much stronger for those communities that are located closer to urban centres or main routes linking rural communities to urban areas, because animals and game fetch higher prices in wider markets. Qualitative interviews with key informants revealed substantial evidence of game meat being sold on the black market in almost all the communities visited during the survey period and, under certain circumstances, poachers transporting game meat to distant markets such as growth points and urban centres.

On the other hand, communities that are located closer to the fences are less likely to conserve biodiversity because they suffer more from wildlife intrusion and interact with wildlife frequently. This is confirmed by the results: the variable measuring distance to the fence is positive and highly significant. In other words, as the distance to the fence increases, biodiversity outcomes improve significantly. The results seem to suggest that biodiversity outcomes are more successful for those communities that are located far away from urban centres or routes connecting rural communities to urban centres, but are not very close to the boundary of the game park.

Social capital is also an important variable explaining the success of biodiversity outcomes. The variable is positive and significant. Social capital may either help to conserve or destroy biodiversity depending on the nature of the relationship. If social capital is high in a community, such that households have links both inside and outside the community, then these households are more likely to get assistance in times of need and hence, less likely to depend on illegal harvesting of wildlife resources. This is true for some communities in the study area because they have more children or relatives working in urban areas or abroad in South Africa. If households are connected at the community level (through common understanding, common interests, respect for each other and the need to maintain a long-term relationship that is beneficial to everyone), they are more likely to make decisions as a community. This minimises the possibility of social deviance while, at the same time, enhancing society's welfare.

The average age of the household head is significant and negatively related to the success of biodiversity outcomes. This implies that older household heads tend to be associated with worse biodiversity outcomes. This also suggests a positive relationship between the age of the household head and income based on resource use. Godoy et al. (1997) reported that the age of household head may be positively related to resource-based income until a point where resource use declines with age, coupled with children moving away to seek new opportunities and start their own households elsewhere. Including a quadratic term for the average age of the household head does not affect the results much. Table B.9 demonstrates that, as the age of the household head increases, biodiversity outcomes deteriorate up to a certain age (51.4 years), when poaching ceases to be an important livelihood activity for the household head due to old age.

The number of years in school and number of years living in the area for the household head are both significant and negatively related to biodiversity outcomes. Again, these results are not immediately intuitive. Expectations were that, as the number of years in school and the number of years living in the area increases, biodiversity outcomes improve. However, that is only realistic in areas with strong institutions. As the descriptive statistics indicated, there is a general weakness of institutions in the study area. The results with respect to age are therefore capturing resource-based income activities only. Highly educated households who have stayed in the area for long seem to be bent on exploiting the resource for income generation in a manner which do not necessarily benefit biodiversity. These trends can be sustained in the medium term through dispersal of wildlife from the park to the communal lands. In the long-run, it is not sustainable and, therefore, needs to be curbed.

An interesting development in the study area is the idea of putting an electric fence around Gonarezhou National Park in order to conserve wildlife and reduce human-wildlife conflict. The project has already started with support from donor funding. Its completion depends heavily on the availability of these funds. Most communities have lost part of their conservation area to the national park and the number is growing with an increase in the area of the park covered by the electric fence. Beneficiaries of conservation payments view this development as a potential threat to the CAMPFIRE project. The results show that the fence reduces biodiversity in the community's conservation area. This is true for two main reasons. Firstly, the fence greatly reduces the number of wild animals moving into the community's conservation area from the park. Secondly, the electric fence greatly reduces the benefits from wildlife conservation, thereby eroding the community's incentives to conserve wildlife resources. As a result, the local community might fight back by increasing their poaching effort. This might lead to resource overexploitation and, finally, exhaustion of all resident species as well as those that cross the fence.[39]

According to Ostrom (2007), information sharing is one of the most important variables that can affect ecological outcomes. During meetings, communities share vital information about past actions, progress updates, general finance matters, fire outbreaks, watering points, poaching and knowledge of the SES, in addition to their usual community agendas. The results show that biodiversity improves when the community is able to share information. Sharing information entails both responsibility and accountability of community leadership. This, in turn, facilitates the development of a relationship based on trust and honesty. In addition, the communities perceive lower cost of organising when users share common knowledge of relevant SES attributes, rules and regulations, and how their actions affect each other (Ostrom, 2009; Berkes & Folke, 1998).

[39] Predators and small plains game that can sneak under the fence in rugged terrain are still able to move inside and outside the park and interact with human beings. Furthermore, elephants can still damage the electric fence if they have mastered the technique of doing so.

3.5. Conclusions and Policy Implications

This study used a sample of 336 households and community-level data from 30 communities to analyse i) the relationship between ability to self-organise and institutions, among other variables identified as affecting collective action; and ii) the relationship between success of biodiversity outcomes and cooperation among variables, identified as enhancing the sustainability of a resource system. To achieve this, the study relied heavily on Ostrom's general framework for analysing complex social-ecological systems. We used ordinary least squares regression analysis and instrumental variables estimation with heteroskedasticity-based instruments. This approach methodologically deals with the endogeneity problem associated with the relationships above.

The results confirmed that sound institutions are indeed an important ingredient for cooperation. Improvements in institutional attributes such as governance (participation and democracy), monitoring and enforcement might lead to increased cooperation, while fairness and clarity of institutions were found to be less important. Community level trust, punishment, number of stakeholders and tenure were also found to be important variables explaining cooperation. With the introduction of disaggregated institutional indices, group size, the size of the resource system and wealth become significant.

Furthermore, cooperation had a positive and significant impact on biodiversity outcomes, suggesting that higher levels of cooperation might translate into a healthy wildlife population. We, therefore, argue that institutions directly affect cooperation, and indirectly influence biodiversity outcomes through cooperation. Cooperation, training, benefits, distance from the nearest urban centre, distance from the park fence, social capital, and information sharing were found to have a positive and significant impact on biodiversity outcomes. The average age of the household head, number of years in school, number of years living in the area and proximity to the park fence had a negative and significant impact on biodiversity.

Both the extent to which communities benefit from wildlife conservation and the extent to which they are allowed to make important decisions about how benefits are distributed and used by the community affect incentives to conserve wildlife. Fencing the national park has a detrimental effect on the CAMPFIRE project and the lives of people whose livelihoods depend on wildlife conservation.

The results of this study helped to shed light on the processes explaining complex social-ecological systems. A number of policy implications obtain from this analysis. The wildlife sector should be re-configured in order to favour the implementation of collective strategies that are endogenous to the community aimed at providing public goods such as wildlife through conservation. Government policy should recognize CAMPFIRE communities as important stakeholders with an important role to play, but not as beneficiaries of wildlife conservation simply because they live, interact with wildlife almost on a daily basis and suffer in the process. There is therefore need for future policy to define the roles of each stakeholder and demarcate boundaries within which each stakeholder can operate.

Future policy reforms should also consider further devolution of natural resource management function from the Rural District Council to CAMPFIRE communities or increased autonomy so that they community members can monitor each other and internalise enforcement, while the state maintains regulation functions. This will allow innovation among communities, while the government create an enabling environment for other stakeholders to operate. For example, the capacity of CAMPFIRE to develop their own rules, rather than just following externally-imposed rules, should not be undermined. The results show that external enforcement of rules and regulations does not necessarily translate into sound ecological outcomes; rather, better outcomes are attainable when punishment is endogenised by local communities. This seems to suggest that communities should be supported in a way that promotes the emergence of robust institutions that are tailor-made to suit local needs. This will, in turn, facilitate good environmental husbandry.

Since wildlife is a fugitive resource that roams freely on communal land, there is therefore a need for coordination among CAMPFIRE communities in order to supply the public good, the required habitat and to benefit from collective. The government, NGOs and private sector should act as catalyst of collective action to ensure that there are incentives for CAMPFIRE communities to self-organize into multiple layers of nested organizations through policies and programmes that allow resource users to participate in decision making process. Development programmes or policy interventions with both a welfare and a conservation component should not be designed as 'one size fits all' but should recognize and understand the differences in community characteristics and incentives to self-organize through policy and development programmes in order to promote conservation and safeguard the livelihood interests of pro-poor communities.

Government programmes can target capacity building in terms of institutional capacity and skills development in order to have a positive impact on biodiversity conservation. Increase spending on training relevant to natural resource management, and particularly wildlife conservation, targeting not only community leadership or project committees but also ordinary community members as a path way to conservation. Therefore, the capacity building efforts of government agencies, NGOs and other stakeholders should complement each other to ensure that the necessary resources are mobilised and all communities receive the necessary training and resources. State authorities should reconsider the way in which they engage with farmers under resettlement schemes, because biodiversity suffers more under this type of arrangement than in communal areas. There is a need for appropriate institutional reforms that allow park authorities to work closely with resettlement schemes, while at the same time giving incentives for plot holders to work together for the improvement of the common pool resource. For example, farmers in resettlement schemes could set aside land for conservation by pooling resources instead of operating at plot level, which works against conservation efforts.

Chapter 4

A Bioeconomic Model of Community wildlife Conservation: CAMPFIRE Community versus the Conservancy Community[40]

4.1 Introduction

Integrated Conservation and Development Projects (ICDPs), commonly referred to as Community-Based Natural Resource Management (CBNRM), are central to future rural development in Southern Africa (Munthali, 2007; and Thomson et al., 2013). Conceptually, CBNRM is a sound idea and seems likely to encourage conservation of wildlife resources and to improve the livelihoods of poor rural households if resources are exploited legally and commercially by local communities. Nevertheless, despite such arrangements, community wildlife conservation in the region still faces some serious challenges, one of them being illegal harvesting[41] of wildlife resources by local people living adjacent to protected areas (Murombedzi, 1999; Fischer et al., 2011; and Gandiwa, 2011).

This paper considers two communities that are involved in wildlife conservation under two different CBNRM arrangements around the Gonarezhou National Park (GNP) in Zimbabwe, but are experiencing very different wildlife conservation outcomes. Moreover, both communities exploit wildlife by selling hunting licenses to foreign hunters but again with different levels of success. For purposes of our analysis, we consider the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) communities and the

[40] A version of this chapter has been desseminated as an EfD Discussion Paper No. DP 15 – 28 and ERSA Working Paper 560.

[41] A distinction is made in this paper between commercial and subsistence poaching. Commercial poaching is presumed to be an open access business usually conducted by outsiders with the help of local communities, while subsistence poaching is mainly done for subsistence by the local communities themselves (Fischer et al., 2011). Local communities contribute to commercial poaching or illegal trophy hunting by supplying information to outsiders about the movements of wild animals in their wilderness area and sometimes provide escort services for a very small fee. The paper studies subsistence poaching by CAMPFIRE communities.

private game farms in the Save Valley Conservancy (SVC) [42]. The park agency plays a central role by authorizing the harvest quota for each community.

Campbell and Shackleton (2001) reported that the conservancy community performs much better than traditional CBNRM arrangements involving local communities with respect to biodiversity conservation and livelihood outcomes. It is argued that the design of benefit-sharing schemes, such as the CAMPFIRE programme, could be responsible for eroding the incentives of local communities to conserve wildlife (e.g., Fischer et al., 2011; and Johannesen and Skonhoft, 2014). In principle, CAMPFIRE aimed at giving indigenous communities co-ownership of local natural resources, particularly wildlife, so that they could generate income through leasing trophy hunting concessions, harvesting resources, game cropping and tourism activities (Fischer et al., 2011; Bond and Frost, 2005; and Balint and Mashinya, 2006). However, this goal has not been achieved, as CAMPFIRE only managed to devolve authority over natural resources from the central government to the Rural District Council (RDC). Murombedzi (1999) argues that, if CAMPFIRE is to be effective, a further devolution of authority is required so that producer communities, those who live directly beside wildlife, are given full control of the natural resources on their land.

Therefore, the main objectives of this paper are to develop a bioeconomic model for each of the three agents identified above (i.e., the park agency, conservancy and CAMPFIRE community); optimize the market problem for each agent and compare the outcomes with the social planner's solution; and finally to suggest appropriate reforms that might encourage the CAMPFIRE community to move from a seemingly suboptimal regime to one that is optimal. Unlike previous studies, this paper seeks to establish the conditions under which the CAMPFIRE community can be incentivized to behave like the conservancy community, which is more successful in revenue generation and stewardship practice. Given the background above, three important questions arise: i) What are the significant differences between the two types of communities that interact with wildlife in Zimbabwe? ii) How and why do their differences affect livelihoods and stewardship practice? iii) Given the apparent superiority of conservancy outcomes in terms of livelihoods and conservation, how can we

[42] The SVC is chosen due to its proximity to the GNP, the fact that the conservancy interacts with neighbouring communities and farmers in the SVC are operating as a community, just as in CAMPFIRE projects. Although the paper utilizes the GNP area as a case study, the results would generally apply to other areas in Zimbabwe.

incentive CAMPFIRE communities to behave like the conservancy community, i.e., what reforms are necessary in CAMPFIRE for it to achieve equivalent outcomes?

The rest of the chapter is organised as follows. Section 4.2 analyses the similarities and differences between the CAMPFIRE and conservancy communities. Section 4.3 develops the bio-economic model and computes estimates of some key parameters of the bio-economic model using data collected from communities around the Gonarezhou National Park (GNP) in Zimbabwe. Section 4.4 presents some comparative statics, including their policy implications, and discusses the results of the optimization problems and model simulation. Finally, section 5.5 concludes.

4.2. A History of Community-based Wildlife Conservation in Zimbabwe

In Zimbabwe, community-based wildlife conservation takes place mainly under two different CBNRM arrangements: conservancies and the CAMPFIRE programme. Zimbabwe has about eight conservancies with over 100 registered private game farms[43], and 37 CAMPFIRE districts, managed by 37 RDCs that comprise 118 wards with over 121,500 households participating in wildlife conservation. The eight conservancies cover an area of 1,140,688 ha in total, while the CAMPFIRE wards cover approximately 2,478,000 ha in total. The potential for crop cultivation is extremely low due to harsh climatic conditions (Gandiwa et al., 2014). This makes livestock rearing and wildlife conservation the most viable private investment options for both the conservancy and CAMPFIRE communities.

According to Bell (1984), emphasis on formal protected areas shifted during the 1970s with the recognition that islands of protection were inadequate for maintaining spatially heterogeneous biodiversity. The Parks and Wildlife Act (1975) gave landowners property rights to wildlife on their land (Murombedzi, 1999). Kreuter et al. (2010) identified groups of landowners within the bounds of the Great Limpopo Trans-frontier Conservation Area that have incorporated their properties into conservancies or private nature reserves, thereby expanding the management scale of common pool wildlife resources. Due to its fugitive character, wildlife is managed as a community, rather than on individual farms, in order to

[43] Because of the FTLRP, the actual number of private game farms and land area is not known due to farm inversions.

supply the required habitat size. This also allows individual farms to specialize in offering services and products that are in line with the resources found on their properties. Conservancies have played a crucial role in protecting wildlife and biodiversity on private farmland outside of formal protected areas since their establishment in Southern Africa during the early 1970s (Muir-Leresche and Nelson, 2000).

The SVC is used in our paper as a case study of CBNRM arrangements involving private game farms in Zimbabwe. Its establishment was meant to rectify the ecological imbalances and environmental degradation caused by excessive cattle ranching during the 1920s, which subsequently forced wildlife to the outskirts of the valley (Fitzgerald, 2012). As a matter of policy, it was suggested that decreasing the number of cattle and introducing the original wildlife back to the area would help to restore the natural balance. The SVC was formed by combining 24 adjoining farms measuring about 3200 km². It is involved in intensive protection of rhinos, private game safaris, limited hunting concession and multi-species research. Farmers in the conservancy receive most of their income from high-quality and low-density tourism, including accommodation for travelers. Furthermore, it supports local communities by supplying jobs, allowing them to sell their arts and crafts, and improving and upgrading the Save Valley area.

Because of this intervention, the environment slowly recovered. Many of the indigenous plants and vegetation have been rehabilitated and the area has been successfully restocked with wildlife. Muir-Leresche and Nelson (2000) reported an increase in the area under conservation and wildlife population on private land. Moreover, the conservancy community exhibits characteristics that enhance CBNRM and coordinated decision-making for wildlife conservation (Krug, 2001; and Kreuter et al., 2010).

Upon independence, the government enacted a new law, the Parks and Wildlife Act of 1982, which gave birth to the CAMPFIRE programme - a benefit-sharing scheme involving local communities. The law aimed to provide democratically elected RDCs the appropriate authority for managing wildlife within their geographical boundaries. This new paradigm entails conferring on local communities, through their respective RDCs, (i) greater control over formerly public wildlife in communal areas in defined territories, (ii) enhanced capacities to add value to local wildlife, and (iii) specific financial rewards linked to the estimated conservation value of wildlife within their territories (Gadgil and Rao, 1994; and Murombedzi, 1999). Provided these commitments are forthcoming, the park agency steps back into the role of regulator and adviser, retaining the right to control wildlife harvesting quotas (Fischer et al., 2011).

Previous studies have described the CAMPFIRE programme as a role model for CBNRM in Southern Africa (Murombedzi, 1999; Logan and Moseley, 2002; Muchapondwa, 2003; and Balint and Mashinya, 2006). The fundamental idea behind such initiatives is that benefits from wildlife conservation should strengthen the incentives of local people in such a way that they treat wildlife as a valuable asset (Songorwa, 1999; Songorwa, 2000; and Balint and Mashinya, 2006). Viewed as an asset, wildlife has the potential to provide local communities with a hedge against agricultural risk associated with extreme weather conditions, by creating employment and generating revenues (Muchapondwa and Sterner, 2012; and Poshiwa et al., 2013).

However, the CAMPFIRE programme has enjoyed very limited success over the entire course of its establishment. Poaching subsided only temporarily after its commencement, as neighbouring communities started to reap economic benefits from legal wildlife utilization, and then rebounded a few years later (Fischer et al., 2011). Evidence of human-wildlife conflict and poaching of elephants and rhinos is well documented by both scholars and International Development Agencies that are involved in wildlife conservation in Zimbabwe such as DFID WWF, UNEP and IUCN. Recent evidence reveals that over 200 elephants were killed by local communities using cyanide poisoning within the past two years. This year alone, fourteen elephants were poisoned by cyanide in Zimbabwe in three separate incidents in Matusadona National Park in the Kariba and most recently 26 elephants in Hwange National Park.

Evidence also shows that CAMPFIRE areas close to human settlements in the northern GNP have low populations of large herbivores such as buffalo, giraffe, and zebra compared to areas further inside the park, possibly due to illegal hunting and competition for forage with livestock (Gandiwa et al., 2013a; Dunham et al., 2013). This evidence demonstrates that poaching incidences have been on the increase for more than a decade now. Considering the same period, poaching incidences in conservancies have stalled due to effective monitoring and enforcement. This has actually seen the population of endangered species such as rhinos and wild dogs improving in some conservancies but not all. The Save Valley Conservancy has reported an increase in the population of white rhinos which is almost facing extinction in Africa. Effective monitoring under the Save Valley Conservancy is made possible by the used of well-trained game guards with firearms and donor support.

Lindsey et al. (2009) documented that from August 2001 to July 2009 in Save Valley Conservancy 10,520 illegal hunting incidents were recorded, 84,396 wire snares removed, 4,148 hunters caught, 2,126 hunting dogs eliminated and at least 6,454 wild animals killed. However, there are no similar reports about CAMPFIRE communities due to lack of coordination, resource need to collect such type of information and local politics or political will but the levels of crime could double or triple thesis figure considering the same period. Furthermore, there are no reports about anti-poaching enforcement exerted by both communities as measured in terms of fines collected from the poachers. Poachers in Zimbabwe are usually surrendered to the police where the possible sentences are: to pay a paltry fine regardless of the crime committed, community service or released free.

Furthermore, both human-wildlife conflict and poaching incidents escalated during the fast-track land reform programme (FTLRP) in Zimbabwe, which spanned more than a decade starting in the year 2000 (Gandiwa et al., 2013b). The FTLRP was also accompanied by severe economic hardships and human settlements encroaching on wildlife habitat. Thus, the CAMPFIRE programme experienced two major setbacks. The increase in poaching incidents seems to suggest that the CAMPFIRE programme failed to generate adequate incentives for local communities to conserve wildlife and that the FTLRP was disruptive, in the sense that it brought in settlers who were not interested in conservation. It is also hard to separate the economic incentives of CAMPFIRE from the general difficulties of the land reforms.

This phenomenon is not unique to Zimbabwe; it has occurred in many other countries in the Southern Africa region (Johannesen and Skonhoft, 2005). To this end, scholars argue that the impact of benefit-sharing schemes such as CAMPFIRE is limited by possible dilemmas in the actual design of the scheme or trade-offs inherent in linking development and conservation objectives (Wells et al., 1999; Fischer et al., 2011; and Johannesen and Skonhoft, 2014). These dilemmas are also closely intertwined with the nature of the community (i.e., the quality of local institutions that are in place) and the benefit-cost structure (incentives) associated with the property rights system.

It is the intention of most governments in the region to improve the living standards of poor rural households living adjacent to national protected areas through wildlife-based land reform. Given this, it is imperative to understand how different types of CBNRM regimes or conservation models work and to use this information not only to take appropriate action to enhance wildlife conservation in existing communal areas but also to undertake reforms to safeguard good stewardship practices in conservancies.[44]

Comparing how communities under CAMPFIRE operate with the conservancy community, we observe both similarities and striking differences in community attributes such as institutions, management and utilization of common pool resources. The differences in characteristics between the CAMPFIRE community and the conservancy could be responsible for driving the discrepancies in outcomes between the two communities. Table C.1 in the Appendix C summarizes some of these differences and similarities.

However, in this paper we will model only those key attributes that we believe matter for conservation and welfare. The main difference between the conservancy and CAMPFIRE community is that the former community is able to exercise anti-poaching enforcement, while the latter community can only engage in poaching. A community that derives benefits from wildlife conservation has adequate incentives to conservation wildlife. It is therefore rational for that community to abstracting from poaching and even engaging in anti-poaching activities. Anti-poaching effort exerted by the park agency and the conservancy has implications on the growth of the wildlife stock on communal and private land, and consequently on the welfare of the communities in question. In contrast, if a community lacks incentives to conserve wildlife, then it is unable to exercise anti-poaching enforcement. As a result, the community might chose to invest its effort in poaching rather than in developing

[44] While there is no statutory definition of a conservancy in Zimbabwe, the working definition is: “Any number of properties, which are amalgamated into a single complex in order to enable more effective management, utilization and protection of the natural resources” (Fitzgerald, 2012).

sound common pool resource institutions, which affects the community's conservation and welfare outcomes[45]. Excessive poaching could have a negative impact on the stock of wildlife on communal land and inside the national park due to the absence of a fence.

Since the park agency is the custodian of wildlife in the country, it cares about the stock of wildlife on both communal and private land, in addition to the stock inside the national park. Potentially, both communities will benefit if they are able to grow the stocks on private and communal land. Based on the knowledge and overall impressions about the community's conservation effort, the park agency plays a central role in deciding the harvest quota for each community. In the next section, we use a bioeconomic model to evaluate the behaviour of various actors in order to propose institutional changes that might move individual decisions closer to the social optimum. Basically, moving from a suboptimal level requires each of these agents to follow the commands of the social planner. Though they have different starting points, there are additional requirements that the social planner also consider in order to differentiate them. Hence, the difference between price taking and price making in this model constitutes one of the differences in institutions. Accordingly, resolving this problem will necessarily not require some market based instrument only, but also institutional reforms.

4.3. The bio-economic model

The analysis focuses on comparing conservation and welfare outcomes for two different communities involved in wildlife management adjacent a formal protected area: the CAMPFIRE community gets part of the proceeds from wildlife conservation which are distributed to them as cash transfers, while the conservancy community manages wildlife and generates revenues directly through hunting and tourism activities. There are three agents: the park agency, the conservancy and the CAMPFIRE community. We formulate optimization problems for each agent representing the baseline scenarios in terms of wildlife management (stock size and anti-poaching enforcement), and wildlife utilization (harvesting effort and subsistence poaching effort). We then compare the outcomes with those of the social planner

45 The park authorities are unable to carry out anti-poaching enforcement on communal land due to limited resources. As a result, local communities are now tasked to continue with the conservation work in their jurisdictions outside the formal protected areas. Anti-poaching enforcement under the CAMPFIRE programme is not effective because it is carried out on a part-time basis by either volunteers in the community (resource monitors) or the wildlife management committee, who usually face resource and time limitations, particularly during the agricultural season.

and suggest reforms required to induce the CAMPFIRE community to behave like the conservancy community, which produces better conservation and welfare outcomes.

We adopt the standard assumption of a homogenous community where decisions are made at a group level, i.e., a community can choose to put effort into either poaching or anti-poaching enforcement, depending on how it weighs the benefits and costs from wildlife conservation. This is a plausible assumption given the nature of decision-making we observe. Local communities use traditional institutions, which normally involve the chief or village headmen, where they meet under a tree and make decisions together as a group. Even though rebellion often occurs in the community, social norms help ensure the prevalence of a certain course of action by all members of the community. The conservancy community has committees and boards in place that make the crucial decisions on behalf of the group.

We assume that agents are managing a single wildlife species (e.g., African elephant) whose stock size is denoted by X_i, where the subscript $\{i = 0, 1\}$ denotes a patch of land. We agree that the issue of relative sizes of the park and various communities is an important consideration, but we do not think that our key results are dependent on relative size[46]. Given the ineffectiveness of the fence between the park and communal lands, we assume that wildlife on that patch is managed as one stock. Intuitively, one could envisage the stock leaving the national park and roaming on communal land during the agricultural season (being attracted by crops) and returning to the protected areas after the season. Let X_0 denote the stock of wildlife shared by the park agency and CAMPFIRE communities and X_1 the stock managed by the conservancy. The following additional implicit assumptions apply: $X_i(t) \geq 0$, $X_i(0)$ at time t=0 is given and $X_i(\infty) < \infty$, i.e., the stock of wildlife will not explode or grow toward infinity as time tends toward infinity because of the carrying capacity of the habitat. Please refer to the appendix for a summary of definitions of symbols and the functional forms used in this paper.

No hunting takes place inside the national park, but it is allowed in areas outside the park. Therefore, in the absence of natural growth, X_0 potentially shrinks when the stock roams on

[46] We abstract from reality and assume that the size of the park is independent of the stock of wildlife, poaching effort and anti-poaching enforcement, so that the variable does not affect our key results. This is a reasonable assumption to make because of the nature of the resource, i.e., its fugitive character.

communal land and is allowed to recover when it returns to the protected areas. Assuming a particular biomass at a specific point in time, $X_i(t)$, the stock grows according to natural growth $F(X_i,\cdot)$ and shrinks due to trophy hunting h_i and poaching $\psi_i(\cdot)$.

$$\dot{X}_i = F(X_i,\cdot) - h_i - \psi_i(\cdot) \qquad 4.1$$

Stock dynamics of wildlife roaming on communal land

$$\dot{X}_0 = F(X_0, L) - h_0 - \psi_0(L, T_0^p) \qquad 4.1(a)$$

Stock dynamics inside the conservancy

$$\dot{X}_1 = F(X_1, T_1^e) - h_1 - \psi_1(T_1^e, T_1^p) \qquad 4.1(b)$$

where L is anti-poaching effort exerted by the park agency, T_1^e denotes anti-poaching effort exerted by the conservancy, T_0^p represents the poaching effort employed by the CAMPFIRE community and $T_1{}^p$ represents the poaching effort of the local communities bordering the conservancy. We assume that the growth function depends not only on stock size, but also on anti-poaching effort, which facilitates growth of the wildlife stock. The natural growth function obeys the usual conditions:

$$\frac{\partial F(X_i,\cdot)}{\partial X_i} > 0, \frac{\partial^2 F(X_i,\cdot)}{\partial X_i^2} \le 0, \frac{\partial F(X_0, L)}{\partial L} > 0 \text{ and } \frac{\partial F(X_1, T_1^e)}{\partial T_1^e} > 0$$

In addition, we assume that $F(0,\cdot) = 0 = F(K,\cdot)$ so that there is no growth if the stock size is either zero or reaches the carrying capacity of the resources system (Fischer et al., 2011).

Unlike Fischer et al. (2011), who emphasized commercial poaching by outsiders but with assistance from the local community and anti-poaching enforcement by a few successful CAMPFIRE communities, we emphasize subsistence poaching activities by the majority of CAMPFIRE communities living adjacent to protected areas[47]. Our approach is in line with Johannesen and Skonhoft (2014). Accordingly, we assume that it is only the park agency that carries out anti-poaching enforcement both inside the park and in the communal areas adjacent to the protected area, but with limited resources at its disposal. The conservancy is

[47] Another point of departure between these two studies is that we are comparing two distinct CBNRM communities or habitats, while Fischer et al. (2011) looked at anti-poaching effort in the CAMPFIRE community

responsible for anti-poaching effort in its area. The CAMPFIRE community perpetrates poaching on communal land, while poaching inside the conservancy is carried out by the non-CAMPFIRE communities living adjacent to the SVC – this is motivated by the need to defray costs from the nuisance effect of wildlife (Johannesen & Skohoft, 2014), for selfish reasons, i.e., hunting for meat and trophies (Marks, 1984; Barrett & Arcese, 1998; Fischer et al., 2011) and also to protest the establishment of conservancies in areas viewed as traditionally belonging to local communities (Wels, 2000).

Poachers do not take into consideration the impact of their actions on the future stock of wildlife. It is natural to assume that the poaching function increases with poaching effort and decreases with anti-poaching enforcement exerted by the park agency and conservancy community, i.e.,

$$\psi_0(L,0)=0, \frac{\partial \psi_0(\cdot)}{\partial T_0^p} > 0 \text{ and } \frac{\partial \psi_0(\cdot)}{\partial L} < 0$$

$$\psi_1(T_1^e,0)=0, \frac{\partial \psi_1(\cdot)}{\partial T_1^p} > 0 \text{ and } \frac{\partial \psi_1(\cdot)}{\partial T_1^e} < 0$$

The second-order derivatives are such that the poaching function is concave with respect to poaching effort and convex with respect to anti-poaching effort. The marginal productivity of poaching effort decreases with anti-poaching enforcement.

$$\frac{\partial^2 \psi_i(\cdot)}{(\partial T_i^p)^2} \le 0, \frac{\partial^2 \psi_0(.)}{\partial L^2} \ge 0, \frac{\partial^2 \psi_1(.)}{\partial (T_1^e)^2} \ge 0, \frac{\partial^2 \psi_0(.)}{\partial T_0^p \partial L} = \frac{\partial^2 \psi_0(.)}{\partial L \partial T_0^p} < 0 \text{ and } \frac{\partial^2 \psi_1(.)}{\partial T_1^e \partial T_1^p} = \frac{\partial^2 \psi_1(.)}{\partial T_1^p \partial T_1^e} < 0$$

It is assumed that there is no relationship between the stock of wildlife in the conservancy and in the national park because the conservancy is enclosed such that wildlife cannot move across borders.[48] The conservancy purchased live animals only once, when it was established, in order to boost its wildlife stock, and thereafter restocking ceased. By contrast, the stock of wildlife on communal land is linked to the population of wild animals in the national park due to the absence of an effective fence. Thus, the wildlife stock on communal land replenishes itself because of its relationship with the park (i.e., wildlife is ordinarily harvested when roaming on communal land and recovers when it returns to the national park).

[48] Of course, in exceptional cases, the conservancy can purchase live animals from the national park when restocking is required due to exogenous forces like severe drought or floods.

Currently, property rights in wildlife belong to the state, both inside and outside the national park (Child, 1996; Murombedzi, 1999). Therefore, both the CAMPFIRE communities and the conservancy landowners have only use rights to wildlife. From this perspective, land tenure ceases to be an important variable in this analysis. For purposes of this model, we can assume that the property rights in wildlife "belong" to the park manager.

The park agency is responsible for allocating hunting quota h_i to other players in the wildlife sector. Let h_1 and h_0 be the quota allocated to the conservancy and CAMPFIRE communities respectively. We assume a modified version of the fixed quota rule[49]. In practice, this could be implemented as $h_i = \bar{h}_i + \varepsilon_i$, where average quota is adjusted on the basis of overall impressions about the community's conservation effort, rather than the actual stock size, because the park agency usually lacks vital information such as animal counts and trophy quality in the study area.

The state has the right to grant appropriation authority (this includes both the right to use and right to income) to any individual or community (Murombedzi, 2003). In the case of the CAMPFIRE programme, the state gave the appropriation rights to the RDC instead of the local communities directly. Therefore, the RDC collects the revenues and makes the decisions about how the proceeds from wildlife conservation are allocated. The RDC generates wildlife income by selling hunting licenses to safari operators who, in turn, sell the licenses at a premium (s) to clients from overseas. The RDC's gross wildlife income is given by the following expression:

$$W = W(h_0) = (P^* - s)h_0 \quad (4.2)$$

The parameter (s) can be interpreted as a premium charged by the safari operator above the fee paid to the RDC and includes his time spent looking for clients, time spent with clients during the actual hunting sessions, his skills, guns, etc. P^* is the fixed price per unit of harvest paid by the trophy hunters, and is exogenous to the communities because there is a competitive environment in the wildlife sector, such that no single community can influence the trophy price. Moreover, the fact that Zimbabwe is only one of the many countries offering

49 In practice, CAMPFIRE communities have generally complained that the quota does not change much and seems irresponsive to stock dynamics. This perception could be true, as quotas for selective or trophy hunting do not change much (Muchapondwa, 2003). Fischer et al. (2011) defined the fixed quotas rule $h_i = \bar{h}_i$, which reflects the behaviour of an agency that is understaffed and does not dare take new initiatives.

sport-hunting opportunities motivates the price-taking assumption (Fischer et al., 2011). A fraction of the income $(0 < \tau < 1)$ goes into the hands of local communities, while the remainder $(1-\tau)$ is retained by the RDC. Hence, the community receives

$$\tau W = \tau W(h_0) = \tau(P^* - s)h_0 \qquad (4.3)$$

The CAMPFIRE community is involved in different production activities, for example, agricultural production, poaching, anti-poaching enforcement and selling hunting licenses. Because the property rights of the park manager are not effectively enforced on communal land, the local people are not effectively prevented from illegally harvesting wildlife. The harvesting of wildlife resources takes place legally through trophy hunting and illegally through poaching. The legal harvesting of wildlife is not carried out by the respective CAMPFIRE communities themselves, but the RDC sell hunting licenses h_0 to the safari operators who, in turn, sell them to overseas clients who eventually utilise the quota.

Both communities allocate a fixed amount of effort $(\overline{T}_i)$ between the two activities, namely agricultural production (T_i^a) and wildlife activities (i.e., either anti-poaching enforcement or illegal harvesting). Assuming a binding time constraint, we have:

$$\overline{T}_i = T_i^a + T_i^j \qquad (4.4)$$

where the superscripts $\{j = p, e\}$ represent poaching effort T_0^p by the CAMPFIRE communities and anti-poaching enforcement T_1^e exerted by the conservancy.

For a fixed size of agricultural land and hence neglecting the possible loss of wildlife habitat through agricultural expansion (Johannesen & Skonhoft, 2014), the agricultural yield function in the absence of wildlife damage depends on effort when all other variable inputs are assumed to be fixed. The agricultural technology is given by:

$$A_i = A_i(T_i^a) = A_i(\overline{T}_i - T_i^j) \qquad (4.5)$$

The agricultural production function satisfies the usual concavity assumptions, i.e., $A(0) = 0$, $A'(.) > 0$ and $A''(.) \le 0$. More wildlife means more nuisances, so that damages are proportional to the amount of wildlife (Carlso & Wetzsten, 1994; Hueth et al., 1998), i.e.,

$$D_i \propto X_i \Rightarrow D_i = \beta X_i \qquad (4.6)$$

where $\beta \geq 0$ is a fixed constant and $D \in [0,1]$. Ideally, β captures the nuisance effect of the wildlife stock (see also Zivin et al., 2000). If the quantity of crops not damaged is $1 - D_i = 1 - \beta X_i$, then the net crop benefits are given by

$$Q_i = A_i(T_i^a) - A_i(T_i^a)\beta X_i = A_i(\overline{T}_i - T_i^j)[1 - \beta X_i] \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots (4.7)$$

Below we discuss the optimization problems of the three agents (i.e., the park agency, conservancy community and the CAMPFIRE community) and the social planner separately.

4.3.1 The park agency

We assume that the park agency gets most of its income from non-consumptive tourism and budget allocation from the state, but not from selling hunting licenses, because trophy hunting is not permitted inside the national park. The park agency employs a small fraction of its anti-poaching effort L outside the national park so that there is a small probability of being caught, $\theta_0(\cdot)$, if the community decides to harvest wildlife illegally. We assume further that the probability of detection is a function of poaching effort and anti-poaching effort of the park agency. The probability of being detected when hunting illegally is assumed to be an increasing function of the time spent hunting illegally, as well as the level of law enforcement exerted by the park agency, i.e., $\partial\theta_0(\cdot)/\partial T_0^P > 0$ and $\partial\theta_0(\cdot)/\partial L > 0$. In addition, we have $\theta_0(L,0) = \theta_0(0,T_0^P) = 0$. The marginal probability of detection increases with the level of anti-poaching effort $\partial\theta_0(\cdot)/\partial L T_0^P > 0$.

The park agency receives four types of benefits: i) budget from the state, $\overline{M}$; ii) revenue from benign tourism $R(X_0)$; iii) the public goods value of wildlife $G(\cdot)$; and iv) proceeds from poaching fines $\theta_0(\cdot)c_0$ imposed on detected perpetrators. It is assumed that $G(0) = 0, G'(\cdot) > 0,$ and $G''(\cdot) \leq 0$. The total cost of managing the park is given by $v_0 L$, where v_0 is the fixed cost per unit of anti-poaching effort. The park agency chooses hunting quotas to allocate to different communities and the level of anti-poaching effort to employ, which is split according to the land sizes inside and outside the park. The agency maximizes net benefits from different sources of income discussed above, subject to stock dynamics of wild animals shared by the park agency and CAMPFIRE community, the stock of wildlife roaming the conservancy, the budget constraints and the participation constraints of the two communities, where the discount rate is given by $\delta > 0$.

$$\underset{\{L,h_1,h_0\}}{Max}\ \pi_0(L,h_1,h_0) = \int_{t=0}^{\infty} [G^{PA}(X_0,X_1) + R(X_0) + \overline{M} + \theta_0(L,T_0^p)c_0 - v_0 L]e^{\delta t}dt \text{.................................}(4.8)$$

$$\begin{aligned} St:\ & \dot{X}_0 = F(X_0,L) - h_0 - \psi_0(L,T_0^P) \\ & \dot{X}_1 = F(X_1,T_1^e) - h_1 - \psi_1(T_1^e,T_1^P) \\ & v_0 L \le \overline{M} \text{ [Budget constraint - assumed to hold with equality]} \\ & P^* h_1 \ge \overline{U}_1 \text{ [Participation constraint for private game farms - assumed to hold with equality]} \\ & (P^* - s)h_0 \ge \overline{U}_0 \text{ [Participation constraint for local community - assumed to hold with equality]} \end{aligned}$$

As in Mukanjari et al. (2013) and Johannesen and Skonhoft (2014), the park agency bases its decision on inter-temporal considerations because it has the property rights to wildlife on both public and private land, and chooses both an optimal amount of effort toward anti-poaching activities and optimal quotas to give to CAMPFIRE communities and the conservancy. However, before park managers can calculate the quota, they need to know the wildlife stock in each community. The current value Hamiltonian is given by:

$$\begin{aligned} H(.) = {} & G^{PA}(X_0,X_1) + R(X_0) + \overline{M} + \theta_0(L,T_0^p)c_0 - v_0 L + \lambda[F(X_0,L) - h_0 - \psi_0(L,T_0^P)] \text{........................}4.8(a) \\ & + \mu[F(X_1,T_1^e) - h_1 - \psi_1(T_1^e,T_1^P)] + \Lambda_0[\overline{M} - v_0 L] + \Lambda_1[\overline{U}_1 - P^* h_1] + \Lambda_2[\overline{U}_0 - (P^* - s)h_0] \end{aligned}$$

with h_1, h_0 and L as the control variables; X_i as state variables; $\lambda > 0$ and $\mu > 0$ as shadow prices[50] for the shared stock X_0 and the stock inside the conservancy X_1 respectively; and Λ_i as Lagrange multipliers. The following expressions can be obtained from the first-order condition:

$$\frac{\partial \theta_0(L,T_0^p)}{\partial L} c_0 + \lambda \frac{\partial F(X_0,L)}{\partial L} = v_o(1 + \Lambda_0) + \lambda \frac{\partial \psi_0(\cdot)}{\partial L} \text{..}4.8(b)$$

According to Equation 4.8(b) above, the park agency employs anti-poaching effort until the benefits of stopping crime and growing the shared stock X_0 equals the value of reduced poaching plus the marginal cost of employing anti-poaching enforcement. Equations 4.8(c)

50 The shadow price measures the approximate decrease in the present value of net benefits resulting from a unit decrease in the wildlife stock.

and 4.8(d) tells us that the park agency allocates hunting quotas until the shadow price of the stock equals the market value.

$$\mu = -\Lambda_1 P^* \qquad 4.8(c)$$

$$\lambda = -\Lambda_2 (P^* - s) \qquad 4.8(d)$$

The portfolio conditions (Equation 4.8(e) and 4.8(f) below) indicates that the sum of the wildlife gain and the net stock effect resulting from maintaining one unit of wildlife must be equal to the marginal benefit of harvesting and putting the proceeds into the bank.

$$\dot{\lambda} + \frac{\partial G(\cdot)}{\partial X_0} + \frac{\partial R(X_0)}{\partial X_0} + \lambda \frac{\partial F(\cdot)}{\partial X_0} = \delta\lambda \qquad 4.8(e)$$

$$\dot{\mu} + \frac{\partial G(\cdot)}{\partial X_1} + \mu \frac{\partial F(\cdot)}{\partial X_1} = \delta\mu \qquad 4.8(f)$$

On the assumption of the functional forms reported in the appendix, the market equilibrium levels of anti-poaching effort by the park agency and the respective stocks of wildlife roaming inside the national park and on communal land and in the conservancy can be computed from the first-order conditions given above. The steady-state off-take of wildlife can be solved for by substituting the optimal wildlife stock and anti-poaching effort into the harvesting function. For comparison purposes, all solutions to the maximization problems presented in this analysis are shown at the end of this section.

4.3.2 The conservancy community

The private game farms employ T_1^a in agricultural production and anti-poaching effort T_1^e in order to grow the wildlife stock. Benefits enjoyed by the private game farms come from agricultural production $A_1(\cdot)$, selling hunting licenses h_1, revenues from tourism activities $R(\cdot)$ and proceeds from poaching fines $\theta_1(\cdot)c_1$ imposed on detected perpetrators on their land. Anti-poaching enforcement is costly, with v_1 as the fixed cost per unit of anti-poaching effort. It is assumed that revenue from non-consumptive tourism $R(X_1)$ increases with the stock of wildlife; that is, $R(0) = 0$, $\partial R(\cdot)/\partial X_1 > 0$ and $\partial^2 R(\cdot)/\partial X_1^2 < 0$.

The decision to be made by the conservancy community is how much anti-poaching effort T_1^e to invest in, while taking the off-take as given, because this is determined by the park agency through quota allocation h_1. We make a crucial assumption that the legal harvest can't exceed quota because it is guided by the quota. However, the overall harvest can exceed the quota, but the extra harvest is illegal. Because the conservancy community has appropriation rights (legal rights to exploit wildlife), they have a long-term view and therefore take the stock dynamics into consideration. For simplicity, we assume that the discount rate used by the park agency is the same as the discount rate used by the conservancy community and the social planner. Thus, the conservancy community's net benefits from agriculture and wildlife conservation are given by:

$$\underset{\{T_1^e\}}{Max}\, \pi_1(T_1^e) = \int_{t=0}^{\infty} [P_a A_1(\overline{T}_1 - T_1^e) + P^* h_1 + R(X_1) + \theta_1(T_1^e, T_1^p)c_1 - v_1 T_1^e] e^{\delta t} dt \text{..............................} 4.9$$

$$\begin{aligned} St:\quad & \dot{X}_1 = F(X_1, T_1^e) - h_1 - \psi_1(T_1^e, T_1^P) \\ & h_1 \le \overline{h}_1 \text{ [Harvesting cannot exceed quota]} \\ & \overline{T}_1 = T_1^a + T_1^e \end{aligned}$$

The production activities inside the conservancy are constrained by stock dynamics, the hunting quota and labour effort. The current value Hamiltonian is given by:

$$\begin{aligned} H(.) = {} & P_a A_1(\overline{T}_1 - T_1^e) + P^* h_1 + R(X_1) + \theta_1(T_1^e, T_1^p)c_1 - v_1 T_1^e + \mu[F(X_1, T_1^e) - h_1 - \psi_1(T_1^e, T_1^P)] \text{................} 4.9(a) \\ & + \Omega_1[\overline{h}_1 - h_1] \end{aligned}$$

The first-order condition with respect to anti-poaching effort is therefore given by:

$$\frac{\partial \theta_1(.)}{\partial T_1^e} c_1 + \mu \frac{\partial F(X_1, T_1^e)}{\partial T_1^e} = P_a \frac{\partial A_1(\cdot)}{\partial T_1^e} + v_1 + \mu \frac{\partial \psi_1(\cdot)}{\partial T_1^e} \text{..} 4.9(b)$$

Equation 4.9(b) tells us that the conservancy community will employ T_1^e until the benefit of catching a poacher and the value of growing the stock of wildlife as a result of a marginal increase in effort equals the value of losing agricultural harvest as a result of employing anti-poaching enforcement, the marginal cost of anti-poaching enforcement and the value of reduced poaching. In other words, the conservancy community allocates time toward anti-poaching activities until the loss from agriculture and anti-poaching equals the value of the growth in stock plus marginal benefits from collecting fines. According to equation 4.9(c), the conservancy community would maintain the stock of wildlife at a level that equates the return from trophy hunting and tourism activities with the return from alternative assets.

$$\dot{\mu} + \frac{\partial R_1(\cdot)}{\partial X_1} + \mu \frac{\partial F(X_1, T_1^e)}{\partial X_1} = \delta \mu \text{...} 4.9(c)$$

On the assumption of the functional forms shown in the appendix, the market equilibrium levels of anti-poaching effort by the conservancy community can be computed from the first-order conditions. The optimal wildlife stock is the same whether computed by the park agency or conservancy.

4.3.3 The CAMPFIRE community

We assume that the CAMPFIRE community takes both stock size and legal off-take h_0 as given. This is consistent with the behaviour we discovered in the survey, which suggests that community institutions are not strong enough to be proactive in conservation. The local community allocates its fixed endowment of labour effort $\overline{T}_0$ between two production activities, namely agriculture T_0^a and poaching T_0^p, with η_0 as the fixed per unit cost of poaching effort. As a result, there is a probability of being caught, denoted by $\theta_0(\cdot)$, if community members engage in illegal harvesting of wildlife resources. If caught, the community is levied a fixed fine $\theta_0(\cdot)c_0$, paid to the park agency.

As usual, the unit price of agricultural output P_a and illegal wildlife off-take ω are assumed to be fixed, where $\omega = P^* - \varepsilon$ and ε is the discount associated with illegal sales. It is reasonable to assume that the price of the illegal off-take could be far less that the market price $(i.e.\ \omega << P^*)$ because of institutional constraints associated with selling on the black market. Poachers sell their trophies at a lower price because they want to attract buyers and dispose of the trophies as quickly as possible to reduce the risk of being caught. The legal benefits from wildlife conservation $\tau(P^* - s)h_0$ are exogenous to the local community. We assume that τ is fixed over time and is decided on by the RDC at the beginning of the CAMPFIRE programme. In a way, this also removes the decision power of the RDC in deciding how much revenue to allocate to CAMPFIRE communities, where all elements are given and not at its discretion. The CAMPFIRE community is taxed twice, first by the safari operator, who charges a total commission of $\tau s h_0$ for the specialised services offered, and then a second tax amount of $(1-\tau)P^* h_0$ by the RDC for general programme administration.

We assume that the CAMPFIRE community maximise short-run gains. Johannesen and Skonhoft (2014) argue that this myopic behaviour is reasonable because, in most cases, the legal benefits from conservation going into the hands of local communities are too small relative to the cost of living with wildlife. Therefore, it is rational for the CAMPFIRE community to harvest as much as possible today, because of perceived risk and uncertainty in the future, i.e., they do not know if they may be effectively prevented from harvesting tomorrow due to, say, improved law enforcement. The issue of time preference or risk affect behaviour, decision marking and the planning horizon of CAMPFIRE communities. Future wildlife stock depends on the current wildlife hunting, which in turn hinges on the agent's time preference and planning horizon.

Furthermore, the benefits derived from wildlife resources are subject to uncertainty associated with production and market risks and variability of benefits from wildlife conservation might affect the decision of the community to exert more effort towards illegal harvesting of wildlife resource or to conserve the resource, depending on the attitude of agents towards risk and how the community perceive wildlife. Wildlife does not generate income for CAMPFIRE communities every hunting season. This depends on a number of factors. For example, the park agent might not to allocate a quota to a particular community, or the animals in a community might not be of good trophy quality. As already alluded to above, if the benefits are small relative to the costs, the community might find it reasonable to invest effort in poaching rather than conserve the resource.

Thus, the CAMPFIRE community chooses poaching effort in order to maximise current net benefits from agriculture and poaching shown in Equation (4.10) subject to the labour constraint.

$$\underset{\{T_0^p\}}{Max}\,\pi_2(T_0^p) = P_a A_0(\overline{T}_0 - T_0^p) + \tau(P^* - s)h_0 + \omega\psi_0(L, T_0^p) - \eta_0 T_0^p - \theta_0(L, T_0^p)c_0 \quad \text{..................4.10}$$

$$\overline{T}_0 = T_0^a + T_0^p \quad \text{..4.10}(a)$$

The first order conditions are given by

$$\omega\frac{\partial\psi_0(\cdot)}{\partial T_0^p} = P_a\frac{\partial A_0(\cdot)}{\partial T_0^P} + n_0 + \frac{\partial\theta_0(\cdot)}{\partial T_0^p}c_0 \quad \text{..4.10}(b)$$

The local communities employ T_0^P until the benefits from employing an additional unit of poaching effort equates to the loss in agriculture, the cost of poaching effort and marginal loss due to paying fines. On the assumption of the functional forms indicated in the appendix, the market equilibrium level of poaching effort by the local communities can be computed from the first-order conditions.

If benefits from wildlife conservation increase, then the welfare of the local community as a whole also increases. This could be achieved when the poaching effort exerted by the CAMPFIRE community is reduced and the population of wildlife on communal land increases, such that the quota allocation given to the local community also increases. Using proceeds from wildlife conservation, local communities around the GNP invest in public goods that benefit the society as whole, such as schools, clinics, electricity and grinding mills, rather than distributing the income to households. Investing in public goods could have a significant impact on community welfare.

4.3.4 The social planner's problem

The social planner chooses anti-poaching enforcement, poaching effort and hunting quotas in order to maximise the present value of net benefits from the activities of all the three agents (e.g., agricultural production, trophy hunting, tourism activities, state budget, the public good value of the wildlife stock and proceeds from poaching fines) subject to stock dynamics, budget, participation and harvesting constraints. The existence value or cultural value of the public goods might be different from what the park agency assumes. The social planner knows about the existence value φ that the local community places on the wildlife stock in their area and, hence, incorporates it in his valuation. The social planner is confronted with the following maximization problem:

$$\underset{\{L,T_1^e,T_0^p,h_1,h_0\}}{Max} \pi_3(.) = \int_{t=0}^{\infty} [G^{SP}(X_0,X_1) + \varphi X_0 + R(X_0) + \overline{M} + P_a A_1(\overline{T}_1 - T_1^e) + P^* h_1 + R(X_1) \ldots\ldots\ldots 4.11$$

$$+ \theta_1(T_1^e, T_1^P)c_1 + P_a A_0(\overline{T}_0 - T_0^p) + \tau(P^* - s)h_0 + \omega\psi_0(L, T_0^p) - \eta_0 T_0^p - v_0 L - v_1 T_1^e]e^{\delta t} dt$$

$$\dot{X}_0 = F(X_0, L) - h_0 - \psi_0(L, T_0^P)$$

$$\dot{X}_1 = F(X_1) - h_1 - \psi_1(T_1^e, T_1^p)$$

$v_0 L \leq \overline{M}$ [Budget constraint- assumed to hold with equality]

$P^* h_1 \geq \overline{U}_1$ [Participation constraint for private game farms- assumed to hold with equality]

$(P^* - s)h_0 \geq \overline{U}_0$ [Participation constraint for local community- assumed to hold with equality]

$h_1 \leq \overline{h}_1$ [Harvesting can not exceed quota]

$h_0 \leq \overline{h}_0$ [Harvesting can not exceed quota]

The current value Hamiltonian is, therefore, given by equation 4.11(a) where λ and μ are the shadow values (co-state variables) for the wildlife stock in the park, outside the park and in the conservancy.

$$\begin{aligned}H(.) = {} & G^{SP}(X_0, X_1) + R(X_0) + \bar{M} + P_a A_1(\bar{T}_1 - T_1^e) + P^* h_1 + R(X_1) + P_a A_0(\bar{T}_0 - T_0^p) + (P^* - s)h_0 \ldots\ldots 4.11(a) \\ & + \omega\psi_0(L, T_0^p) + \theta_1(T_1^e, T_1^p)c_1 - n_0 T_0^p - v_0 L - v_1 T_1^e + \lambda[F(X_0, L) - h_0 - \psi_0(L, T_0^P)] + \mu[F(X_1, T_1^e) - h_1 - \psi_1(T_1^e, T_1^p)] \\ & + \varphi X_0 + \Lambda_0[\bar{M} - v_0 L] + \Lambda_1[\bar{U}_1 - P^* h_1] + \Lambda_2[\bar{U}_0 - (P^* - s)h_0] + \Omega_1[\bar{h}_1 - h_1] + \Omega_2[\bar{h}_0 - h_0]\end{aligned}$$

The first-order conditions with respect to anti-poaching effort are given by:

$$\lambda \frac{\partial F(X_0, L)}{\partial L} = (\lambda - \omega)\frac{\partial \psi_0(\cdot)}{\partial L} + v_o(1 + \Lambda_0) \ldots\ldots 4.11(b)$$

$$\frac{\partial \theta_1(\cdot)}{\partial T_1^e} + \mu \frac{\partial F(\cdot)}{\partial T_1^e} = P_a \frac{\partial A_1(\cdot)}{\partial T_1^e} + \mu \frac{\partial \psi_1(\cdot)}{\partial T_1^e} + v_1 \ldots\ldots 4.11(c)$$

According to Equation 4.11(b), the social planner employs anti-poaching effort until the benefit of growing the stock in and outside the park equals the value of reduced poaching, plus the cost of employing the anti-poaching effort. Equation 4.11(c) says that the social planner will allocate labour between agriculture and anti-poaching enforcement in the conservancy until the benefit of catching a poacher and the value of growing the wildlife stock as a result of a marginal increase in anti-poaching effort equate to the value of losing agricultural harvest as a result of employing anti-poaching enforcement, the marginal cost of anti-poaching enforcement and the value of forgone poaching.

The first-order condition with respect to poaching effort is given by:

$$(\omega - \lambda)\frac{\partial \psi_0(\cdot)}{\partial T_0^P} = P_a \frac{\partial A_0(\cdot)}{\partial T_0^P} + n_0 \ldots\ldots 4.11(d)$$

The social planner would allocate labour between agriculture and poaching until the benefits of reduced poaching due to an additional unit of poaching effort (corrected for market distortions) equates to the loss in agriculture and the cost of poaching effort. Equation 4.11(e) and 4.11(f) states that the social planner will allocate hunting licenses until the shadow price and market price are equated, again correcting for market distortions.

$$\mu = P^*(1-\Lambda_1) - \Omega_1 \qquad 4.11(e)$$
$$\lambda = (P^* - s)(\tau - \Lambda_2) - \Omega_2 \qquad 4.11(f)$$

Equations 4.11(g) and 4.11(h) show the evolution of the co-state variables over time. The social planner would, therefore, maintain the wildlife stock at a level that equates the return from wildlife conservation with the return from alternative assets. The return from the stock of wildlife is in terms of the change in the marginal valuation of the stock and stock effects on revenue from wildlife tourism (in the case of wildlife stock in the conservancy) and natural growth of the wildlife stock.

$$\dot{\lambda} + \frac{\partial G(\cdot)}{\partial X_0} + \frac{\partial R(\cdot)}{\partial X_0} + \lambda \frac{\partial F(\cdot)}{\partial X_0} + \varphi = \delta\lambda \qquad 4.11(g)$$
$$\dot{\mu} + \frac{\partial G(\cdot)}{\partial X_1} + \frac{\partial R(\cdot)}{\partial X_1} + \mu \frac{\partial F(\cdot)}{\partial X_1} = \delta\mu \qquad 4.11(h)$$

The social planner's explicit solution can be computed from the first-order conditions using the functional forms assumed and presented in the appendix. From the maximization problems discussed above, one can solve for the optimal anti-poaching effort by the park agency and conservancy community, i.e. L^* and T_1^{e*}, respectively; poaching effort exerted by the CAMPFIRE communities, i.e. T_0^{p*}; the steady state wildlife stock, i.e. X_0^* and X_1^*; and the optimal quota allocation, i.e. h_1^* and h_0^*, from all the equations presented in section 4.3.1 through 4.3.4. Table 4.2 shows the comparison between the market equilibrium and the social planner's solution.

Table 4.2: Comparison of the market solution and the social planner's solution

Market Solution	**Sign**	**Social Planner**
$L^* = \left[\dfrac{\lambda\gamma b_0}{v_o(1+\Lambda_0)+\lambda b_0 - g_0 c_0}\right]^{\frac{1}{1-\gamma}}$	$\substack{\leq \\ >}$	$L^* = \left[\dfrac{\gamma\lambda b_0}{v_o(1+\Lambda_0)+\lambda b_0 - \omega b_0)}\right]^{\frac{1}{1-\gamma}}$
$X_0^* = K_0 \dfrac{\lambda(r-\delta)+2q}{2r\lambda}$	$<$	$X_0^* = K_0 \dfrac{\lambda(r-\delta)+2q+\varphi}{2r\lambda}$
$X_1^* = K_1 \dfrac{\mu(r-\delta)+q}{2r\mu}$	$<$	$X_1^* = K_1 \dfrac{\mu(r-\delta)+2q}{2r\mu}$
$h_1^* = F\lfloor X_1^* \rfloor - \psi_1(T_1^{e*}, T_1^{P*})$	$<$	$h_1^* = F\lfloor X_1^* \rfloor - \psi_1(T_1^{e*}, T_0^{P*})$
$h_0^* = F\lfloor X_0^* \rfloor - \psi_0(L^*, T_0^{P*})$	$<$	$h_0^* = F\lfloor X_0^* \rfloor - \psi_0(L^*, T_0^{P*})$
$T_1^{e*} = \left[\dfrac{\alpha\mu b_1}{P_a A_1 + v_1 + \mu b_1 - g_1 c_1}\right]^{\frac{1}{1-\alpha}}$	$=$	$T_1^{e*} = \left[\dfrac{\alpha\mu b_1}{P_a A_1 + v_1 + \mu b_1 - g_1 c_1}\right]^{\frac{1}{1-\alpha}}$
$T_0^{p*} = \left[\dfrac{\phi a \omega}{P_a A_0 + n_0 + g_0 c_0}\right]^{\frac{1}{1-\phi}}$	$>$	$T_0^{p*} = \left[\dfrac{\phi a(\omega-\lambda)}{P_a A_0 + n_0}\right]^{\frac{1}{1-\phi}}$

Source: own calculations 2015

4.4. Discussion of the results

In this section, we start by commenting on the differences in Table 4.2 and then narrow them down to the key roles of poaching and institutions in influencing biodiversity outcomes. We also consider some comparative statics, including the related policy implications, and numerical illustrations of the theoretical model.

4.4.1 Comparative statics

The results are consistent with theoretical expectations. The results above show that the stock of wildlife roaming in and outside the protected area, i.e., the shared stock X_0^* and the stock managed by the conservancy X_1^*, could be less than what the social planner would prescribe.

This also implies that the optimal harvest on communal land and in the conservancy is less than that of the social planner; see Table 2 above. The solution of anti-poaching enforcement by the park agency L^* is ambiguous. The market solution is greater than the social planner's prescription if $g_0 c_0 > \omega b_0$ and vice versa. The fact that the market solution differs from the social planner's outcome suggests that anti-poaching effort exerted by the park agency is suboptimal. Duffy (1999) reported some inefficiency associated with anti-poaching enforcement in Zimbabwe. Anti-poaching effort exerted by the park agency decreases with the cost of employing that effort, i.e., $\partial L^* / \partial v_o < 0$, while increasing anti-poaching enforcement increases the probability of being caught, i.e., $\partial L^* / \partial g_o > 0$. Given the latter result, it might be beneficial for the park agency to increase L in order to grow the shared wildlife stock X_0^*.

If the conservancy community values wildlife as much as the social planner does, then the level of anti-poaching enforcement T_1^{e*} exerted by the conservancy achieves social optimality, i.e., both the market and the social planner's solution are the same. This is the case when a market efficient outcome is equal to a socially optimal level. With this level of anti-poaching enforcement, poaching activities are kept at their lowest level and, hence, the stock inside the conservancy will grow. We argue that the off-take inside the conservancy is efficient because harvesting is determined by the quota set by the park agency, and poaching is contained through the employment of an efficient level of effort, which increases the probability of being caught, i.e., $\partial T_1^{e*} / \partial g_1 > 0$. Therefore, starting from a lower level of stock, if the anti-poaching effort exerted by the conservancy is both efficient and socially optimal, then this could drive the wildlife stock inside the conservancy toward optimality, provided that harvesting does not exceed the maximum sustainable yield.

The CAMPFIRE communities exert more poaching effort than the level the social planner would recommend. The role of poaching is to reduce the stock of wildlife when it is roaming on communal land. The off-take on communal lands is suboptimal, since harvesting is not only determined by the quota set by the park agency, but also by communities through poaching. Again, starting from a lower level, the wildlife stock in the community could diverge from the social planner's recommendation due to resource overexploitation. The differences between the market and the social planner's solutions are driven by externalities.

Given the fact that we are considering non-marketed goods, the market solution suffers from externalities, while the social planner takes externalities into account. To capture the deviation between the market and social equilibria, a parameter ω is included in the model to take into account the potential for the divergence to worsen under market equilibrium. Thus, as the price of the illegal harvest increases, the community increases its poaching effort (i.e., $\partial T_0^{p*} / \partial \omega > 0$) in order to increase net benefits. This behaviour could lead to overexploitation of wildlife resources on communal land as the community seeks to maximize net benefits.

The results show a negative relationship between poaching effort and the price of agricultural output $\partial T_0^{p*} / \partial P_a < 0$, the discount associated with illegal sales $\partial T_0^{p*} / \partial \varepsilon < 0$ and the probability of being caught $\partial T_0^{p*} / \partial g_0 < 0$. From the analysis, it is evident that the CAMPFIRE communities suffer a double tax; initially, the safari operator charges a commission S for the services rendered, and then the community loses a fraction, $1-\tau$, which goes to the RDC. Effectively, the price faced by the local community becomes $\tau(P^* - s)$, while the conservancy community gets P^*. Consequently, anything that deviates from the social planner's solution is not optimal and, thus, must be corrected.

The real result from this analysis is the uncovering of the policy instrument to improve outcomes in the CAMPFIRE communities. That policy instrument should not be imposed exogenously. We focus on policy because it is crucial on the conservation side through its effect on stock dynamics, and also crucial on the welfare side through its effect on economic benefits. The following policy interventions could potentially benefit the CAMPFIRE communities if they were to be implemented.

i. Reducing taxation on CAMPFIRE communities: From a policy standpoint, local communities would benefit if they could operate with the same self-sufficiency as the conservancy community because both taxes could be avoided. This could be achieved by hiring a manager or building internal capacity to match that in the conservancy community. The differences in the level of education between these two communities are revealing: the average number of years in school in the CAMPFIRE community is 7 compared to 15 in the conservancy.

ii. Reduce the price of illegal offtake: Likewise, a policy instrument that increases the risk premium ε could decrease the effective price of the illegal off-take and, hence, poaching effort, i.e., $\partial T_0^{p^*} / \partial \varepsilon < 0$. Reducing the effective price of the illegal off-take discourages the community from poaching by eroding the incentives, because $\partial T_0^{p^*} / \partial \omega > 0$. It is possible to integrate the risk premium into CPR institutions by carefully designing policy instruments that are adapted to local conditions.

iii. Appropriate institutional reforms: As reported earlier, the survey results for the CAMPFRE community point to weak institutions that are not supportive of proactive conservation. This is dramatically opposite to the behaviour observed in the conservancy community. As a result, it motivates us to explore whether an institutional reform in the CAMPFIRE community could move its welfare and conservation outcomes closer to those of the seemingly successful conservancy community. Because the CAMPFIRE community exerts more poaching effort in the market solution than the level that is socially optimal, we investigate the transition to social optimality by introducing an institutional variable which portrays a constraint on poaching behaviour. We argue that institutions affect biodiversity indirectly through constraining human behaviour (see chapter 3).

Thus, we introduce an institutional variable ρ, which enters the model through the poaching function $\psi_0(\cdot)$. For this purpose, we will consider a variable ρ that measures lack of cooperation, such that, when ρ is zero, the community has sound institutions and cooperation is also high. When ρ is one, then institutions are very weak and there is no cooperation in the community. Accordingly, $\rho = 0$ produces zero poaching, while $\rho = 1$ produces maximum poaching. Assuming the following explicit form, $\psi(.) = a[\rho T_0^P]^\phi - b_0 L, b_0 > 0 \; and \; \rho \in [0,1]$, the modified solution for poaching effort is thus given by:

$$T_0^{p^*} = \left[\frac{\rho^\phi \phi a \omega}{P_a A_0 + n_0 + g_0 c_0} \right]^{\frac{1}{1-\phi}}$$

Most importantly, improvement in institutions might have significant impact on growth of the wildlife stock through its role in constraining behaviour, i.e., $\partial T_0^{p*} / \partial \rho > 0$. Thus, constraining poaching effort might drive the stock shared by the park agency and CAMPFIRE communities toward the social planner's solution and avert a tragedy of the commons. The institutional variable ρ is a function of several other variables, such as governance, monitoring and enforcement, community level trust and endogenous punishment (see chapter 3). As a matter of policy, we want ρ to be a number which is low and very close to zero.

One way to iron out all issues with the current CAMPFIRE setup is to give local communities autonomy and to empower the wildlife management committees, so that they are able to effectively discharge their duties. This entails building local level institutions that will, in turn, set the community agenda on new social norms which are pro-conservation. For instance, an improvement in governance structures at the community level, monitoring and enforcement, and community level trust might contribute toward the attainment of a healthy biodiversity outcome as well as contain poaching activities. This could also be achieved through capacity building (institutional capacity) or training and funding to equip CAMPFIRE communities with much-needed resources. Moreover, if the community is allowed to endogenize punishment, then poaching might subside to socially optimal levels.

4.4.2 Numerical illustrations

The theoretical model will now be illustrated using functional form assumptions and data which fit well with the exploitation of the African elephant population by communities around the Gonarezhou ecosystem. African elephants are threatened by local communities because they cause more damage to agricultural crops than do other wild animals (Fischer et al., 2011). Each year, quite a significant proportion of elephants leaves the national park and visits the nearby communal areas during the agricultural season. Using MATLAB[51], we compute the optimal solutions from all the optimization problems presented above, and then proceed to show the stock dynamics as we vary anti-poaching effort and poaching effort and constrain the poaching effort while holding other variables constant. Model simulation was done using the following stock dynamic equation in discrete form.

[51] A MathWorks programming tool in MATLAB known as Simulink was used to do graphical programming, i.e., modeling or developing the algorithms, simulating and analyzing dynamic systems.

$$X_i(t+1) = X_i(t) + F(X_i,\cdot) - h_i(X_i(t)) - \psi_i(\cdot, T_i^P) \text{ given } \mathrm{X_i}(0) \text{ at time} t = 0 \text{..................} 4.12$$

Following Johannessen and Skonhoft (2000), we normalize the catch-ability coefficient η to one so that the Schaefer harvesting function becomes $h_i(t) = \sigma X_i(t)$. The harvesting effort now belongs to the interval $0 < \sigma < 1$ so that the off-take cannot exceed the available resources. The natural growth function is specified as shown in the appendix, and again we normalise the size of the stock by setting the carrying capacity equal to one, i.e., $K = 1$. The size of the wildlife stock (measured in biomass level) is thus expressed as a fraction of the carrying capacity and must be in the interval $0 \leq X_i(t) \leq 1$. Furthermore, the intrinsic growth rate r is set equal to 0.3 (Caughley & Sinclar, 1994; Johannessen & Skonhoft, 2000). In line with other studies, we also force both poaching and anti-poaching effort to lie between 0 and 1, i.e., $0 \leq L \leq 1, 0 \leq T_1^e \leq 1 \text{ and } 0 \leq T_0^p \leq 1$.

The model identifies three strategies (reducing taxation, reduce the price of illegal offtake and institutional reform) as part of the solution for sustainable wildlife management by CAMPFIRE communities. In an idea setting, we believe that the strategies should be deployed simultaneously. The model simulations are therefore based on the simultaneous deployment of these three strategies. If they were mutually exclusive and a choice of the most effective strategies was required, then the simulations would have to be conducted using a different model and data set.

The model simulation results confirm the theoretical predictions in section 4.3. In equilibrium, the anti-poaching effort by the park agency is less than the level of effort recommended by the social planner, while anti-poaching effort exerted by the conservancy community is just the same as that prescribed by the social planner. The poaching effort employed by the CAMPFIRE community is twice as much as that required for social optimality. The equilibrium stocks under the market solution are less than the social planner's solution.

Table 4.3: Numerical illustration – optimal solutions

Markert Equilibrium	Social Equilibria
$L^* = 0.6957$	$L^* = 0.6231$
$X_0^* = 0.5321$	$X_0^* = 0.5877$
$X_1^* = 0.5622$	$X_1^* = 0.5901$
$T_1^{e*} = 0.5992$	$T_1^{e*} = 0.6012$
$T_0^{P*} = 0.5183$	$T_0^{p*} = 0.2640$

Source: model simulation results

Figure 4.1 shows the changes in the next period stock (i.e. for the shared stock) as we vary anti-poaching enforcement exerted by the park agency between 0 and 1 while holding other variables constant. The figure shows that size of the wildlife stock on communal land increases as the park agency increases anti-poaching effort up to a certain point, and later on stabilizes at a slightly lower level than the social planner's recommendations. The gap between the market solution and the social planner's prescription does not completely iron out due to resource limitations. The numerical illustrations show that anti-poaching enforcement might grow the stock on communal land up to a certain level of effort $L^* = 0.62$ that is socially optimal, beyond which the stock ceases to grow due to other factors beyond the park agency's control.

Figure 4.1: Market solution for the park agency versus the social planner

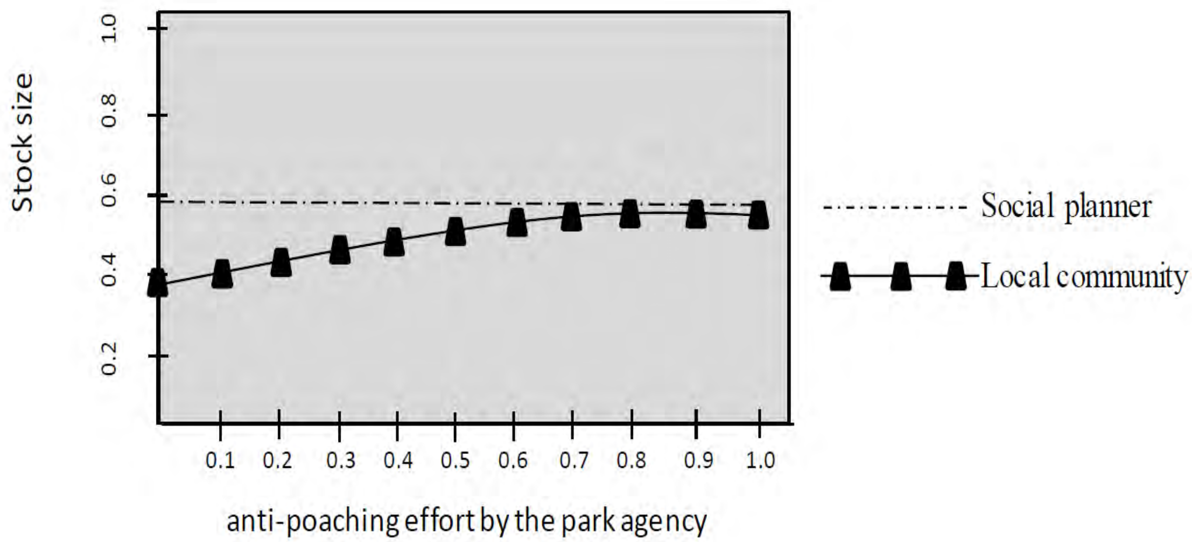


Figure 4.2 shows the changes in the next period stock inside the conservancy as we vary anti-poaching effort exerted by the conservancy community between 0 and 1 while holding other variables constant. The diagram shows that if anti-poaching effort by the conservancy community is optimal, then the stock size prescribed by the social planner and the market solution will eventually coincide with each other. This is the case when market solution is equal to social optimality. The convergence of the two solutions is very fast in the case of the conservancy community.

Figure 4.2: Market solution for the conservancy community versus the social planner

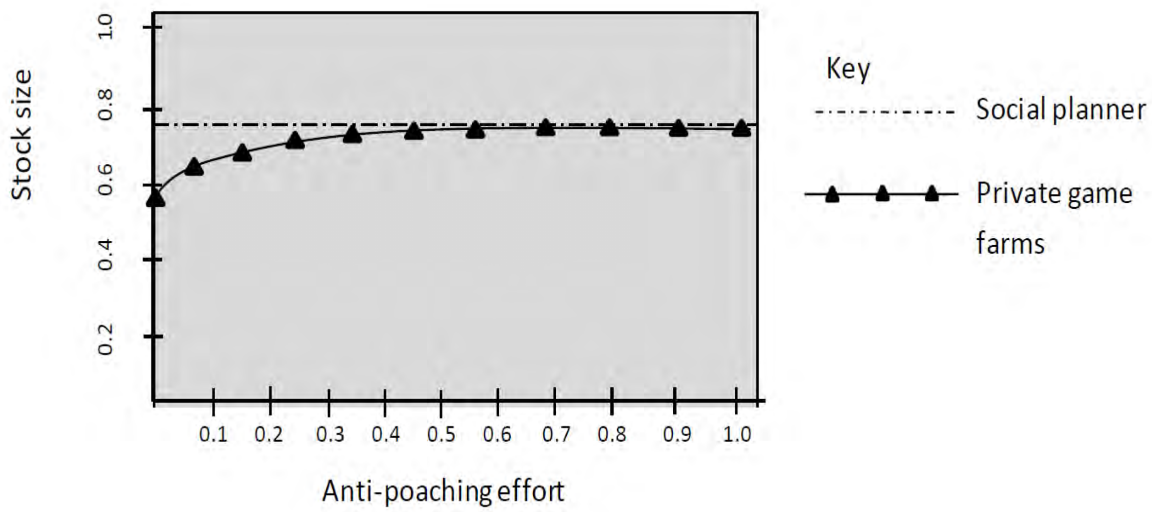


Figure 4.3 shows the changes in the next period stock (i.e., shared stock) as we vary the poaching effort exerted by the CAMPFIRE community between 0 and 1 while holding other variables constant. The figure shows that the stock size on communal land diverges from the social planner's solution if poaching continues unabated. Initially, the wildlife stock outside the park increases with very low levels of poaching up to a certain point (about 0.26), then starts to decrease tremendously. If local communities continue to increase the level of poaching effort beyond this point, then this could drive the resource system toward economic or physical extinction. Beyond a certain level of stock, again, the wildlife stock will not be able to regenerate itself without human intervention.

Figure 4.3: Market solution of CAMPFIRE community versus the social planner

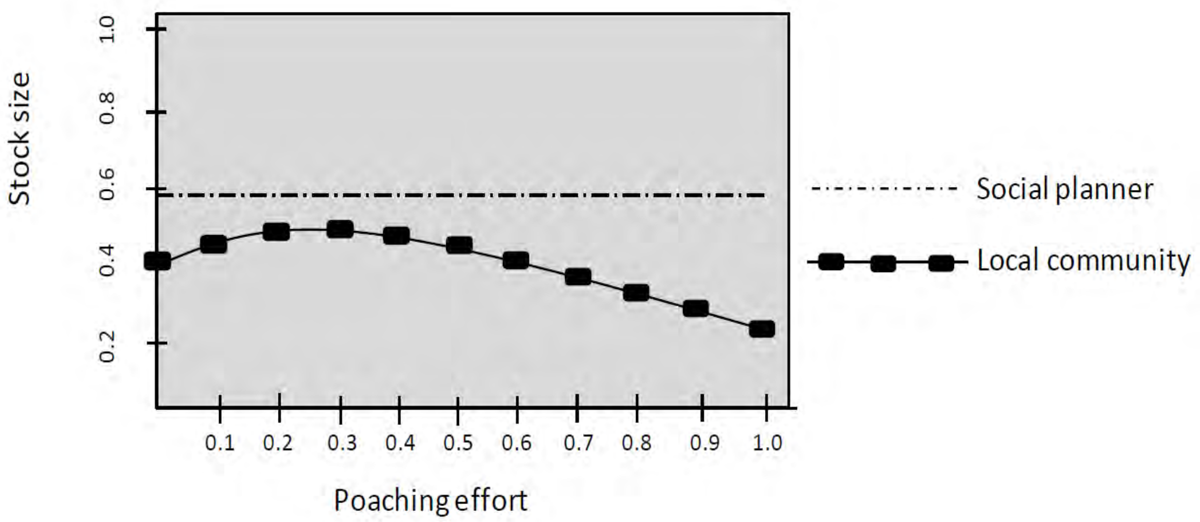
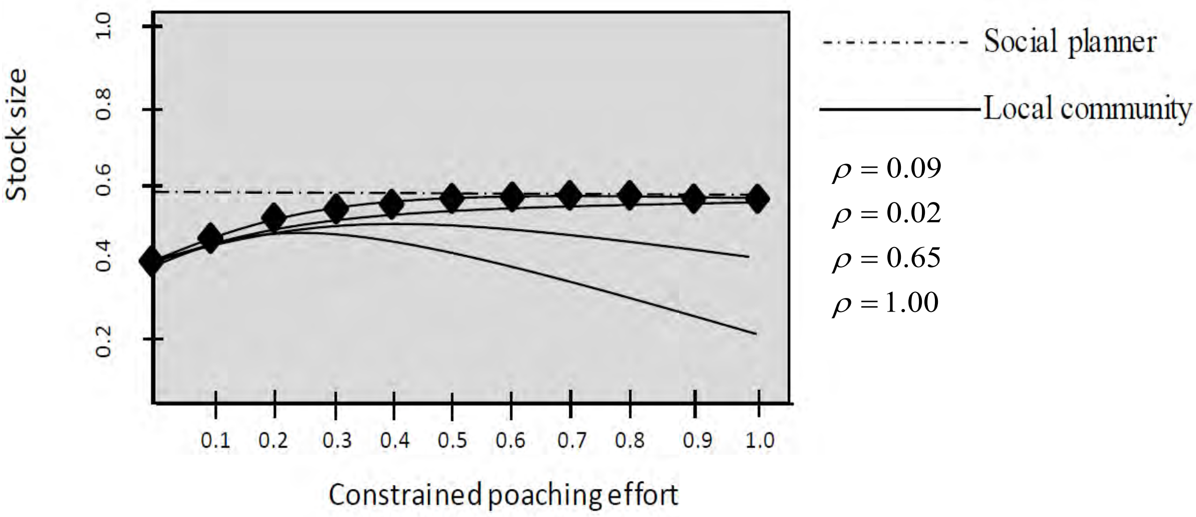


Figure 4.4 shows the changes in the next period stock of wildlife (i.e., shared stock) as we introduce a variable for institutions and vary constrained poaching effort exerted by the CAMPFIRE community between 0 and 1 while holding other variables constant. Starting from a lower level, the stock size in the local community's conservation area grows in a nonlinear fashion[52] until the solution coincides with the prescriptions of the social planner, if the poaching effort is constrained. Ideally, we would want parameter ρ to be some number which is close to zero for faster convergence. The figure below shows that, starting with $\rho = 1$ (i.e., with the situation in Figure 4 above), an improvement in institutions, such that the parameter rho is forced toward zero $(i.e.\ \rho \to 0)$, means that the shared stock will mimic the dynamics of the wildlife stock inside the conservancy. The sensitivity analysis demonstrates how stronger institutions help to bridge the divide between the market and social planner's solution more quickly than do weaker institutions. The graph below is drawn for the different values of rho, i.e., ρ =1.00, 0.65, 0.02, 0.09 and $\phi = 0.3$.

[52] Non-linearity occur when the relationship between variables is not simply static, direct or exact, for instance, y = ax + b. This is a property of chaotic systems, characterized by random behavior and uncertainty such as in modelling extraction (growth) of natural resources or a social ecological system. If the resource base features non-lenearities, which is highly likely, that might also give rise to non-linearities in the growth path of the resource. Non-linearity and uncertainty requires pro-active and adaptive management rather than reactive management because we don't know, *a priori*, what will happen if the resource system is subjected beyond its limit, e.g., harvested beyond its threshold. We also don't know what will happen to the ecological balance if one or more species were to disappear from the system.

Figure 4.4: Market solution of local communities (with institutions) versus the social planner



4.5. Conclusions and policy implications

Integrated Conservation and Development Projects are central for future rural development in Southern Africa. However, the impact of benefit-sharing schemes, such as the CAMPFIRE programme in Zimbabwe, is limited by possible dilemmas in the actual design of the scheme or trade-offs inherent in linking development and conservation objectives. The objectives of this paper are to compare wildlife management and utilization under the CAMPFIRE communities and conservancy community, and to consider the possibility of wildlife tenure and institutional reforms that might replicate conservancies' successful outcomes on communal areas implementing CAMPFIRE. Therefore, unlike previous studies, this paper seeks to establish the conditions under which a CAMPFIRE community can be incentivized to behave like the conservancy community, which is more successful in revenue generation and stewardship practice.

To achieve the objectives above, we used a bio-economic model. We developed and compared the problems for benefit-sharing arrangements under CAMPFIRE and the conservancy communities operating adjacent to Gonarezhou National Park. Firstly, the chapter demonstrated that the conservancy community is superior to the pure benefit-sharing scheme in terms of employment of effort and the long-run wildlife stock. Secondly, the chapter analysed wildlife management and utilization under the assumption that the communities in question are given a greater degree of autonomy so that they are able to invest in stronger CPR institutions.

The results show that the level of anti-poaching enforcement by the park agency could be lower than the social planner's prescription. It might not be optimal for the park agency to provide anti-poaching enforcement inside the national game park and in communal areas. This result seems to support policy or institutional reforms that convey greater control of natural resources through devolution and decentralization of NRM functions, and decision-making to the community's grass roots level, since the community incurs lower cost of monitoring and enforcement.

The social planner recommends higher levels of wildlife stock in the conservancy and on communal land, i.e., shared stock. If the conservancy community values wildlife to the same degree as does the social planner, then their level of anti-poaching enforcement achieves social optimality. This could drive the wildlife stock in the conservancy toward the social planner's solution, starting from a lower stock level. CAMPFIRE communities exert more poaching effort than what the social planner would recommend. As a result, the size of the shared stock might diverge over time from the social planner's prescription, starting from a lower level.

Because both the CAMPFIRE and conservancy communities are carrying out similar activities, they should potentially be able to achieve similar results. The differences in observed outcomes between them could be a result of the differences in community institutions. The results confirm that an improvement in community institutions might have significant impact on growth of the wildlife stock through its role of constraining behaviour. Resolving the problem faced by CAMPFIRE communities will necessarily not require market based instrument alone, but also institutional reforms. This study calls for three main strategies to allow campfire communities to move from a seemingly inferior outcome to one that is optimal.

Firstly, this result calls for policy instruments that will facilitate the development of sound CPR institutions that are tailored to suit local conditions and endogenous to the community. Specifically, to strengthen the incentives in CAMPFIRE communities, we propose that the

RDC should transfer wildlife management functions and benefits to sub-district producer communities. This also implies that government policy should aim at building local level institutions that will, in turn, set the tone for community agenda on new social norms which are pro-conservation. This could be achieved through capacity building, investment in institutional building blocks such as governance structures at the local level, democracy, monitoring and enforcement, and community level trust, and funding to equip CAMPFIRE communities with much-needed resources.

Since wildlife is a fugitive resource, there is also need for CAMPFIRE communities to coordinate their effort and resources in order to supply the required habitat and effort to fight illegal harvesting of wildlife resource on a broader scale. This calls for CAMPFIRE communities to invest in collective action beyond the borders of each project or community in the form of multi-layered enterprises or organizations that are endogenous to the programme. The government should therefore allow CAMPFIRE projects to be innovative so that they can learn from each other, through their past experiences and mistakes, and to develop and experiment with different models of conservation.

Secondly, CAMPFIRE communities suffer double taxation from the premium charged by the safari operators and a significant proportion of wildlife income that remains in the hands of the Rural District Council. From a policy standpoint, CAMPFIRE communities would benefit immensely if they could learn from the model of the conservancy community and operate with the same autonomy and self-sufficiency as the conservancy because both taxes could be avoided. This could be achieved by hiring a manager or building internal capacity to match that in the conservancy community. This implies a different model or an improvement of the current benefit-sharing scheme so that CAMPFIRE communities can mimic the business model or behaviour of the conservancy community and in the process receive full benefits from conservation and pay taxes to the state and levies to the Rural District Council, instead of receiving cash transfers. Innovative policy instruments are required to assist CAMPFIRE communities in various activities of wildlife conservation so that they can integrate into the main stream economy and fully commercialize their operations.

Thirdly, designing policy instruments that increases the risk premium could decrease the effective price of the illegal off-take. Reducing the effective price of the illegal off-take discourages the community from poaching by eroding the incentives. It is possible to integrate the risk premium into local institutions by carefully designing policy instruments that are adapted to local conditions. We recommend that these three strategies should be deployed simultaneously for an effective and fast solution to the challenges faced by CAMPFIRE projects. This implies that policy makers should use a combination of both market-based instruments and institutional reforms since they seem to complement each other.

Result show that the employment of anti-poaching enforcement by the park agency is suboptimal. As a matter of policy, it is optimal for the park agency to withdraw anti-poaching effort from the local community and offer regulatory services. Withdrawing effort from the community will allow the park managers to use the limited resources effectively and efficiently on a relatively smaller area. The finding support policies that favour increased devolution or decentralisation of natural resource management functions, decision making and authority to the community's grassroots level. We recommend that the park agency should leave the society to its own device so that it can develop its own solutions to the environmental problems faced.

Chapter 5

Conclusions and Policy Implications

5.1. Summary of the findings

This thesis examined the economics and institutional aspects of community wildlife conservation in the context of local communities living adjacent to the Gonarezhou National Park in Zimbabwe. Several challenges threaten conservation efforts at both local and higher levels. This impedes its ability to bring about development that might improve the welfare of poor rural communities participating in wildlife conservation. The most pressing issues in the wildlife sector include; i. the inability to extract resource rents from wildlife conservation that in turn affects household welfare in terms of total household income, and reduction in poverty and inequality, ii. the lack of capacity by local communities to solve collective action problems, and iii. the lack of comparable successful outcomes in CBNRM communities such as the wildlife conservancy communities.

The thesis contributes to the broad research agenda on the human-environment relationship by analysing the issues highlighted above in three separate papers each corresponding to a thesis chapter. The first paper presented in chapter 2 analyses the effects of wildlife resources in the portfolio of environmental income on household welfare. The paper seeks to enhance our understanding of the relationship between wildlife income in the portfolio of environmental income versus poverty and inequality. The results show that the contribution of environmental income (including wildlife) to total household income is quite substantial. Wealthier households consumed more non-wildlife resources and wildlife products in total than relatively poor households. However, poorer households derived greater benefit from the consumption of both wildlife and non-wildlife resources than wealthier households. Excluding wildlife from the analysis compromised the relative contribution of environmental resources, while at the same time increased the relative contribution of farm and wage income. Environmental income (with and without wildlife) had more impact in terms of poverty reduction in the lower income quintiles than in the upper quintiles. Wildlife income alone accounted for about 5.5% reduction in the proportion of people living below the poverty line. Furthermore, wildlife income had an equalising effect, bringing about a 5.4% reduction in measured inequality.

The regression results revealed that the likelihood of belonging to a wealthier category of income increased with an increase in environmental income. However, the marginal effect of environmental income is very small. It suggests that only households that are positioned on the boundary will be able to move to the next income quintile because of the increase in environmental income. Its impact may be less pronounced for households that are located further away from the boundary. As expected, household wealth significantly and positively affect environmental income generated by households. Evidence of the relationship between benefits and quality of the resource system suggest that households generated more environmental income in areas with good biodiversity than in areas where there is an unhealthy population of wild animals. It is actually a good outcome that poor people benefit from wildlife conservation in the study area. However, conservation outcomes in the CAMPFIRE communities fall short of expectations. Therefore, there was need to investigate the drivers of conservation.

The second paper presented in chapter 3 analyses the role of institutions in community wildlife conservation in Zimbabwe, based on Ostrom's framework for analysing complex social ecological systems. The paper used two regression models to examine the association between institutions and cooperation (defined as the ability to self-organise) and the relationship between cooperation and success of biodiversity outcomes. Overall, the results demonstrated that institutions are indeed an important ingredient for cooperation and that cooperation is necessary for the success of biodiversity outcomes. Therefore, the story supported in this paper is that biodiversity conservation is forged by cooperation and cooperation is directly driven by institutions.

Other important variables explaining cooperation in the first model were community level trust, number of stakeholders and punishment. Cooperation was higher in those communities where the level of trust was higher, in communities that had more stakeholders involved in wildlife conservation and in cases where punishment was endogenised by the community. In the second model; training, benefits, distance from the nearest urban centre, distance from the park fence, social capital average age of household head, new fence and information sharing were also found to be important variables explaining the success of biodiversity.

From a theoretical point of view, it is agreed that institutional attributes, such as monitoring and enforcement, fairness, governance and democracy, are important ingredients for the community's ability to self-organisation and to manage natural resources sustainably. The third paper presented in chapter 4 uses bio-economic modelling with numerical illustrations to confirm theoretically the importance of institutions for community wildlife conservation in the context of poor rural communities living adjacent to the GNP in Zimbabwe. To be able to deal effectively with institutions in the model, there was a need to talk about a different type of community, other than the CAMPFIRE community. Thus, for the purposes of this study, we chose the the Save Valley Conservancy. Both communities are conserving wildlife in a community-based fashion, but their level of success is different – the performance of the CAMPFIRE community is much worse.

The paper formulated bio-economic models for the park agency, the conservancy and CAMPFIRE communities that represent the baseline scenarios in terms of wildlife management, wildlife utilisation, poaching effort and anti-poaching enforcement, and compare the outcome with the social planner's solution. We then proceed to suggest institutional reforms for the CAMPFIRE community so that they behave like the conservancy community in the Save Valley Conservancy (SVC).

The results show that the level of anti-poaching enforcement by the park agency is suboptimal, while anti-poaching enforcement exerted by the game farming community in the conservancy achieves social optimality. CAMPFIRE communities exert more poaching effort than what the social planner would recommend. Our model shows that an improvement in community institutions might have a significant impact on growth of the wildlife stock through their role in constraining behaviour. Thus, institutional reforms in benefit-sharing schemes such as CAMPFIRE could see the local community behaving like the game farming communities such as the one in the Save Valley Conservancy.

5.2 Policy implications of the research findings

The results of this study help to shed light on the link between wildlife income (in the context of environmental income) and welfare, the processes explaining complex social-ecological systems and suggest appropriate institutional reforms for the CAMPFIRE programme. The intended beneficiaries of the study are local communities, wildlife agencies, development practitioners and policy makers who will acquire relevant and empirically grounded evidence. Our research findings will allow them to interrogate their wildlife management strategies and policies while at the same time identifying areas that need to be improved. A number of policy implications obtain from the analysis in this thesis.

Chapter 2: Effects of Wildlife on Community Welfare: Income Poverty and Inequality

The main policy message to take home in Chapter 2 is that, wildlife conservation has an important role in mitigating poverty and income inequality in CAMPFIRE communities. Several other policy implications can be drawn from the analysis in this chapter. Reduced access to wildlife income could have substantial impact on welfare in CAMPFIRE communities and might potentially increase poverty and inequality in the study area. There is therefore a need to design policies that increase access to wildlife income by poor rural households living adjacent to national parks because this could have an impact on their welfare. To avoid further marginalization of the poor, attention to equity in resource management and access to resources should be a prime consideration, particularly with valuable resources such as wildlife. Increased devolution of wildlife management function from the Rural District Council into the hands of local communities could allow more access to wildlife income and potentially contribute towards reducing poverty and inequality. The relationship between environmental income and wealth has some interesting policy implications given that wealthier households accumulate more assets with which to harvest more environmental resources. This implies that wildlife-based land reform also needs to empower poor households in the area of capital accumulation while imposing restraint on harvesting by well-off households.

In line with our goal of using wildlife resources to bring about community welfare, we believe that policy experiments of increasing wildlife income could highlight the importance of wildlife resources for redistributive policy targeting. The results illustrate that increases in wildlife income can greatly impact poverty, but not inequality because inequality is less responsive to policy induces increments in wildlife income. Thus policies which seek to reduce poverty in rural communities should consider increasing access to wildlife income by these communities. The results demonstrate that a policy induced increment in wildlife income of slightly more than 15% is needed to reduce poverty in the whole population to zero.

There is need for a policy shift away from the resource itself to encompass the broader local economy in order to enhance livelihoods. Quick policy interventions such as formal and informal employment in the craft industry are required to increase household earnings from wildlife conservation in the short to medium term. In the long-run, household welfare in CAMPFIRE communities could benefit through policies and programs that stimulate an increase in wildlife earnings from activities such as non-consumptive tourism, tourism business ventures such as hotels or accommodation, investment in infrastructure, filming and live animal sales.

Chapter 3: The Role of Institutions in Community Wildlife Conservation in Zimbabwe

A number of policy implications obtain from this analysis in Chapter 3. To begin with, the wildlife sector should be re-configured in order to favour the implementation of collective strategies that are endogenous to the community aimed at providing public goods such as wildlife through conservation. Government policy should recognize CAMPFIRE communities as important stakeholders with an important role to play, but not as mere beneficiaries of wildlife conservation simply because they live, interact with wildlife almost on a daily basis and suffer in the process. There is therefore need for future policy to define the roles of each stakeholder and delineate boundaries within which each stakeholder can operate.

Future policy reforms should also consider further devolution of natural resource management function from the Rural District Council to CAMPFIRE communities or increased autonomy so that they community members can monitor each other and internalise enforcement, while the state maintains regulation functions. This will allow innovation among communities, while the government create an enabling environment for other stakeholders to operate. For example, the capacity of CAMPFIRE to develop their own rules, rather than just following externally-imposed rules, should not be undermined. The results show that external enforcement of rules and regulations does not necessarily translate into sound ecological outcomes; rather, better outcomes are attainable when punishment is endogenized by local communities. This seems to suggest that communities should be supported in a way that promotes the emergence of robust institutions that are tailor-made to suit local needs. This will, in turn, facilitate good environmental husbandry.

Since wildlife is a fugitive resource that roams freely on communal land, there is therefore a need for coordination among CAMPFIRE communities in order to supply the public good, the required habitat and to benefit from collective. The government, NGOs and private sector should act as catalyst of collective action to ensure that there are incentives for CAMPFIRE communities to self-organize into multiple layers of nested organizations through policies and programmes that allow resource users to participate in decision making process. Development programmes or policy interventions with both a welfare and a conservation component should not be designed as 'one size fits all' but should recognize and understand the differences in community characteristics and incentives to self-organize through policy and development programmes in order to promote conservation and safeguard the livelihood interests of pro-poor communities.

Government programmes can target capacity building in terms of institutional capacity and skills development in order to have a positive impact on biodiversity conservation. Increase spending on training relevant to natural resource management, and particularly wildlife conservation, targeting not only community leadership or project committees but also ordinary community members as a path way to conservation. Therefore, the capacity building efforts of government agencies, NGOs and other stakeholders should complement each other to ensure that the necessary resources are mobilised and all communities receive the necessary training and resources. State authorities should reconsider the way in which they engage with farmers under resettlement schemes, because biodiversity suffers more under this type of arrangement than in communal areas. There is a need for appropriate institutional reforms that allow park authorities to work closely with resettlement schemes, while at the same time giving incentives for plot holders to work together for the improvement of the common pool resource. For example, farmers in resettlement schemes could set aside land for conservation by pooling resources instead of operating at plot level, which works against conservation efforts.

Chapter 4: A Bioeconomic Model of Community Wildlife Conservation in Zimbabwe

Resolving the problem faced by CAMPFIRE communities will necessarily not require market based instrument alone, but also institutional reforms. This study calls for three main strategies to allow campfire communities to move from a seemingly inferior outcome to one that is optimal.

Firstly, this result calls for policy instruments that will facilitate the development of sound CPR institutions that are tailored to suit local conditions and endogenous to the community. Specifically, to strengthen the incentives in CAMPFIRE communities, we propose that the RDC should transfer wildlife management functions and benefits to sub-district producer communities. This also implies that government policy should aim at building local level institutions that will, in turn, set the tone for community agenda on new social norms which are pro-conservation. This could be achieved through capacity building, investment in institutional building blocks such as governance structures at the local level, democracy, monitoring and enforcement, and community level trust, and funding to equip CAMPFIRE communities with much-needed resources.

Since wildlife is a fugitive resource, there is also need for CAMPFIRE communities to coordinate their effort and resources in order to supply the required habitat and effort to fight illegal harvesting of wildlife resource on a broader scale. This calls for CAMPFIRE communities to invest in collective action beyond the borders of each project or community in the form of multi-layered enterprises or organizations that are endogenous to the programme. The government should therefore allow CAMPFIRE projects to be innovative so that they can learn from each other, through their past experiences and mistakes, and to develop and experiment with different models of conservation.

The results show that the employment of anti-poaching enforcement by the park agency is suboptimal. As a matter of policy, it is optimal for the park agency to withdraw anti-poaching effort from the local community and offer regulatory services. Withdrawing effort from the community will allow the park managers to use the limited resources effectively and efficiently on a relatively smaller area. The finding support policies that favour increased devolution or decentralisation of natural resource management functions, decision making and authority to the community's grassroots level. We recommend that the park agency should leave the society to its own device so that it can develop its own solutions to the environmental problems faced.

Secondly, CAMPFIRE communities suffer double taxation from the premium charged by the safari operators and a significant proportion of wildlife income that remains in the hands of the Rural District Council. From a policy standpoint, CAMPFIRE communities would benefit immensely if they could learn from the model of the conservancy community and operate with the same autonomy and self-sufficiency as the conservancy because both taxes could be avoided. This could be achieved by hiring a manager or building internal capacity to match that in the conservancy community. This implies a different model or an improvement of the current benefit-sharing scheme so that CAMPFIRE communities can mimic the business model or behaviour of the conservancy community and in the process receive full benefits from conservation and pay taxes to the state and levies to the Rural District Council, instead of receiving cash transfers. Innovative policy instruments are required to assist CAMPFIRE communities in various activities of wildlife conservation so that they can integrate into the main stream economy and fully commercialize their operations.

Thirdly, designing policy instruments that increases the risk premium could decrease the effective price of the illegal off-take. Reducing the effective price of the illegal off-take discourages the community from poaching by eroding the incentives. It is possible to integrate the risk premium into local institutions by carefully designing policy instruments that are adapted to local conditions. We recommend that these three strategies should be deployed simultaneously for an effective and fast solution to the challenges faced by CAMPFIRE projects. This implies that policy makers should use a combination of both market-based instruments and institutional reforms since they seem to complement each other.

5.3 Suggestions for future research

In terms of the direction for future research, the analysis in this thesis suggests that further understanding of the role of local institutions in community wildlife management requires a cross-country analysis – i.e., a comparison of common pool resource institutions in two or more countries where communities are involved in wildlife conservation. Cross–country studies are also needed to enhance our understanding of the effects of wildlife income on household welfare, especially for those households living adjacent to national protected areas. In particular, it has become rather imperative to recognize the relative contribution of different income sources in the portfolio of environmental income in driving inter-household poverty and inequality.

The issue of time preference affect the behaviour, decision making process and the planning horizon of local communities that are involved in managing common pool wildlife such as CAMFIRE communities. The future stock of wildlife depends on the current wildlife hunting, which in turn hinges on the agent's time preference and planning horizon. The issue of benefits is closely related to the issues of time preference and planning horizon. These are topical issues currently receiving attention in behavioural economics. The issues of time preference and planning horizon are very relevant and deserve to be studied further. So we suggest these for future work.

The success of CAMPFIRE projects and biodiversity conservation does not only depend on community characteristics but also on the external environment, which includes government policy, the economic environment (local, national and global) and other stakeholders. The external environment is a black box that needs to be unpacked. For future research we recommend studies to look at the role of these variables and how they simultaneously affect the behaviour of the community.

It is still a challenge to operationalize Ostrom's framework for analysing complex social-ecological systems due to unavailability of data and lack of knowledge about how to collect information on some of the variables, for examples, variables such as institutions, cooperation social capital, collective action and information. There is no consensus among resource economists about the type of questions that should be ask to measure such variables. There is need for more research into this area in order to harmonize the different methods that are used by economists to measure such variables. Thanks to recent developments in experimental economics, it is now possible to solicit information for some of these variables using experiments.

There is also need to investigate some of the relationships among the variables suggested by Ostrom's theory, i.e., those variables that are assumed to influence collective action on one hand, and those that are believed to affect biodiversity outcomes. For example, the knowledge of a resource system is believed to have a positive impact on biodiversity conservation, according to the theory. It is now possible to solicit such information through public goods experiments and then establish the relationship. Future research should consider such relationships from ostrom's theory.

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APPENDIX A

Appendix for Chapter 2

Description of the IV estimation with heteroskedasticity-based instruments

Challenges in employing the standard IV methods

The standard IV method is employed in linear regression models, e.g., $Y = X\beta + \mu$, where we come across violations of the zero conditional mean assumption $E[\mu \mid X] = 0$.

Dependence on IV methods generally entails that suitable instruments are available to identify the model via exclusion restrictions.

The instruments, Z, must satisfy three conditions:

a) they must satisfy the orthogonality conditions, i.e., $E[\mu Z] = 0$;

b) they must be correlated with the Xs; and

c) they must be properly excluded from the model, so that their effect on the response variable is only indirect.

Textbook treatments of IV methods stress their usefulness in dealing with endogenous regressors.

Finding suitable instruments which concurrently satisfy these three conditions are often problematic and one of the major obstacle to the use of standard IV approach in many studies.

Lewbel's Approach

Consider observed endogenous variables Y_1 and Y_2, X a vector of observed exogenous regressors, and $\varepsilon = (\varepsilon_1, \varepsilon_2)$ as unobserved error processes. Consider a structural model of the form:

$Y_1 = X_0\beta + Y_2\gamma_1 + \varepsilon_1$

$Y_2 = X_0\beta + Y_1\gamma_2 + \varepsilon_2$

This system is triangular when $\gamma_2 = 0$ (or, with renumbering, when $\gamma_1 = 0$). Otherwise, it is fully simultaneous. The errors $\varepsilon_1, \varepsilon_2$ may be correlated with each other.

If the exogeneity assumption, $E(\varepsilon X) = 0$ holds, the reduced form is identified, but in the absence of identifying restrictions, the structural parameters are not identified. These restrictions often involve setting certain elements of β_1 or β_2 to zero, which makes instruments available.

Identification in Lewbel's approach is achieved by restricting correlations of $\varepsilon\varepsilon'$ with X. This relies upon higher moments, and is likely to be less reliable than identification based on coefficient zero restrictions. However, in the absence of plausible identifying restrictions, this approach may be the only reasonable strategy.

The parameters of the structural model will remain unidentified under the standard homoskedasticity assumption: that $E(\varepsilon\varepsilon' | X)$ is a matrix of constants. However, in the presence of heteroskedasticity related to at least some elements of X, identification can be achieved.

In a fully simultaneous system, assuming that $\text{cov}(X, \varepsilon_j^2) \neq 0$; j = 1; 2 and $\text{cov}(Z, \varepsilon_1\varepsilon_2) = 0$ for observed Z will identify the structural parameters. Note that Z may be a subset of X, so no information outside the model specified above is required.
The key assumption that $\text{cov}(Z, \varepsilon_1\varepsilon_2) = 0$ will automatically be satisfied if the mean zero error processes are conditionally independent: $\varepsilon_1 \perp \varepsilon_2 | Z = 0$. However, this independence is not strictly necessary.

Source: Extracted from Lewbel (2012)

Table A.1: Quantity consumed by gender

Variables	**Gender**		**Total**
	Female	*Male*	
Quantity of major wild foods items consumed			
vegetables	10.37	3.73	**7.056**
mushroom	0.28	0.84	**0.586**
insects	13.21	5.41	**9.310**
fruits	0.94	2.72	**1.831**
honey	0.30	1.16	**0.730**
Sub-total	**25.10**	**13.86**	**19.51**
Wildlife products only			
bush meat	4.69	12.64	**8.676**
small animals	4.25	4.45	**4.355**
fish	4.07	8.23	**6.148**
birds	1.34	1.56	**1.457**
Sub-total	**14.35**	**26.88**	**20.64**
Quantity of major non-food items consumed			
timber	23.76	30.90	**27.33**
firewood	595.29	849.99	**722.64**
thatch grass	52.65	50.45	**51.54**
basket	1.89	3.61	**2.750**
livestock fodder	5.23	16.81	**11.02**
Sub-total	**678.82**	**951.74**	**815.271**
Total consumed			
Total	**718.27**	**992.48**	**855.437**

Source: survey results 2013

Table A.2: Quantity consumed by farm land (F)

Variables	Farm size (ha)					Total
	F ≤ 0.5 ha	*0.5<F ≤ 1*	*1 <F ≤ 2.5*	*2.5 <F ≤ 5*	*F > 5*	
Quantity of major wild foods items consumed						
vegetables	8.98	7.86	6.73	5.96	5.72	**7.056**
mushroom	0.44	0.47	0.56	0.65	0.78	**0.586**
insects	13.61	11.30	9.18	7.34	4.12	**9.310**
fruits	3.42	2.07	1.72	1.53	0.41	**1.831**
honey	0.24	0.67	0.75	0.84	1.15	**0.730**
Sub-total	**26.29**	**22.37**	**18.94**	**16.32**	**13.18**	**19.51**
Wildlife products only						
bush meat	5.57	6.74	8.62	9.47	12.94	**8.676**
small animals	3.11	3.76	4.21	5.03	5.69	**4.355**
fish	2.64	3.43	6.46	7.58	10.43	**6.148**
birds	2.80	1.64	1.43	0.93	0.45	**1.457**
Sub-total	**14.12**	**15.57**	**20.73**	**23.01**	**29.51**	**20.64**
Quantity of major non-food items consumed						
timber	19.49	22.25	25.37	29.86	39.68	**27.33**
firewood	576.51	651.28	726.54	745.50	913.37	**722.64**
thatch grass	60.84	55.64	50.45	46.37	44.45	**51.54**
basket	0.63	1.85	2.67	3.78	4.82	**2.750**
livestock fodder	4.88	8.76	10.22	11.53	19.71	**11.02**
Sub-total	**662.35**	**739.78**	**815.25**	**837.04**	**1022.03**	**815.271**
Total consumed						
Total	**703.16**	**777.72**	**854.92**	**876.37**	**1064.72**	**855.437**

Source: survey results 2013

Table A.3: Percent household purchased or sold resources by environmental resource

Type of environmental resources	% Household purchased or sold environmental resources	
	Purchased	*Sold*
wild food items		
wild vegetables	74.7	76.2
mushroom	22.5	35.4
insects	78.9	82.6
fruits	28.1	43
honey	27.8	36.5
Subtotal	**46.4**	**54.7**
Wildlife products		
bush meat	70.3	54.8
small animals	43.6	89.6
fish	58.4	54.3
birds	55.2	62.1
Subtotal	**56.9**	**65.2**
Non-food items		
furniture	69.3	56.4
timber	57.7	47.2
firewood	70.5	66.5
thatch grass	61.2	73.8
Basket	52.9	45.5
kitchen utensils	50.1	70.6
agricultural implements	56.4	68.2
door mat	46.3	54.9
hat	48.6	59.1
pottery	47.5	52.3
Subtotal	**56.1**	**59.5**
Grand total		
Total	**53.1**	**59.7**

Source: survey results 2013

Table A.4: Poverty analysis results

Alpha	Poverty line	Standard income	Total income without wildlife		Total household income		Effect of wildlife
		Estimate	*Estimate*	*% reduction*	*Estimate*	*% reduction*	*% reduction*
α=0	360	0.476367	0.247451	48.1	0.221038	53.6	5.5
	450	0.643188	0.442539	31.2	0.369323	42.6	11.4
α=1	360	0.165532	0.054980	66.8	0.042633	74.2	7.5
	450	0.246873	0.112226	54.5	0.093163	62.3	7.7
α=2	360	0.074261	0.016523	77.8	0.011573	84.4	6.7
	450	0.122178	0.040607	66.8	0.031729	74.0	7.3

Source: survey results 2013

Table A.5: Comparison of headcounts, poverty gap and poverty severity indices

Poverty line	Units of analysis	Mean income	Headcount ratio (%)	Poverty gap	Poverty severity
	Standard household income				
360	All households (N=336)	452.97	47.6	16.6	7.4
	1st quintile	204.62	100.0	42.5	20.1
	2nd quintile	301.06	100.0	17.1	6.2
	3rd quintile	367.75	9.5	9.2	4.2
	4th quintile	527.22	0	0	0
	5th quintile	867.93	0	0	0
450	All households (N=336)	452.97	64.3	24.7	12.2
	1st quintile	204.62	100.0	54.0	30.8
	2nd quintile	301.06	100.0	31.8	12.9
	3rd quintile	367.75	100.0	16.9	7.2
	4th quintile	527.22	0	0	0
	5th quintile	867.93	0	0	0
	Total household income without wildlife resources				
360	All households (N=336)	595.50	24.7	5.5	1.7
	1st quintile	279.95	100.0	22.7	6.9
	2nd quintile	405.59	100.0	0.5	0
	3rd quintile	502.44	0	0	0
	4th quintile	668.29	0	0	0
	5th quintile	1125.96	0	0	0
450	All households (N=336)	595.50	44.3	11.2	4.1
	1st quintile	279.95	100.0	38.1	15.7
	2nd quintile	405.59	100.0	9.9	1.6
	3rd quintile	502.44	87.2	0.5	0
	4th quintile	668.29	0	0	0
	5th quintile	1125.96	0	0	0
	Total household income with wildlife resources				
360	All households (N=336)	634.69	22.1	4.3	1.2
	1st quintile	300.12	100.0	17.9	4.8
	2nd quintile	432.59	0	0	0
	3rd quintile	542.76	0	0	0
	4th quintile	710.13	0	0	0
	5th quintile	1194.00	0	0	0
450	All households (N=336)	634.69	36.9	9.3	3.2
	1st quintile	298.12	100.0	34.2	12.8
	2nd quintile	430.59	100.0	5.7	0
	3rd quintile	542.76	0	0	0
	4th quintile	713.13	0	0	0
	5th quintile	1193.88	0	0	0

Source: survey results 2013

Table A.6: Comparison of FGT indices assuming a poverty line of US$360.00 per capita

Units of analysis	Mean income	Headcount ratio (%)	Poverty gap	Poverty severity
Standard household income				
All households (N=336)	452.97	47.6	16.6	7.4
1st quintile	204.62	100.0	42.5	20.1
2nd quintile	301.06	100.0	17.1	6.2
3rd quintile	367.75	25.9	9.2	4.2
4th quintile	527.22	0	0	0
5th quintile	867.93	0	0	0
Total household income without wildlife resources				
All households (N=336)	595.50	24.7	5.5	1.7
1st quintile	279.95	100.0	22.7	6.9
2nd quintile	405.59	9.6	0.5	0
3rd quintile	502.44	0	0	0
4th quintile	668.29	0	0	0
5th quintile	1125.96	0	0	0
Total household income with wildlife resources				
All households (N=336)	634.69	22.1	4.3	1.2
1st quintile	300.12	100.0	17.9	4.8
2nd quintile	432.59	0	0	0
3rd quintile	542.76	0	0	0
4th quintile	710.13	0	0	0
5th quintile	1194.00	0	0	0
Policy experiments				
Total household income with wildlife resources (increase by 5%)				
All households (N=336)	647.76	8.8	4.3	1.2
1st quintile	296.25	85.6	11.7	2.3
2nd quintile	443.48	0	0	0
3rd quintile	543.65	0	0	0
4th quintile	685.24	0	0	0
5th quintile	1140.37	0	0	0
Total household income with wildlife resources (increase by 10%)				
All households (N=336)	664.48	2.1	1.3	0.3
1st quintile	458.36	61.4	4.8	0.7
2nd quintile	443.48	0	0	0
3rd quintile	567.74	0	0	0
4th quintile	699.35	0	0	0
5th quintile	1164.83	0	0	0
Total household income with wildlife resources (increase by 15%)				
All households (N=336)	687.72	0.3	0.2	0.0
1st quintile	471.52	29.5	3.3	0.1
2nd quintile	464.36	0	0	0
3rd quintile	586.74	0	0	0
4th quintile	729.15	0	0	0
5th quintile	1183.41	0	0	0

Source: survey results 2013

Table A.7(a): Test for significance difference between Gini coefficients

Index	Estimate	Std.Err.	t	P>|t|	95% Conf.	Interval
GINI Dis1|	0.333	0.0165	20.21	0.0000	0.301	0.366
GINI Dis2|	0.296	0.0150	19.67	0.0000	0.266	0.325
diff.|	-0.0376	0.00408	-9.203	0.0000	-0.0456	-0.0295

Source: survey results 2013

Table A.7(b): Test for significance difference

Index	Estimate	Std.Err.	t	P>|t|	95% Conf.	Interval
GINI Dis1|	0.333	0.0165	20.21	0.0000	0.301	0.366
GINI Dis2|	0.280	0.0137	20.47	0.0000	0.253	0.307
diff.|	-0.0535	0.00706	-7.584	0.0000	-0.0674	-0.0397

Source: survey results 2013

Table A.7(c): Test for significance difference

Index	Estimate	Std.Err.	t	P>|t|	95% Conf.	Interval
GINI Dis1|	0.296	0.0150	19.67	0.0000	0.266	0.325
GINI Dis2|	0.280	0.0137	20.47	0.0000	0.253	0.307
diff.|	-0.0160	0.00454	-3.522	0.0005	-0.0249	-0.00705

Source: survey results 2013

Table A.8: Gini indices for standard income and total income with & without wildlife

Original results

Standard income		**Total income (without wildlife)**		**Total household income**	
Estimate	*Std. Error*	*Estimate*	*Std. Error*	*Estimate*	*Std. Error*
0.333176	0.016490	0.295609	0.015028	0.279633	0.013661

Effect of including non-wildlife resources on total household income

Standard income	**Total income (without wildlife)**	**% reduction**
0.333176	0.295609	11.28

Effect of including wildlife resources on total household income

Standard income	**Total household income**	**% reduction**
0.333176	0.279633	16.07

Contribution of wildlife resources

Total income (without wildlife)	**Total household income**	**% reduction**
0.295609	0.279633	5.4

Results when wildlife income increases by 5%

Effect of including wildlife resources on total household income

Standard income	**Total household income @ 5%**	**% reduction**
0.333176	0.26218	21.31

Contribution of wildlife resources

Total income (without wildlife)	**Total household income @ 5%**	**% reduction**
0.295609	0.26218	11.30

Results when wildlife income increases by 10%

Effect of including wildlife resources on total household income

Standard income	**Total household income @ 10%**	**% reduction**
0.333176	0.24845	25.4

Effect of including wildlife resources on total household income

Standard income	**Total household income @ 10%**	**% reduction**
0.295609	0.24845	15.9

Results when wildlife income increases by 15%

Effect of including wildlife resources on total household income

Standard income	**Total household income @ 10%**	**% reduction**
0.333176	0.22461	32.6

Effect of including wildlife resources on total household income

Standard income	**Total household income @ 10%**	**% reduction**
0.295609	0.22261	24.7

Source: survey results 2013

Table A.9: Comparison of the Gini indices by income quintile

Group	Standard income		Total income (without wildlife)		Total household income	
	Index	Std. Err	Index	Std. Err	Index	Std. Err
1st quintile	0.156	0.0124	0.0992	0.00676	0.0921	0.00670
2nd quintile	0.137	0.0129	0.0502	0.00358	0.0426	0.00215
3rd quintile	0.144	0.0187	0.0493	0.00417	0.0403	0.00256
4th quintile	0.132	0.0167	0.0558	0.00431	0.0479	0.00307
5th quintile	0.275	0.0302	0.176	0.0255	0.168	0.0244
Population	0.333	0.0165	0.282	0.0141	0.200	0.0137

Source: survey results 2013

Table A.10: Decomposition of Gini Index by Incomes Sources – Shapley decomposition

Source	**With wildlife**			**Without wildlife**		
	Income share	*Absolute Contribution*	*Relative Contribution*	*Income share*	*Absolute Contribution*	*Relative Contribution*
employment	0.194	0.0650	0.160318	0.258086	0.0739	0.279398
agricultural income	0.401	0.133	0.493624	0.399289	0.148	0.500797
capital income	0.0732	0.0157	0.058299	0.072833	0.0176	0.059284
transfers	0.0194	0.00280	0.010360	0.019336	0.00226	0.007630
remittances	0.0580	0.00988	0.036596	0.057723	0.00697	0.023516
environmental income	0.254	0.0433	0.240803	0.192732	0.0472	0.159375
Total	**1.000**	**0.270**	**1.000000**	**1.000000**	**0.296**	**1.000000**

Source: survey results 2013

Table A11: VIF test results

Model 1 - Ordered logit regression model			*Model 2 - Determinants of environmental income generation*		
Variable	**VIF**	**1/VIF**	**Variable**	**VIF**	**1/VIF**
age of household head	2.100	0.476	tenure	2.320	0.431
education of the household head	1.860	0.538	age of household head	2.100	0.477
household head employed	1.250	0.800	area under cultivation	2.040	0.491
Tenure	1.180	0.849	household head live on the farm	1.510	0.660
household size	1.160	0.864	household head born in this area	1.360	0.734
household head born in this area	1.150	0.869	distance to the market	1.310	0.765
environmental income	1.090	0.920	education of the household head	1.230	0.810
household head is a Christian	1.040	0.958	household head employed	1.180	0.846
Mean	**VIF**	**1.350**	household head is a Christian	1.150	0.867
			household size	1.140	0.877
			number of dogs	1.130	0.886
			biodiversity	1.230	0.725
			Mean	**VIF**	**1.500**

Source: survey results 2013

Table A.12: Brant test for the ordered logit regression model

Estimated coefficients from regressions

	Quintile 1	quintile 2	quintile 3	quintile 4	quintile 5
environinc	0.00001	0.00049	0.00023	0.00215	-0.00282
age	-0.00119	0.02837	0.00647	0.00014	0.00059
educ	0.09538	0.03770	0.09545	0.12689	0.26755
employ	0.06493	0.01581	0.13632	0.25321	0.13344
hhsize	-0.03204	-0.04024	-0.05238	0.00706	0.00202
bornarea	-0.01587	-0.00769	-0.02374	0.02718	0.00267
religion	0.00954	0.00312	0.00440	0.00133	0.00453
gender	-0.12646	-0.3445	-0.22923	-0.43772	-0.1372

Brant Test of Parallel Regression Assumption

Variable	chi2	p>chi2	df
All	**4.34**	0.227	7
environinc	0.13	0.716	1
age	3.44	0.064	1
educ	1.26	0.672	1
employ	0.18	0.590	1
hhsize	2.73	0.424	1
bornarea	1.39	0.771	1
religion	4.50	0.183	1
gender	0.08	0.235	1

A significant test statistic provides evidence that the parallel regression assumption has been violated.

Table A.13: Ordered logit regression model

Ordered logit estimates	Number of obs	= 336
	LR chi2(8)	= 99.41
	Prob>chi2	= 0.000
Log likelihood = -491.2	Pseudo R2	= 0.0922
Quintile	**Coef.**	**Std. Err.**
environmental income	0.00217	0.0026***
age of household head	0.00729	0.0328
education of household head	0.582	0.2652***
household head employed [0, 1]	0.466	0.1474*
household size	-0.324	0.0253***
household head born in area [0, 1]	-0.143	0.0636
religion of household head [0, 1]	0.0580	0.2203**
gender [0,1]	0.0042	0.0436
tenure [0, 1]	-0.695	0.4577**
chamabvuwani [0, 1]	0.04749	0.0227*
chehondo [0, 1]	0.05270	0.0963
chihosi [0, 1]	0.57344	0.3201
chingele [0, 1]	-0.96421	0.5237**
chipachini [0, 1]	-0.5300	0.3971
chitete [0, 1]	-5.69564	1.0517
chitsanzeni [0, 1]	0.05587	0.0024
chizvirizvi [0, 1]	0.0680	0.0080
dhumisa [0, 1]	0.9941	0.3685
dopi [0, 1]	3.15709	0.1775*
gonakudzingwa Area 1 [0, 1]	2.6904	0.1024
gonakudzingwa Area 2 [0, 1]	0.02580	0.0660
gonakudzingwa Area 3 [0, 1]	0.00053	0.0019**
gondweni [0, 1]	0.07411	0.0853
hlarweni [0, 1]	-0.8652	0.1396
kotsvi/Sengwe [0, 1]	-27.270	6.2931
lisesa [0, 1]	-0.8243	0.2498
machiloli [0, 1]	-1.0794	0.2817
machindu [0, 1]	-0.0872	0.2434
mahenye [0, 1]	-0.3822	0.5341***
malifumune [0, 1]	1.32451	0.1940
malipati [0, 1]	0.07653	0.0985
muchingwizi [0, 1]	-0.4348	0.5831
mugiviza [0, 1]	0.0529	0.0295*
mutandahwe [0, 1]	0.6211	1.0622
muthlanguleni [0, 1]	0.1073	0.4539
nyangambe [0, 1]	11.4623	2.8081
samu [0, 1]	0.74520	0.0911*
sibizaphanzi[0, 1]	0.29211	0.0479
tinhongeni [0, 1]	-0.6195	0.3272**
cut1	-2.236	0.8283
cut2	-0.878	0.7505
cut3	0.0567	0.8564
cut4	1.485	0.8402

Source: survey results 2013

Table A.14: Determinants of environmental income generation

Environmental income	Ordinary Least Squares		IV Estimation	
	Coef.	*Std. Err.*	*Coef.*	*Std. Err.*
wealth index	8.649***	3.472	7.26***	3.813
age of household head	3.265**	7.516	3.678***	7.354
education of the household head	-62.82**	87.84	-73.70**	77.58
household head employed [0, 1]	-287.4**	165.0	-256.2***	179.0
household head live on the farm [0, 1]	7.053	52.56	4.435	44.02
household head born in this area [0, 1]	89.0	43.02	20.18	51.76
household head is a Christian [0, 1]	-117.3*	60.35	-125.2**	72.34
household size	54.34**	20.57	47.63***	33.21
distance to the market	4.381**	2.139	5.565**	3.411
number of dogs	31.36	49.18	53.07*	48.42
biodiversity	125.6*	77.57	87.3**	98.73
tenure [0, 1]	-531.7***	225.4	-535.3***	242.6
constant	346.1	541.6	324.0	653.5
chamabvuwani [0, 1]	0.5000	0.261	0.0374	0.074
chehondo [0, 1]	-1.632	0.963	-0.0527*	0.032
chihosi [0, 1]	0.3538	0.191	0.5914	0.964
chingele [0, 1]	0.1960	0.227	1.8530	1.076
chipachini [0, 1]	0.0573	0.023	0.1933	0.043
chitete [0, 1]	6.9517	0.537	8.6609	0.583
chitsanzeni [0, 1]	0.6801	0.244	0.5587	0.695
chizvirizvi [0, 1]	0.3685	0.157	0.6080	0.412
dhumisa [0, 1]	0.0258	0.693	0.0107	0.017
dopi [0, 1]	-0.111	0.242	-0.1568	0.277
gonakudzingwa Area 1 [0, 1]	0.0227**	0.036	2.5577**	0.314
gonakudzingwa Area 2 [0, 1]	-10.71*	4.223	-0.3417*	0.334
gonakudzingwa Area 3 [0, 1]	-0.968*	0.235	-3.0000**	1.005
gondweni [0, 1]	-0.075	0.142	-0.1568	0.314
hlarweni [0, 1]	0.0662*	0.557	0.3423**	0.734
kotsvi/Sengwe [0, 1]	-0.714	0.223	-0.2277	0.036
lisesa [0, 1]	0.0023	0.001	0.0053	0.010
machiloli [0, 1]	0.2950	0.529	-0.6211	0.268
machindu [0, 1]	0.4623	0.453	0.07317	0.066
mahenye [0, 1]	0.0479	0.040**	0.0914	0.178***
malifumune [0, 1]	0.0619	0.019	0.3272	0.292
malipati [0, 1]	3.0912	0.353	0.6374	0.100
muchingwizi [0, 1]	0.0007	0.018	0.3963	0.632
mugiviza [0, 1]	0.0814	0.182	0.6498	0.270
mutandahwe [0, 1]	0.5341	0.087	0.2434	0.079
muthlanguleni [0, 1]	0.0985	0.032	0.1947	0.003
nyangambe [0, 1]	0.2921**	0.278	1.0653***	0.647
samu [0, 1]	0.0479	0.056	0.2952	0.096
sibizaphanzi[0, 1]	-1.7477	0.453	-0.621	0.235
tinhongeni [0, 1]	0.0910	0.031	0.9291	1.000
0bservations	336		336	
R-squared	0.655		0.774	
Underidentification test (Kleibergen-Paap rk LM statistic):			Chi-square	27.827
			P-value	0.0065
Weak identification test (Cragg-Donald Wald F statistic):			F-statistic	9.1053
Hansen J statistic (overidentification test of all instruments):			Chi-square	9.2462
			P-value	0.6170

*** p<0.01, ** p<0.05, * p<0.1

Source: survey results 2013

Table A.15: Marginal effects for the ordered logit model

Average marginal effects	Number of obs = 336	
Model VCE:	OIM	
Expression:	environmincome b4_age1 b5a_educ1 b7_occup1 b_hhsize b11_born	
dy/dx w.r.t:	b12_religion a10_tenure	
		Delta-method
	Pr(quintile==1),	predict()
	dy/dx	*Std.Err.*
environmental income	0.0000167***	0.00000385
age of household head	0.00119	0.00151
education of household head	0.0953**	0.0236
household head employed	0.0649*	0.0343
household size	-0.0320***	0.00684
household head born in area	-0.0158	0.0333
religion of household head	0.00954**	0.0295
tenure	-0.126**	0.0565
	Pr(quintile==2),	predict()
	dy/dx	*Std.Err.*
environmental income	0.00000049**	0.00000011
age of household head	0.0283	0.0326
education of household head	0.0377***	0.0059
household head employed	0.0158*	0.0225
household size	-0.0402***	0.0034
household head born in area	-0.0076	0.0083
religion of household head	0.00312**	0.0140
tenure	-0.344**	0.0071
	Pr(quintile==3),	predict()
	dy/dx	*Std.Err.*
environmental income	0.0000023	0.00000463
age of household head	0.0064	0.0034
education of household head	0.0954***	0.0230
household head employed	0.1363*	0.0215
household size	-0.0523***	0.0024
household head born in area	-0.0237	0.0013
religion of household head	0.0044**	0.0027
tenure	-0.229*	0.0644
	Pr(quintile==4),	predict()
	dy/dx	*Std.Err.*
environmental income	0.0000215	0.0000062
age of household head	0.00014**	0.0021
education of household head	0.1268***	0.0166
household head employed	0.2532**	0.0036
household size	0.0070**	0.0022
household head born in area	0.0271	0.0214
religion of household head	0.0013*	0.0153
tenure	-0.437	0.0545
	Pr(quintile==5),	predict()
	dy/dx	*Std.Err.*
environmental income	0.0000282	0.00000233
age of household head	0.00059**	0.0007
education of household head	0.2675***	0.1345
household head employed	0.1334**	0.0042
household size	0.0020*	0.0146
household head born in area	0.0026	0.0042
religion of household head	0.0045	0.0113
tenure	-0.137	0.0338

Source: survey results 2013

Bootstrapping results

Table A.16: Ordered logit regression model

Bootstrap replications (200)

------1-------- --------2-------- --------3------- --------4----- --------5---- 200

	OLS
Number of obs	336
Replications	200
Wald chi2 (10)	412.2
Prob>chi2	0.000
Pseudo R-sq	72.50

Quintile	Observed Coef.	Bootstrapped Std. Err.
environmental income	0.01439	0.0004***
age of household head	0.02774	0.1471
education of household head	0.52727	0.2157***
household head employed [0, 1]	0.67756	0.2629*
household size	-0.0277	0.5612***
household head born in area [0, 1]	-0.0146	0.1263
religion of household head [0, 1]	0.45991	0.2955**
gender [0,1]	0.11045	0.2147
Tenure [0, 1]	-0.1647	0.5786**
Cons	1.4476**	0.9969

Source: survey data Aug 2013

Table A.17: Determinants of environmental income generation

Bootstrap replications (200)

------1-------- --------2-------- -----3------ ------4------ ------5----- 200

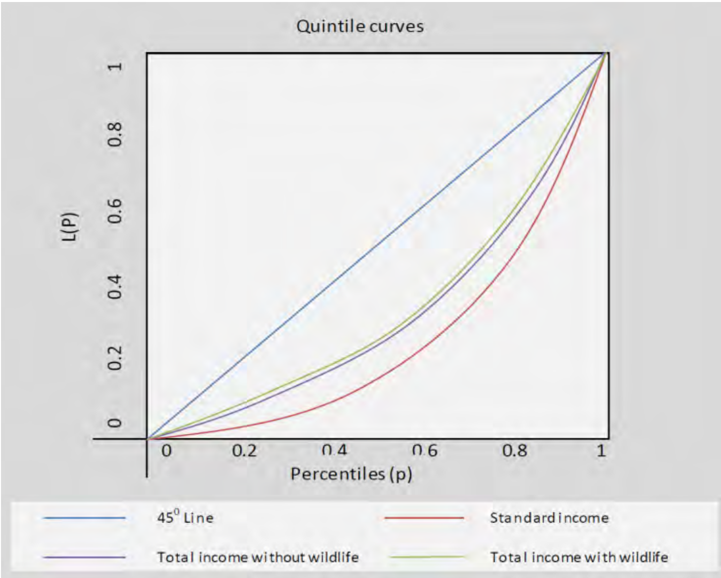
	OLS		IV	
Number of obs	336		336	
Replications	200		200	
Wald chi2 (10)	284.7		-	
Prob>chi2	0.000		-	
R-squared	0.792		-	
F (13, 322)	-		300.37	
Prob>F	-		0.0000	
Centered R2	-		0.7562	
Uncentered R2	-		0.8432	

environmental income	Observed Coef.	Bootstrapped Std. Err.	Observed Coef.	Bootstrapped Std. Err.
wealth index	7.773***	2.375	10.326***	8.113
age of household head	1.1570**	1.335	4.564***	2.541
education of the household head	-5.695**	7.233	-15.23**	7.258
household head employed [0, 1]	-2.212**	4.350	-5.223***	5.240
household head live on the farm [0, 1]	9.0741	52.56	6.346	3.021
household head born in this area [0, 1]	1.0762	0.432	2.217	1.763
household head is a Christian [0, 1]	-13.27*	6.735	-12.52**	7.343
household size	23.076**	11.25	17.22***	23.21
distance to the market	2.353**	5.139	5.114**	9.217
number of dogs	6.9648	4.184	9.727*	8.402
biodiversity	42.345*	29.45	30.23**	28.13
tenure [0, 1]	-22.53***	13.45	-35.23***	42.62
constant	3.2460	1.255	7.250	3.545

Underidentification test (Kleibergen-Paap rk LM statistic):			54.272
	Chi-sq(10) P-val	=	0.0000
Weak identification test (Cragg-Donald Wald F statistic):			34.261
Hansen J statistic (overidentification test of all instruments):			8.362
	Chi-sq(9) P-val	=	0.5912
Breusch-Pagan test for heteroskedasticity	chi2(1)	=	47.195
	Prob > chi2	=	0.0000

Source: survey data Aug 2013

Figure A.1: Lorenz curves for standard income and total household income



Source: survey results 2013

APPENDIX B

Appendix for Chapter 3

Table B.1: List of variables identify as relevant for studying complex SESs

Social, Economic, and Political Settings (S)	
S1- Economic development. S2- Demographic trends. S3- Political stability. S4- Government resource policies. S5- Market incentives. S6- Media organisation.	
Resource Systems (RS)	**Governance Systems (GS)**
RS1- Sector (e.g., wildlife, forests, pasture, fish)	GS1- Government organisations
RS2- Clarity of system boundaries	GS2- Nongovernment organisations
RS3- Size of resource system*	GS3- Network structure
RS4- Human-constructed facilities	GS4- Property-rights systems
RS5- Productivity of system*	GS5- Operational rules
RS6- Equilibrium properties	GS6- Collective-choice rules*
RS7- Predictability of system dynamics*	GS7- Constitutional rules
RS8- Storage characteristics	GS8- Monitoring and sanctioning processes
RS9- Location	
Resource Units (RU)	**Users (U)**
RU1- Resource unit mobility*	U1- Number of users*
RU2- Growth or replacement rate	U2- Socioeconomic attributes of users
RU3- Interaction among resource units	U3- History of use
RU4- Economic value	U4- Location
RU5- Number of units	U5- Leadership/entrepreneurship*
RU6- Distinctive markings	U6- Norms/social capital*
RU7- Spatial and temporal distribution	U7- Knowledge of SES/mental models*
	U8- Importance of resource*
	U9- Technology used
ACTION SITUATIONS [Interactions (I) → Outcomes (O)]	
I1- Harvesting levels of diverse users	O1- Social performance measures (e.g., efficiency, equity, accountability, sustainability)
I2- Information sharing among users	
I3- Deliberation processes	
I4- Conflicts among users	O2- Ecological performance measures (e.g., overharvested, resilience, biodiversity, sustainability)
I5- Investment activities	
I6- Lobbying activities	
I7- Self-organising activities	
I8- Networking activities	O3- Externalities to other SESs
Related Ecosystems (ECO)	
ECO1- Climate patterns. ECO2- Pollution patterns. ECO3- Flows into and out of focal SES.	

Source: Source: Extracted from Elinor Ostrom (2007: 15182)

*NB: *Subset of variables found to be associated with self-organisation.*

Table B.2: Sample size by community

Ward	Name of community	No. of interviews	Target	Households
Ward 10	Gondweni	10	20	400
	Muthlanguleni	10	20	420
Ward 12	Gonakudzingwa (Area 1)	1	2	6
	Gonakudzingwa (Area 2)	2	5	14
	Gonakudzingwa (Area 3)	4	10	23
Ward 13	Chamabvuwani	9	30	625
	Malifumune	6	10	236
Ward 14	Kotsvi/Sengwe	9	12	492
Ward 15	Dhumisa	18	15	320
	Hlarweni	25	15	108
	Malipati	10	10	260
	Mugiviza	3	10	178
	Samu	8	10	89
Ward 22	Chizvirizvi	10	10	81
Ward 23	Nyangambe	13	15	181
Ward 29	Mutandahwe	19	35	2000
Ward 30	Mahenye	29	30	960
Ward 4	Sibizaphanzi	10	10	350
Ward 5	Chitete	10	20	1000
	Chitsanzeni	12	10	311
	Tinhongeni	10	10	278
Ward 8	Chehondo	30	20	350
	Chihosi	8	10	250
	Chipachini	3	10	200
	Dopi	14	10	220
	Lisese	10	10	144
	Machiloli	10	10	71
Ward 9	Chingele	9	15	360
	Machindu	15	25	557
Total		336	419	10560

Source: Survey data Aug 2013

Table B.3: Important stakeholders and their role in community wildlife conservation

Variable	Obs	Mean	Std. Dev.	Min	Max
Important stakeholders					
Rural District Council	336	0.961	0.193	0	1
National Parks	336	0.875	0.331	0	1
Professional Hunter	336	0.732	0.444	0	1
Zimbabwe Republic Police	336	0.289	0.454	0	1
Traditional leaders	336	0.262	0.440	0	1
PARSEL	336	0.330	0.471	0	1
Environmental Management Agency	336	0.313	0.464	0	1
Veterinary Department	336	0.0595	0.237	0	1
AGRITEX	336	0.0565	0.231	0	1
Community Development Association	336	0.161	0.368	0	1
Malilangwe Trust	336	0.134	0.341	0	1
Hippo Valley Conservancy	336	0.0298	0.170	0	1
Save Conservancy	336	0.0387	0.193	0	1
Africa Wildlife Foundation	336	0.0387	0.193	0	1
Mean number of stakeholders	336	4.280	1.698	2	9
Role of the Rural District council in the CAMPFIRE project (RDC)					
RDC has the appropriation authority	336	0.732	0.444	0	1
Monitoring & enforcement/patrols	336	0.390	0.488	0	1
Major decision making organ	336	0.886	0.500	0	1
Select PH and issue licenses	336	0.813	0.391	0	1
Enact bylaws	336	0.881	0.324	0	1
Who is involved in setting the rules?					
Local communities	336	0.461	0.499	0	1
Rural District Council	336	0.789	0.409	0	1
National Parks	336	0.548	0.498	0	1
Wildlife Management Committee	336	0.310	0.463	0	1
Traditional leaders	336	0.265	0.442	0	1

Source: survey data Aug 2013

Table B.4: Community involvement in wildlife management

How community is involved?	Obs	Mean	Std. Dev.	Min	Max
Anti-poaching activities	336	0.595	0.492	0	1
Whistleblowing	336	0.363	0.482	0	1
Employ game guards with guns	336	0.223	0.417	0	1
Use resource monitors (volunteers)	336	0.244	0.430	0	1
Veldt fire management	336	0.423	0.495	0	1
Awareness campaigns	336	0.967	0.178	0	1
Quota setting	336	0.324	0.469	0	1
Live cropping	336	0.125	0.331	0	1

Source: survey data Aug 2013

Table B.5: Community characteristics

Variables	No		Yes		Total	
	Freq	%	Freq	%	Freq	%
Bundle of rights enjoyed by the community						
Use rights	206	61.3	130	38.7	336	100.0
Decision making rights	223	66.4	113	33.6	336	100.0
Ability to enter or exit and to exclude unentitled parties						
Are you able to enter or exist WMC	306	91.3	30	8.7	336	100.0
Are you able to exclude *unentitled parties*	223	66.4	113	33.6	336	100.0
Benefits from wildlife conservation						
Is the community entitled to benefits?	22	6.5	314	93.5	336	100.0
Household received cash dividends	259	77.1	77	22.9	336	100.0
Existence of a wildlife management plan and constitution						
Wildlife Management Plan	250	74.4	86	25.6	336	100.0
Constitution	47	14.0	259	86.0	336	100.0

Source: survey data Aug 2013

Table B.6: Number of rules, meetings and activities

Variable	Obs	Mean	Std.	Min	Max
Number of WMC meetings	336	3.58	1.62	0	12
Number of WMC activities	336	0.860	1.141	0	8
Number of WMC major rules	336	13.89	10.90	0	32
Number of WMC minor rules	336	33.04	27.44	0	78
Total number of WMC rules & regulations	336	46.87	38.09	0	110
Number of other NRM rules	336	6.846	5.304	0	19.25
Number of NRM meetings	336	2.48	1.50	0	5.25
Number of NRM activities	336	3.24	3.16	0	12.5
Number of non-NRM rules	336	14.34	4.57	7.5	25
Number of non-NRM meetings	336	4.00	1.45	2	7
Number of non-NRM activities	336	8.59	3.83	2	15
Extent to which rules are recognized by society	336	3.51	2.75	0	8
Extent to which rules are recognized by authorities	336	6.10	4.32	0	10
Number of poaching incidence	336	7.955	6.192	0	22

Source: survey data Aug 2013

Table B.7: Durbin-Wu-Hausman test for endogeneity

Model 1 - Cooperation and Institutions		
inst_res	=	0
F (1, 322)	=	0.540
Prob > F	=	0.461
Model 2 - Biodiversity and Cooperation		
cooperation_res	=	0
F (1, 322)	=	11.53
Prob > F	=	0.0008

Source: survey data Aug 2013

Table B.8: VIF test result for the models below

Variable	**VIF**	**1/VIF**
Model 1- with overall institutional index		
year of establishment	5.720	0.175
Trust	4.340	0.230
inst index	3.170	0.316
number of stakeholders	2.520	0.397
punishment [0 = Exo, 1 = Endo]	2.330	0.428
group size	1.910	0.523
Tenure	1.860	0.538
resource size	1.350	0.740
Ethnicity	1.310	0.763
wealth index	1.150	0.869
Mean	**VIF**	**2.570**
Model 1- with disaggregated institutional index		
Governance	8.000	0.125
Clarity	6.300	0.159
Year of est.	5.700	0.175
Monitoring	4.900	0.204
Trust	4.490	0.223
Fairness	4.100	0.244
punishment	2.900	0.345
number of stakeholders	2.680	0.373
group size	1.900	0.526
Tenure	1.860	0.538
Ethnicity	1.550	0.646
wealth index	1.360	0.735
Mean	**VIF**	**3.750**
Model 2 – cooperation and biodiversity		
Cooperation	5.060	0.198
Benefits-hat	4.650	0.215
average number of years in school	2.980	0.336
Fence	2.580	0.388
number of poaching incidents	2.240	0.447
average age of household head	1.950	0.512
distance to nearest urban centre	1.850	0.541
information sharing index	1.770	0.565
Training [0=no, 1=yes]	1.630	0.613
distance to the fence	1.600	0.625
average number of years living in the area	1.120	0.891
social capital index	1.080	0.926
Mean	**VIF**	**2.430**

Source: survey data Aug 2013

Table B.9: Model 2 - Relationship between biodiversity and cooperation with age squared

biodiversity	OLS	IV
cooperation	0.00575***	0.00758***
	(0.00346)	0.00415
training [0=yes, 1=no]	0.215***	0.208***
	0.0745	0.0762
Benefits-hat	1.61e-05***	1.35e-05***
	4.56e-06	5.37e-06
distance to nearest urban centre	0.0132***	0.0136***
	0.00145	0.00140
distance to the fence	0.0145***	0.0133***
	0.00342	0.00254
social capital index	0.00724***	0.00730***
	0.00221	0.00232
average age of household head	-0.347***	-0.312***
	0.123	0.135
age2	0.00347***	0.00368***
	0.00125	0.00138
average number of years in school	-0.0825***	-0.0791***
	0.0216	0.0223
average number of years living in the area	-0.00527**	-0.00586**
	0.00207	0.00236
Fence	-0.00434***	-0.00582***
	0.00199	0.00172
Information sharing index	0.252***	0.2302***
	0.0408	0.0443
Cons	8.520***	10.251***
	3.034	3.456

Source: survey data Aug 2013
NB: Standard errors shown in brackets
* Significant at 10% ** Significant at 5% *** Significant at 1%

Bootstrapping results

Table B.10: Model 1 - Relationship between cooperation and overall institutional index

Bootstrap replications (200)

------1-------- --------2-------- --------3------- --------4----- --------5---- 200

	OLS		IV	
Number of obs	336		336	
Replications	200		200	
Wald chi2 (10)	3533.93		-	
Prob>chi2	0.000		-	
R-squared	85.37		-	
F (10, 325)	-		223.22	
Prob>F	-		0.0000	
Centered R2	-		0.8485	
Uncentered R2	-		0.9479	
cooperation	Observed Coef.	Bootstrapped Std. Err.	Observed Coef.	Bootstrapped Std. Err.
institutional index	0.129***	0.0501	0.595***	0.109
group size	0.00121	0.0016	0.00452**	0.00188
Trust	3.120***	0.5523	1.205***	0.662
Ethnicity	-1.172	0.9825	0.223	0.849
wealth index	0.0673**	0.0318	0.0364	0.0280
year of establishment	1.672***	0.2556	1.151***	0.260
punishment [0 = Exo, 1 = Endo]	12.99***	1.7365	11.311***	1.781
resource size	-0.000150**	7.58e-05	-0.000267***	8.33e-05
number of stakeholders	1.552**	0.7267	1.832**	0.730
Tenure	-8.657***	1.9852	-7.046***	1.663
Cons	-12.04**	5.0241	-12.25**	5.158

Underidentification test (Kleibergen-Paap rk LM statistic):			68.935
	Chi-sq(10) P-val	=	0.0000
Weak identification test (Cragg-Donald Wald F statistic):			12.194
Hansen J statistic (overidentification test of all instruments):			18.252
	Chi-sq(9) P-val	=	0.3528
Breusch-Pagan test for heteroskedasticity	chi2(1)	=	18.543
	Prob > chi2	=	0.0000

Source: survey data Aug 2013

Table B.11: Model 1 - Cooperation and disaggregated institutional index

Bootstrap replications (200)

------1--------　--------2--------　--------3-------　--------4-----　--------5----　200

	OLS		IV	
Number of obs	336		336	
Replications	200		200	
Wald chi2 (10)	5371.36		-	
Prob>chi2	0.000		-	
R-squared	0.8838		-	
F (13, 322)	-		300.48	
Prob>F	-		0.0000	
Centered R2	-		0.8671	
Uncentered R2	-		0.9543	

cooperation	Observed Coef.	Bootstrapped Std. Err.	Observed Coef.	Bootstrapped Std. Err.
clarity index	0.023*	0.0539	0.109	0.2753
fairness index	-0.185***	0.0254	-0.524**	0.0411
governance	0.381***	0.1535	0.712***	0.1323
monitoring & enforcement index	0.451***	0.0182	0.336***	0.2143
group size	0.00428*	0.00126	0.00314*	0.1026
Trust	0.740***	0.5580	2.587***	0.8456
Ethnicity	-0.535	0.8854	-0.493	0.9210
wealth index	0.0647**	0.0236	0.0534**	0.0369
year of establishment	1.346***	0.1905	1.057***	0.2442
punishment [0 = Exo, 1 = Endo]	7.284***	1.9253	10.726***	2.4393
resource size	-0.000354**	0.0001	-0.000475**	0.0044
number of stakeholders	0.823*	0.5330	0.648**	0.6874
Tenure	-7.526**	3.6375	-15.28**	4.7651
Cons	-10.194***	3.8330	-2.127	4.1830

Underidentification test (Kleibergen-Paap rk LM statistic):			43.361
	Chi-sq(10) P-val	=	0.0000
Weak identification test (Cragg-Donald Wald F statistic):			25.372
Hansen J statistic (overidentification test of all instruments):			7.473
	Chi-sq(9) P-val	=	0.4804
Breusch-Pagan test for heteroskedasticity	chi2(1)	=	38.283
	Prob > chi2	=	0.0000

Source: survey data Aug 2013

Table B.12: Model 2 - Relationship between biodiversity and cooperation

Bootstrap replications (200)

------1-------- --------2-------- --------3------- --------4----- --------5---- 200

	OLS		IV	
Number of obs	336		336	
Replications	200		-	
Wald chi2 (10)	2804.46		-	
Prob>chi2	0.0000		-	
R-squared	0.7238		-	
F (13, 322)	-		283.38	
Prob>F	-		0.0000	
Centered R2	-		0.7217	
Uncentered R2	-		0.9221	
Biodiversity	Observed Coef.	Bootstrapped Std. Err.	Observed Coef.	Bootstrapped Std. Err.
Cooperation	0.00513***	0.00195	0.00902***	0.00300
training [0=no, 1=yes]	0.224***	0.0762	0.198***	0.07650
Benefits-hat	1.46e-05***	3.58e-06	1.12e-05***	4.43e-06
distance to nearest urban centre	0.0120***	0.00137	0.0129***	0.00160
distance to the fence	0.0140***	0.00213	0.0126***	0.00262
social capital index	0.00720***	0.00205	0.00705***	0.00218
average age of household head	-0.343***	0.10923	-0.403***	0.00149
average no. of years in school	-0.0821***	0.02745	-0.00934***	0.02354
average no. of years living in the area	-0.00567**	0.00267	-0.00615**	0.00201
Fence	-0.00474***	0.00210	-0.00499***	0.00179
Information sharing index	0.229***	0.04992	0.198***	0.04743
Cons	9.352***	4.18635	10.92***	3.62650
Underidentification test (Kleibergen-Paap rk LM statistic):				138.181
		Chi-sq(10) P-val	=	0.0000
Weak identification test (Cragg-Donald Wald F statistic):				24.338
Hansen J statistic (overidentification test of all instruments):				0.0641
		Chi-sq(9) P-val	=	0.7463
Breusch-Pagan test for heteroskedasticity		chi2(1)	=	21.94
		Prob > chi2	=	0.0000

Source: survey data Aug 2013

Table B.13: Model 2 - Relationship between biodiversity and institutions

Biodiversity	**OLS**		**IV**
Number of obs	336		336
Prob>F	0.7205		0.1027
R-squared	0.4223		-
F (7, 314)	-		9.47
Centered R2	-		0.507
Uncentered R2	-		0.534
institutions	0.00531		0.00517
	(0.01490)		(0.0105)
training [0=yes, 1=no]	0.0243*		0.317
	(0.00836)		(0.06305)
Benefits-hat	0.0000405**		0.02505*
	(0.00740)		(0.03706)
distance to nearest urban centre	0.1236		0.0364
	(0.00351)		(0.00307)
distance to the fence	0.0237***		0.0242***
	(0.01323)		(0.00126)
social capital index	0.00615**		0.00629*
	(0.00310)		(0.00301)
average age of household head	0.0306**		0.274**
	(0.2456)		(0.3520)
average number of years in school	-0.00817*		-0.0764
	(0.01362)		(0.03452)
average number of years living in the area	0.00429**		0.01473**
	(0.01307)		(0.00056)
Fence	-0.00046		-0.00381
	(0.00308)		(0.00270)
Information sharing index	0.2512**		0.02137**
	(0.03165)		(0.0645)
Cons	5.742		7.3054
	(2.00347)		(4.2172)
Underidentification test (Kleibergen-Paap rk LM statistic):			5.2440
	Chi-sq(9) P-val	=	0.2143
Weak identification test (Cragg-Donald Wald F statistic):			2.5308
Hansen J statistic (overidentification test of all instruments):			0.7437
	Chi-sq(7) P-val	=	0.0346
Breusch-Pagan test for heteroskedasticity	chi2(1)	=	0.0942
	Prob > chi2	=	0.2270

Source: survey data August 2013
NB: Standard errors shown in brackets
* Significant at 10% ** Significant at 5% *** Significant at 1%

Table B.14: Correlation matrix for the variables in the first model

(obs=336)

	Biodiversity	Capacity	Training	Benefits	Distance to the market	Distance to the fence	Social capital	Poaching incidents	Age of household head	Age2	No of years in school	Residence status of head	Head born in this area	No of years living in this area
Biodiversity	1													
Capacity	0.350	1												
Training	0.446	0.346	1											
Benefits	0.592	0.444	0.414	1										
Distance to market	0.375	-0.157	-0.092	0.135	1									
Distance to fence	0.326	0.281	0.213	-0.018	-0.089	1								
Social capital	0.214	0.202	0.194	0.195	-0.042	-0.043	1							
Poaching incidents	-0.143	-0.332	-0.050	-0.068	0.122	-0.175	0.032	1						
Age of household head	-0.034	-0.073	-0.038	-0.082	0.134	-0.054	0.026	0.033	1					
Age2	-0.026	-0.076	-0.023	-0.085	0.124	-0.041	0.022	0.039	0.99	1				
No. of years in school	0.066	0.192	0.067	0.191	-0.113	0.068	0.116	0.045	-0.718	-0.704	1			
Residence status of head	0.00	-0.092	0.052	-0.087	0.010	0.061	-0.039	-0.062	0.210	0.207	-0.334	1		
Head born in this area	-0.069	0.040	-0.033	-0.036	-0.136	-0.090	-0.013	-0.195	-0.283	-0.270	0.144	0.039	1	
No. of living in this area	-0.093	-0.093	-0.091	-0.133	0.200	-0.104	0.009	-0.070	0.482	0.465	-0.371	0.082	0.094	1

Source: survey data Aug 2013

Table B.15: Correlation matrix for the variables in the second model

(obs=336)

	Biodiversity	Capacity	Training	Benefits	Distance to the market	Distance to the fence	Social capital	Poaching incidents	Age of household head	Age2	No of years in school	Residence status of head	Head born in this area	No of years living in this area
Biodiversity	1													
Capacity	0.350	1												
Training	0.446	0.346	1											
Benefits	0.592	0.444	0.414	1										
Distance to market	0.375	-0.157	-0.092	0.135	1									
Distance to fence	0.326	0.281	0.213	-0.018	-0.089	1								
Social capital	0.214	0.202	0.194	0.195	-0.042	-0.043	1							
Poaching incidents	-0.143	-0.332	-0.050	-0.068	0.122	-0.175	0.032	1						
Age of household head	-0.034	-0.073	-0.038	-0.082	0.134	-0.054	0.026	0.033	1					
Age2	-0.026	-0.076	-0.023	-0.085	0.124	-0.041	0.022	0.039	0.99	1				
No. of years in school	0.066	0.192	0.067	0.191	-0.113	0.068	0.116	0.045	-0.718	-0.704	1			
Residence status of head	0.00	-0.092	0.052	-0.087	0.010	0.061	-0.039	-0.062	0.210	0.207	-0.334	1		
Head born in this area	-0.069	0.040	-0.033	-0.036	-0.136	-0.090	-0.013	-0.195	-0.283	-0.270	0.144	0.039	1	
No. of living in this area	-0.093	-0.093	-0.091	-0.133	0.200	-0.104	0.009	-0.070	0.482	0.465	-0.371	0.082	0.094	1

Source: survey data Aug 2013

APPENDIX C

Appendix for Chapter 4

Table C.1: Summary of key differences and similarities

Differences	
Communal farmers in CAMPFIRE projects	*Private game farms in the conservancy*
- Degree of state interference is high	- Minimal state interference
- Operating on communal (ancestral) land	- Operating on private land
- Land is owned communally	- Pooled land together/removing internal fence
- Appropriation rights belong to RDC	- Appropriation rights belong to the conservancy
- Revenue sharing plan is such that the RDC gets 47%, 3% goes to CAMPFIRE association and 50% goes to the producer community	- The conservancy retains 100% of the revenues and pays tax and levies to the state and the RDC
- Wildlife is managed by the RDC and ZNPWA	- Wildlife is managed by the conservancy
- Decisions about wildlife utilization are made by the RDC	- The conservancy make some important decisions about wildlife management and utilization except harvesting
- There is a relationship with the park, i.e., wildlife free to roam inside and outside the park	- No relationship with the park because the conservancy is fenced
- Contributions/shares are not well defined	- Contributions and shares are well defined in terms of provision rules
- Poaching in CAMPFIRE projects is perpetrated by CAMPFIRE communities	- Poaching in conservancies is done by non-CAMPFIRE communities
- Anti-poaching enforcement is done by community/ unpaid volunteers who lack incentives (anti-poaching not effective)	- Anti-poaching enforcement is done by trained and armed game guards (anti-poaching is effective)
- Marketing is done by safari operators on behalf of local communities	- Marketing activities are carried out at both individual farm level and as a group
- Most CAMPFIRE communities are not involved in non-consumptive tourism (potentially they can benefit through tourism, research, filming, live animal sales and meat cropping)	- Income is generated from non-consumptive tourism (accommodation, research, filming, live animal sales and meat cropping)

Similarities

- Same geographical region and located adjacent to GNP
- Similar activities (i.e., agricultural production and wildlife conservation)
- Decision about how much to harvest are made at national level by ZNPWA
- Wildlife in Zimbabwe is property of the state and no one individual or group owns it
- Wildlife is managed as a Common Property Resource (CPR) due to the absence of internal boundaries
- Wildlife is harvested both legally and illegally
- Anti-poaching enforcement is done at group level
- The clients who eventually utilize the quota are the same
- Income from trophy hunting

Table C.2: Summary of definitions of symbols used in this paper

$G(.)$	Public goods value
X_0	Stock of wildlife roaming on communal land (shared stock)
X_1	Stock of wildlife in the conservancy
h_0	Hunting quota for the CAMPFIRE community
h_1	Hunting quota for the conservancy
$\psi_i(.)$	Poaching function
$R(.)$	Revenue from tourism activity
L	Anti-poaching enforcement by the park agency
$\overline{T}_i$	Total production effort
T_i^a	Amount of effort toward agricultural production
T_1^e	Anti-poaching enforcement by the conservancy
T_i^p	Poaching effort
θ_i	Probability of being caught
ρ	Effectiveness parameter/lack of cooperation/institutional variable
π_i	Net benefits
λ	Shadow value of the stock of wildlife roaming on communal land (shared stock)
μ	Shadow value of the stock of wildlife in the conservancy
τ	Proportion of income going into the hands of CAMPFIRE communities
η_0	Fixed per unit cost of poaching effort
v_i	Per unit cost of anti-poaching effort
c_i	Fine imposed on poachers
A_i	Agricultural technology
β	Nuisance parameter
P^*	Average fixed marginal valuation (price) of legal off-take
P_a	Price of agricultural produce
s	Premium charged by the safari operator for their services
ε	Risk premium charged on illegal off-take
$\overline{M}$	Budget/income from the state

Maths C.3: The Park Agency's problem

The First Order Conditions (maximum principle) are given by

$$\frac{\partial H(.)}{\partial L}=0\Rightarrow \frac{\partial \theta_0(L,T_0^p)}{\partial L}c_0-v_o+\lambda\frac{\partial F(X_0,L)}{\partial L}-\lambda\frac{\partial \psi_0(.)}{\partial L}-\Lambda_0 v_0=0$$

$$\Rightarrow \frac{\partial \theta_0(L,T_0^p)}{\partial L}c_0+\lambda\frac{\partial F(X_0,L)}{\partial L}=v_o(1+\Lambda_0)+\lambda\frac{\partial \psi_0(.)}{\partial L}......(b)$$

$$\frac{\partial H}{\partial h_1}=-\mu-\Lambda_1 P^*=0\Rightarrow \mu=-\Lambda_1 P^*..........(c)$$

$$\frac{\partial H}{\partial h_0}=-\lambda-\Lambda_2(P^*-s)=0\Rightarrow \lambda=-\Lambda_2(P^*-s).........(d)$$

$$\dot{\lambda}=\delta\lambda-\frac{\partial H(.)}{\partial X_0}=\delta\lambda-\frac{\partial G(.)}{\partial X_0}-\frac{\partial R(.)}{\partial X_0}-\lambda\frac{\partial F(.)}{\partial X_0}..........(e)$$

$$\dot{\mu}=\delta\mu-\frac{\partial H(.)}{\partial X_1}=\delta\mu-\frac{\partial G(.)}{\partial X_1}-\mu\frac{\partial F(.)}{\partial X_1}...........(f)$$

Solving for X_0^*

$$\Rightarrow \dot{\lambda}=\delta\lambda-\frac{\partial G(.)}{\partial X_0}-\frac{\partial R(.)}{\partial X_0}-\lambda\frac{\partial F(.)}{\partial X_0}$$

$$\Rightarrow \dot{\lambda}=\delta\lambda-q-q-r\lambda\left[1-\frac{2X_0}{K_0}\right]$$

In equilibrium

$$r\lambda\left[1-\frac{2X_0}{K_0}\right]=\delta\lambda-2q$$

$$X_0^*=K_0\frac{\lambda(r-\delta)+2q}{2r\lambda}$$

Solving for X_1^*

$$\Rightarrow \dot{\mu}=\delta\mu-\frac{\partial G(.)}{\partial X_1}-\mu\frac{\partial F(.)}{\partial X_1}$$

$$\Rightarrow \dot{\mu}=\delta\mu-q+\mu r\left[1-\frac{2X_1}{K_1}\right]$$

In equilibrium

$$\mu r\left[1-\frac{2X_1}{K_1}\right]=\delta\mu+q$$

$$X_1^*=K_1\frac{\mu(r-\delta)+q}{2r\mu}$$

Solving for L^*

$$\Rightarrow \frac{\partial \theta_0(L,T_2^p)}{\partial L}c_0+\lambda\frac{\partial F(X_0,L)}{\partial L}=v_o(1+\Lambda_0)+\lambda\frac{\partial \psi_0(.)}{\partial L}$$

$$\Rightarrow g_0c_0+\lambda b_0\gamma L^{\gamma-1}=v_o(1+\Lambda_0)+\lambda b_0$$

$$\Rightarrow \lambda\gamma b_0 L^{\gamma-1}=v_o(1+\Lambda_0)+\lambda b_0-g_0c_0$$

$$\Rightarrow L^{\gamma-1}=\frac{v_o(1+\Lambda_0)+\lambda b_0-g_0c_0}{\lambda\gamma b_0}$$

$$\Rightarrow L^*=\left[\frac{v_o(1+\Lambda_0)+\lambda b_0-g_0c_0}{\lambda\gamma b_0}\right]^{\frac{1}{\gamma-1}}$$

$$\Rightarrow L^*=\left[\frac{\lambda\gamma b_0}{v_o(1+\Lambda_0)+\lambda b_0-g_0c_0}\right]^{\frac{1}{1-\gamma}}$$

Solving for h_i^*

$$F(X_1,T_1^e)-h_1-\psi_1(T_1^e,T_1^P)=0$$

$$\Rightarrow h_1=F(X_1^*,T_1^e)-\psi_1(T_1^e,T_1^P)$$

$$\Rightarrow h_1^*=rX_1^*\left[1-\frac{X_1^*}{K_1}\right]+b_1(T_1^{e*})^\alpha-\psi_1(T_1^e,T_1^P)$$

$$\Rightarrow h_1^*=rX_1^*\left[1-\frac{X_1^*}{K_1}\right]+b_1(T_1^{e*})^\alpha-(a[T_1^{P*}]^\phi-b_1T_1^{e^*})$$

$$F(X_0,L)-h_0-\psi_0(L,T_0^P)=0$$

$$\Rightarrow h_0=F(X_0^*,L^*)-\psi_0(L^*,T_0^P)$$

$$\Rightarrow h_0^*=rX_0^*\left[1-\frac{X_0^*}{K_0}\right]+b_0(L^*)^\gamma-(a[T_0^{P*}]^\phi-b_0L^*)$$

$$\Rightarrow h_0^*=rX_0^*\left[1-\frac{X_0^*}{K_0}\right]+b_0(L*)^\gamma+b_0L*-a[T_0^{P*}]^\phi$$

Maths C.4: Conservancy community

The First Order Conditions are given by:

$$\frac{\partial H}{\partial T_1^e}=0\Rightarrow -P_a\frac{\partial A_1(.)}{\partial T_1^e}+\frac{\partial \theta_1(.)}{\partial T_1^e}c_1-v_1+\mu\frac{\partial F(X_1,T_1^e)}{\partial T_1^e}-\mu\frac{\partial \psi_1(.)}{\partial T_1^e}=0$$

$$\Rightarrow \frac{\partial \theta_1(.)}{\partial T_1^e}c_1+\mu\frac{\partial F(X_1,T_1^e)}{\partial T_1^e}=P_a\frac{\partial A_1(.)}{\partial T_1^e}+v_1+\mu\frac{\partial \psi_1(.)}{\partial T_1^e}$$

$$\dot{\mu}=\delta\mu-\frac{\partial H(.)}{\partial X_1}=\delta\mu-\frac{\partial R_1(.)}{\partial X_1}-\mu\frac{\partial F(X_1,T_1^e)}{\partial X_1}$$

Solving for T_1^{e*}

$$\frac{\partial \theta_1(.)}{\partial T_1^e}c_1+\mu\frac{\partial F(X_1,T_1^e)}{\partial T_1^e}=P_a\frac{\partial A_1(.)}{\partial T_1^e}+v_1+\mu\frac{\partial \psi_1(.)}{\partial T_1^e}$$

$$\Rightarrow g_1c_1-P_aA_1-v_1+\alpha\mu b_1(T_1^e)^{\alpha-1}-\mu b_1=0$$

$$\Rightarrow \alpha\mu b_1(T_1^e)^{\alpha-1}=P_aA_1+v_1-g_1c_1+\mu b_1$$

$$\Rightarrow (T_1^e)^{\alpha-1}=\frac{P_aA_1+v_1+\mu b_1-g_1c_1}{\alpha\mu b_1}$$

$$\Rightarrow T_1^e=\left[\frac{P_aA_1+v_1+\mu b_1-g_1c_1}{\alpha\mu b_1}\right]^{\frac{1}{\alpha-1}}$$

$$\Rightarrow T_1^{e*}=\left[\frac{\alpha\mu b_1}{P_aA_1+v_1+\mu b_1-g_1c_1}\right]^{\frac{1}{1-\alpha}}$$

Maths C.5: The CAMPFIRE community

The First Order Conditions are given by:

$$\frac{\partial \pi_0}{\partial T_0^P} = 0 \Rightarrow -P_a \frac{\partial A_0(.)}{\partial T_0^P} + \omega \frac{\partial \psi_0(.)}{\partial T_0^p} - v_0 - \frac{\partial \theta_0(.)}{\partial T_0^p} c_0 = 0$$

$$\Rightarrow \omega \frac{\partial \psi_0(.)}{\partial T_0^p} = P_a \frac{\partial A_0(.)}{\partial T_0^P} + v_0 + \frac{\partial \theta_0(.)}{\partial T_0^p} c_0$$

Solving for T_0^{P*}

$$\Rightarrow \omega \frac{\partial \psi_0(.)}{\partial T_0^p} = P_a \frac{\partial A_0(.)}{\partial T_0^P} + \eta_0 + \frac{\partial \theta_0(.)}{\partial T_0^p} c_0$$

$$\Rightarrow \phi a \omega [T_0^p]^{\phi-1} = P_a A_0 + \eta_0 + g_0 c_0$$

$$\Rightarrow [T_0^p]^{\phi-1} = \frac{P_a A_0 + \eta_0 + g_0 c_0}{\phi a \omega}$$

$$\Rightarrow T_0^{p*} = \left[\frac{P_a A_0 + \eta_0 + g_0 c_0}{\phi a \omega}\right]^{\frac{1}{\phi-1}}$$

$$\Rightarrow T_0^{p*} = \left[\frac{\phi a \omega}{P_a A_0 + \eta_0 + g_0 c_0}\right]^{\frac{1}{1-\phi}}$$

Maths C.6: Introducing the institutional variable

$$\Rightarrow \omega \frac{\partial \psi_0(.)}{\partial T_0^p} = P_a \frac{\partial A_0(.)}{\partial T_0^P} + \eta_0 + \frac{\partial \theta_0(.)}{\partial T_0^p} c_0$$

$$\Rightarrow \phi \rho^{\phi} a \omega [T_0^p]^{\phi-1} = P_a A_0 + \eta_0 + g_0 c_0$$

$$\Rightarrow [T_0^p]^{\phi-1} = \frac{P_a A_0 + \eta_0 + g_0 c_0}{\rho^{\phi} \phi a \omega}$$

$$\Rightarrow T_0^{p*} = \left[\frac{P_a A_0 + \eta_0 + g_0 c_0}{\rho^{\phi} \phi a \omega}\right]^{\frac{1}{\phi-1}}$$

$$\Rightarrow T_0^{p*} = \left[\frac{P_a A_0 + \eta_0 + g_0 c_0}{\rho^{\phi} \phi a \omega}\right]^{\frac{1}{\phi-1}}$$

$$\Rightarrow T_0^{p*} = \left[\frac{\rho^{\phi} \phi a \omega}{P_a A_0 + \eta_0 + g_0 c_0}\right]^{\frac{1}{1-\phi}}$$

$$\Rightarrow T_0^{p*} = \left[\frac{\rho^{\phi} \phi a \omega}{P_a A_0 + \eta_0 + g_0 c_0}\right]^{\frac{1}{1-\phi}}$$

Maths C.7: The social planner

First Order Conditions

$$\frac{\partial H(.)}{\partial L}=0\Rightarrow \omega\frac{\partial\psi_0(.)}{\partial L}+\lambda\frac{\partial F(X_0,L)}{\partial L}-\lambda\frac{\partial\psi_0(.)}{\partial L}-v_o-\Lambda_0 v_0=0$$

$$\Rightarrow \lambda\frac{\partial F(X_0,L)}{\partial L}=(\lambda-\omega)\frac{\partial\psi_0(.)}{\partial L}+v_o(1+\Lambda_0)\ldots\ldots\ldots\ldots\ldots\ldots.(13b)$$

$$\frac{\partial H}{\partial h_1}=P^*-\mu-\Lambda_1P^*-\Omega_1=0\Rightarrow P^*(1-\Lambda_1)-\mu-\Omega_1=0\Rightarrow \mu=P^*(1-\Lambda_1)-\Omega_1\ldots\ldots\ldots\ldots.(13c)$$

$$\frac{\partial H}{\partial h_2}=\tau(P^*-s)-\lambda-\Lambda_0(P^*-s)-\Omega_0=0\Rightarrow (P^*-s)(\tau-\Lambda_0)-\lambda-\Omega_2=0\Rightarrow \lambda=(P^*-s)(\tau-\Lambda_2)-\Omega_2\ldots\ldots.(13d)$$

$$\frac{\partial H}{\partial T_1^e}=\frac{\partial\theta_1(.)}{\partial T_1^e}c_1-P_a\frac{\partial A_1()}{\partial T_1^e}-v_1+\mu\frac{\partial F(.)}{\partial T_1^e}-\mu\frac{\partial\psi_1(.)}{\partial T_1^e}=0\ldots\ldots\ldots\ldots\ldots.(13e)$$

$$\frac{\partial H}{\partial T_0^p}=-P_a\frac{\partial A_0()}{\partial T_0^P}+\omega\frac{\partial\psi_0(.)}{\partial T_0^P}-v_0-\lambda\frac{\partial\psi_0(.)}{\partial T_0^P}=0\ldots\ldots\ldots\ldots\ldots\ldots.13(f)$$

$$\dot\lambda=\delta\lambda-\frac{\partial H(.)}{\partial X_0}=\delta\lambda-\frac{\partial G(.)}{\partial X_0}-\frac{\partial R(.)}{\partial X_0}-\lambda\frac{\partial F(.)}{\partial X_0}-\varphi\ldots\ldots\ldots\ldots\ldots\ldots\ldots.13(g)$$

$$\dot\mu=-\frac{\partial H(.)}{\partial X_1}=\delta\mu-\frac{\partial G(.)}{\partial X_1}-\frac{\partial R(.)}{\partial X_1}-\mu\frac{\partial F(.)}{\partial X_1}\ldots\ldots\ldots\ldots\ldots.13(h)$$

Solving for L^*

$$\Rightarrow \lambda\frac{\partial F(X_0,L)}{\partial L}=(\lambda-\omega)\frac{\partial\psi_2(.)}{\partial L}+v_o(1+\Lambda_0)$$

$$\Rightarrow \gamma\lambda b_0L^{\gamma-1}=(\lambda-\omega)b_0+v_o(1+\Lambda_0)$$

$$\Rightarrow \gamma\lambda b_0L^{\gamma-1}=(\lambda-\omega)b_0+v_o(1+\Lambda_0)$$

$$\Rightarrow L^{\gamma-1}=\frac{(\lambda-\omega)b_0+v_o(1+\Lambda_0)}{\gamma\lambda b_0}$$

$$\Rightarrow L^*=\left[\frac{(\lambda-\omega)b_0+v_o(1+\Lambda_0)}{\gamma\lambda b_0}\right]^{\frac{1}{\gamma-1}}$$

$$\Rightarrow L^*=\left[\frac{\gamma\lambda b_0}{v_o(1+\Lambda_0+(\lambda-\omega)b_0)}\right]^{\frac{1}{1-\gamma}}$$

Solving for X_0^*

$$\Rightarrow \dot\lambda=\delta\lambda-\frac{\partial G(.)}{\partial X_0}-\frac{\partial R(.)}{\partial X_0}-\lambda\frac{\partial F(.)}{\partial X_0}-\varphi$$

$$\Rightarrow \dot\lambda=\delta\lambda-2q-\varphi-r\lambda\left[1-\frac{2X_0}{K_0}\right]$$

In equilibrium

$$\Rightarrow r\lambda\left[1-\frac{2X_0}{K_0}\right]=\delta\lambda-2q-\varphi$$

$$\Rightarrow 1-\frac{2X_0}{K_0}=\frac{\delta\lambda-2q-\varphi}{r\lambda}$$

$$\Rightarrow \frac{2X_0}{K_0}=1-\frac{\delta\lambda-2q-\varphi}{r\lambda}$$

$$\Rightarrow X_0^*=K_0\frac{\lambda(r-\delta)+2q+\varphi}{2r\lambda}$$

Solving for X_1^*

$$\Rightarrow \dot{\mu} = \delta\mu - \frac{\partial G(.)}{\partial X_1} - \frac{\partial R(.)}{\partial X_1} - \mu\frac{\partial F(.)}{\partial X_1}$$

$$\Rightarrow \dot{\mu} = \delta\mu - q - q - r\mu\left[1 - \frac{2X_1}{K_1}\right]$$

In equilibrium

$$\Rightarrow r\mu\left[1 - \frac{2X_1}{K_1}\right] = \delta\mu - 2q$$

$$\Rightarrow 1 - \frac{2X_1}{K_1} = \frac{\delta\mu - 2q}{r\mu}$$

$$\Rightarrow \frac{2X_1}{K_1} = 1 - \frac{\delta\mu - 2q}{r\mu}$$

$$\Rightarrow X_1^* = K_1\frac{\mu(r-\delta)+2q}{2r\mu}$$

$$\Rightarrow X_1^* = K_1\frac{\mu(r-\delta)+2q}{2r\mu}$$

Solving for T_1^{e*}

$$\frac{\partial H}{\partial T_1^e} = \frac{\partial\theta_1(.)}{\partial T_1^e}c_1 - P_a\frac{\partial A_1()}{\partial T_1^e} - v_1 + \mu\frac{\partial F(.)}{\partial T_1^e} - \mu\frac{\partial\psi_1(.)}{\partial T_1^e}$$

$$\Rightarrow \frac{\partial\theta_1(.)}{\partial T_1^e}c_1 - P_a\frac{\partial A_1()}{\partial T_1^e} - v_1 + \mu\frac{\partial F(.)}{\partial T_1^e} - \mu\frac{\partial\psi_1(.)}{\partial T_1^e} = 0$$

$$\Rightarrow g_1c_1 - P_aA_1 - v_1 + \alpha\mu b_1(T_1^e)^{\alpha-1} - \mu b_1 = 0$$

$$\Rightarrow \alpha\mu b_1(T_1^e)^{\alpha-1} = P_aA_1 + v_1 - g_1c_1 + \mu b_1$$

$$\Rightarrow (T_1^e)^{\alpha-1} = \frac{P_aA_1 + v_1 + \mu b_1 - g_1c_1}{\alpha\mu b_1}$$

$$\Rightarrow T_1^e = \left[\frac{P_aA_1 + v_1 + \mu b_1 - g_1c_1}{\alpha\mu b_1}\right]^{\frac{1}{\alpha-1}}$$

$$\Rightarrow T_1^{e*} = \left[\frac{\alpha\mu b_1}{P_aA_1 + v_1 + \mu b_1 - g_1c_1}\right]^{\frac{1}{1-\alpha}}$$

Solving for T_0^{P*}

$$\frac{\partial H}{\partial T_0^P} = -P_a\frac{\partial A_0()}{\partial T_0^P} + \omega\frac{\partial\psi_0(.)}{\partial T_0^P} - \eta_0 - \lambda\frac{\partial\psi_0(.)}{\partial T_0^P}$$

$$\Rightarrow -P_a\frac{\partial A_0()}{\partial T_0^P} + \omega\frac{\partial\psi_0(.)}{\partial T_0^P} - \eta_0 - \lambda\frac{\partial\psi_0(.)}{\partial T_0^P} = 0$$

$$\Rightarrow -P_aA_0 + \phi\omega a[T_0^p]^{\phi-1} - \eta_0 - \phi\lambda a[T_0^p]^{\phi-1} = 0$$

$$\Rightarrow \phi\omega a[T_0^p]^{\phi-1} - \phi\lambda a[T_0^p]^{\phi-1} = P_aA_0 + \eta_0$$

$$\Rightarrow \phi a[T_0^p]^{\phi-1}(\omega-\lambda) = P_aA_0 + \eta_0$$

$$\Rightarrow [T_0^p]^{\phi-1} = \frac{P_aA_0 + \eta_0}{\phi a(\omega-\lambda)}$$

$$\Rightarrow T_0^p = \left[\frac{P_aA_0 + \eta_0}{\phi a(\omega-\lambda)}\right]^{\frac{1}{\phi-1}}$$

$$\Rightarrow T_0^{p*} = \left[\frac{\phi a(\omega-\lambda)}{P_aA_0 + \eta_0}\right]^{\frac{1}{1-\phi}}$$

Maths C.8: Functional forms

$$G(.) = qX_0 + qX_1 = q(X_0 + X_1)$$
$$R(.) = qX_i$$
$$A_0(.) = AT_0^a = A_0[\overline{T}_0 - T_0^p]$$
$$A_1(.) = AT_1^a = A_1[\overline{T}_1 - T_1^e]$$
$$\psi_0(.) = a[T_0^P]^\phi - b_0 L, b_0 > 0 \; and \; 0 < \phi < 1$$
$$\psi_1(.) = a[T_1^P]^\phi - b_1 T_1^e, b_1 > 0 \; and \; 0 < \phi < 1$$
$$\theta_0(L, T_0^P) = g_0(L + T_0^P)$$
$$\theta_1(T_1^e, T_1^P) = g_1(T_1^e + T_1^P)$$
$$F(X_0, L) = rX_0\left[1 - \frac{X_0}{K_0}\right] + b_0 L^\gamma \text{ [growth inside the national game park]}$$
$$F(X_1, T_1^e) = rX_1\left[1 - \frac{X_1}{K_1}\right] + b_1 (T_1^e)^\alpha \text{ [growth inside the conservancy]}$$

Maths C.9: Numerical illustration of stock dynamics

$$X_i(t+1) - X_i(t) = F(X_i, L) - \psi_i(\cdot, T_i^P)$$

Shared stock

$$X_0(t+1) = X_0(t) + F(X_0, L) - \psi_0(L, T_0^P) \text{ given } X_0^{(0)} \text{ and t} = 0$$

Shared stock (with constrained poaching)

$$X_0(t+1) - X_0(t) = F(X_0, L) - h_0 - \psi_0(L, \rho T_0^P) \text{ given } X_0^{(0)} \text{ and t} = 0$$

In the conservancy

$$X_1(t+1) - X_1(t) = F(X_1, T_1^e) - h_1 - \psi_1(T_1^e, T_1^P) \text{ given } X_1^{(0)} \text{ and t} = 0$$

APPENDIX D

A. HOUSEHOLD QUESTIONNAIRE

B. GUIDELINE FOR KEY INFORMANT INTERVIEWS

Number__________

HOUSEHOLD QUESTIONNAIRE

Introduction

Good morning/afternoon. My name is **Herbert Ntuli**, and I am currently studying for a PhD in Economics at the University of Cape Town. As part of my degree programme, I am currently carrying out a research study on **Community Wildlife Management involving local communities and private game farms around South Africa's Kruger National Park and Gonarezhou in Zimbabwe**. The major objective of this study is to enhance our understanding of the role of local institutions in promoting sustainable management of social-ecological systems, compare the performance of wildlife management under private and common property systems and to learn from private systems, how local communities can adapt to commercial park activities. Villagers from this area have been randomly selected to represent other members of the community in this study. From this area, a small sample of households have been selected and I am now in the process of discussing with people like you to get information about household participation in community-based wildlife management and many other activities including agricultural and non-agricultural activities. This information is **confidential** and will only be used by myself for the purposes of this study which will not make reference by name to any one respondent. I will be grateful if you could assist me in filling out this questionnaire in as honest a manner as possible.

Section A: Questionnaire Identification

A1	Date of Interview ____________________	**A5**	Enumerator code ____________________
A2	Household ID ____________________	**A6**	Country 1 = South Africa 2 = Zimbabwe
A3	District ____________________	**A7**	Village ____________________
A4	Ward ____________________	**A8**	Name of Game park ____________________

Section B: Household Demographics and Village Characteristics

B1 What is your household composition and characteristics?

	Name (start with the HH head)	**Relation to HH head**	**Sex** 0=F 1=M	**Age (No. of years)**	Educational Level	No. of years in School	Marital status	Occupation (Specify)	Is HH member still in school? (6 – 18 yrs)	School dropout 0=No 1=Ye	Is HH member resident on the farm? 0=No 1=Yes 2 = DK	If not, where does he or she live?
		B2	**B3**	**B4**	**B5(a)**	**B5(b)**	**B6**	**B7**	**B8**(a)	**B8(b)**	**B9**	**B10**
1.												
2.												
3.												
4.												
6.												
7.												
8.												
9												
10.												

Codes

B2	1 = Self	2 = Spouse	3 = Child	4 = Relative	5 = Labourer	6 = Other ____________
B3	0 = None	1 = Primary	2 = Secondary	3 = Tertiary	4 = Vocational	6 = Other ____________
B6	1 = Single	2 = Married	3 = Divorced	4 = Widowed		
B7	0 = None	1 = Farmer	2 = Casual	3 = Self	4 = Employed	5 = Other ____________
B11	1 = nearby fams	2 = urban	3 = other rural areas	4 = Abroad	5 = Other ____________	

B11 Was household head born in this area? 0=No 1=Yes

B12 Religion of the household head?
0=None 1=Christianity 2=Traditional 3=Muslim 4=Other (______________________________)

B13 Major family language?
1=Shangani 2=Venda 3=Shona 4=Other (______________________)

B14 Type of toilet facility used by the household?
0=None/bush 1=Flash toilet 2=Blair toilet 3= Pit latrine 5=Neighbour 6=Other (__________)

B15 Walling material of household's main residential house?
1=Burned bricks 2=Unburned bricks 3=Stones 4=Pole and dagga 5=Other (__________________)

B16 Roofing material of household's main residential house?
1=Grass thatch 2=Asbestos 3=Iron sheet 4=Tiles 5=Other (__________________________)

B17 What is the household's main source of drinking water?
1=Piped 2=Borehole 3= Protected well 4=Unprotected well 5=Stream 6=River
7=Dam/lake 8=Ponds 9=Other (________________________)

Section C: Land holding and general crop production

C1 What was the household's land holding during the 2012/2013 agricultural season?

C2 Farm size (ha) ___________________

C4 Rented out (ha) ___________________

C3 How land acquired? ____

C5 Rented in (ha) ___________________

C6 Farming experience of the household head (number of year) ______________________________________

C7 Did you grow any summer crop(s) during the 2012/13 agricultural reason?

Crop(s)	Tick	Area (ha)	Harvest (kg)	Value of harvest (US$)	Quantity retained (kg)	Quantity Sold (kg)	Amount realised (US$)	Crop loss due to wildlife (Kg)	Crop loss postharvest (Kg)
Maize									
Sorghum									
Millet									
Groundnuts									
Dry beans									
Soybean									
Sunflower									
Rapoko									
Sweet potatoes									
Cotton									
Cassava									
Cowpea									
Round nuts									
Watermelon									
Other (________)									
Other (________)									
Other (________)									

C8 Production costs for the summer crop(s) during the 2012/13 agricultural reason?

Crop		Basal fertilizer (kg)	Cost of Basal fertilizer (US$)	Top dressing fertilizer (kg)	Cost of top dressing (UD$)	Chemicals (mls)	Chemicals (US$)	Manure (Kg)	Cost of Manure (US$)	Draught Power (no. of times hired)	Cost of Draught Power (US$)
Maize											
Sorghum											
Millet											
Groundnuts											
Dry beans											
Soybean											
Sunflower											
Rapoko											
Sweet potatoes											
Cotton											
Cassava											
Cowpea											
Round nuts											
Watermelon											
Other (________)											
Other (________)											
Other (________)											

C9 Labour and storage costs for the summer crop(s) during the 2012/13 agricultural reason?

	Land Preparation			**Planting**			**Weeding**			**Harvesting**			**Protection**		**Storage cost**
	Number of people	Number of days	Cost of labour/ day	Number of people	Number of days	Cost of labour/ day	Number of people	Number of days	Cost of labour/ day	Number of people	Number of days	Cost of labour/ day	Number of days	Cost per day	
Maize															
Sorghum															
Millet															
Groundnuts															
Dry beans															
Soybean															
Sunflower															
Rapoko															
Sweet potatoes															
Cotton															
Cassava															
Cowpea															
Round nuts															
Watermelon															
Other (________)															
Other (________)															
Other (________)															
Other (________)															

C10 Did you grow any vegetable crop(s) during the 2012/13 agricultural season?

Crop(s)		Area (ha)	Harvest (kg)	Value of harvest (US$)	Quantity retained (kg)	Quantity Sold (kg)	Amount realised (US$)	Crop loss due to wildlife (Kg)	Crop loss postharvest (Kg)
Leafy vegetables									
Tomatoes									
Pumpkin									
Okra									
Onions									
Carrots									
Cabbage									
Fruit trees***									
Other (______)									
Other (______)									
Other (______)									

*** Fruits include bananas, pine apples, mangoes, paw paws, oranges e.tc

C11 Production costs for vegetable crop(s) during the 2012/13 agricultural reason?

Crop(s)		Fertilizer (kg)	Cost of fertilizer (US$)	Manure (Kg)	Cost of Manure (US$)	Chemicals (mls)	Chemicals (US$)	Labour cost	Transport cost	Storage cost
Leafy vegetables										
Tomatoes										
Pumpkin										
Okra										
Onions										
Carrots										
Cabbage										
Fruit trees										
Other (______)										
Other (______)										
Other (______)										

Section D: Information about livestock and other household assets

D1 What were your livestock holding for the past twelve months (*e.g., since July last year*)?

Type of livestock	Own livestock 0=No 1=Yes	Initial stock	Number bought	Amount (US$)	Number of births	Number of deaths	Number sold	Amount (US$)	Maintenance costs (US$)**	Closing stock
Cattle										
Draught power										
Sheep										
Goats										
Donkeys										
Pigs										
Chicken										

Turkeys										
Guinea fowls										
Ducks										
Rabbit										
Local bee hives										
Other (_________)										
Other (_________)										

*** Maintenance costs include the estimated costs of veterinary services, drugs and labour

D2 What type of dwelling does household own, number of rooms, year built and estimated value?

Type of dwelling	Type of dwelling	Own 0=No 1=Yes	Number of rooms	Year built	Est. cost of construction
Burned brick under asbestos/iron sheets					
Unburned brick under asbestos/iron sheets					
Burned brick under grass					
Unburned brick under grass					
Pole and dagga under asbestos/iron sheets					
Pole and dagga under grass					
Other (____________)					

D3 Which types of farm mechanization or assets does household own?

Type of asset	Own 0=No 1=Yes	Number owned	Condition (see codes)	Year bought	Estimated value (US$)
Cars					
Trucks/lorry					
Tractor					
Trailer					
Plough					
Cultivator					
Planter					
Harrow					
Cart					
Wheelbarrow					
Pick					
Machete					
Knapsack sprayer					
Axe					
Hand hoes					
Spade/shovel					
Water pump/hand pump					
Other (________)					

Condition: 1 = Obsolete 2 = Very bad 3 = Bad 4 = Good 5 = Very good 6 = Excellent

D4 Animal handling facilities and grain storage facilities

Facility	Own (0=No 1=Yes)	Number owned	Year built	Est. cost of construction
Paddocks				
Cattle kraal				
Pigsty				
Fowl run				
Granary				
Shade/storeroom				
Other (_______)				

D5 household assets owned

Asset	Own 0=No 1=Yes	Number owned	Condition (see codes)	Year bought	Estimated value (US$)
Kitchen utensils					
Television					
Radio					
Motorbike					
Bicycle					
Bed					
Sofa					
Table					
Chairs					
Wood stove					
Kerosene stove					
Stone grain mill					
Grinding mill					
Cell phone					
Gun					
Solar panel					
Other (_______)					

Condition: 1 = Obsolete 2 = Very bad 3 = Bad 4 = Good 5 = Very good 6 = Excellent

Section E: Livelihoods strategies of the household

E1 What were the household's sources of income for the past 12 months?

Income sources		Estimate for the past year (US$)	Please rank the top ten income sources during the past 12 months
Fulltime employment	0=No 1=Yes		
Pension, Government & NGO Transfers	0=No 1=Yes		
Income generating projects**	0=No 1=Yes		
Agriculture	0=No 1=Yes		
Rented out land	0=No 1=Yes		
Rented out oxen for ploughing	0=No 1=Yes		
Livestock sales (cattle, goats e.tc)	0=No 1=Yes		
Animal products (e.g., milk, eggs e.tc)	0=No 1=Yes		
Remittances	0=No 1=Yes		
Wildlife conservation & tourism activities	0=No 1=Yes		
Gold panning	0=No 1=Yes		
Brick making	0=No 1=Yes		
Construction/builder/brick layer	0=No 1=Yes		
Craft	0=No 1=Yes		
Casual labour	0=No 1=Yes		
Drought relief/food for work	0=No 1=Yes		
Cross border/buying and selling/Shops	0=No 1=Yes		
Gifts (including marriage gifts)	0=No 1=Yes		
Manure/dung cake/crop residues/hay	0=No 1=Yes		
Interest from deposits	0=No 1=Yes		
Other (________________________)	0=No 1=Yes		
Other (________________________)	0=No 1=Yes		

** Include community projects, gardens, women's groups e.tc
*** Includes bush meat

E2 Household estimated expenditure for the past 12 months?

Expenditure item (should include own production)		Quantity bought (kg)	Unit price (US$)	Total (US$)
Maize grain	0=No 1=Yes			
Mealie meal	0=No 1=Yes			
Other grain	0=No 1=Yes			
Grain milling	0=No 1=Yes			
Groceries (bread, cooking oil, salt, sugar, tea leaves…)	0=No 1=Yes			
Meat (chicken, beef, pork e.tc)	0=No 1=Yes			
Animal products (skin, milk, eggs, …)	0=No 1=Yes			
Clothing (including blankets)	0=No 1=Yes			
Hospital bills	0=No 1=Yes			
Paraffin	0=No 1=Yes			
Electricity bills	0=No 1=Yes			
Rent (house, ox plough, land e.tc)	0=No 1=Yes			
Debt payments (including input loan)	0=No 1=Yes			
Construction (material, labour, repairs..)	0=No 1=Yes			
Household furniture/appliances/kitchen utensils	0=No 1=Yes			
School fees (including books, transport ….)	0=No 1=Yes			
Vegetables (leafy vegetables, onions, tomatoes …)	0=No 1=Yes			
Fruits (mangoes, oranges, e.tc)	0=No 1=Yes			
Ceremonies (including marriage gifts)	0=No 1=Yes			
Recreation (beer/cigarettes/hair saloon …)	0=No 1=Yes			
Purchase of vehicles and farm machinery	0=No 1=Yes			
Repairs (vehicles, farm machinery, fuel …)	0=No 1=Yes			
Public transport	0=No 1=Yes			
Mobile phone air time	0=No 1=Yes			
Contribution to wildlife associations	0=No 1=Yes			
Contribution to farmer associations	0=No 1=Yes			
Other contributions (e.g., church, burial society…)	0=No 1=Yes			
Other (____________________)	0=No 1=Yes			
Other (____________________)	0=No 1=Yes			

Section F: Environmental resource utilization (past 12 months)

Type of resource	Harvest				Purchases		Sold	
	*Technology**	*Frequency***	*Quantity*	*Value*	*Quantity*	*Value*	*Quantity*	*Value*
Wild vegetables								
Mushroom								
Bush meat (specify ____________)								
Small animals								
Insects								
Fish								
Birds								
Flavours/cinnamon								
Wild fruits/berries								
Honey								
Medicines								
Pesticides and drugs for livestock								
Timber/construction material								
Firewood								
Furniture								
Household utensils								
Agricultural implements								
Leaf litter as manure/mulch								

Thatch grass								
Fibres/ropes								
Glue/Adhesive/Gum/ resins								
Baskets								
Mats								
Hats								
Pottery								
Sculpture carving								
Livestock fodder								
Common pastures								
Watering points								
Other (____________________)								
Other (____________________)								

* Tech: 1 = Machete 2 = Cart 3 = Basket 4 = Gun 5 = Bucket 6 = Bow & arrows 7 = Catapult 8 = 50kg bag 9 = Sickle 10 = Snare
** 1M = once a month, 1Y = once a year, 2M = twice a month, 2Y = twice a year and so on.

Section G: Wildlife and Natural Resource Management (NRM) Organisations

G1 Is there a wildlife management committee? 0 = No 1 = Yes 2 = DK

Is the committee functional? ____

G2 Does household head or any member of the family belong to a wildlife management organisation or committee? 0 = No 1 = Yes 2 = DK

a) Current position ____________________

1 = Chairperson 2 = Vice Chairperson 3 = Secretary 4 = Vice secretary 5 = Treasurer
6 = Committee member 7 = Ordinary member 8 = Security 9 = Other __________________

G3 Is household entitled to benefits from wildlife management? 0 = No 1 = Yes

Nature of benefits ____

Value of benefits (US$) ____________________

G4 Did you suffer any loss to wildlife during the past 12 months? 0 = No 1 = Yes

Area of crop destroyed (ha) _________________ Estimated vale (US$) _________

Number of domestic animals killed ____________ Estimated value (US$) ________

G5 Were you compensated for the loss? 0 = No 1 = Yes

a) What was the mechanism of compensation? ____________________________________

b) Estimated value of the compensation (US$) ____________________________________

G6 Do you consider poaching of wildlife resources to be a problem in this area? 0 = No 1 = Yes

Explain ____

G7 Has household consumed game meat within the past 12 months? 0 = No 1 = Yes

Number of times _______ Problem animals (kg) _______ Own (kg) ________

Trophy hunting (kg) _______ Buying (kg) _______

G8 How does household create and capture value through wildlife conservation and tourism activities? (*Please indication income by activity*)

____

____

____

G9 How frequent does the community hold meetings in a year? _______________

How frequent did you attend such meetings? ____________________________

G10 How many activities within the past 12 months? __________________________

How many activities did you attend within the past 12 months? _____________

G11 How would you rate the following issues?
(*Please use a scale from 0 to 10*)

	Rating
a) Extent to which household enjoy use rights (right to harvest)	
b) Existence of clearly defined boundaries/effective exclusion of external un-entitled parties	
c) Collective choice arrangements that allow resource users to participate in decision making process	
d) Extent to which decision makers are accountable to the community	
e) Existence of structures/organisations in the form of multiple layers of nested enterprises	
f) Extent to which the process of making rules is fair and transparent	
g) Community structures/organizations and rules recognised by higher level authorities	
h) Effectiveness of rules in managing wildlife resources and allowing access	
i) Existence of punishment/sanctions for members who violate rules	
j) Effective monitoring by people who are part of or accountable to community	
k) Extent to which village elites or groups members influence these rules in their favour	
l) Extent to which people responsible for punishing offenders are accountable to the community	
m) The mechanism of conflict resolution is fair and transparent	
n) The mechanism of conflict resolution is cheap and accessible to all members	
o) The mechanism of compensation for losses due to wildlife intrusion is fair and transparent	
p) Level of confidence with community leadership and committee (skills, trust, e.tc)	
q) The criteria for selecting committee members is transparent and fair	
r) Extent to which the disadvantaged members are represented adequately in wildlife committees	

G12 Are there any NRM organizations or committees (other than wildlife management committees) in your community that you know of?
0 = No 1 = Yes 2 = DK

G13 Does household head or any member of the family belong to any NRM organisations or committee?
0 = No 1 = Yes 2 = DK

G14 Provide details of the NRM organisations

	1st Organisation	**2nd Organisation**	**3rd Organisation**
Description (name, function, activities, year established e.tc)			
Existence of Rules and regulations			
Joining fee			
Constitution	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK
No. of meeting past 12months			
No. of Meetings attended			
No. of activities past 12months			
Activity participation rate			

Section H: Social capital and networking

H1 Have you or any member of the household joined any community group (whether formal or informal organisations) in the past three years other than NRM organisations? 0 = No 1 = Yes 2 = DK

H2 Provide details of the non-NRM organisations

	1st Organisation	2nd Organisation	3rd Organisation
Description (name, function, activities, year established e.tc)			
Existence of Rules and regulations			
Joining fee			
Constitution	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK
No. of meeting past 12months			
No. of Meetings attended			
No. of activities past 12months			
Activity participation rate			

H3 Numbers of year living in the area ____________

H4 On a scale of 0 – 10, to what extent could you say that most members of your community/village are trustworthy? ________

H5 Number of people that you can rely on for critical support in times of need?

a) Inside this village

Relatives _______________________

Non-Relatives ___________________

b) Outside this village

Relatives _______________________

Non-Relatives ___________________

H6 Are any of your relatives or friends in leadership positions in formal or informal institutions inside this village? 0 = No 1 = Yes 2 = DK

H7 Are any of your relatives or friends in leadership positions in formal or informal institutions outside this village? 0 = No 1 = Yes 2 = DK

H8 Have you or any member of your household exchanged gifts, labour, borrowed money or production inputs with other members of the community within the past 12 months? 0 = No 1 = Yes 2 = DK

	In			out		
What was the nature of this transaction?						
Number of times transaction occurred						
Labour days/time allocated to collective activities						
Quantity						
Value						

Nature: 1 = Food items 2 = Clothes 3 = Draught power 4 = Labour 5 = Fertilizer 6 = Manure 7 = Money 8 = Other _____

H9 Number of grain traders that you know in this village who could buy your grain _______

H10 Number of grain traders that you know outside this village who could buy your grain _______

H11 Generally speaking, can you say that most traders in your village can be trusted
1=Strongly disagree 2=Disagree 3=Agree 4=Strongly Agree

Number___________

KEY INFORMANT INTERVIEW

Introduction

Good morning/afternoon. My name is **Herbert Ntuli**, and I am currently studying for a PhD in Economics at the University of Cape Town. As part of my degree programme, I am currently carrying out a research study on **Community Wildlife Management involving local communities and private game farms around South Africa's Kruger National Park and Gonarezhou in Zimbabwe**. The major objective of this study is to enhance our understanding of the role of local institutions in promoting sustainable management of social ecological systems, compare the performance of wildlife management under private and common property systems and to learn from private systems, how local communities can adapt to commercial park activities. From this area, key informants (resource persons) have been selected and I am now in the process of discussing with people like you to get information about wildlife management in your community. This information is **confidential** and will only be used by myself for the purposes of this study which will not make reference by name to any one respondent. I will be grateful if you could assist me in filling out this questionnaire in as honest a manner as possible.

Section A: Identification

A1	Date of Interview ____________________	**A6**	Identification code ________________________
A2	Country 1 = South Africa 2 = Zimbabwe	**A7**	Enumerator code ________________________
A3	District ____________________	**A8**	Name (optional) ________________________
A4	Ward ____________________	**A9**	Capacity/Institution ________________________
A5	Village ____________________		

Section B: Wildlife Management

B1 How is your community/group/ward involved in wildlife management?

____

____

____

B2 Is there a joining fee to participate in wildlife management? 0 = No 1 = Yes Amount____________

B3 Number of ethnic groups in the community/ward/group (please identify them)

____

B4 Are individuals or members of the community able to enter wildlife management committee (group) and/or exit voluntarily? 0 = No 1 = Yes 2 = DK

B5 Is community able to exclude external or untitled parties? 0 = No 1 = Yes 2 = DK

B6 Which bundle of rights does the community/group possess? ____________________________________

a) Use rights 0 = No 1 = Yes 2 = DK

b) Management (decision making) rights 0 = No 1 = Yes 2 = DK

c) Both use rights and management rights 0 = No 1 = Yes 2 = DK

B7 Is the community/ward entitled to benefits from wildlife management? 0 = No 1 = Yes 2 = DK

Nature of the benefits ____

Value of benefits in 2012/2013 Hunting ______________

Tourism ______________

Other ______________

B8 Do you have a wildlife management plan as a group? 0 = No 1 = Yes 2 = DK

a) How was wildlife management plan developed? ____

b) Are members of your community/ward/group bound by a constitution to adhere to a wildlife management plan? 0 = No 1 = Yes 2 = DK

B9 How do you normally communicate important information regarding wildlife activities?
1=Meetings 2=Word of Mouth or face-to-face 3=Other (___________________)

B10 a) What type of information is normally communicated during meeting?

			b) What type of information is difficult access and why?
Information about past actions	0 = No	1 = Yes	________________________________
Financial reports	0 = No	1 = Yes	________________________________
Progress reports	0 = No	1 = Yes	________________________________
Knowledge of the SESs	0 = No	1 = Yes	________________________________
Other (_______________________)	0 = No	1 = Yes	

B11 How frequent does the community hold meetings in a year? ______________

Meeting participation rate withing past 12 months ______________

B12 Number of activities within the past 12 months ______________

Meeting participation rate withing past 12 months ______________

B13 Who makes the vital decisions about the utilization and management of wildlife?
0 = DK 1 = Local communities 2 = Government/State agencies 3 = Private sector
4 = NGOs 5 = RDC 6 = Committee 7 = All stakeholders 8 = Other __________

a) Decisions about tourism development in the community

____

b) Who is involved in negotiating tourism business ventures

____

c) Harvesting levels or allocation of hunting licences

____

d) Decisions about how revenue is shared among stakeholders

____

e) Decision about how wildlife income is used in the community

____

f) How much was allocated to the community last year (US$) ________________

B14 How are decisions made/what processes are used for choosing actions? _________________________

____

B15 Tell me about the rules/regulation that the committee has developed for managing wildlife resources

____

____

a) Number of rules and regulations _________________

b) Are rules recognised by all members of the community? 0 = No 1 = Yes 2 = DK

c) Are rules recognised by higher level authorities? 0 = No 1 = Yes 2 = DK

B16 Who is involved in setting the rules for access and management of wildlife within the community?

0 = DK 1 = Local communities 2 = Government/State agencies 3 = Private sector

4 = NGOs 5 = RDC 6 = Committee 7 = All stakeholders 8 = Other ________

B17 How are rules regarding access and management of wildlife resources **set** and who is responsible for **monitoring** & **enforcing** the rules?

____

____

____

____

B18 Is there punishment to people who violate wildlife management rules? 0 = No 1 = Yes 2 = DK

a) Who is responsible for monitoring and sanctioning offenders? ________________________________

b) What is the nature of punishment? ____

B19 Does community have mechanisms in place for conflict resolution? 0 = No 1 = Yes 2 = DK

Explain ____

B20 How do you deal with offenders who are not members of your community, group or outsiders (not involved in wildlife management)?

____

Section C: Organisational structures and the role of other stakeholders

C1 How is the community actually organised?

a) Structures at village level

Number of people in the committee _______________

Positions/offices

1. ___________________________ 2. ______________________ 3. ________________________

4. ___________________________ 5. ______________________ 6. ________________________

7. ___________________________ 8. ______________________ 9. ________________________

Number of Men _________________, *Women* _______________ *and Youths* _________________

b) Structures at ward/district level

Number of people in the committee _______________

Positions/offices

1. __________________________ 2. ______________________ 3. _______________________

4. __________________________ 5. ______________________ 6. _______________________

7. __________________________ 8. ______________________ 9. _______________________

Number of Men _________________, *Women* _______________ *and Youths* _________________

C2 What role does the committee play in community wildlife management?

a) At village level

Functions ____

Activities ____

b) At ward/district level

Functions ____

Activities ____

How are community members selected? ____

C3 Do you think local communities have the capacity to manage wildlife efficiently?
(*Please comment on the institutional capacity, organizational skills and information processing capabilities*)

____

____

____

C4 Please identify all the important stakeholders and how they are involved in community wildlife management (**CWM**)

Stakeholder	**Their role (function) in CWM**	**Main activities**	**How does their involvement affect the community?**
1.			
2.			

3.			
4.			

C5 Identify the major institutional and legal constraints facing local communities in the wildlife sector?

Nature of constraint	How does this affect local communities?	What could be done?
1.		
2.		
3.		
4.		

Section D: Business models, investments and community infrastructure

D1 How does the community create and capture value from wildlife conservation?
(*e.g., lodge, hotels, hunting, types of tourism products, business venture, e.tc*)

____

____

D2 What type of investments have the community made so far?

____

____

D3 Description of assets and infrastructure belonging to the community ___________________________

____

D4 How did you finance these investments? ____

Does community have access to credit? ____

D5 Does the community have the means to advertise their products, attract and track tourists from overseas markets? 0 = No 1 = Yes 2 = DK

Explain ____

Section E: Enumeration of organisations in the community
(*other than wildlife management organisations***)**

E1 Natural resource management (NRM) organizations/committees in the community

	Org 1	**Org 2**	**Org 2**	**Org 3**	**Org 4**
Type of organisation (name, function, year registered)					
How was organisation created?					
Membership					
Management structure					
Rules and regulation					
Activities					
No. of meetings past 12 months					
Meeting participation rate					
No. of activities past 12 months					
Activity participation rate					
Constitution	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK

E2	Non NRM organizations/committees in the community				
	Org 1	**Org 2**	**Org 2**	**Org 3**	**Org 4**
Type of organisation (name, function, year registered)					
How was organisation created?					
Membership					
Management structure					
Rules and regulation					
Activities					
No. of meetings past 12 months					
Meeting participation rate					
No. of activities past 12 months					
Activity participation rate					
Constitution	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK	0 = No 1 = Yes 2 = DK

Section F: Local Ecological Knowledge (LEK) and harvesting levels

F1 How big is the area under the community/ward/village's jurisdiction or resource system?

Area (ha) ____________________

Is the boundary clearly defined? 0 = No 1 = Yes 2 = DK

Has the areas under your jurisdiction reduced, remained constant or increased over the past five years?

1 = Reduced 2 = Remained constant 3 = Increased

By how much has the area reduced or increased? ha ________________

F2 What is the distance to the nearest game park? km ________________

F3 What is your perception of the value of the resource system to the community/ward/village?

Use the scale

0	1	2	3	4	5	6	7	8	9	10

F4 Which commercial species or animals are very common in this area?

____

____

F5 Which species or wild animals are more productive?

____

____

F6 Generally speaking, can you say these species have exceeded the carrying capacity of the resource system or the resource system is in equilibrium?

____

____

F7 Which species or wild animals are less productive?

____

____

F8 Which animals can you classify as endangered species in this area?

____

____

F9 What is the subjective probability of extinction for each species identified by respondent and endangered?

____

F10 Do you consider illegal harvesting of wildlife resources to be a problem in this area? 0 = No 1 = Yes 2 = DK

Explain ____

____

a) Number of incidences past 12 months ____________________

b) What type of wild animals are normally killed by poachers and why?

____

c) Has illegal harvesting of wildlife resources decreased, remained constant or increased over the past five years?
1 = Decreased 2 = Remained constant 3 = Increased

d) Are the people involved members of the community or not 0=No 1=Yes 2=DK

e) What mechanism does the community have in place to curb illegal harvesting of wildlife resources?

____

____

F11 When we consider animals that are killed through tourism activities and poaching, do you think the current harvesting levels are sustainable for different species?

____

____

F12 Which species are more resilient to shocks such as overharvesting and climatic variables?

____

F13 What has been happening to commercial wildlife species in your area/community/ward/village over the past five years?

Species	**General Observation**	**Animal Population**		% Change
		2009	2013	
Elephants				
Black Rhino				
White Rhino				
Buffalo				
Giraffe				
Zebra				
Wildebeest				
Eland				
Kudu				
Antelope				
Waterbuck				
Bushbuck				
Springbok				
Warthog				
Lion				
Leopard				
Hyena				
Cheater				
Wild dogs				
Other				

General observation: 0=Don't Know 1=Increased 2=Decreased 3=Constant

F14 Harvesting levels of diverse users past 12 months

Species	tourism activities		Illegal harvesting by community members**		Illegal harvesting by outsiders	
	No. killed	Technology	No. killed	Technology	No. killed	Technology
Elephants						
Black Rhino						
White Rhino						
Buffalo						
Giraffe						
Zebra						
Wildebeest						
Eland						
Kudu						
Antelope						
Waterbuck						
Bushbuck						
Springbok						
Warthog						
Lion						
Leopard						
Hyena						
Cheater						
Wild dogs						
Other						

Technology: 1=Gun 2=Bow & arrow 3=Wire snares 4=Trap nets 5=Pitfall traps 6=foothold traps 7=Spears 8=Other _____
** Including problem animals killed by the community

F15 Other comments ____

____

____