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THE ECONOMICS OF COMMUNITY-BASED WILDLIFE CONSERVATION IN ZIMBABWE

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Edwin Muchapondwa

To Cheryl Tatenda, with love

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List of Abbreviations

AA	Appropriate Authority
AERSG	African Elephant and Rhino Specialist Group
AGRITEX	Agricultural, Technical and Extension Services Department
BCA	Benefit-Cost Analysis
CA	CAMPFIRE Association
CAMPFIRE	Communal Areas Management Programme for Indigenous Resources
CASS	Centre for Applied Social Sciences
CCG	CAMPFIRE Collaborative Group
CDF	CAMPFIRE Development Fund
CITES CoP	Convention of International Trade in Endangered Species of the Flora and Fauna Conference of Parties
COP 11	11^{th} Conference of Parties for CITES
CVM	Contingent Valuation Method
DNPWLM	Department of National Parks and Wildlife Management
DNR	Department of Natural Resources
ETIS	Elephant Trade Information System
FPL	Food Poverty Line
IC	Institutional Contractor
ITRG	Ivory Trade Review Group
IUCN-ROSA	World Conservation Union-Regional Office for Southern Africa
LIMDEP	Limited Dependent Variables Computer Software
MIKE	Monitoring of Illegal Killing of Elephants
MLGRUD	Ministry of Local Government, Rural and Urban Development
NGO	Non-Governmental Organisation
NOAA	National Oceanic and Atmospheric Administration
ODA	British Overseas Development Association
PAC	Problem Animal Control
RBZ	Reserve Bank of Zimbabwe
RD	Rural District
RDC	Rural District Council
SAFIRE	Southern Alliance for Indigenous Resources
TCPL	Total Consumption Poverty Line
UMP	Uzumba Maramba Pfungwe
USA	United States of America
USAID	United States Agency for International Development
VIDCO	Village Development Committee
WADCO	Ward Development Committee
WINDFALL	Wildlife Industries New Development For All
WTA	Willingness To Accept
WTP	Willingness To Pay
WWF SARPO	World Wide Fund for Nature Southern Africa Regional Programme Office
ZANU PF	Zimbabwe African National Union Patriotic Front
ZIMTRUST	Zimbabwe Trust
ZW	Zimbabwe

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Abstract

This thesis deals with the economics of community-based wildlife conservation in Zimbabwe. In Zimbabwe, community-based wildlife conservation takes place under the banner of the communal areas management programme for indigenous resources (CAMPFIRE). The thesis consists of an introductory chapter and four self-contained papers, which make up the rest of the thesis chapters.

Chapter 1 gives the general geographical and ecological features of Zimbabwe and spells out the history of wildlife management. An overview of CAMPFIRE is given together with an examination of the implications of wildlife trade bans on community-based wildlife conservation. The chapter concludes by highlighting challenges facing community-based wildlife conservation in Zimbabwe and research issues that will be the subject matter for the rest of the thesis.

Chapter 2 formulates a bio-economic model with two agents (the parks agency and the local community) and two land uses (wildlife conservation and livestock production) to analyse the conflict between wildlife conservation and livestock production and welfare implications in a typical rural area in Zimbabwe, where a local community lives adjacent to a safari area. The parks agency has a fixed amount of land, which is the permanent residence of wildlife, while the local community has user rights over the remaining land. Wildlife tends to roam around the adjacent land imposing a negative externality on the local community's livestock production. Thus a conflict arises in that the parks agency desires to expand its enterprise thereby encroaching onto the local community's share of land creating a nuisance. Some locals tolerate poaching in order to reduce the number of wildlife. We analyse the wildlife-livestock conflict and the resultant welfare in three resource use regimes: (i) the local community does not reap any benefits from wildlife, (ii) the local community gets profit shares from wildlife hunting and tourism, and (iii) a socially optimal arrangement. Wildlife conservation is shown to be more successful under regimes in which the local community gets profit shares from hunting and tourism but at a potential cost of the local community's welfare. Policies that could enhance wildlife conservation and social welfare are suggested. Relaxing the assumption of fixed and exogenous profit shares shows that optimal profit shares from hunting and tourism ought to exceed unity. Thus, devolution of wildlife conservation to the local community should be augmented by inflows of external funding.

If the local communities who live side by side with the elephant see it as valueless nuisance then they cannot be trusted to be its good stewards. To assess their valuation of it, **Chapter 3** presents a contingent valuation study that was conducted for the case of one CAMPFIRE district, Mudzi, in Zimbabwe. An approach that can evaluate projects that generate both winners and losers is used. The study shows that the median willingness to pay for the preservation of an elephant population of 200 is ZW$300 (*US$5.45*) for the respondents who consider the elephant a public good while the same statistic is -ZW$98 (-*US$1.78*) for the respondents who consider the elephant a public bad. The preservation of an elephant population of 200 in Mudzi yields an annual net worth of ZW$123,771 (*US$2,250*) to the households living in CAMPFIRE wards. However, the majority of households do not support the preservation of the current

elephant population since 62% of them would rather not have the elephant because they view it as a nuisance. This is one argument against devolution of elephant conservation to the local communities. The rural communities' perceptions of the elephant are generally useful for other species of wildlife since the elephant is considered a keystone species and, most importantly, an umbrella species in the African Savannas. Adequate economic incentives must be extended to the local communities if a majority of them is to be persuaded to partake in sound elephant conservation. External transfers constitute one way of providing additional economic incentives to encourage elephant conservation by local communities such as Mudzi. Co-management should be the preferred mode of communities' involvement in wildlife conservation.

Chapter 4 notes that Zimbabwe faces an increasing incidence of poverty with the poorest areas being wildlife-abundant rural districts where the sustainable use of wildlife and other natural resources could greatly reduce rural poverty. Despite significant gains that CAMPFIRE has recorded it has not significantly alleviated rural poverty because of the current low levels of monetary benefit and local participation, among other problems. With reforms, CAMPFIRE could enhance sustainable wildlife conservation and consequently reduce rural poverty. Our starting point in search for potentially beneficial reforms in CAMPFIRE is an investigation of the extent to which the design principles that are shared by the institutions of the world's long-enduring common pool resources are satisfied. Our investigation suggests that the large-scale and irreversible nature of wildlife ecologies require co-management for effective long-term sustainable resource management. Most importantly, increased local communities' contestations should be promoted. The potentially beneficial reforms needed in CAMPFIRE consist of specific actions that honour and encourage the formation of institutions satisfying the design principles such as: congruence between clearly defined resource and governance boundaries; congruence between appropriation and provision rules and local conditions; collective choice arrangements and localised monitoring.

Chapter 5 focuses on risk management in agricultural production. Risk faced by rural farmers in agricultural production could potentially be managed in two ways. Firstly, adding wildlife conservation as a land use in the framework of CAMPFIRE could diversify and subsequently reduce risk, particularly where evidence suggests that wildlife conservation is a feasible hedge asset. Risk management through diversification into wildlife conservation could help farmers but it could also help efforts to conserve wildlife. Secondly, establishing a wildlife damage insurance programme would assist farmers, particularly those living in less marginal areas where the benefits of diversification into wildlife conservation are likely to be low. A complement to the insurance programme could be an investment in electric fences and buffer zones to reduce the likelihood and severity of loss. Without detailed empirical investigations we can only speculate that highly marginal and wildlife-abundant districts would benefit more from diversification into wildlife conservation as a risk management strategy while the remaining wildlife-endowed districts would benefit more from the wildlife damage insurance.

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Edwin Muchapondwa
Gothenburg, Sweden, 1 June 2003

THE PRECAUTIONARY PRINCIPLE:

If we live as if it matters and it does not matter, then it does not matter
If we live as if it does not matter and it matters, then it matters.

(anonymous)

AN OVERVIEW OF COMMUNITY-BASED WILDLIFE CONSERVATION IN ZIMBABWE

Edwin Muchapondwa[1]

Abstract
In this paper we give the general geographical and ecological features of Zimbabwe and spell out the history of wildlife management. In Zimbabwe, community-based wildlife conservation takes place under the banner of the communal areas management programme for indigenous resources (CAMPFIRE). An overview of CAMPFIRE is given together with an examination of the implications of wildlife trade bans on community-based wildlife conservation. The paper concludes by highlighting challenges facing community-based wildlife conservation in Zimbabwe and research issues that will be the subject matter for the rest of the thesis.

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1. Introduction

Zimbabwe has a total land area of about 39 million hectares of which 33.3 million hectares are suitable for agricultural purposes and the remaining 6 million hectares have been reserved for national parks, wildlife reserves and urban settlements. The country first became a political entity as Rhodesia, under British influence, towards the end of the 19th century. It was the British South Africa Company, under Cecil John Rhodes, that negotiated with the native leader, Lobengula, for the right to mine and settle; formally annexing the country from 1890 (Child 1995). The Republic of Zimbabwe got independence from Britain on 18 April 1980, after 15 years of international ostracism and economic sanctions on the Rhodesian regime, and a protracted civil war (Child 1995). At independence agricultural land was divided along racial lines as follows: (i) about 15.5 million hectares, which is almost half the total agricultural land in the country, controlled by 6,000 white large-scale commercial farmers and (ii) about 16.4 million hectares controlled by 840,000 communal farmers. This uneven land distribution pattern between the large-scale commercial and small-scale communal areas also extended to the suitability of land for agricultural purposes.

Water is a major constraint to national development since about 80% of Zimbabwe is semi-arid averaging less than 800mm of rainfall a year (Child 1995). Frequent mid-season dry spells and annual droughts have a major influence on natural productivity. The division of Zimbabwe into five agro-ecological regions of generalized land use potential is based mainly on the amount and reliability of the rainfall (*see table below*). Of the total communal land, more than three-quarters is located in low rainfall regions IV and V, where the potential for crop agriculture is limited. It implies that rural economies in about 80% of the country should be based on harvesting the natural vegetation using either wild or domestic herbivores, unless crops can be grown with supplementary irrigation (Child 1995). In the large-scale commercial sector, over half of the total land is located in the high rainfall regions I, II and III, where agricultural potential is very high.

Table 1: Natural Agro-Ecological Regions in Zimbabwe

Natural Region	Annual Rainfall	Parks & Wildlife Estate (Hectares)	Forest Estate (Hectares)	Zimbabwe (Hectares)	%
I	above 1000mm reliable	50,000	70,000	705,000	1.8
II	750 to 1000mm reliable	25,000	20,000	5,875,000	15.0
III	650 to 750mm erratic	545,000	145,000	7,290,000	18.7
IV	450 to 650mm very erratic	2,510,000	620,000	14,770,000	37.8
V	below 450mm unreliable	1,840,000	70,000	10,450,000	26.7
TOTAL	-	4,970,000	925,000	39,090,000	100.0

Source: Child 1995, table 1, page 15

In an effort to redress the inherited imbalance in the land distribution pattern, in 1980 the government embarked on a resettlement programme. Land acquisition proceeded in the spirit of the 1979 Lancaster House Constitution's "willing seller, willing buyer" clause, which could not be changed within 10 years since independence in 1980. The targeted beneficiaries were people in communal areas, war-displaced people and Zimbabwean refugees. As a result of the implementation of the resettlement programme, about 73,000 families out of a target of 162,000 families were resettled on 3.5 million hectares, 0.5 million hectares of which was former state land in the large and small-scale commercial sectors. By the late 1980s, the land distribution pattern was as follows; (i) 1.4 million hectares owned by 10,000 small-scale commercial farmers, (ii) 11 million hectares of land owned by the large-scale commercial farmers, (iii) 0.5 million hectares owned by the state farming sector, (iv) about 16.4 million hectares controlled by 1 million communal farmers, and (v) 3.5 million hectares controlled by about 73,000 resettled families. In spite of the progress made, there was still a huge demand for land among the rural peasant and other landless groups, with over 524,890 families awaiting resettlement.

Most commercial farmers were plainly unwilling to sell any excess land because, among other things, they could not repatriate all proceeds from the sale of land due to foreign exchange controls while others overpriced their land twice or thrice over. The government was powerless in the face of the farmers' resistance because of the "willing seller, willing buyer" clause. Britain, who had, from the onset of the land reform programme, been a partner who had agreed to fund 50% of the land purchases contributed about £44 million. The Land Acquisition Act of 1992 was enacted to speed up the land reform process by removing the "willing seller, willing buyer" clause. The Act empowered the government to buy land compulsorily for redistribution, and a fair compensation was to be paid for the acquired land. Landowners were given the right to go to court if they did not agree with the compensation set by the acquiring authority. In an apparent protest to the Land Acquisition Act of 1992, Britain withdrew her aid to the land reform programme, accusing the government of giving the land exclusively to high ranking government and ruling party officials. The Act had a limited impact largely because the government, going it alone, did not have the money to pay compensation to landowners and most compulsory land acquisitions were contested in court.

In 1998 the government made a decision to compulsorily purchase 5 million hectares of land over five years as part of the second phase of the land reform and resettlement programme. Land was to be identified on the basis of the following criteria: (i) under utilization, (ii) dereliction, (iii) multiple ownership, (iv) absentee ownership and (v) proximity to congested communal areas. 841 farms were served with acquisition orders but the government failed to make the applications for compensation orders within the legally stipulated time. As a result land reverted to its owners. The Commercial Farmers Union, an organisation that represents, protects and advances the interests of commercial farmers and furthers the development of an economically viable and sustainable agricultural industry in Zimbabwe, subsequently offered 1.5 million hectares of land from its members for sale to the government for redistribution. The land reform programme had to move on but landowners once again dragged their feet in offering more land to the government. As frustration set in on both sides, in 2000, the government drafted a new constitution with a clause to compulsorily acquire land for redistribution without paying compensation, except with respect to improvements made on the land. The proposed Constitution failed to win by 55% of the votes in a referendum. Immediately after the defeat of the proposed Constitution, the government quickly moved to amend the old Constitution and the Land Acquisition Act (1992) with a clause to compulsorily acquire land for redistribution without paying compensation, except with respect to improvements made on the land. Following that amendment, a long list of farms became the initial target for a new mode of acquisition where compensation is only paid with respect to improvements made on the land, rather than for the land itself. This, coupled with decisive demonstrations by the Zimbabwe "liberation war veterans" and the land hungry peasants who spontaneously occupied commercial farms throughout the country, caused a huge outcry in the country and internationally. A proper audit of the land redistribution that was carried out in this wave of farm occupations is yet to be finalised. Initial reports indicate that the government acquired close to 10 million hectares of land and more than 352,000 households were resettled, with priority having been given to people who were living in congested rural areas in the country. About 453 commercial farmers, out of 4,137 who worked the land before the start of the controversial land reforms three years ago, are reported to be still fully operating their farms, while another 666 are partially operational. Aggregate agricultural output has been scaled down by approximately 50% and about 200,000 farm jobs have been reported lost in 2002. Zimbabwe's food security and foreign currency earnings have been hit by the decline in agricultural output. Close to eight million Zimbabweans are in need of emergency food aid because of the combination of the impacts of the land reform programme and drought that hit southern Africa in 2001/2002.

The dominant factors leading to land hunger in agrarian economies such as Zimbabwe include rapid population growth, deteriorating fertility of the land and the declining terms of trade for most agricultural commodities. In Zimbabwe, the human population has burgeoned from around 0.5 million in 1900 to over 12 million at present. Over half of the population lives in poverty in overcrowded rural areas. Agriculture will not be able to support the majority of the people and innovative solutions will have to be found to enhance rural production without further environmental damage. The wildlife-based[2] industry, especially tourism, has emerged as an increasingly important asset in the national economy of Zimbabwe. The value of this industry has not been quantified, but a conservative estimate is US$125 million annually (Child 1995). Its potential is probably much larger but the severe conflicts concerning land are a constant threat to the industry too. The land reform programme brought in its wake an increase in poaching over the past two years, particularly in private wildlife conservancies. In some cases this has resulted in the loss of highly endangered and economically valuable species such as the rhinoceros. It has been reported that the Government will introduce a Wildlife Land Reform Policy to provide newly resettled indigenous farmers with options of utilising wildlife in a way that is economical and environmentally sustainable[3].

Sound environmental management and land use are crucial for the sustainable development of Zimbabwe. Resource depletion and environmental degradation are accelerating threats to the quality of life in Zimbabwe. Despite a tradition of caring which goes back over 50 years, the destructive processes are accelerating and placing all renewable resources under threat (Child 1995). In particular, the challenges of wildlife and protected areas management in Zimbabwe must be resolved if the resources and the biological diversity they represent are to survive and prosper. There are also difficult issues concerning wildlife management itself. Of particular concern is the speed at which some of the habitats in protected areas are being modified by wildfire and overpopulation of elephants. Elephant damage is most apparent in Chizarira, Gonarezhou, Hwange, Mana Pools, Matusadona and Zambezi.

Species conservation in Zimbabwe and elsewhere has relied too heavily on passive legal protection of a given portion of species' range. Preserving genetic diversity requires a much more innovative and versatile approach. Advice emerging from genetic theory warns that many populations have a minimum size below which there is fatal genetic depression if a population remains at low numbers (Child 1995). Clearly, protected areas can never be large enough to contend with all such eventualities. The only hope

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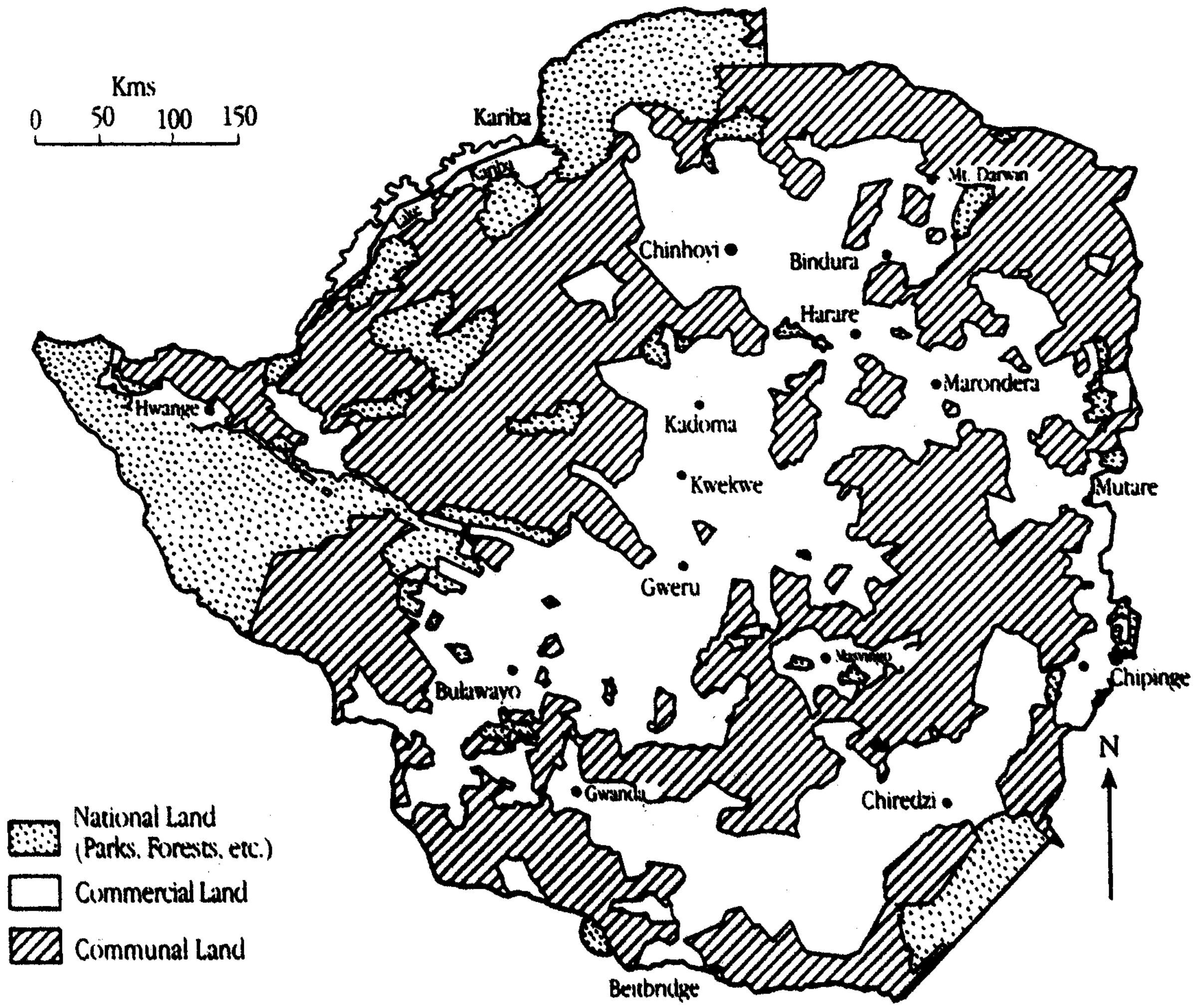
[2] While the term 'wildlife' generally means animals and plants that grow independently of people, usually in natural conditions, we use it in this thesis to mean wild animals, unless otherwise stated.
[3] The Herald, 16 May 2003.

lies in extending the areas in which wild animals can survive beyond the borders of parks and reserves (Child 1995). This will be possible only if the people owning that land are prepared to tolerate the animals and preserve their habitats. Conservation of wildlife requires the active co-operation of adjacent rural landholders. For wildlife and protected areas to survive on a significant scale they must be socio-politically acceptable, economically viable and ecologically sustainable (Child 1995, Child *et al* 1997). If wildlife is to become a viable and alternative land use, those people who live with it and therefore bear its "accommodation costs" must get some economic benefit out of it. The resource will benefit on a large scale only where these people have an incentive to conserve it. It is expected that this would require that the rights to use the resource are allocated to them and they have freedom to use and trade the products generated by the resource.

The three main land use categories, that all have good wildlife populations are depicted in the figure below.

Figure 1: Land Classification in Zimbabwe



Source: Child 1995, figure 3, page 16

The rest of the paper is arranged as follows: Section 2 gives an overview of the history of wildlife management in Zimbabwe, putting into perspective the role of communities in wildlife conservation. Section 3 briefly examines the implications of wildlife trade bans on community-based wildlife conservation and Section 4 concludes by highlighting challenges facing community-based wildlife conservation in Zimbabwe and research issues that will be the subject for the rest of the thesis. For detailed analyses of the wildlife policy in Zimbabwe, the interested reader is referred to Child (1995), Duffy (2000), and Hulme and Muphree (2001).

2. The History of Wildlife Conservation in Zimbabwe

2.1. Before 1890

Prior to colonial settlement in 1890, the human population was small and wildlife and other renewable resources were relatively plentiful. Apart from traditional management by humans, wildlife populations were kept in check by diseases such as Rinderpest[4] and the absence of adequate surface water (WWF SARPO 2000). Wildlife utilization was an integral part of community life. It provided an important source of meat for subsistence farmers, especially in times of drought. Commercial elephant hunting for ivory had been going on since the 1500s, mainly in the Zambezi Valley (WWF SARPO 2000)[5]. Elaborate measures to regulate the use of the resources were unnecessary and the institutions relating to wildlife were non-complex, mostly customs and beliefs. There was, for example, a taboo against people eating the meat of their totem species and the parts of some animals were reserved for the traditional Chief. Anyone coming into possession of elephant ivory, pangolin or a leopard skin was required to deliver it to the ruler (Child 1995). Other species such as hammerkops, chameleons and hyaena were protected through religious respect or superstitious fear. Most traditionally protected areas were for religious purposes. Although non-complex, these mechanisms were sufficient to protect wildlife while it remained plentiful and could be hunted only with primitive weapons and traps by a sparse population moving about on foot. It is estimated that there were not more than 3,000 elephants throughout the country (WWF SARPO 2000).

2.2. 1890 to 1977

Wildlife also became an important source of food for early explorers, missionaries and colonizers upon colonization in 1890. Wildlife immediately became the responsibility of the British Crown on behalf of the people in Zimbabwe (Child 1995). With time, more and more species could be hunted only under a permit issued by a local bureaucrat

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[4] Rinderpest is a contagious viral disease of cattle, domestic buffalo, and some species of wildlife. It is characterized by fever, oral erosions, diarrhoea, lymphoid necrosis, and high mortality.

[5] For the interested reader, an extensive discussion of the elephant is given in the appendix.

on behalf of the British Monarch. The first wildlife legislation, the Game Law Amendment Act (1891), provided protection to certain animals. Further legislation, especially the Game and Fish Preservation Act (1929) and the Wildlife Conservation Act (1960) alienated all farmers from wildlife and vested its control and management with the state. Farmers with wildlife received no compensation for any losses they suffered from having the Crown's animals on their land. The Game and Fish Preservation Act (1929) allowed for the creation of the first protected areas in Zimbabwe namely Wankie, Victoria Falls and Urungwe Game Reserves (WWF SARPO 2000). Between 1931 and 1935, a total of nine game reserves were established, forming the basis of the current network of protected areas in Zimbabwe. The systems of protected areas emerged from this desire to have more control over wildlife use. Consequently many farmers developed very negative attitudes to wildlife as a productive and useful resource.

The State's implicit message was thus clear – wildlife should be eliminated outside protected areas because of its negative impact on agricultural production. However, significant wildlife populations survived in remote, sparsely populated, communal areas and on a number of farms especially if it did not impinge heavily on the farmers' livelihoods. Tolerance was limited mainly to herbivores, which conflicted least with human interests. As the human population grew and agriculture spread, easily accessible wildlife became scarcer and the law[6] more comprehensive and restrictive (Child 1995). This wildlife conservation strategy tended to focus on elimination of subsistence demand of local communities as a major conservation measure, a focus that brought in its wake serious conflicts (Gadgil and Rao 1995). Thus between 1890 and the mid-1970s, all wildlife was regarded as state property in Zimbabwe. There were problems managing this resource under the state property rights regime because, wildlife being a fugitive resource, would entail enormous monitoring costs to guard against

[6] Instruments in the management of wildlife and protected areas in Zimbabwe include the Trapping of Animals (Control) Act, which aims to control the trapping of both wild and domestic animals; the Bees Act, which provides the legal framework to encourage, regulate and safeguard the lucrative bee-keeping industry; the Quelea Control Act, which facilitates the control of the highly gregarious quelea finches that are a serious pest to grain farmers; the Natural Resources Act; the Forest Act, for the protection of vegetation and the control of fire; the Museums and Monuments Act, for the protection of archaeological and historical sites; and the Development of Tourism Act, which allows for the regulation of outfitters and others offering services based on wildlife or protected areas.

encroachment or poaching. In particular, there was little support to state-sponsored conservation efforts by the local communities.

While in other areas, during the 1940s and 1950s, the total extermination of wildlife was the main way of clearing tsetse fly, large game animals were becoming increasingly scarce in other areas and existence became confined to the few game reserves that had been created (WWF SARPO 2000). Even these surviving wild animals were under threat from agricultural expansion by such means as land clearance, increased competition and disturbance from livestock, fencing, or limiting access to water (Child 1995). Protectionist legislation that sought central Government control over wildlife and relied on enforcement was failing. Failure of the old game laws to accept that conserving wildlife involved costs, introduced inequities that discriminated against landholders with wildlife, by obliging them to protect it. In the case of wildlife conservation, Child (1995) argues that the requirement that a portion of society should bear a cost on behalf of the whole society is unjust and likely to lead to resentment. The old legislation discriminated against people with wildlife on their land by imposing the costs of conservation on them, rather than rewarding them for protecting the resource.

In 1975 the government allowed private property holders to claim ownership of wildlife on their land and to benefit from its use through the Parks and Wildlife Act of 1975. The private property holders were given what is called the Appropriate Authority (AA) status. The Parks and Wildlife Act (1975) did not accord any claims of ownership to farmers in communal lands because of the nature of their land tenure. The rights that existed under communal tenure for a community to use the common property resources anywhere in the communal area were a serious constraint against the devolution of the AA status. That reality negated the ability of a community to allocate the resources in its part of the communal area exclusively to its members so that the resources could be managed better.

As a direct result of being allowed to exploit wildlife, many commercial ranchers chose to replace livestock monocultures with a diversity of species, and prior to the current phase of land reform some 75% of ranches in drought prone areas incorporated wildlife as a farming enterprise, usually alongside cattle but increasingly on its own (Child, *et al* 1997). The 1975 legislative change did not eliminate the need for a central wildlife agency as there are many regulatory and co-ordinatory functions that can only be performed centrally, such as the regulation of trade. Wildlife outside protected areas in Zimbabwe began to recover due to the Parks and Wildlife Act (1975) initiative as wild populations became as asset instead of a liability for the private property holders. Effectively, the protected area expanded from the 15% of the country's land that is designated for parks. Wildlife in the Communal Areas was however in rapid decline mainly due to poaching and habitat loss to crops and livestock.

2.3. 1978 to 1981

In the late 1970s an operation named Wildlife Industries New Development For All (WINDFALL) was launched under which revenues from safari hunting in Communal Areas, that were still managed by the State through the Department of National Parks and Wildlife Management (DNPWLM), and meat from elephant culling in certain protected areas such as Chizarira and Chirisa and ex-Communal Areas, were to be given back to the adjacent communal land administrative authorities, Rural District Councils[7] (RDCs) for distribution to their inhabitants. WINDFALL as an experimental programme in the wildlife abundant areas of Binga, Gokwe and Nyaminyami, belonging to an informal 4 million hectare region lying to the south of Lake Kariba known as Sebungwe, was aimed at eliminating the conflict between people and wildlife. In principle, the Parks and Wildlife Act (1975) had made the necessary provision for returning revenues to communal area administrators also but that had been blocked politically until the propagation of the WINDFALL initiative (Child 1995). The revenue from wildlife safari hunting went into a central fund administered by the Central Treasury and at times benefited communal areas anywhere in the country. The RDCs had to come up with community developmental projects that would be financed from the central fund. By making such a requirement Central Treasury was effectively saving

[7] The terms RD, RDCs and local communities are not necessarily interchangeable. The term RD is used to denote the territory of communal area inhabitants (10,000 to 50,000 households) while RDC is the communal area inhabitants' administrative body, which is made up of representatives elected from sub-district structures called wards. The RDC is a legal institution created by an Act of Parliament while local communities have no legal status at all.

itself the cost of these projects and denying payments to the individual peasants who bore the opportunity costs of the revenue generated. Use of wildlife unlike the use of other communal land resources, was being implicitly taxed.

Wildlife funded development projects were welcomed and WINDFALL did briefly reduce poaching, but eventually the projects were seen as a rightful expectation from government, instead of as a product from wildlife. Furthermore, as revenues from wildlife remained with the RDCs without a significant trickle-down effect, the people living alongside the wildlife had trouble appreciating the benefits, and moreover, as they were excluded from the management process, wildlife was still of no interest to them. Direct benefits to individuals did not exceed the opportunity costs of having wildlife given the periodic nature of elephant culls, and the connection between these benefits and the game animals were insufficiently well defined to change attitudes towards wild animals (Child 1995).

2.4. 1982 to present : CAMPFIRE in Zimbabwe

In many cases, when the parks and other protected areas were established since 1929, local communities were evicted from their homes and told that they were not allowed to harvest wild animals and plants, as they had done for centuries. However, these same animals ruined their livelihoods by destroying their crops and livestock, and from time to time injuring or killing their relatives and friends. To be complete, the social costs with regard to wildlife management are high and take the following forms (i) crop damage, (ii) livestock crowd-out, injury and predation, (iii) human threat, injury and death especially by lion, leopard, buffalo and elephant, (iv) opportunity costs of the land on which they live, (v) social instability due to fear of wild animals, (vi) direct management costs, and (vii) loss of leisure time as people have to sleep in fields guarding against wildlife intrusions during cropping seasons. Conflicts arose between rural people and national parks staff, and some rural dwellers supported illegal harvesting of wildlife either in order to reduce damages they suffered or to profit from illegal sale of wildlife products. Local people treated poachers as heroes, particularly those who killed animals raiding their crops or competing with their livestock (Child *et* *al* 1997). The rapidly increasing rural population resulted in competition for land between agriculture and wildlife. Communities converted natural habitat to crops and livestock pasture. Conservation was therefore forced to rely on guns and guards, making it a very expensive exercise (Gadgil and Rao 1995).

In an attempt to resolve the human-wildlife conflict and to restrict damages, losses, injuries or deaths occasioned by wildlife, the following measures have been used (i) erection of game fences or the use of other deterrents, and this has been viewed as an optimal strategy though it is costly both with regards to the initial outlay and maintenance, (ii) eradication of wildlife has been applied mainly to communities whose lives are directly most threatened by wildlife, (iii) problem animal control (PAC), which means eradication of "problem animals" only that are usually a few species of large and potentially dangerous wild animals (WWF SARPO 2000). This has been difficult to implement in most cases due to lack of transport facilities to the area with the problem animal and it can potentially be unsustainable as communities might mislabel animals as problematic simply with the intention of eliminating them as they consider them as pests, and (iv) translocation of animals from the community – this is also a costly method.

Despite the state establishing a public agency to carry out the preventive strategies and making rules regarding use of the resource, the agency had problems of enforcement of such rules. Wildlife went into rapid decline, particularly in the communal areas (Child, *et al* 1997). There was a realisation that allowing landholders in the communal areas considerable freedom to use and benefit from wildlife and curb abuses primarily at a local level using social pressures in the first instance would be socially acceptable and cost effective. In this case rights are closely linked to accountability, and management is efficient because it is more sensitive to day to day variations affecting the resource than what any form of centralised decision making can hope to be (Child 1995).

The Parks and Wildlife Act of 1982 was enacted to give provision for the democratically elected RDCs to become the appropriate authority for managing wildlife within their geographical boundaries. The lack of financial resources, political will and expertise delayed the operationalisation of this legislative provision (Duffy 2000). At the initiative of the DNPWLM in 1989[8] the property rights regime under which wildlife in communal areas is managed effectively changed substantially under the program now commonly known as CAMPFIRE. CAMPFIRE is an acronym for Communal Areas Management Programme for Indigenous Resources. This new paradigm attempts to involve the masses of rural people as partners, to marry conservation with development, and to employ positive rewards in place of bureaucratic regulations as the main instrument of conservation (Gadgil and Rao 1994, 1995). It conceded to the view that the assumption that all human use is detrimental to conservation was evidently invalid (Gadgil and Rao 1995). The new paradigm entails local communities being conferred, through their RDCs, (a) greater control over formerly public wildlife in communal areas in defined territories, (b) enhanced capacities to add value to local wildlife, and (c) specific financial rewards likened to alleged conservation value of wildlife within their territories (Gadgil and Rao 1995).

Essentially the RDCs get the AA status upon satisfactory demonstration to the DNPWLM that they are capable of managing the resources in their area in a sustainable way and that they can satisfy two key conditions: (a) RDCs must disburse at least 50% of the CAMPFIRE revenues to the sub-district producer communities (with a disbursement target of 80% while the remaining 20% would be used to manage CAMPFIRE in the area (15%) and for general council administration and development (5%)), and (b) they must undertake to devolve management functions to those communities over time. These conditions are expected to be fulfilled from the standpoint of the RDCs' moral obligations. Provided these commitments are forthcoming the parks agency steps back into the role of regulator and adviser, retaining the right to control wildlife harvesting quotas. Under CAMPFIRE people living in Zimbabwe's marginalized communal areas essentially claim the same right of proprietorship as private landholders, but through their RDCs. While a private landholder may use a land title to claim ownership of natural resources, a village on communal land only has statutory rights to use such resources as part of a local authority i.e. the RDC that has been granted AA by the DNPWLM (Child *et al* 1997).

[8] Though efforts started in 1985, CAMPFIRE is officially recognized to have emerged in 1989.

CAMPFIRE was conceived in the harsh Sebungwe region as an innovative extension of the Parks and Wild Life Act (1975) induced successful game ranching on commercial lands to communal areas, with the comfort of the successful human-wildlife conflict resolution of the small Mahenye community experience in the background (Child *et al* 1997). The Mahenye community lives on the border with Mozambique on the banks of Save River across from the Gonarezhou National Park. Hunting was a way of life for these people and they resented the Park for denying them rights to use the resources and for isolating them from others of their tribe. Poaching was severe and in one fortnight in 1982 there were about 80 convictions against people in the community, which did nothing to reduce their antagonism towards the Park. A safari hunter and rancher brokered an agreement between the DNPWLM and the Mahenye people, whereby he would shoot a small quota of elephant, buffalo and nyala crossing out of the Park (Child 1995). The people would receive the meat and all the net revenue in exchange for not poaching. As a result of these measures poaching decreased sharply.

The Mahenye community was persuaded by the net revenue earned from wildlife to move some of its villages away from the prime wildlife habitats along the river. The villages vacated a small, but highly fertile area of their land contiguous with the Gonarezhou National Park, and gave it over to wildlife, mainly the elephant. High proceeds from sale of hunting rights meant that an elephant was more valuable than the crop damage it caused. With the DNPWLM's concurrence, the community sold a small quota of animals to a safari operator, from a population they now shared with the Park. The proceeds were used by the community to build the much needed local social infrastructure: a school, a road, a borehole and a grinding mill. The off-take increased while the community's wildlife earnings grew and they extended the amount of their land allocated to wildlife. The community started controlling poaching and people became hesitant to kill wildlife to protect crops. Instead the community set up a scheme funded from wildlife profits to compensate members for crop losses, as they preferred to use the wildlife to generate greater income. This resolution of the human-wildlife conflict experience gave the DNPWLM confidence in taking far-reaching decisions to devolve authority over wildlife to communities elsewhere, through their RDCs.

CAMPFIRE was a program designed to allocate the rights to use communal resources to small communities, providing an incentive to use the resources better. The program needed to be acceptable to participants from different tribal cultures, under differing ecological and economic circumstances, and conform with government policy. In addition, it was necessary to generate sufficient incentives to promote good conservation and to create disincentives to inhibit abuse of the resources. Improved returns were needed to cover the costs of the new institutions, leaving enough over to provide a strong incentive to rightholders to invest in the conservation and development of their resources. It was preferable that sanctions for the misuse of the shared resource base should be through local social pressures, at least in the first instance. CAMPFIRE aimed to internalize the costs and benefits of resource management to the individuals in defined communities, removing externalities and systems of open access. At the outset the program envisaged a transitional period when abuse of resources might intensify. It was feared that this might occur before the merits of new approaches to old problems were appreciated and could become effective, and while communities relearned the art of managing a resource from which they had been alienated for many years. The DNPWLM expected that during this period communities might need assistance in acquiring new skills and expertise.

The program was meant to involve all natural resources but so far the principles of CAMPFIRE have been applied only to wildlife management in Zimbabwe. Wildlife has been the main focus largely due to the reason that the program originated in the DNPWLM. Other reasons have to do with the importance of wildlife to tourism and the need to provide tangible benefits to those who live with wildlife. CAMPFIRE emerged with the recognition that as long as natural resources, particularly wildlife, remained the property of the state then no one would invest in them as resources. CAMPFIRE has not replaced the DNPWLM but it has simply enhanced the joint planning between rural communities and the DNPWLM. Such co-management may indeed hold the best hope for the future of national parks across Africa.

No single organization runs CAMPFIRE at the national level. There is a collaborative group for CAMPFIRE (CCG), now commonly referred to as CAMPFIRE Service

Providers, which is responsible for co-ordinating the various inputs, including policy, training, institution building, scientific and sociological research, monitoring and international advocacy. The CAMPFIRE association, which is a body made up of the RDCs with AA status and whose sole task is to co-ordinate the needs and services of the membership takes a 2% levy of all CAMPFIRE revenues from each RDC and is the lead agency in the CCG. As the programme expands in geographical area and scope, new partners are continually being added but the original members of the CCG are the following organisations.

Figure 2: Original members of the CAMPFIRE collaborative group

The **CAMPFIRE Associatio**n represents rural district councils and therefore the interests of the rural communities involved in CAMPFIRE. The Association is the lead agency and co-ordinator of the programme. It chairs the CAMPFIRE Collaborative Group.
The Department of National Parks and Wildlife Management originally devolved its responsibilities for wildlife to communities and now provides those communities with technical advice on wildlife management.
The Ministry of Local Government, Rural and Urban Development is responsible for the supervision of the rural district councils, to whom the authority for wildlife has been decentralized.
Zimbabwe Trust focuses on training, institution building, and the development of skills among community members and representatives.
The Africa Resources Trust monitors external policy and regulation that effects CAMPFIRE and provides information to decision-makers worldwide.
World Wide Fund for Nature (WWF) provides ecological and economic research, monitoring, and advisory services to CAMPFIRE and also assists in training.
ACTION is best known for providing environmental education, training and materials to schools in CAMPFIRE districts.
The Centre for Applied Social Sciences at the University of Zimbabwe is involved in socio-economic research and monitoring within CAMPFIRE communities.

Source: CAMPFIRE Association

Some of the more recent organizations to join the CCG are:

- The Department of Natural Resources (DNR)

- The Southern Alliance for Indigenous Resources (SAFIRE)
- The Forestry Commision
- The Agricultural, Technical and Extension Services Department (AGRITEX)

Both the government and the Non-Governmental Organisations (NGOs) now support the idea that benefits from wildlife and other natural resources should go to the local communities. A significant part of CAMPFIRE focuses on recreating natural resources in general, and wildlife in particular, as a common property resource as opposed to state property. As environmental concerns are integrated into narrowly developmental ones there is widespread recognition that for CAMPFIRE to be a success it also needs to incorporate pasture, forests, water and other natural resources. The returns to shareholders from a resource based production system become linked to environmental inputs; profits are related directly to how people use or abuse their resources to influence their own future earnings. Thus people who look after their impala herds and the habitats can continue to harvest more impala. Less formal implementation of CAMPFIRE allowed the concept to shift from a strategy for conserving wildlife to one aimed at developing rural communities and their capacity to manage natural resources, using wildlife as the catalyst. The goal is self-sufficiency in self-supporting resource conservation with minimal dependence on any kind of external direction.

CAMPFIRE begins at a district level when the RDC asks the DNPWLM to grant it the legal authority to manage its wildlife resources, and demonstrates its capacity to do so. Once accorded this legal authority, the projects that the RDCs devise to undertake vary. Over the years, the RDCs have harvested their natural resources and earned income in the following ways: leasing trophy hunting concessions, utilizing forestry and forest products, leasing eco-tourism sites and making live animal sales.

- ***Leasing trophy hunting concessions***. Over 90% of CAMPFIRE revenues earned by rural communities come from foreign hunters who come to Zimbabwe to hunt elephants, buffaloes, lions or other wild animals. Hunters are considered the ultimate ecotourists as they have a much lower impact on the environment than other tourists. In addition, their presence in remote areas acts as a poaching deterrent, and hunters pay much higher fees than other tourists. With the increasing participation of RDCs and communities, the DNPWLM determines the sustainable off-take quota of wildlife available on communal areas for hunting. The RDCs then lease hunting concessions to professional safari hunting companies. 65% of CAMPFIRE hunting revenue comes from elephants. There are several reasons why elephants play such a significant role: (i) their trophy fee is far higher than for other species; (ii) most sportsmen who come to hunt in Zimbabwe are after an elephant trophy; (iii) very few private landowners are in a position to offer elephant trophies, so communal areas have the majority of the elephant trophy hunting market and (iv) with a trophy fee of up to US$12,000 or more, together with a daily hunting fee of US$1,000, one elephant can realise US$33,000 over the course of an average 21-day hunt, in which they are usually packaged.

- ***Harvesting natural resources***. Communities harvest and sell natural products such as crocodile eggs, timber, river-sand and caterpillars. Unfortunately for some communities, the CITES ban on international trade in elephant products prevented them from selling hide and ivory from 'problem animals' – some of which persistently raid crops or threaten local residents. Fortunately the ban was conditionally lifted in June 1997.
- ***Tourism***. Tourists have visited Zimbabwe's rural areas for many years, although the local communities were rarely involved (or benefited from) tourism until a few pilot projects were set up by CAMPFIRE in the early 1990's. Most revenues from tourism in Zimbabwe's Communal Areas are generated through the leasing of sites for nature tourism, although in some cases local residents run basic tourist facilities and act as guides. Many more tourism plans are in the pipeline, including cultural tourism, bird-watching and access to natural hot springs.
- ***Live animal sales***. In areas where wildlife populations are high, RDCs have begun to sell live animals to commercial game reserves or national parks. In 1994, Guruve RDC sold ten roan antelope, earning some US$50,000.
- ***Meat cropping***. Meat from wildlife is sometimes sold to neighbouring communities or towns.

The tables below give an overview of the generation and expenditure of revenues from the various CAMPFIRE activities (with small rounding-off differences).

Table 2: Rural District Councils' Annual Income from CAMPFIRE Activities (US$)

Year	Sport Hunting	Tourism	PAC Hides & Ivory	Other	TOTAL
1989	326,798	28	5,294	17,690	349,811
1990	453,424	2,865	42,847	57,297	556,433
1991	638,153	15,904	20,859	101,105	776,021
1992	1,154,082	18,951	9,429	34,216	1,216,678
1993	1,394,060	21,095	14,988	53,730	1,483,873
1994	1,553,543	39,985	2,770	46,373	1,642,671
1995	1,476,812	54,866	11,685	48,204	1,591,567
1996	1,656,338	23,275	39,869	36,429	1,755,912
1997	1,708,234	71,258	44,331	13,615	1,837,438
1998	1,787,977	40,871	25,205	37,713	1,891,766
1999	1,940,366	78,709	720,440	14,442	2,753,958
2000	1,919,980	55,668	116,075	13,482	2,105,204
2001	2,142,306	41,439	111,914	32,793	2,328,452
TOTAL	**18,152,074**	**464,915**	**1,165,706**	**507,090**	**20,289,784**

Source: WWF SARPO, Harare

Notes:

1. Sport hunting - income earned from lease and trophy fees paid by safari operators
2. Tourism - income earned from the lease of wild areas for non-consumptive tourism
3. PAC Hides & Ivory - income from the sale of animal products primarily from problem animal control
4. Other - income from the sale of live animals, collection of ostrich eggs and crocodile eggs, etc
5. Mean annual exchange rate based on RBZ end of month exchange rates

Table 3: Allocation of Revenue from CAMPFIRE Activities by Year (US$)

Year	Disbursed to Communities	Wildlife Mgt.	Council Levy	Other	Not Detailed	TOTAL
1989	186,268	81,458	28,404	12,032	41,651	349,811
1990	206,308	121,485	52,530	22,501	153,609	556,433
1991	320,894	219,526	120,444	56,930	56,884	774,678
1992	601,385	207,291	115,398	17,837	274,767	1,216,678
1993	851,732	357,055	251,082	32,172	-14,216	1,477,824
1994	949,138	314,572	148,517	42,514	187,889	1,642,631
1995	946,777	353,772	193,080	26,214	71,723	1,591,565
1996	833,025	405,755	301,091	7,796	191,792	1,739,458
1997	858,357	29,661	26,746	12,415	915,884	1,843,063
1998	910,200	521,373	70,666	82,939	306,589	1,891,766
1999	1,341,853	608,678	253,252	29,477	520,698	2,753,958
2000	1,025,586	320,973	491,411	127,276	139,958	2,105,204
2001	858,869	538,596	454,265	210,388	278,156	2,340,274
TOTAL	**9,890,392**	**4,080,194**	**2,506,885**	**680,491**	**3,125,382**	**20,283,343**

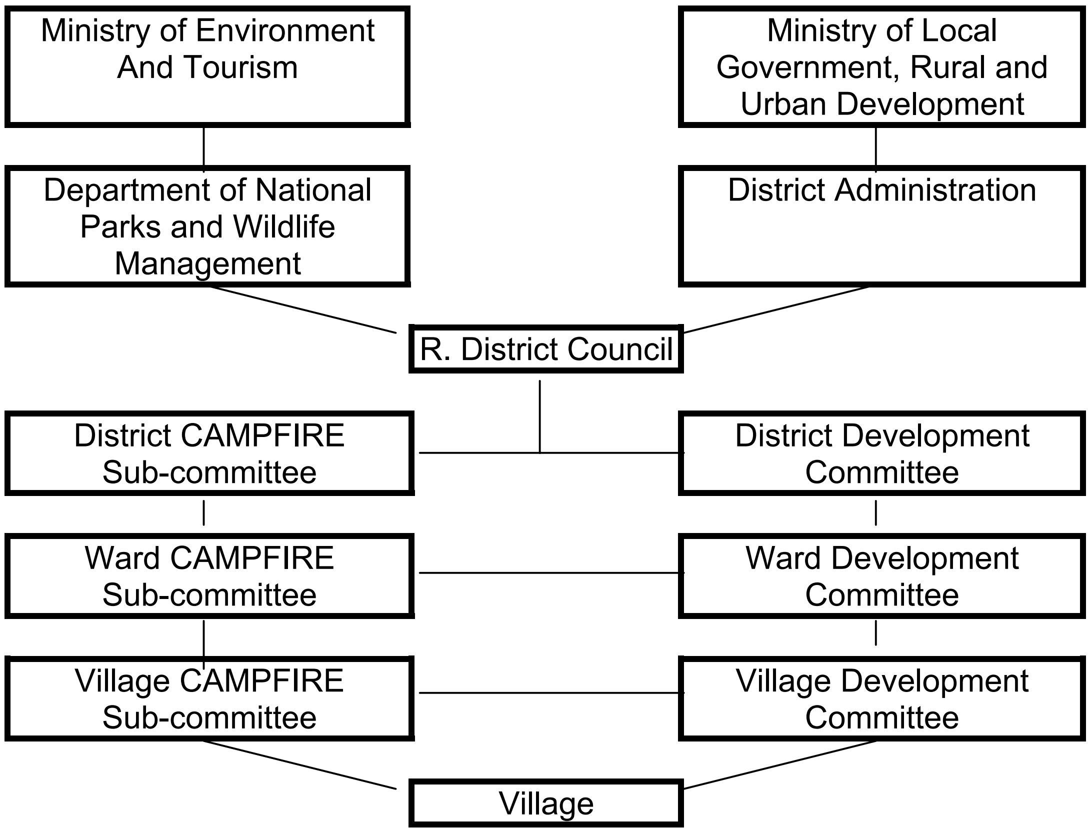
Source: WWF SARPO, Harare

Notes:

1. Disbursed to communities - revenue allocated to sub-district CAMPFIRE institutions
2. Wildlife Management - revenue allocated for wildlife and programme management
3. Council Levy - revenue allocated to district council general account
4. Other - revenue invested in capital development projects and RDC levy to CAMPFIRE Association
5. Amount Not Detailed - revenue not allocated but retained by RDC for general account
6. Mean annual exchange rate based on RBZ end of month exchange rates

At independence in 1980 several new political administrative structures were created namely village, ward and RDC. A village consists of approximately 100 to 150 households, or about 1,000 people. A Ward represents about 6 to 10 villages, or around 6,000 individuals. Each ward is represented in the RDC by an elected councillor. Owing to several trouble spots in operating through the government's administrative structures such as the lack of a legal hierarchy for statutes dealing with natural resource management, conflicts between institutions, confusion and disagreements of jurisdiction in communal areas and clashes between customary law and statute law, CAMPFIRE created its own parallel structures. The figure below shows the generalised organisational structure developed under CAMPFIRE.

Figure 3: Generalised organisational structure of CAMPFIRE

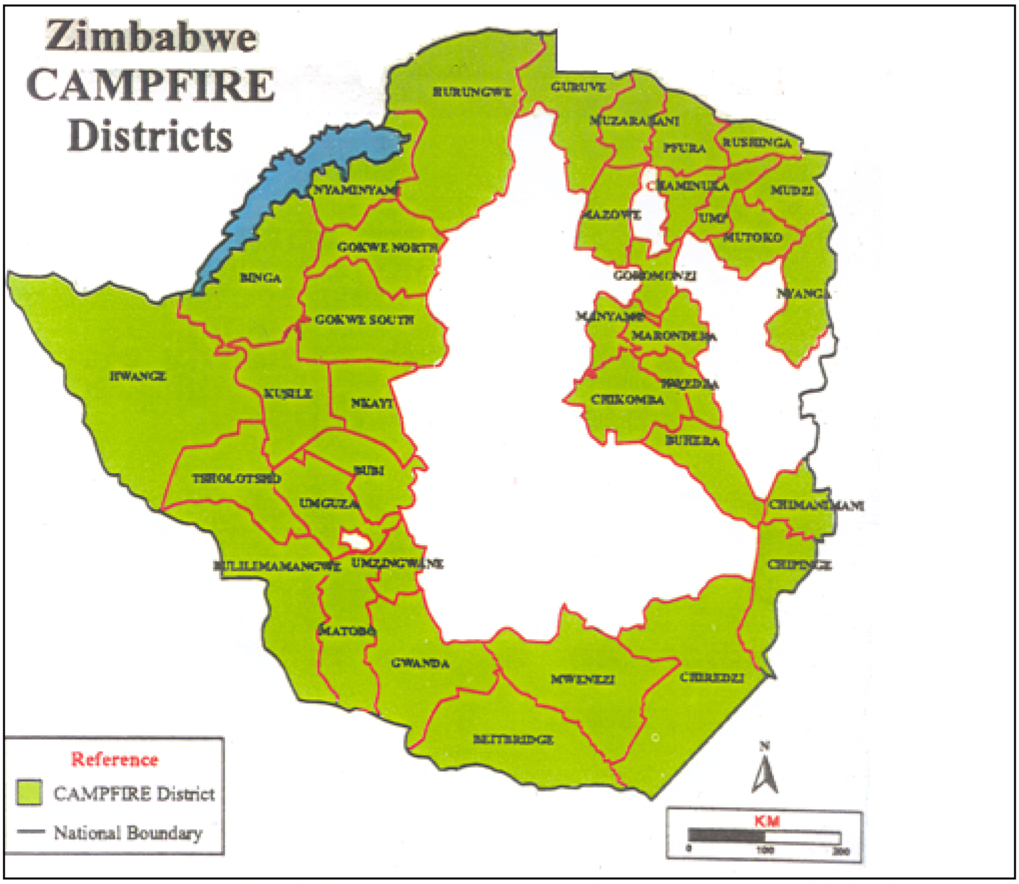


Source: CAMPFIRE Association

At the grassroots level, each village with a significant wildlife population elects six members to sit on a Village CAMPFIRE sub-committee. Representatives of these committees sit on the Ward sub-committee, which is chaired by the councillor. The councillor sits on the District sub-committee, which also includes the RDC chairman and vice-chairman. In addition, one other representative, usually the secretary or treasurer, from each Ward sub-committee is also co-opted as a non-voting member of the District sub-committee. Other advisors include council executive staff and representatives from other government departments, such as the Forestry Commission, Department of Natural Resources, and Ministry of Agriculture. CAMPFIRE and its administrative structures are tailored to suit local needs.

The CAMPFIRE concept first germinated in Nyaminyami, a communal area surrounding the Matusadona National Park, with an extensive frontage on Lake Kariba. The area has good game populations, including some 2,500 elephants. In 1989 the Nyaminyami RDC was given the AA for its wildlife, in recognition of the progress the district had made towards managing the resource sustainably and the Council could now collect and use the revenue. Similar authority was granted to Guruve RDC, further east in the Zambezi valley. Many other RDCs joined the club. CAMPFIRE spread rapidly and by 1993 it covered 70 wards in 12 RDCs. So far 36 RDCs, out of the country's 57 RDCs, have been accorded AA status (*see figure below*).

Figure 4: Map showing the CAMPFIRE districts.



Source: CAMPFIRE Association

Plumtree and Tsholotsho pioneered open competitive marketing and drove the price of elephant up from ZW$7,500 to ZW$30,000 in a single season. They used a tender system, to replace the government prices based on the fees levied in safari concessions. Beitbridge RDC set two precedents: the return of all the money to the villages from which it was earned, and democratic procedure for apportioning dividends. It was noted that cash is normally scarce in the rural areas. Dividends from CAMPFIRE programmes were found to be one of the chief contributors to household incomes in some areas. In wards like Masoka where cash dividends per household are high the idea and essence of CAMPFIRE is well appreciated. In wards 2, 3 and 4 of Guruve it is apparent that the cash dividends are low and thus the appreciation of CAMPFIRE is low (Child *et al* 1997). Also people were seen to be more interested in cash dividends. The concepts of community projects was found to be still not well engrained in the community. The cash dividends were seen to be used freely in most cases. At times locals were given a chance to choose projects they were interested in, but these projects were seen to have resulted in conflicts or general flops.

CAMPFIRE has relied heavily on donor funds (mainly US government grants but also from European Union, United Kingdom, Norway, Netherlands, Germany and Japan) to pay for administrative expenses (Patel 1998), usually occurring at the level of CAMPFIRE Service Providers. Only a small fraction of CAMPFIRE running expenses is covered by its activities. The aggregate income from CAMPFIRE activities between 1989 and 1999 was about US$13 million (*see table 3*) compared to aggregate donor funds of at least US$27 million (*see table below*). The external funding can be viewed as payment for the international portion of existence and option values of wildlife. The donor funds may have helped CAMPFIRE to kick off without the problems of inadequate incentives that it could have faced in the absence of donor funds. The donor funds were channelled mostly to supporting organisations at the national level managed to provide incentives by way of community development, applied research, regional communication, project management, project evaluation and wildlife conservation (*see table below*). These overheads would otherwise have been paid from CAMPFIRE revenues thereby reducing disbursements to producer communities and negatively affecting stewardship practice. The US government grants were not renewed after the completion of the second 5-year phase in 1999. The donor-funded activities in CAMPFIRE have continued to run either from residual funds or from new sources of funding. The perverse political climate has seen a lot of donors withdrawing financial support to many sectors in Zimbabwe. CAMPFIRE has not been spared. Operations of the CAMPFIRE service providers are the ones that have been seriously affected since they relied so much on foreign aid. The impact on the local communities has either been negligible or is yet to be felt.

Table 4: Allocation of donor funds for CAMPFIRE Phases I (1989-94) and II (1994-99)

Activity	**Implementing Agency**	**Total Phase Amount (US$)**	
		***1989-1994**	**1994-1999**
Wildlife conservation	WWF DNPWLM	1,481,500	1,570,000 1,500,000
Community development	ZIMTRUST CAMPFIRE Association MLGRUD CDF	4,180,500	1,965,000 1,750,000 200,000 6,000,000
Regional Communication & Project Management	USAID Africa Resources Trust IC	899,295	1,530,000 1,200,000 2,675,000
Applied research	CASS Action Magazine	699,000	1,050,000 1,060,000
Audit/Evaluation		100,000	Nil
Contingency		239,705	Nil
Total		**7,600,000**	**20,500,000**

Source: Patel (1998); * = joint budget

3. Implications of wildlife trade bans on CAMPFIRE

Increased ivory prices, especially since about 1970, have been blamed for the widespread slaughter of the elephant in Africa and led to an international ban on the trade in ivory in 1989 (CITES 1990). Environmentalists claim that the ban is necessary to curb an unsustainable level of elephant off-take. They argue that managed trade is an unrealistic option given that history shows that prior to 1989 issuance of CITES ivory export permits was poorly controlled and smuggled ivory tended to find its way into the legal trade through various loopholes in trade regulations. It was hoped that the ban would eliminate the market for ivory. With no market for ivory, demand would dwindle, the price of ivory would skydive and the incentive to kill elephant would be erased. Sutton (2001) points out three problems associated with the ivory ban approach to conservation. Firstly, the ban does not recognise the heterogeneity in the conservation status across the African range states. The four southern African countries namely Botswana, Namibia, South Africa and Zimbabwe have large and growing elephant populations which are a fruit of effective management programs instituted even before the ban. Enforcement of anti-poaching laws has largely depended on revenues from consumptive uses as well as benign tourism. It has been estimated that adequate protection of the elephant requires US$200 per km^2 annually (Child 1995). The ban may have brought in its wake a drastic decline in anti-poaching budgets, a situation which cannot be assisted by the cash strapped African governments.

Secondly, the ban assumes that poaching is the primary cause of the elephant decimation. Instead, poaching could simply be a symptom of a fundamental human-wildlife conflict. Thus, the competitive exclusion principle could perhaps help to account for the decline in the elephant population, by recognising that two species seek simultaneously an essential resource of the environment that is scarce.

Thirdly, the assumption that the desired decrease in the price of ivory will result in an increase in the stock of the elephant is flawed. The elephant generates large negative externalities for the society and yet it requires investments from the same society in the form of land. The lowering of the price of ivory could in fact make the elephant less valuable for the society to have an incentive to invest in its existence.

Kenya has claimed that the ban has assisted in reducing the illegal slaughter of elephants in that country. A parallel explanation for the improved Kenyan situation is that her stance on the ban earned her vastly more resources and these have been devoted to law enforcement. Bell, *et al* (1992) found that they were able to bring elephant poaching in Zambia down to acceptable levels (10% of annual recruitment) when they increased expenditure on law enforcement from US$15 to US$65 per square km.

Parker and Graham (1989) argue that species such as the giraffe that are unaffected by international trade are declining in the face of burgeoning human demands for land, and as such there is no reason to link the downward trend in the elephant to increasing ivory prices. Rather the problem facing the elephant 20 to 25 years ago was that it was valuable but not yet valuable enough to the people on the land. This was due to relatively low market prices for ivory, hide and hunting, and institutions that accentuated the negative value of the species to the local communities.

The legal protection of cheetah and rhino has done little, if anything to reduce poaching of the two species in Zimbabwe and may have exacerbated the problem. Cheetah preyed on domestic stock. There was vociferous condemnation of cheetah and more than 400 were killed illegally by conservation-minded ranchers, including in those parts of the country from which the DNPWLM could have expected reports of any illegal activities (Child 1995). The DNPWLM did not secure a single conviction during the spate of illegal cheetah killings. There was little if any illegal trade in cheetah skins and the species was destroyed by ranchers simply to avoid the opportunity costs imposed on them by its legal protection.

At the 1993 CITES meeting Zimbabwe concurred with the imposition of a complete embargo on trading in rhino parts and derivatives. Despite this the illegal trade has flourished. The DNPWLM had increased its surveillance. Anti-poaching operations assumed the proportions of moderately intensive anti-insurgence warfare, employing the same tactics and equipment, including automatic weapons, sophisticated radio and intelligence networks, vehicles, boats, helicopters and fixed-wing aircraft (Child 1995). Law enforcement was, however, tackling the effect rather than the cause of the problem. Poaching was motivated by the high price of horn on the illegal market, which had been handed a monopoly by the prohibition of legal trade.

In general, hunting and trade bans are seldom effective for conserving wildlife. They can do nothing to prevent animals disappearing through loss of habitat, which is the most common cause of decline where wildlife cannot compete with other land uses. Furthermore, enforcing a legal ban is difficult and costly, with the result that it is often ignored by both the hunters and the authorities, but remains on the statutes, driving the industry underground where it cannot be managed (Child 1995). Records needed for a sustainable enterprise are neglected, and the State cannot support an illegal activity by providing extension, marketing or other services. With no legal sellers or buyers, the value of wildlife is depressed. If the peasants are to stay on the land and without any benefits forthcoming from wildlife, they must generally abandon wildlife. The alternative use of the land invariably replaces wildlife and its habitats, and is seldom as environmentally friendly as harvesting wild animals.

The harvestable wildlife gives the habitats a value to the people and enables them to live there without clearing the forest for agriculture. Wildlife uses less natural forage and so places less stress on the natural vegetation and retains better herbaceous cover thereby providing better ecological resilience to droughts through increased plant production and reduced variability in available forage (Child 1995). Furthermore, wildlife can potentially bring substantially higher economic returns than normal agriculture as a form of land use in marginal areas and therefore profits from tourism, hunting rights, trophies and sales of game meat could be used to significantly improve the local standards of living. The adoption of the scheme would result in the economic utilization of marginal land. The rarity of the animals gives them a value as huntable trophies, far in excess of that as meat, which could be exploited to persuade the locals to allow wild populations to build up so as to attract the benefits from the hunting (Child 1995). Since there is still a large wildlife population in many communal areas particularly the keystone and umbrella species, the elephant, whose existence on privately owned land is still insignificant to the total population outside protected areas, it consequently means that the fate of over half the potential elephant population in Zimbabwe depends on the decision of peasants who have a very low annual income.

Allowing wildlife an economic value is a force for its conservation, provided it can be traded freely (Child 1995). The notion that denying wildlife a monetary value and a legitimate market removes the incentive to poach or over-exploit the resource cannot be right in the light of high negative externalities to the local communities from some species of wildlife such as the elephant. The wildlife trade ban is only justified if the law enforcement it aims to assist is very effective, and centralized prohibition and law enforcement are more efficient than developing legal markets, thereby encouraging local people to protect their own resources. In all other cases, we believe that wildlife trade bans provide a disincentive to community-based wildlife conservation.

4. Which way forward with CAMPFIRE?

Significant gains have been recorded in community-based wildlife conservation in Zimbabwe under CAMPFIRE. These gains include the broad scale of implementation of the project, increased share of land devoted to wildlife management, establishment of monitoring of wildlife populations in communal areas, building up of institutional and administrative capacity at rural district (RD) and local community level, democratisation by proposing alternative models to the centralised control political culture, development of social infrastructure, influencing sensible regional wildlife policy reform, opening markets for trade to enable Africans to accrue revenue from their wildlife resources, build stewardship for the resources, promoting wildlife resource utilisation as a complimentary land use strategy on marginal agricultural lands for local people, provide a social, political and economic context in which wildlife resource use can be discussed, debated and decisions implemented.

However, many researchers (for example Child 1995, Child *et al* 1997, Patel 1998, and Hasler 1999) have cautioned against over-celebrating the gains of community-based wildlife conservation in Zimbabwe arguing that it is still too early to call CAMPFIRE a success. There is an array of challenges still facing CAMPFIRE. Under the current legal set-up in Zimbabwe all funds generated by CAMPFIRE projects go first to the RDCs, and it is then at the RDCs' discretion to determine how much goes to the producer communities. Weaknesses still exist in the system, notably in the area of accountability of revenues. For example, a cash-strapped RDC might be tempted to misappropriate wildlife revenues destined for communities to cover its own administrative costs. In fact, the management of game animals as a State resource left a legacy of treating them as a communal asset and this results in wildlife being taxed unfairly compared with other resources. The temptation for RDCs to use the revenue from communal wildlife for community projects, especially elsewhere in the district rather than where the revenue was earned, is fraught with danger as it disadvantages wildlife in the local economy by comparison with other resources.

The first 12 districts to implement CAMPFIRE were the best endowed, in terms of wildlife. Some districts with less wildlife have started different programmes based on such resources as cultural heritage, scenery, forests, rare birds (all non-consumptive uses) and game farming. The programme's resource base is expanding, and many CAMPFIRE districts have sought to promote non-consumptive eco-tourism projects. However, most of these projects are mainly leased to private operators and local communities play at best a minor role, and often no role, in their management. Also RDCs currently tend to categorize revenues from these operations as non-CAMPFIRE revenues, and thus the revenues have not been distributed to communities.

The availability of huge external funding in the presence of a small size of programme-generated revenue could potentially undercut the capabilities of local institutions to be sustained over time. The rationale for such external funding is that the global society must pay for its portion of existence and option values to raise the level of conservation towards globally desirable levels. In most cases huge external funding makes it difficult to build upon indigenous knowledge and institutions (Gadgil, Berkes, and Flokes 1993). This is unfortunate especially if traditional institutions are more likely to lead to greater conservation than modern institutions. By providing the requisite operating capital and sponsoring skilled labour, external aid resulted in the development of sophisticated top-heavy management structures at the district level aimed at managing wildlife, carrying out PAC and other crop protection measures and entering into wildlife exploitation joint ventures with safari operators. Such structures resulted in increased technical management capacity for the RDCs to manage the resource at the expense of the basic tenet of CAMPFIRE, namely local communities' participation in the management of the resource. In the presence of external aid, sub-district devolution did not and might never take place. This could lead to the persistence of the current scenario in which sub-district communities receive insignificant dividends without expending any conservation effort beyond the damages suffered from wildlife (Murombedzi 1997). On the other hand, donor funds may have helped CAMPFIRE to kick off without the problems of inadequate incentives that it could have faced in the absence of donor funds. Where short-term external aid is indispensable it should be channelled directly to the producer communities so that they respond to it as increased demand for conservation.

Lack of devolution of responsibility for wildlife and other renewable resources to the sub-district producer community level has been cited (for example Patel 1998) as one of the problems that still remain in CAMPFIRE. Despite the enactment of the Parks and Wildlife Act (1982), sub-district communities are still not legal entities. As such CAMPFIRE had to be content with devolving authority to the legally constituted RDCs. Governments often hesitate to devolve and confirm proprietorship over wildlife to local people for two reasons: (i) lack of faith in the ability of the people to manage the resource correctly. There is a fear that the people living with the resources and already using them will abuse them if government suddenly relinquishes control, and (ii) an unhealthy inclination among bureaucrats to cling to power in an effort to control the wildlife industry, rather than encourage the corporate strength of stakeholders to optimize opportunities as they emerge (Child 1995). Wherever possible responsibility for wild resources should be devolved to the producer or producer community. The person or group of people who benefit from the primary use of a resource should be held fully responsible for managing it correctly. This implies full proprietorship of the resource by landholders. So there is still need to (i) consolidate the devolution of responsibility for wildlife and other renewable resources to the producer community level, (ii) continue to strengthen the inherent institutional capabilities for developing and managing the resource, at this level, and (iii) ensure that the revenues generated are managed carefully to avoid reintroducing distorted accountability (Child 1995).

Many people seek to discredit programmes such as CAMPFIRE because these programmes dispute the right of the center to control the periphery, and because they involve the killing of animals for profit (Child 1995). CAMPFIRE is portrayed as a paradox because you have to kill animals in order for them to survive.

There are many potential research issues that come to mind after an examination of how the community-based wildlife conservation in Zimbabwe has been working. Of interest to know is how communities should respond to the various incentives put forward to induce them to partake in wildlife conservation. It is important for communities to know what action they should take with respect to wildlife conservation and to what length they should pursue that action. Does the inception of CAMPFIRE mean that local communities should vote for a motion that seeks to reduce the wildlife populations or should they vote for a motion that seeks to increase the wildlife populations? What is the direction of welfare change emanating from changes in wildlife populations, given the reality of human-wildlife conflicts? Chapter 2 tries to give a theoretical insight to potential answers of these questions through the use of bio-economic modeling of wildlife-livestock conflicts and welfare in three resource use regimes.

The issue of whether wildlife is a public good or public bad from a local communities' perspective, given the benefits and costs that it imposes on them, is ultimately an empirical matter. CAMPFIRE has not been able to eliminate the human-wildlife conflicts. Also, the lack of agreed measure of damage has prevented the compensation of aggrieved households who suffer damages from wildlife. Wildlife poses a particular problem in that ownership and control are usually unclear. In the pre-CAMPFIRE period compensation could be awarded to some local farmers on privately held land for crop damages mainly because they were not owners of the wildlife. With CAMPFIRE, the government surrenders the control over wildlife to the local communities, through their RDC, hence it would seem consistent that the community could be held responsible to compensate individual households for the losses incurred by the wildlife falling under its authority. With this in the background, however, community shared revenue has been found to be difficult to use to finance compensation if the majority of residents or households are unwilling to accept that the communities' money ought to be used in this manner. In Guruve, for the first 2 years since the acquisition of AA status, the RDC used to set aside a certain sum of money for compensation purposes. This exercise was discontinued due to problems of moral hazard. Some people wanted to be compensated even for unsubstantiated damages and for those damages that were suffered several years before. So there was a long list of intended beneficiaries from the compensation fund that was presented to the RDC every year. The money set aside each year could not be enough for everybody to be compensated no matter how high the compensation allocation was raised. The RDC then decided to remove the compensation burden off itself and asked the wards to compensate their own people as they saw fit. Thus the funds that used to be kept by the RDC in the compensation fund were released to the wards and it was up to them to decide whether or not to give compensation. Since the compensation bill was always larger than the disbursed revenues the wards themselves simply did away with the compensation scheme.

If the local communities who live side by side with wildlife, and to whom conservation responsibilities are in the process of being devolved under CAMPFIRE, see wildlife as a nuisance then they cannot be trusted to be its good stewards. First and foremost, there is a need to assure the society and government of the genuine desire of the local communities to manage the wildlife resource correctly before any devolution is undertaken. Chapter 3 on the contingent valuation of the elephant in a typical rural district seeks to establish the economic value that communities living adjacent to wildlife reserves attach on the elephant as a representative of wildlife. If on the one hand, empirical evidence could show that the economic value of the elephant relative to the communities' other economic activities were larger then it would imply that wildlife conservation might be enhanced through devolution of wildlife user rights to the local communities. If on the other hand, evidence would suggest that the local communities view the elephant as a liability then it would give the government the assurance that heeding the current calls for devolution of conservation responsibilities and confirming proprietorship over wildlife to local communities might be detrimental to wildlife.

The national and global society considers wildlife a public good. Its provision depends on the ability of the local communities, who live adjacent to wildlife and to whom conservation responsibilities are in the process of being devolved under CAMPFIRE, to manage it correctly. With the promulgation of CAMPFIRE, the first step towards devolution of conservation responsibilities and confirmation of proprietorship over wildlife to local communities has been taken. Under this new paradigm of resource use, wildlife conservation could be promoted by enhancing the institutions at the local community level, so that they favour sustainable wildlife conservation. Significant gains have been recorded in CAMPFIRE. However, literature (for example Halser 1999, Patel 1998, Murombedzi 1992) indicates an array of problems that have emanated from or have not been resolved by CAMPFIRE. With reforms CAMPFIRE could potentially become a strong strategy blending wildlife conservation and rural development. The starting point in search for reform that should be made in CAMPFIRE could be an investigation of the extent to which Ostrom's institutional design principles that are shared by the world's long-enduring common pool resources are satisfied. Such an investigation could give the direction of reform thrusts that should be encouraged. The investigation of the extent to which Ostrom's institutional design principles are satisfied is the subject matter of Chapter 4.

As we have alluded to before, the presence of wildlife potentially introduces some risk and also potentially eliminates other risks. Empirical investigation of the dominant risk impacts is needed in order to create effective economic incentives for community-based wildlife conservation. Effective community-based wildlife conservation in the twenty first century will necessarily need to involve the masses of rural people as partners and to utilise economic incentives instead of bureaucratic regulations as the main instrument of conservation. The last chapter (5) is a paper on risk management through community-based wildlife conservation and wildlife damage insurance which sets out the theoretical foundations for future empirical research on how rural farmers in Zimbabwe could manage the risk that they face in agricultural production due to unpredictable climatic conditions and wildlife intrusions into agricultural production.

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APPENDIX A

The Elephant in Africa and Zimbabwe

Elephant, popularly known as the king of all beasts, has existed on the earth for millions of years. The word "elephant" comes from the Greek word *elephas*, which means ivory and refers to the animal's prominent tusks that are actually elongated incisor teeth. There are two species – the African elephant (*loxodonta africana*) and the Asian elephant (*elephas maximus*) – that descended from a long line of giant mammals, including the mammoth and mastodon. Both the male and female African elephant carry tusks, while it is only the male Asian elephant that carries tusks. However, due to the hunting pressure on tusked animals brought about by poaching for ivory, tusklessness is an increasingly common condition in African elephants. While Asian and African elephants have a lot in common, each species looks a bit different and each faces different threats to its survival. Asian elephants are mainly threatened by destruction and fragmentation of their habitat having been saved from trade when the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) put them on Appendix I in 1976[9]. Remaining herds of the Asian elephant are located in small numbers in India, Indonesia (Sumatra), Burma (Myanmar), Sri Lanka and Thailand and number between 35,000 and 50,000 due to severe over-hunting in the past[10].

From this point we shift our attention exclusively to the African elephant. While the rest may relate to elephants, in general, it relates to the African elephant, in particular. Besides the tusks, elephants only have four molar teeth that are replaced, as they wear out, up to six times throughout an elephant's life. The elephant's unique trunk is used for drinking (by sucking up and squirting water into the mouth), bathing, smelling, breathing, feeling, grasping food, greeting, caressing, threatening, and throwing dust

[9] CITES was formally established on 1 July 1975 and now has a membership of 148 countries. CITES acts by banning commercial international trade in an agreed list of endangered species and by regulating and monitoring trade in others that might become endangered. Appendix I lists species that are the most endangered among CITES-listed animals and plants. These are threatened with extinction and CITES generally prohibits commercial international trade in specimens of these species. However, trade may be allowed under exceptional circumstances, e.g. for scientific research, by the granting of both an export permit (or re-export certificate) and an import permit.

[10] Source: WWF's Asian Rhino and Elephant Action Strategy http://www.wwf-areas.net/ (31 May 2003)

over the body. At the end of the trunk are sensitive ‘fingers’ for grasping things as small as a berry or as large as a branch. African elephants have two fingers (while the Asian elephant has only one). The trunk is also used as a snorkel when crossing deep rivers. To help protect themselves from the heat in the hot climates of Africa where they live, elephants have large ears with prominent veins that they can flap to cool their blood. Elephants usually stay near water, not only for drinking, but also for bathing and cooling. In addition to mud baths, elephants also take dust baths to keep cool and deter insect attacks.

The elephant's lifespan is up to 70 years. Elephants have a highly developed social system with supportive family units of about 10 members consisting of a group of females and their calves, which are led to feed and drink by an older experienced cow called the matriarch. The age (denoting experience and leadership) of the matriarch has been found[11] to be a significant predictor of the number of calves produced by the family per female productive year because the inability by younger matriarchs to distinguish between potential threats forces families to spend too much time being defensive at the expense of reproduction. Several family units may join together to form a clan consisting of up to 70 members. Large associations are thought to be temporary, forming when food is abundant or when the normal pattern of social life is disrupted by human interference. Some males form bachelor herds, joining the females only to mate, while other bulls are loners. Both male and female elephants reach sexual maturity at about 10 years of age although males become sexually active much later owing to social constraints, which are usually such that males and females do not mate until they are about 20 – 30 years and 15 years old respectively. Mating takes place throughout the year but peaks during the rainy season. After a 22-month gestation period, elephants usually give birth to a single calf weighing an average of 113 kilograms and standing almost 0.9 metres tall every 2 to 9 years, depending on habitat conditions and population densities. Most births occur during the rainy season to ensure that the mother has plenty to eat while suckling her calf. While the calf will begin eating vegetation

[11] This result was reported (see http://iucn.org/themes/ssc/sgs/afesg/faq/elefaq.html) by researchers from the Institute of Zoology at the University of Sussex and the Amboseli Elephant Research Project in Kenya after they used high-powered hi-fi equipment to play back the sounds of elephant calls.

within a few months, it continues to nurse on its mother's milk until it is at least 2 years old. Females can remain fertile until 55 – 60 years old.

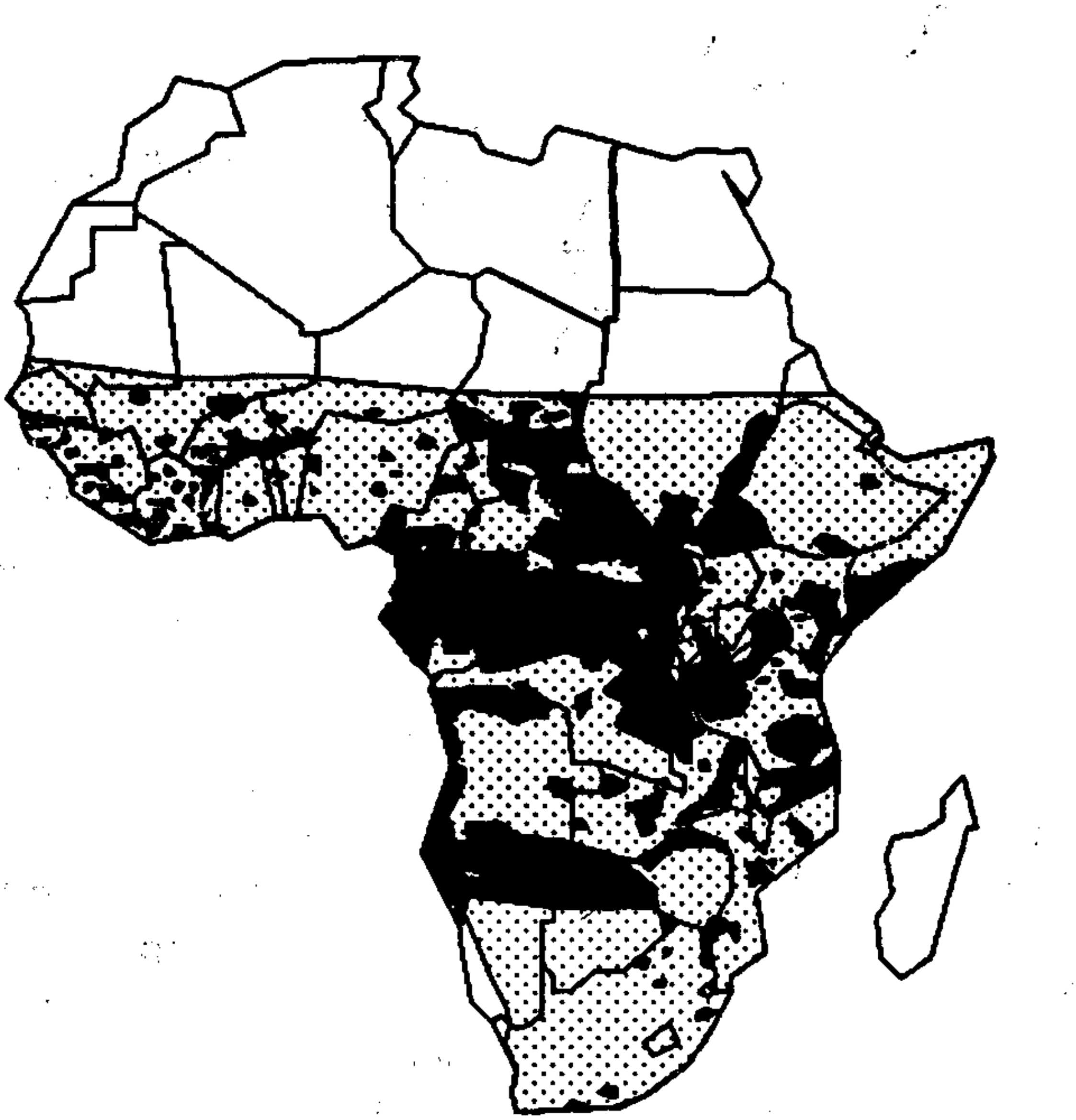
Elephants are highly intelligent and have remarkable memory, particularly for places to drink and find food. Senses of sight, scent, sound and touch are all important in elephant communication about health, sexual condition and other information. Elephant vocalizations range from high-pitched squeaks to deep rumbles, two-thirds of which are emitted at a frequency too low for the human ear to detect but are detectable by other elephants at distances of about 10 kilometres. Elephants also appear to mourn their dead by covering them with twigs and leaves and staying at the gravesite for hours. Even though elephants are not territorial, they utilize specific home areas during particular times of the year. Home ranges vary in size between $14km^2$ and $3,120km^2$, depending on food and water resources.

Elephants are herbivores and their diet consists mainly of leaves and branches of bushes and trees, but also grass, fruit, bark, shoots, root and twigs. Elephants do not have efficient stomachs because they cannot digest cellulose, the substance that makes up much plant matter, hence they spend about three-quarters of their time selecting, picking, preparing and eating food. An adult elephant eats an average of 150 kilograms of vegetation per day. An elephant's choice of food-plants will be determined by what grows locally, what was learnt from its mother, and what it has discovered by trying new food items. Elephants love the pod of the apple ring acacia tree. Because of its trunk, an elephant can reach up to six metres high, making it the most versatile herbivore. Even though food crops are not the primary food for elephants they have been known to like water melons (*manwiwa*), pumpkins (*manhanga*), maize cobs (*chibage*), sorghum (*mapfunde*), millet (*zviyo*) and sweet cane (*ipwa*) which they eat amid huge trampling (*see figure A2 in appendix for typical elephant damage*). In addition, an adult elephant drinks about 225 litres of water each day and this can sometimes be drunk during a single visit to the water hole. When water becomes scarce, elephant will strip and destroy even the strongest and oldest baobab trees. Although elephants are thought of as ‘gentle giants’, they will destroy crops, fences and homes in their search for food and trample on anyone who stands in their way.

The African elephant is the largest land mammal. While calves may fall prey to lions and hyenas, adult elephants have no natural predators except man. 500 years ago 10 million elephants roamed the African continent. Not only have elephants been slaughtered for their ivory tusks but their populations have declined significantly because of habitat destruction and fragmentation. Although ivory trade has experienced sustained growth since the 1940s, the huge increase that occurred during the 1970s was the result of automatic weapons availability and widespread government corruption in many exporting countries. Organized gangs of poachers laundered tons of elephant tusks through several African countries to destinations in Eastern and Western countries. In 1975, the price of raw ivory reached US$110.23 per kilogram, up from between US$6.61 and US$22.05 per kilogram in the 1960s, because ivory was perceived as a valuable hedge against rising inflation apart from the traditional uses such as jewellery, carvings, piano keys, personal name seal stamps, traditional medicines and aphrodisiacs (Sack 1992). CITES added the African elephant to Appendix II[12] in 1977 thereby keeping a close eye on it. The continued poaching trend decimated elephant populations across Africa and in 1979 there were just 1.3 million elephants left (*see map on distribution of elephants on next page*). It was only in the 1980s that most African countries joined CITES but this did not help to improve the elephant population trends. In 1985, reports of rising elephant poaching levels led CITES to establish quotas for ivory exports for African range states and propagate an ivory monitoring unit, the Ivory Trade Review Group (ITRG). The ITRG was responsible for setting up and ensuring enforcement of ivory trade quotas in exporting countries. According to the quota system, each tusk had to be marked and coded by country of origin and then entered into an international database that was used to monitor trade. Authorities were alerted when discrepancies arose. Immediately prior to the ivory quota system becoming effective in January 1986, there was a general amnesty on illegal ivory stockpiles.

[12] Appendix II lists species that are not necessarily now threatened with extinction but that may become so unless trade is closely controlled. It also includes so-called look-alike species, i.e. species of which the specimens in trade look like those of species listed for conservation reasons. International trade in specimens of Appendix II species may be authorized by granting an export permit or re-export certificate; no import permit is necessary. Permits or certificates should only be granted if the relevant authorities are satisfied that certain conditions are met, above all that trade will not be detrimental to the survival of the species in the wild.

Figure A1: Distribution of the African Elephant



Key	
Present Range	Black
Historic Range	Shading

Source: WWF

The ITRG found that CITES controls were relatively easy to evade since illegal ivory traders simply altered their trade routes in order to circumvent Appendix II restrictions. For example, raw ivory was carved to meet the minimum requirements for reclassification as worked ivory before being exported to key manufacturing centres in East Asia. In 1986, approximately 75% of all raw ivory was derived from illegal sources translating to about 89,000 illegally hunted elephants (Sack 1992). Furthermore, sanctions imposed on smugglers were not severe enough to inhibit illegal trade. The ITRG singled out Burundi and South Africa as being havens for illegal ivory traders. Between 1976 and 1986, tusks from an estimated 200,000 elephants were exported from Burundi (Sack 1992). Kenya, Zambia and Tanzania who lost about 80% of the herds were particularly badly hit by poaching. Since the early 1970s Zambia lost almost 90% of its elephants from 250,000 to around 25,000 (Sack 1992). In Luangwa Valley, between 1973 and 1987, an estimated 56,000 elephants were lost to poachers (WWF 1998). Despite the reports that only one live elephant survived in Burundi as of 1988, this country's traffickers exported approximately one-third of the world's annual total of raw ivory (Sack 1992). It is believed that a major ivory smuggling network was being coordinated in South Africa to launder ivory through legitimate CITES channels[13].

The African Elephant and Rhino Specialist Group (AERSG) reported in 1987 that the world demand for ivory superseded competition over land resources as the key factor contributing to the demise of the African elephant. By 1987, the price of raw ivory was US$275.58 per kilogram (Sack 1992). The relative price inelasticity of ivory also fuelled demand. New manufacturing techniques, which enabled the mass production of ivory carvings, along with rising demand in East Asia led to increased elephant kills (Sack 1992). Hong Kong was the primary consumer of raw ivory from 1979 to 1987. Japan was the second largest consumer in this time, followed by Taiwan. For both Hong Kong and Taiwan there were probably significant trans-shipment of the product to China. The European Union's consumption share dropped to 4% by 1987. At the same time, the United States' consumption share rose from 1% in 1979 to 6% in 1987 (see Table 1 in Sack 1992). The larger share of revenue earned from ivory trade was not received by the countries or communities from which the ivory originated, but by professional poaching organizations. For example, ivory in Zaire, Congo, Gabon, and Cameroon sold for only between 10% and 20% of the value obtained upon resale in Hong Kong (Sack 1992).

[13] The possibility of smuggling drives the industry underground where it cannot be monitored and managed hence makes data very uncertain.

Although the tonnage of ivory on the international market remained relatively stable between 1979 and 1989, the size of the tusks decreased significantly. In 1979, the mean weight of traded tusks was 9.8 kilograms, but by 1987 the average tusk weighed only 4.7 kilograms (Sack 1992). This meant that elephants with larger tusks had become very rare and poachers were possibly killing younger animals as well as females, which generally have smaller tusks. Decreasing tusk size also meant that more elephants had to be killed to obtain the same volume of ivory as before. The problems created by the unabated killing of elephants were that (i) the species' genetic resources were being reduced and weakened, and (ii) elephant behaviour was compromised as herd composition was fundamentally altered. Sack (1992) reports that by 1989 only 625,000 elephants survived on the African continent while Kenya's elephant population, for instance, is estimated to have been 170,000 in 1963 and by 1989 the population had declined to a mere 16,000. The United States which was then one of the largest importers of worked ivory in the world, following behind Japan and the European Union who had world consumption shares of 38% and 18% respectively, banned the importation of ivory for commercial purposes (Sack 1992). The major cause of the rapidly declining African elephant population, particularly in the two decades before 1989, was seen to be illegal exploitation of elephant for its ivory, although the decline over the longer run had been attributed to direct competition for habitat between man and elephant. Thus conversion of certain areas to crop-land to feed an ever-growing human population led to an increased amount of human-elephant conflicts, as elephants ate crops planted on lands that were once their feeding grounds. Elephant migratory routes had been interrupted and habitats fragmented by highways, rural developments and human settlements. Fragmentation isolated herds, preventing unrelated elephants from mating with one another – a vital necessity if elephants are to maintain their genetic diversity and survive in perpetuity.

Since 1989 the African elephant, threatened with extinction, is theoretically protected from international trade by its listing on Appendix I of CITES. The success of enforcement of this ban, the level of compliance adhered to by CITES Parties, the response of non-CITES members, and the policy question of the extent to which trade interventions are effective in serving the environmental objective of species preservation, are key issues that have surrounded the debate on the appropriateness of the CITES ban.

Six southern African countries - Botswana, Malawi, Mozambique, South Africa, Zambia, and Zimbabwe opposed the 1989 CITES ban because they did not consider their elephant populations to be threatened with extinction, and because they reckoned that being unable to sell their ivory would seriously limit their investment in conservation. These countries were significant because they were estimated to hold about 40% of Africa's elephants. And it was true that some of their herds were well-managed and flourishing, relative to those north of the Zambezi. The southern African countries actually had too many elephants, and inevitably there had been conflict with humans. Two things had changed that particularly in Zimbabwe. First, farmers had been given legal ownership of wildlife on their land. And second, the value of elephants had risen enormously - mainly because of safari hunting. The more wildlife was worth, the more space farmers gave it in lieu of livestock and the better they looked after it. Trade in elephant products had helped conservation in that it provided funds for investment and it provided financial incentives to land owners thereby reducing the impact of human-elephant conflicts.

Realising that the threats of elephants consists of commercial poaching due to demand of tusks, loss of natural habitat due to increased demand of arable land in light of increased population growth, demand for use in trophy hunting, and the fact that elephants reproduce and grow slowly, the WWF has suggested five priority areas in elephant conservation (WWF 1998).

- slowing the loss of the elephant's natural habitat - mainly by providing support to protected areas and by helping local communities to develop economic activities which benefit both people and elephants on the land they share.
- strengthening activities against ivory poachers and the illegal ivory trade.
- reducing conflict between human and elephant populations through sensible and sustainable approaches.

- determining the status of elephant populations through more and improved surveys and range assessments.
- increasing technical and financial support from the industrialized world to enhance the capacities of local wildlife authorities in all aspects of elephant management and conservation - including the ability to draft enabling legislation and to review, reform, and implement relevant national and international policies.

Several African countries have strengthened their elephant conservation programmes, many of which include setting aside preservation areas and hiring wildlife rangers to protect elephants from poachers. However, limited resources and the eminent danger of poaching operations, as well as the political instability of many African countries, makes it very challenging to implement effective, long-term elephant conservation programmes in Africa. The situation is aggravated by war and civil unrest that is a common feature on the continent. Where firearms are easily obtainable, soldiers unpaid, refugees unfed, and the forces of law and order preoccupied elsewhere, all forms of wildlife are put acutely at risk. Not only middlemen, but corrupt game wardens and government officials might expect a pay-off. Kenya and Zimbabwe instigated a shoot-to-kill policy for poachers. There has been a recovery of elephant populations in some countries due to conservation programmes and, some also attribute it to, the international ban on ivory trade (*see elephant populations in figure below*). Kenya attributed the dramatic recovery of its elephant population to 26,000 elephants in 1996 to the ban in international trade in ivory and other elephant products.

The first legal international sales of ivory after the 1989 ban occurred in May 1999 following decisions made in Harare at the 10th CITES Conference of Parties (COP 10) of June 1997, where the African elephant populations of Botswana, Namibia and Zimbabwe were down-listed from Appendix I to Appendix II and a limited, one-time trade in ivory would be allowed to take place from these three countries to Japan. Ivory auctions were subsequently held in the three countries with Japanese buyers in April 1999. The entire stock eligible for the one-off sale was 49,574 kilograms of ivory, representing 5,446 tusks. It was purchased for approximately US$5 million. The

Nairobi COP 11 of April 2000 voted to put a hold on the international trade in ivory and other elephant products. The elephant population of South Africa was down-listed to Appendix II, under a zero quota trade condition, to join the populations of Botswana, Namibia and Zimbabwe. At the Santiago COP 12 of November 2002, Botswana, Namibia, South Africa and Zimbabwe petitioned to maintain their elephant populations on Appendix II with the stipulation that they be allowed to trade ivory, hides, leather goods, trophies and live animals. Zambia also joined in the petition.

Building on an earlier consensus amongst most African elephant range states, CITES also agreed on a rigorous regime for controlling any eventual trade in ivory stockpiles. It conditionally accepted proposals from Botswana, Namibia and South Africa that they be allowed to make one-off sales of 20, 10 and 30 tonnes of ivory, respectively[14] (see table A1 for ivory stocks and elephant populations of these countries). The ivory is held in existing legal stocks that have been collected from elephants that died of natural causes or as a result of government-regulated problem-animal control. The agreement requires any future one-off sales to be supervised through a strict control system. The sales cannot occur before May 2004 to provide time for baseline data to be gathered on population and poaching levels and for the CITES Secretariat to confirm whether any potential importing countries can effectively regulate their domestic ivory markets and are thus eligible for importing the ivory. The aim of these controls is to prevent any illegal ivory from entering into legal markets and to discourage an upsurge in poaching.

Another protection built into the system is that trade can be suspended if the CITES Secretariat and Standing Committee find either an exporting or an importing country to be in non-compliance. In addition, trade can be stopped if there is evidence that trade negatively affects elephant populations in other regions of Africa. Two monitoring systems were established to track the illegal killing of elephants and illegal sales of ivory i.e. Monitoring of Illegal Killing of Elephants (MIKE) and Elephant Trade Information System (ETIS). These monitoring systems will be critical in ensuring that countries relying on tourism are not harmed by ivory sales from countries that also rely on trade. Thus as of today elephants are protected from trade, save for the positive

[14] Similar proposals from Zambia and Zimbabwe for 17 and 10 tonnes, respectively, were not accepted.

quotas granted to Botswana, Namibia, South Africa and Zimbabwe to be used in trophy hunting.

Table A1: Ivory in five southern African countries as of November 2002 (metric tonnes)

Country	Existing Stocks	Recent Annual Stock growth	Future Potential Annual Stock Growth*	Elephant Population
Botswana	33.0	7.7	10-50	120,000
Namibia	39.0	3.5	1-5	9,000
South Africa	32.0		1-4.5	13,000
Zambia	17.0			29,000
Zimbabwe	20.9	20.0	8.5-42.5	88,000
TOTAL	141.9			259,000

Source: http://www.cites.org/eng/news/press/021112_ivory_update.shtml
* Based on 1-5% natural mortality and low crude average combined tusk weights of 10 kg per individual.

The number of elephants in Zimbabwe was probably less than 5,000 at the turn of the twentieth century. Through protection and sound management the number has now risen to over 88,000 as reported in June 2002[15]. Since the first public concern over vegetation damage by elephants in the 1960s, Zimbabwe has culled elephants over the years in an attempt to limit the severe changes which elephants were, and are still, causing in the Parks and Wildlife Estate, and beyond. With an annual growth rate of 5%, the elephant population is probably the single greatest factor influencing ecosystem conservation in protected areas. The Zimbabwean government has said that it views the elephant as one of many wild species to be conserved in the Parks and Wildlife Estate, and will act to limit elephant numbers whenever scientific evidence indicates that their own numbers are threatening their own habitats and those of other species or producing changes in vegetation which are incompatible with the declared objectives for any given protected area. Outside the Parks and Wildlife Estate, the government has said it recognises the rights of Appropriate Authorities to manage and utilise elephants in accordance with their objectives for land use.

[15] The Herald, Zimbabwe, Monday 3 June 2002. “Jumbo population shoots up to 88,000.

Table A2: Summary of Elephant Estimates in Africa[16].

REGION	Country	Number of elephants				TOTAL AREA (km^2)	RANGE AREA (km^2)
		Definite	Probable	Possible	Speculative		
CENTRAL	Cameroon	1,071	5,285	8,704	675	475,440	229,195
	Central Afr. Rep.	2,515	1,600	6,605	8,000	622,980	314,274
	Chad	0	0	1,600	300	1,284,000	219,130
	Congo	0	0	0	0	342,000	255,373
	D.R. of Congo	3,736	20,219	5,618	120	2,345,410	1,476,560
	Equator. Guinea	0	0	0	80	28,050	14,559
	Gabon	0	0	7,500	54,294	267,670	263,306
	TOTAL	7,322	27,104	27,613	63,469	5,365,550	2,772,397
EASTERN	Eritrea	2	0	0	0	121,320	2,967
	Ethiopia	321	0	0	985	1,127,127	59,717
	Kenya	14,364	11,350	4,882	100	582,650	112,988
	Rwanda	39	0	20	10	26,340	1,019
	Somalia	0	0	130	120	637,660	11,783
	Sudan	0	0	0	0	2,505,810	404,908
	Tanzania	67,416	12,196	12,078	0	945,090	458,315
	Uganda	215	565	1,662	280	236,040	11,872
	TOTAL	83,770	22,698	17,216	1,495	6,182,037	1,063,569
SOUTHERN	Angola	0	0	0	170	1,246,700	678,785
	Botswana	76,644	13,414	13,414	0	600,370	81,486
	Malawi	647	1,569	1,649	20	118,480	7,968
	Mozambique	6,898	1,946	4,496	0	801,590	467,062
	Namibia	6,263	1,421	1,421	0	825,418	145,015
	South Africa	11,905	0	0	0	1,219,912	25,847
	Swaziland	39	0	0	0	17,360	188
	Zambia	15,863	6,179	6,964	0	752,610	208,123
	Zimbabwe	63,070	8,034	10,185	0	390,580	109,563
	TOTAL	196,845	17,057	22,263	190	5,973,020	1,724,037
WEST	Benin	0	0	400	0	112,620	13,036
	Burkina Faso	1,616	606	1,486	0	274,200	18,198
	Ghana	476	218	1,185	443	238,540	30,202
	Guinea	0	0	108	140	245,860	2,277
	Guinea-Bissau	0	0	0	35	36,120	331
	Côte d'Ivoire	51	0	495	645	322,460	35,543
	Liberia	0	0	0	1,783	111,370	22,003
	Mali	0	0	950	50	1,240,000	37,024
	Niger	0	0	817	100	1,267,000	2,694
	Nigeria	157	0	860	236	923,770	34,383
	Senegal	9	0	11	10	196,190	8,428
	Sierra Leone	0	0	0	0	71,740	2,914
	Togo	0	0	96	0	56,790	5,430
	TOTAL	2,489	644	6,628	3,442	5,096,660	212,463
	CONTINENTAL ESTIMATES	301,773	56,196	61,180	68,596	22,729,887	5,785,502

***Source: African Elephant Database 1998. See website:* http://www.iucn.org/themes/ssc/aed/home**

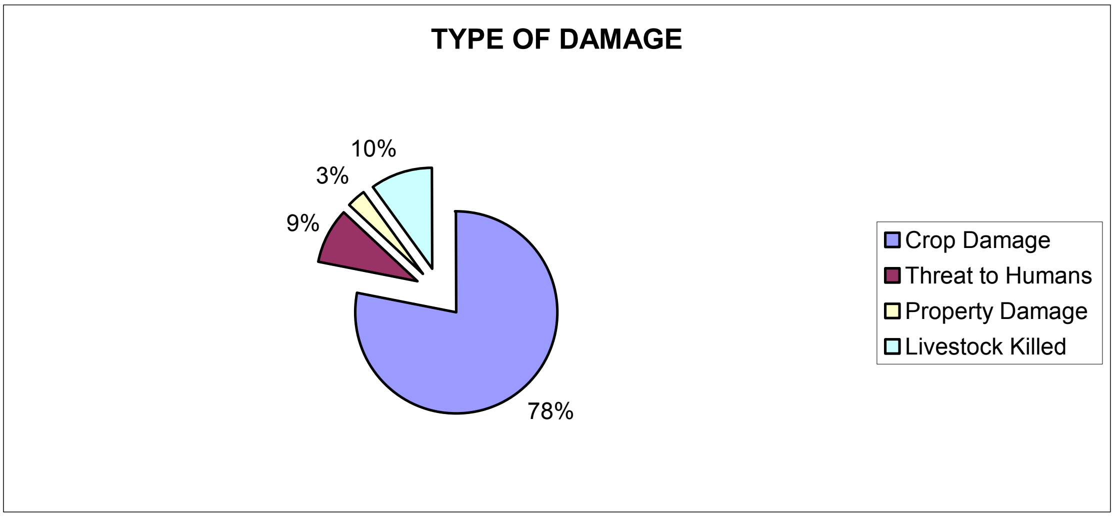
[16] (Explanation: For each country, definite, probable, possible, and speculative numbers of elephants derived from a variety of survey methods and are of decreasing quality are given. Country totals are minimum estimates, based on the estimates for the areas that have been surveyed in that country. Areas of elephant range that have not been censused are not included in these totals. If all of the elephant range is listed, then totals are national estimates).

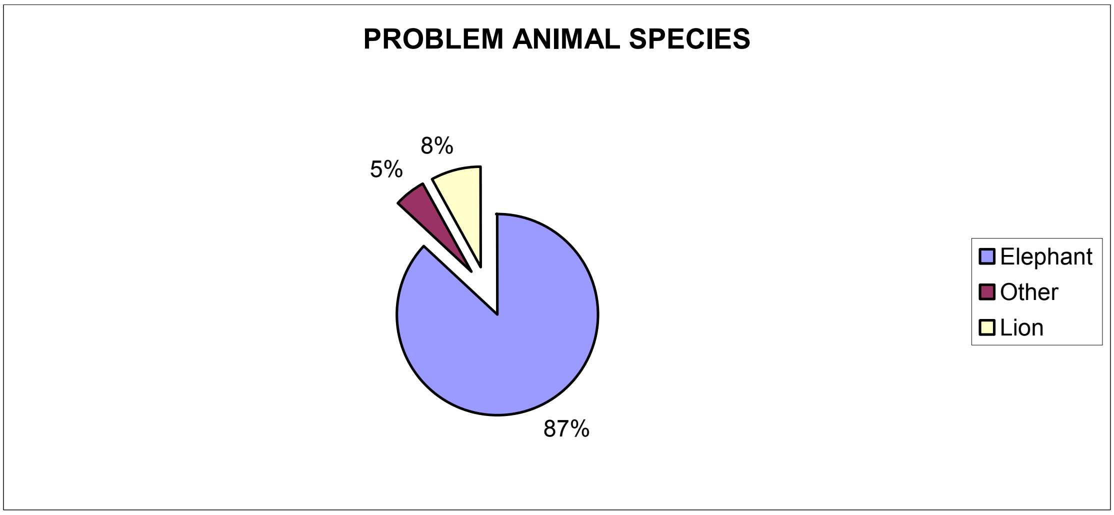
In general, Zimbabweans consider elephants to have an intrinsic value by being a heritage from previous generations. Elephants have lived with humanity for quite a long time that humans have recognised them as equally entitled to live where they are. They are sovereign in their own sense and this is recognised by the society that considers it an abomination to deny elephants of this right of theirs. Nevertheless elephants have significant adverse ecosystem effects. With the lack of efficiency in the elephant stomach, the food requirements of up to 150 kilograms a day entail that too many of them living in one area can easily result in serious damage to the trees and plants. Subsistence farmers constitute about 60% of Zimbabwe's population. The farmer's fields, on which everything depends, could be devastated in a single night by marauding elephants. With such a situation it is difficult to trust the subsistence farmers who are usually victims of elephant perpetrated damage to cooperate in elephant conservation if there are no adequate incentives put forward for them. Consideration of wildlife costs thus forms a central part of the economics of community-based wildlife conservation and strengthens the argument for building community benefit-sharing arrangements into wildlife conservation (Emerton 2001).

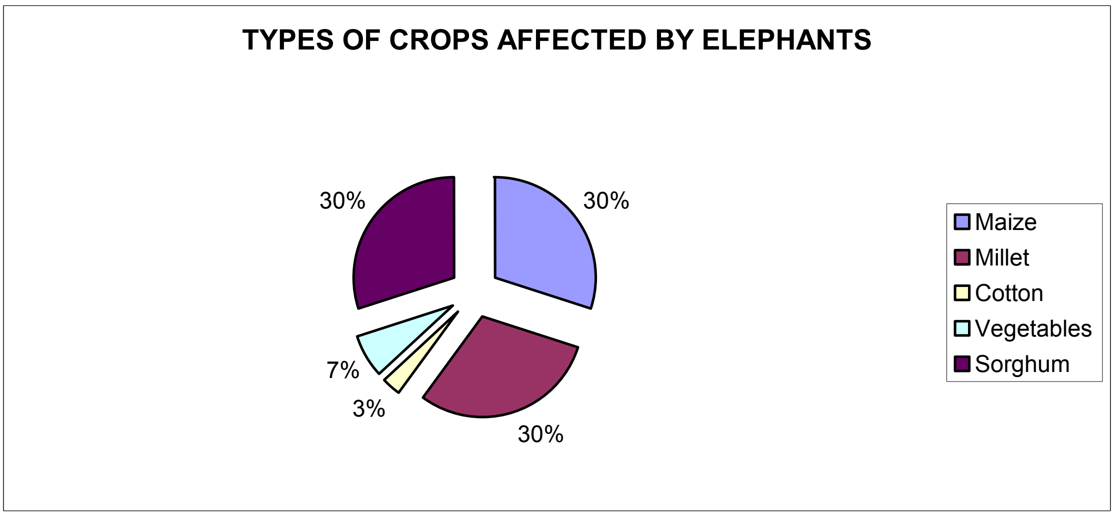
Due to the CITES ban on elephant products, motivated by the need to conserve the elephant, the Zimbabwean elephant population has become an issue not only for Zimbabweans but for the global community as well. This means that Zimbabwe, as the case is, is expected to maintain a population that may be nationally above optimal because such optimality valuations have to be taken at the global level. Overpopulation of the elephant, apart from destroying the habitat of other fauna and flora species that are part of the elephant ecosystem, creates ever-increasing human-elephant conflicts for landholders. Given that most of the elephants will have to live outside parklands with the local communities because (i) there are simply not enough parklands to let there be all the elephants anybody might ever want, and (ii) a contented, protected elephant population can increase by 5% a year and will soon outgrow any area in which it is confined hence move on to greener pastures lest there is trouble, their survival depends on the goodwill of the communities living side by side with them.

No government conservation programme operating in the areas of the communities can be effective without those communities support since it could well be that the more people resent the presence of elephants and feel that the government is preserving them at their expense, the more they will be inclined to sidestep government controls. What is at any rate clear is that the survival of the African elephant outside protected areas now depends to a very great extent on the goodwill of the communities with which it must share living space. And the best way to ensure that goodwill is by fostering schemes that turn elephants from liabilities into assets (*see diagram showing the typical elephant nuisance below*). Without that, the gap between perceptions of elephants internationally and locally will continue to widen, with increasing numbers of local people regarding the revered animals of Western fantasy and wonder as irredeemable agricultural pests, and obstacles to their development. A more definite link has been found between elephant deaths and lack of investment in protection than in elephant deaths and the ban on trade. The illegal killing of elephants cannot properly be considered in isolation from problems of funding and management.

Figure A2: Typical Problem Animal Incidents Recorded In Binga District







Adapted from (Jones 1994)

A BIO-ECONOMIC MODEL OF WILDLIFE-LIVESTOCK CONFLICT AND WELFARE IMPLICATIONS IN THREE RESOURCE USE REGIMES

Edwin Muchapondwa[17,18]

Abstract

This paper formulates a bio-economic model with two agents and two land uses to analyse the conflict between wildlife conservation and livestock production and welfare implications in a typical rural area in Zimbabwe, where a local community lives adjacent to a safari area. The agents are the parks agency and the local community. The parks agency has a fixed amount of land, which is the permanent residence of wildlife, while the local community has user rights over the remaining land. Wildlife tends to roam around the adjacent land imposing a negative externality on the local community's livestock production. Some locals tolerate poaching in order to reduce the number of wildlife. We analyse the wildlife-livestock conflict and the resultant welfare in three resource use regimes in which: (i) the local community does not reap any benefits from wildlife, (ii) the local community gets fixed and exogenous profit shares from wildlife hunting and tourism, and (iii) the social planner undertakes unified resource management. Wildlife conservation is shown to be more successful under regimes in which the local community gets profit shares from hunting and tourism but at a potential cost of the local community's welfare. Policies that could enhance wildlife conservation and social welfare are suggested. Relaxing the assumption of fixed and exogenous profit shares shows that optimal profit shares from hunting and tourism ought to exceed unity. Thus, devolution of wildlife conservation to the local community should be augmented by inflows of external funding.

JEL Classification: H41, Q20
Keywords: bio-economic, CAMPFIRE, community, poaching, wildlife-livestock conflict

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1. Introduction

In this paper we seek to formulate a bio-economic model to analyse the conflict between wildlife[19] conservation and livestock production and welfare implications in a typical wildlife-abundant rural area in Zimbabwe, where a local community lives adjacent to a safari area. Ordinarily, hunting is disallowed in national parks hence we assume that the protected area in question is a safari area, where hunting as well as benign tourism are allowed[20]. Zimbabwe has 17 safari areas covering a total of 1,892,724 hectares, examples of which are Chewore, Chirisa, Matetsi and Sapi (Child 1995). The establishment of national parks, game reserves and safari areas in the late 1920s, after colonisation in 1890, may have helped to avert biodiversity and wildlife loss through development and land conversion but it also displaced rural communities from land that was traditionally theirs. Cultivation and grazing land was lost and the old practice of subsistence hunting became illegal. The newly created parkland became the permanent residence for the wildlife but it could also roam freely in surrounding areas destroying crops and threatening the lives of livestock and people. The Department of National Parks and Wildlife Management (DNPWLM) owned the wildlife in trust for the country and collected the economic benefits produced by wildlife from sale of licences for consumptive wildlife use (hunting) and non-consumptive wildlife services (benign tourism). The local people simply engaged in livestock production and marginal crop agriculture, despite wildlife intrusions. Thus the creation of parklands bore a conflict between wildlife conservation and agricultural development.

Shulz and Skonhoft (1996) and Skonhoft and Solstad (1998) have formulated models similar to the one in this paper for the East African case. The model in this paper extends that work by incorporating (i) poaching that is conducted by people who are external to the local community, (ii) anti-poaching effort exerted by the local community and (iii) the public good effect of wildlife. Due to the prominence that this paper assigns to poaching we will briefly discuss poaching, anti-poaching activities and

[19] While the term ‘wildlife’ generally means animals and plants that grow independently of people, usually in natural conditions, we use it in this paper to mean wild animals.
[20] Our definition of safari area is almost equivalent to the IUCN category VI of protected areas where a protected area is managed to also provide a sustainable flow of natural products and services to meet community needs.

community-based wildlife conservation in Zimbabwe, before we set up the model. The arrangement of the rest of the paper is as follows. Section 2 briefly discusses poaching, anti-poaching activities, and community-based wildlife conservation in Zimbabwe while Section 3 sets up the model. Thereafter the paper will analyse the conflict between wildlife conservation and livestock production and welfare implications in three resource use regimes where: (i) the local community does not reap any benefits from wildlife (Section 4), (ii) the local community gets fixed shares of profit from wildlife hunting and tourism (Section 4), and (iii) there is a social planner who undertakes unified resource management (Section 5). Policies that could be taken to enhance wildlife conservation and social welfare by transforming the current CAMPFIRE resource use regime close to social optimality are suggested in section 6. In section 7 the social planner determines the optimal levels of the profit shares. After the analysis of the comparative statics in Section 8, the paper concludes in Section 9.

2. Poaching, anti-poaching activities and CAMPFIRE in Zimbabwe

Poaching is probably the most important and widespread form of wildlife utilization throughout much of Africa in terms of the quantity of the resource utilised and the economic value of the industry (Milner-Gulland and Leader-Williams 1992a). Generally, poaching could be subsistence or commercial. In the former, wildlife products are for consumption while commercial poachers hunt trophy primarily for sale at a market. Subsistence hunters have close historical, traditional and cultural ties to wildlife and use primitive firearms, spears, snares and dogs (Marks 1984, Skonhoft and Solstad 1996). Commercial poachers, who are usually outsiders employed by dealers, include carriers and professional hunters armed with automatic weapons.

In Zimbabwe, the local communities are themselves perpetrators of small-scale subsistence poaching. Subsistence poachers usually hunt smaller game such as spring-hare, bushbuck, and guinea fowl, which generally have large stocks and high growth rates. This type of illegal harvest has tended to be overlooked by the parks agency to the extent that it remains poaching for the pot. Besides, it is difficult for the parks agency to enforce anti-poaching laws against local communities who themselves permanently live closer to the wildlife resource and stubbornly claim traditional ownership of the wildlife resource, at least for consumption purposes.

Commercial poaching, particularly of larger game, has been the major focus of anti-poaching enforcement. Commercial poachers have been portrayed to be black, poverty-stricken and foreigners. In Zimbabwe, the public perception is that commercial poachers are Zambians and, occasionally, Mozambicans (Duffy 2000). The media and the parks agency in Zimbabwe have reinforced this perception. However, for commercial poachers to succeed they usually make use of a few local informers and accomplices whom they remunerate for their assistance. Thus economic benefits from commercial poaching are largely dissipated from the local communities.

Poaching is thought to have been prevalent particularly outside the protected areas, where wildlife strays and the parks agency has been largely an absentee wildlife owner due to the vastness of habitats, preoccupation with anti-poaching enforcement inside the protected areas and financial constraints. In practice, some anti-poaching activities have been carried out on the parkland by the parks agency in collaboration with the police and defence forces. Outsiders such as the police, defence forces and game ranchers have great difficulty in enforcing anti-poaching measures, given that they do not permanently live close to the safari area. State-sponsored anti-poaching activities have tended to be highly centralised such that it is difficult to apportion the effort exerted on a particular safari area. The parks agency has largely withdrawn its limited services in anti-poaching enforcement outside protected areas, particularly in areas where community-based wildlife conservation has been initiated, though it still sets the hunting quotas for those areas.

In some areas where community-based wildlife conservation has been initiated, the communities, through their RDCs, have embarked on monitoring and protecting wildlife as they take up their new role as co-managers of wildlife. Some RDCs, for example Muzarabani, Guruve, Chipinge, Gokwe North, UMP Zvataida, Binga, Hwange and Nyaminyami, have employed game guards who have been trained and equipped to monitor the state of the resource, carry out problem animal control, carry out anti-poaching campaigns, and monitor the interaction between local communities, safari operators, safari clients and the resource. Most people in the local communities will abide by the local norms and peer enforcement is very important. In a situation where the local communities are alienated from making decisions about wildlife conservation, activities such as the indiscriminate harming, snaring or shooting of wildlife might be tolerated and even seen as reasonable expressions of some traditional right of locals as distinct from the right of outsiders. The local communities also have an interest in getting rid of the nuisance from wildlife, particularly large game. If the local communities are integrated in making decisions about wildlife conservation and receive shares of benefits, then they will be more negative to poaching and thus more supportive of anti-poaching measures. With integration, the local communities could in fact perceive activities such as the indiscriminate harming, snaring or shooting of wildlife as poaching. This very fact alienates the poachers and their accomplices and makes their poaching more difficult and expensive because they cannot count on support from neighbours and the communities. In fact, it eventually becomes natural for neighbours to report that kind of behaviour or discourage it in other ways. All such activities of monitoring, discouraging and reporting unbecoming behaviour have an economic value, which we interpret as the cost of the anti-poaching effort. The local communities decide the extent to which they should provide anti-poaching effort in the framework of maximisation of the economic means of livelihood.

Thus, the local communities have exerted anti-poaching effort through at least four modes: (i) encouraging members to withdraw their services to the commercial poachers as informers and accomplices, (ii) active anti-poaching enforcement by monitoring and protecting wildlife, (iii) reporting unbecoming behaviour with respect to wildlife conservation, and (iv) employing and equipping the anti-poaching units through district administrative structures. Our model will therefore put emphasis on the anti-poaching effort of the local communities rather than the parks agency. We will assume that the parks agency does not carry out any anti-poaching activity.

Evidence from some areas in Zimbabwe shows that poaching was rampant prior to the introduction of community-based wildlife conservation, formally known as the communal areas management programme for indigenous resources (CAMPFIRE). Since the introduction of CAMPFIRE, poaching has been drastically reduced in some areas as the neighbouring communities started reaping economic benefits from legal wildlife utilisation and consequently began to make public arrests of commercial poachers (Child *et al* 1997). However, in other areas poaching only subsided upon inception of CAMPFIRE and reverted back to its previous trend a couple of years afterwards.

Even though wildlife continues to impose a nuisance on the rural communities they now reap economic benefits from it under CAMPFIRE. They generate the economic benefits by selling hunting (licences) to hunters, and collecting fees from tourists engaging in benign tourism. The parks agency, which is still the guardian of all wildlife in Zimbabwe, sets hunting quotas with respect to which hunting (licences) will be sold by the rural communities and itself, each in their designated area of control. Of late, there has been an increasing input from the rural communities in the quota setting process. Under CAMPFIRE, the rural communities engage in wildlife conservation so as to benefit themselves as communities rather than individuals (Patel 1998). The proceeds from hunting (licences) and tourism fees, in their designated area of control, go to the rural communities who will utilise them in a way they decide themselves. Despite the new wildlife conservation task the rural communities still engage in agricultural activities.

3. The model

The bio-economic model comprises two agents and two land-uses. The agents are the parks agency and a local community. The alternative land-uses are wildlife conservation and livestock production. The regions in which national parks, game reserves and safari areas are situated are ecologically best suited for extensive cattle and wildlife ranching. We assume that livestock production is the only feasible agricultural land-use. The parks agency has a fixed amount of land, which is the permanent residence of wildlife, while the local community has user rights over the remaining land. Though the parkland is its permanent residence, wildlife tends to roam around the adjacent land where the local community is practising livestock production. Thus, wildlife and livestock compete for the scarce grazing land outside the safari area. Here, we are thinking of species such as cattle and buffaloes that are capable of co-existing without predation i.e. we limit our analysis to the exploitative competition, rather than interference competition or predation or the threat that the buffalo might infect livestock with diseases such as the foot and mouth disease and brucellosis (for definitions, see Ehrlich and Roughgarden 1987). This analysis does not focus on the threat that wildlife imposes on humans or their other assets.

The ecological interaction between wildlife and livestock is assumed in this paper to be unidirectional, a negative effect from wildlife to livestock, but not vice versa. The justification of this assumption is that while wildlife would generally roam about from the safari area to the neighbouring rangeland, the local community is not allowed to take its livestock into the safari area. Thus conflict over land and the subsequent competition takes place on the rangeland, as opposed to the parkland. The effect of this competition on livestock would be depicted by a depression of livestock growth as the stock of wildlife increases while the growth of wildlife is unaffected by the stock of livestock. Thus the extent of the wildlife-livestock conflict in this setting is simply depicted by the stock of wildlife. The higher the stock of wildlife the greater the negative impact it brings on livestock production.

Assuming that one grazing species of wildlife represents the entire wildlife then the biomass of the stock of wildlife at a specified point in time (where the time index is omitted) is given by $\boldsymbol{W}$. The growth of the biomass of the stock of wildlife is given by equation (1) where $\boldsymbol{F(W)}$ is the natural growth function of the stock of wildlife, $\boldsymbol{h^W}$ is the rate of hunting wildlife, $\boldsymbol{Q(W,\theta)}$ represents the rate of loss of wildlife due to poaching, and $\boldsymbol{\theta}$ is the anti-poaching effort applied by the local community which comes with a cost of $\boldsymbol{s(\theta)}$.[21,22] (***Note:*** superscripts are used to classify variables and parameters

[21] Analysing the problems of poaching, Milner-Gulland and Leader-Williams (1992b) and Skonhoft and Solstad (1996) used the poaching function of the form $\boldsymbol{Q=Q(W,\theta;N); Q_W>0; Q_\theta<0; Q_{\theta\theta}>0;}$ where $\boldsymbol{Q, W}$ and θ are as previously defined while $\boldsymbol{N}$ is a vector reflecting additional factors determining poaching such as the penalty rate, the opportunity costs of poaching, cultural and historical factors. In our own formulation we have dropped the vector $\boldsymbol{N}$ since θ is endogenously determined.

while subscripts denote derivatives; for the convenience of the reader, the symbol and function definitions are in the appendix).

$$\frac{dW}{dt} = F(W) - h^W - Q(W,\theta) \tag{1}$$

One of the potential specifications of the natural growth function $\boldsymbol{F(W)}$ is the logistic i.e. $\boldsymbol{F_W(W)>0}$ and $\boldsymbol{F_{WW}(W)<0}$. Accordingly, the population dynamics of wildlife would be given as:

$$\frac{dW}{dt} = r^W W\left(1 - \frac{W}{K^W}\right) - h^W - Q(W,\theta) \tag{1a}$$

where $\boldsymbol{r^W}$ is the intrinsic rate of increase and $\boldsymbol{K^W}$ is the carrying capacity in the absence of livestock.

The poaching term, $\boldsymbol{Q(W,\theta)}$, is such that no poaching takes place with a zero stock of wildlife, $\boldsymbol{Q(0,\theta)=0;}$ more wildlife means a higher illegal off-take, $\boldsymbol{Q_W(W,\theta)>0;}$ and more anti-poaching effort increases the possibility of detection hence results in lower illegal off-take, $\boldsymbol{Q_\theta(W,\theta)<0}$. The second order derivatives are such that $\boldsymbol{Q_{WW}(W,\theta)<0;}$ $\boldsymbol{Q_{\theta\theta}(W,\theta)>0;}$ $\boldsymbol{Q_{\theta W}(W,\theta)<0}$ and $\boldsymbol{Q_{W\theta}(W,\theta)<0}$ while the cost of anti-poaching effort is such that $\boldsymbol{s_\theta(\theta)>0}$ and $\boldsymbol{s_{\theta\theta}(\theta)>0}$. It is important to note that poaching will not necessarily be zero however large $\boldsymbol{\theta}$ becomes because there is also a certain amount of poaching that cannot be detected even with the cooperation of the local community. Also poaching will have a maximum bound despite having a case where $\boldsymbol{\theta}$ is zero.

The parks agency generates revenue from wildlife by selling hunting (licences) to hunters and collecting fees from tourists who engage in non-consumptive tourism. We assume that the market price of hunting (licences) per unit of harvest has been fixed. The fact that Zimbabwe is only one of the many countries offering sport-hunting opportunities substantiates the price taking assumption. We assume that the costs of hunting (licences) do not fall as the stock of wildlife increases. Instead the average costs

[22] The effects of subsistence poaching have tended to be relegated into the natural growth function by most model builders.

of hunting (licences) are constant. As such $\boldsymbol{p^W}$ is the fixed profit per unit of hunting (licences). The profit from hunting at a rate of $\boldsymbol{h^W}$ is $\boldsymbol{p^W h^W}$. Costs and demand conditions are also assumed to be constant through time. Revenue from non-consumptive tourism, ***T(W)***, will increase with the stock of wildlife i.e. $\boldsymbol{T_W(W)>0}$; $\boldsymbol{T(0)=0;\ T_{WW}(W)<0}$. We abstract from the notion that non-consumptive tourism is a function of an index of biodiversity. The simplifying assumption we made about representing the whole wildlife biomass with one stock necessitates this abstraction. Also we do not explicitly model for the number of tourists engaging in non-consumptive tourism.

In terms of modelling, the issuing of a hunting quota to the local community under CAMPFIRE translates to giving the local community a share of the hunting profits, α, since the parks agency incorporates the quota in its overall optimal harvest decision. Since the local community also has rights to engage in non-consumptive use of wildlife roaming outside the parkland under CAMPFIRE they effectively also get a share, β, of the profits from benign tourism. We assume that the local community's profit shares, α and β, are fixed through time and satisfy $0<\alpha<1$ and $0<\beta<1$. The intuition is that the local community could simply be given profit shares equal to the proportions of the usable wildlife populations that are observed outside the parkland at a specified reference time. These profit shares would then be fixed through the legal system. The profit going to the parks agency from the consumptive and non-consumptive wildlife uses would therefore be given as in equation (2).

$$\Pi^W = (1-\alpha)p^W h^W + (1-\beta)T(W) \tag{2}$$

The public good value of wildlife is defined in (3). ***E(W)*** captures the value of wildlife to the general public in the form of contribution to biodiversity, option value and existence value. The public good value of wildlife will depend on the population for which it is evaluated since ***E(W)*** may comprise local, national and international components. The national component of ***E(W)*** obliges the national government to partake in wildlife conservation. The international component of ***E(W)*** gives room for international transfers to help locals with and reward locals for wildlife conservation.

We would expect that $E(0)=0$; $E_W(W)>0$ and $E_{WW}(W)<0$ for a stock of wildlife that is regarded as a public good. Even though safari areas are also created due to the public good considerations of natural resources, audits are rarely carried out to ascertain whether an adequate amount of the public good component has been supplied. The common observation has been that decisions about stocks of wildlife on safari areas are guided by profit maximisation objectives. We therefore assume that while the benefit from wildlife in the form of fees from tourists engaging in non-consumptive tourism is captured by the parks agency in $T(W)$, $E(W)$ is not captured.

$$E = E(W) \tag{3}$$

The only agricultural activity of the local community is livestock production. The population dynamics of livestock are given by equation (4), where $G(L,W)$ is the natural growth function of the stock of livestock and h^L is the rate of harvesting livestock. As we pointed out earlier, the effect of wildlife grazing competition on livestock would be depicted by a depression of livestock growth as the stock of wildlife increases.

$$\frac{dL}{dt} = G(L,W) - h^L \tag{4}$$

One of the potential specifications[23] of $G(L,W)$ is an extension of the logistic so that it covers interspecific competition, rather than just the intraspecific competition that it already represents. This specification is included in equation (4a) and $G(L,W)$ is such that $G_L(L,W)>0$; $G_{LL}(L,W)<0$; $G_W(L,W)<0$; $G_{LW}(L,W)<0$ and $G_{WL}(L,W)<0$.

$$\frac{dL}{dt} = r^L\left(K^L - L - \eta W\right)\frac{L}{K^L} - h^L \tag{4a}$$

where r^L is the maximum specific growth rate, K^L is the carrying capacity of the area outside the parkland, in the absence of wildlife, and η is the coefficient of interaction between wildlife and livestock.

[23] The wildlife and livestock natural growth functions use a variant of the Lotka-Volterra model where the interaction coefficient in the wildlife equation is suppressed to zero. The Lotka-Volterra model is the simplest model capable of handling the dynamics of the biological interactions between species, particularly predator-prey interactions, and was developed independently by Lotka (1925) and Volterra (1926).

The benefit of livestock production comes from sale and utilisation of consumptive and non-consumptive livestock uses. The consumptive products include meat and skins. The average cost of livestock production, i.e. the cost of fodder, vaccines, veterinary fees, etc, is assumed to be constant. The price or marginal valuation of the livestock off-take is also assumed fixed because there exists an international competitive market for livestock products. The profit from consumptive use of livestock at a rate of h^L is $p^L h^L$, given that the fixed profit per unit is p^L.

When not slaughtered, livestock provides draught power, milk, status, insurance and dung for manure and fuel. In general, the amount of non-consumptive products generated would depend on the stock of livestock. The imputed earnings from the local community's use of livestock products is $D(L)$ where $D(0)=0$; $D_L(L)>0$ and $D_{LL}(L)<0$.

In resource use regimes where profit sharing exists, the local community's share of profit from hunting is $\alpha p^W h^W$ while its share of profit from benign tourism is $\beta T(W)$. As we have indicated earlier, the local community engages in anti-poaching activities at a cost of $s(\theta)$. While some individuals in the local community may also get economic benefits from the poaching activity such income does not enter the local community's welfare function, which the community council uses for decision-making. Therefore the profits received by the local community from both livestock rearing and wildlife activities are given by equation (5).

$$\Pi^L = p^L h^L + D(L) + \alpha p^W h^W + \beta T(W) - s(\theta) \qquad (5)$$

4. The Market Regimes

In the market-based resource use regimes the two agents, the parks agency and the local community, ignore the externalities that they either cause to each other or jointly derive from wildlife conservation. The maximisation problems of these agents are presented and subsequently followed by analyses of the wildlife-livestock conflict and welfare implications in the resource use regimes where: (i) the local community does not reap any benefits from wildlife, and (ii) the local community gets fixed shares of profit from wildlife hunting and tourism. The problems are presented in the context of the resource use regime where profit sharing exists.

4.1 The Parks Agency Optimisation

The parks agency would seek to maximise the present value (PV) of its share of profit from consumptive and non-consumptive wildlife uses by choosing $\boldsymbol{h^W}$ subject to the dynamics of the stock of wildlife (1).

$$\underset{\{h^W\}}{Max}\; PV^W = \int_0^\infty \left[\Pi^W\right] e^{-\delta t}\,dt \tag{6}$$

$$s.t. \quad \frac{dW}{dt} = F(W) - h^W - Q(W,\theta) \tag{1}$$

We will solve this dynamic optimisation problem using Pontryagin's Maximum Principle, where $\boldsymbol{H^{CV}}$ is the current value Hamiltonian.

$$H^{CV} = \Pi^W + (1-\beta)T(W) + \mu 1\left[F(W) - h^W - Q(W,\theta)\right] \tag{7}$$

$$\frac{\partial H^{CV}}{\partial h^W} = (1-\alpha)p^W - \mu 1 \le 0 \qquad \Rightarrow \quad \mu 1 \ge (1-\alpha)p^W \tag{8}$$

$$\frac{\partial \mu 1}{\partial t} - \delta\mu 1 = -\left[(1-\beta)T_W(W) + \mu 1 F_W(W) - \mu 1 Q_W(W,\theta)\right] \tag{9}$$

Assuming the steady state (where $\partial\mu 1/\partial t=0$) the co-state equation yields:

$$\mu 1\left[\delta - F_W(W) + Q_W(W,\theta)\right] = (1-\beta)T_W(W) \tag{10}$$

$$\mu 1 = \frac{(1-\beta)T_W(W)}{\left[\delta - F_W(W) + Q_W(W,\theta)\right]} \ge (1-\alpha)p^W \tag{10a}$$

In an equilibrium with both hunting and tourism:

$$\left[F_W(W) - Q_W(W,\theta) - \delta\right] = -\frac{(1-\beta)T_W(W)}{(1-\alpha)p^W} \tag{10b}$$

4.2 The Local Community Optimisation

Assuming the goal of long-term utility maximisation, the local community maximises the present value (PV) of profits by choosing $\boldsymbol{h^L}$ and θ subject to the dynamics of the stock of wildlife (1) and livestock (4). The local community takes into account the dynamics of the stock of wildlife because its anti-poaching effort affects them.

$$\underset{\{h^L,\theta\}}{Max}\ PV^L = \int_0^\infty \left[\Pi^L\right] e^{-\delta t}\, dt \tag{11}$$

$$s.t.\quad \frac{dW}{dt} = F(W) - h^W - Q(W,\theta) \tag{1}$$

$$\frac{dL}{dt} = G(L,W) - h^L \tag{4}$$

We will solve this dynamic optimisation problem using Pontryagin's Maximum Principle, where $\boldsymbol{H^{CV}}$ is the current value Hamiltonian.

$$H^{CV} = \Pi^L + \lambda 1\left[G(L,W) - h^L\right] + \mu 2\left[F(W) - h^W - Q(W,\theta)\right] \tag{12}$$

$$\frac{\partial H^{CV}}{\partial h^L} = p^L - \lambda 1 \le 0 \qquad \Rightarrow \quad \lambda 1 \ge p^L \tag{13}$$

$$\frac{\partial H^{CV}}{\partial \theta} = -s_\theta(\theta) - \mu 2 Q_\theta(W,\theta) \le 0 \qquad \Rightarrow \quad s_\theta(\theta) \ge -\mu 2 Q_\theta(W,\theta) \tag{14}$$

$$\frac{\partial \lambda 1}{\partial t} - \delta\lambda 1 = -\left[D_L(L) + \lambda 1 G_L(L,W)\right] \tag{15}$$

$$\frac{\partial \mu 2}{\partial t} - \delta\mu 2 = -\left[\beta T_W(W) + \lambda 1 G_W(L,W) + \mu 2 F_W(W) - \mu 2 Q_W(W,\theta)\right] \tag{16}$$

Assuming the steady state (where $\partial\mu2/\partial t=\partial\lambda1/\partial t=0$) and that we always have an interior solution for livestock the co-state equations yield

$$p^L\left[\delta - G_L(L,W)\right] = D_L(L) \tag{17}$$

$$\mu2 = \frac{\left[\beta T_W(W) + p^L G_W(L,W)\right]}{\left[\delta - F_W(W) + Q_W(W,\theta)\right]} \tag{18}$$

Assuming that some hunting occurs we have that:

$$\mu2 = \frac{\left[\beta T_W(W) + p^L G_W(L,W)\right]}{\left[\delta - F_W(W) + Q_W(W,\theta)\right]} = \frac{\left[\beta T_W(W) + p^L G_W(L,W)\right]}{\left[\frac{(1-\beta)}{(1-\alpha)p^W} T_W(W)\right]} \tag{19}$$

Equation (19) is derived from equation (10).

4.3 The Pre-CAMPFIRE Regime

In this sub-section we analyse the conflict between wildlife conservation and livestock production and welfare implications in a resource use regime where the local community does not reap any benefits from wildlife i.e. the profit shares, α and β, are set to zero. This resource use regime, in which the local community does not have any user rights over wildlife, is synonymous with that which existed before the inception of CAMPFIRE i.e. the pre-CAMPFIRE regime.

The necessary conditions for the parks agency and local community maximisation problems reduce to (8), (10), (13), (14), (17) and (18), given that $\alpha=\beta=0$. Firstly, we note that the local community has its own valuation of wildlife that differs from the valuation of wildlife by the parks agency. From equation (19) we get the result that the shadow value of wildlife from the point of view of the local community, $\mu2$, is negative since $\beta T_W(W)=0$ and $p^L G_W(L,W)<0$. Condition (14) shows that the local community will apply anti-poaching effort up to a level at which the marginal cost of such effort is equal to its marginal benefit. The marginal benefit consists of the value of the biomass of wildlife saved from poaching as a result of anti-poaching effort. With a negative shadow value of wildlife, $\mu 2$, the "marginal cost is equal to marginal benefit" optimisation rule depicted in condition (14) will only be satisfied with an optimal anti-poaching effort of zero. Thus the local community applies no anti-poaching effort in the pre-CAMPFIRE regime.

Equation (9) shows that the parks agency will continue to expand the stock of wildlife up to the point where the marginal cost of that expansion is equal to its marginal benefit. The benefit of expansion comprises of the additional tourism profit, the increase in the value of the wildlife resource (capital gain) and the stock effect (a higher stock of wildlife yields increased growth) that comes with an increase in the stock of wildlife. The opportunity cost of conservation of wildlife is the additional poaching that would have been avoided and the foregone interest receipt on proceeds from sale of wildlife that would have been realised had wildlife not been conserved. The opportunity cost of capital, δ, is on the one hand mitigated by the fact that the natural capital (wildlife) itself grows and on the other hand increased by the fact that the stock of wildlife decays due to poaching.

Equation (10) solves for the optimal stock of wildlife, W^n, (the superscript n denotes the non-presence of the local community's user rights on wildlife) given the value of θ^n from the local community's optimisation. Condition (10a) allows for a corner solution. In that case the shadow value of wildlife from the parks agency's perspective would be high enough to exceed $(1-\alpha)p^W$, making optimal hunting zero. This is particularly true when α is close to unity. In a corner solution, the parks agency has no choice since hunting would not be worthwhile. Tourism and the stock of wildlife would evolve according to equation (1). In the equilibrium with both hunting and tourism, the optimal stock of wildlife, W^n, would be derived from equation (10b). Equation (10b) shows that $F_W - Q_W - \delta < 0$ implying that the marginal growth net of poaching is slower than the interest rate to the extent that tourism is relatively more valuable than hunting.

According to equation (15), the local community will expand livestock production until the marginal cost is equal to the marginal benefit. The marginal benefit of conservation of livestock consists of the additional non-market livestock products, the increase in the value of livestock (capital gain) and the stock effect (a higher stock of livestock yields increased growth) that comes with an increase in the stock of livestock. The opportunity cost of the local community's livestock, δ, is mitigated by the natural growth of livestock[24]. The optimal stock of livestock is solved for in equation (17) as L^n, where W^n is given from equation (10). Given that the stocks of wildlife and livestock are ecologically related, in a unidirectional way, the optimal stock of livestock primarily depends on the stock of wildlife. The optimal wildlife hunt, h^{Wn}, and livestock harvest, h^{Ln}, if nonzero, can be obtained from the stock evolution equations (1) and (4) respectively, by assuming the steady state.

The effect of including poaching in the model is the reduction of the optimal stock of wildlife. Thus the presence of poaching discourages wildlife conservation by the parks agency. If the ultimate effect of poaching on the stock of wildlife were to reduce it, it would be the case that in the long run, poaching reduces grazing competition and hence allows the local community to increase the stock of livestock.

The level of wildlife-livestock conflict under the pre-CAMPFIRE regime is indicated by $G(L^n,W^n)$. The welfare of the local community is indicated by their profit from livestock production, $p^L h^{Ln}+D(L^n)$, and that for the parks agency by the profit from wildlife conservation, $p^W h^{Wn}+T(W^n)$.

As Clark (1990) and Skonhoft and Solstad (1998) noted, the dynamics of this two species model will be such that one hunts as much as possible when the initial stock sizes are above the long term optimum while one stops hunting when the initial stock sizes are below the long term optimum. This is the essence of the so-called Most Rapid Approach Path (MRAP) strategy, which basically says approach the steady state as fast as possible. The curious reader is referred to Clark (1990) for more details.

[24] Contrary to usual arguments that the private rate of discount and the rate of discount for most communities in Africa are higher than the social discount rate, for simplicity, we assume throughout this analysis that the discount rate is the same for the two agents and the society.

4.4 The CAMPFIRE Regime

In this sub-section we analyse the conflict between wildlife conservation and livestock production and welfare implications in a resource use regime where the local community gets fixed profit shares, α and β, from wildlife utilisation. This resource use regime is synonymous with CAMPFIRE.

The major change in the maximisation problem of the local community from the pre-CAMPFIRE scenario is that the local community now gets additional profit from wildlife utilisation. It must be the case that when community-based wildlife conservation schemes are put in place, they are structured so as to adequately reward the local community to exert anti-poaching effort. In line with this reasoning, we assume that $\boldsymbol{\beta T_W(W)>-p^L G_W(L,W)}$. Since the local community now gets compensated for the negative externality from wildlife it now places a positive shadow value on wildlife, $\boldsymbol{\mu 2}$, as determined in equation (19). The local community gets a greater incentive to exert anti-poaching effort the higher the share of profit from benign wildlife use. Condition (14) uses the result of the local community's positive shadow value on wildlife to show that the optimal anti-poaching effort, θ^u (the superscript *u* denotes the presence of the local community's user rights on wildlife through profit shares), would be positive. The local community must be sufficiently rewarded in order to secure its cooperation in wildlife conservation.

The differences from the parks agency's pre-CAMPFIRE and CAMPFIRE solutions are the nonzero anti-poaching effort and the nonzero shares of hunting and tourism profits going to the local community which feature under CAMPFIRE. It has been shown in condition (14) and equation (19) that anti-poaching effort would be positive under CAMPFIRE. The effect of increased local community cooperation in wildlife conservation in the CAMPFIRE regime, as shown by $\theta^u > 0$, is to reduce the level of poaching. This in turn enhances the stock of wildlife such that, *ceteris paribus*, the level of conservation is expected to be greater under CAMPFIRE. The positive-ness of θ^u

suggests the possibility of designing CAMPFIRE to enlist the support of the local community in the fight against poaching through the use of economic incentives.

In the equilibrium with both hunting and tourism, the optimal stock of wildlife, $\boldsymbol{W^u}$, would be derived from equation (10b). We denote the optimal stock of livestock from equation (17) by $\boldsymbol{L^u}$, given $\boldsymbol{W^u}$ from equation (10). The effect of CAMPFIRE on the optimal stock of livestock would depend on what happens to the stock of wildlife because the two stocks are ecologically related. If the optimal stock of wildlife increases then in the long run the optimal stock of livestock necessarily has to decrease, and the converse is true. The optimal wildlife hunt, $\boldsymbol{h^{Wu}}$, and livestock harvest, $\boldsymbol{h^{Lu}}$, if nonzero, can be obtained from the stock evolution equations (1) and (4) respectively, by assuming the steady state.

The level of wildlife-livestock conflict under the CAMPFIRE regime is indicated by $\boldsymbol{G(L^u,W^u)}$. The welfare of the local community is now indicated by their profit from livestock production and wildlife conservation, $\boldsymbol{p^L h^{Lu}+D(L^u)-s(\theta^u)+\alpha p^W h^{Wu}+\beta T(W^u)}$, and that for the parks agency by the profit from wildlife conservation, $\boldsymbol{(1-\alpha)p^W h^{Wu}+ (1-\beta)T(W^u)}$. The optimal stock of wildlife under CAMPFIRE will increase from the pre-CAMPFIRE level.

If $\boldsymbol{\alpha \leq \beta}$ then we would expect a greater increase in the optimal stock of wildlife under CAMPFIRE. Both agents have more conservation incentives with profit from tourism. If hunting is more valued by one or the other, then they have less incentive to conserve wildlife. The ultimate direction of change in wildlife conservation would depend on the relative strength of $\boldsymbol{Q_{W\theta}}$ and $\boldsymbol{\mu 1}$. If $\alpha=\beta$, then the parks agency's incentives for wildlife conservation remain the same as under pre-CAMPFIRE but the local community's incentives for wildlife conservation become strictly stronger. Overall, the optimal stock of wildlife would increase under the CAMPFIRE regime than the pre-CAMPFIRE regime. The effect of CAMPFIRE in this respect would have been to increase the stock of wildlife and decrease the stock of livestock i.e. $\boldsymbol{W^n < W^u}$ and $\boldsymbol{L^n > L^u}$. In this situation the local community gets, on one hand, a financial reward from wildlife but, on the other hand, the increase in the stock of wildlife increases the nuisance from wildlife.

Thus according to our narrow definition of conflict, CAMPFIRE results in increased wildlife-livestock conflict since it promotes more wildlife conservation i.e. $\boldsymbol{G(L^u,W^n) > G(L^u,W^u)}$.

It should be noted that, in our model, an increase in wildlife-livestock conflict does not necessarily entail a reduction in welfare. Neither does a decrease in wildlife-livestock conflict necessarily entail an increase in welfare. Using the terminology of Skonhoft (1998), if the 'preference effect' of wildlife is greater than the 'nuisance effect' as shown by the society wishing to have more wildlife and less livestock than in the market regimes (i.e. $\boldsymbol{W^n < W^s}$; $\boldsymbol{W^u < W^s}$ and $\boldsymbol{L^n > L^s}$; $\boldsymbol{L^u > L^s}$ where the superscript *s* denotes social optimality), then an increase in the stock of wildlife would be going in the direction of the social planner's solution hence the local community's welfare must have increased above the pre-CAMPFIRE level. The converse is true. If on the other hand, the 'preference effect' of wildlife is lower than the 'nuisance effect' as shown by the society wishing to have less wildlife and more livestock than in the market regimes (i.e. $\boldsymbol{W^n > W^s}$; $\boldsymbol{W^u > W^s}$ and $\boldsymbol{L^n < L^s}$; $\boldsymbol{L^u < L^s}$), then an increase in the stock of wildlife would be going against the direction of the social planner's solution hence the local community's welfare must have decreased below the pre-CAMPFIRE level. The converse applies. Statements about welfare implications of changes in wildlife-livestock conflicts would therefore require knowledge about socially optimal stocks.

5. The Social Optimality

As can be noted from the above optimisation problems, the parks agency and the local community are ignoring the public good effect of wildlife while at the same time the park agency does not fully take into account the negative externality which wildlife imposes on the local community. Ignoring externalities will necessarily mean that the socially optimal solution would differ from the two market solutions given above. We now assume that a social planner who knows the society's valuation of the costs and benefits of the different land uses and with powers to dictate what the parks agency and the local community should each do carries out unified resource management. The social planner maximises the present value of wildlife and livestock profits while taking into account the nuisance costs and the public good effect of wildlife, ***E(W)***, by choosing $\boldsymbol{h^W}$, $\boldsymbol{h^L}$ and θ subject to the dynamics of the stock of wildlife (1) and livestock (4).

$$\underset{\left\{h^W, h^L, \theta\right\}}{Max} \quad PV^S = \int_0^{\infty} \left[\Pi^L + \Pi^W\right] e^{-\delta t} dt \tag{20}$$

$$s.t. \quad \frac{dW}{dt} = F(W) - h^W - Q(W, \theta) \tag{1}$$

$$\frac{dL}{dt} = G(L, W) - h^L \tag{4}$$

We will solve this dynamic optimisation problem using Pontryagin's Maximum Principle, where $\boldsymbol{H^{CV}}$ is the current value Hamiltonian.

$$H^{CV} = \Pi^L + \Pi^W + \mu 3\left[F(W) - h^W - Q(W, \theta)\right] + \lambda 2\left[G(L, W) - h^L\right] \tag{21}$$

$$\frac{\partial H^{CV}}{\partial h^W} = p^W - \mu 3 \le 0 \qquad \Rightarrow \quad \mu 3 \ge p^W \tag{21a}$$

$$\frac{\partial H^{CV}}{\partial h^L} = p^L - \lambda 2 \le 0 \qquad \Rightarrow \quad \lambda 2 \ge p^L \tag{21b}$$

$$\frac{\partial H^{CV}}{\partial \theta} = -s_\theta(\theta) - \mu 3 Q_\theta(W, \theta) \le 0 \quad \Rightarrow \quad s_\theta(\theta) \ge -\mu 3 Q_\theta(W, \theta) \tag{21c}$$

$$\frac{\partial \mu 3}{\partial t} - \delta \mu 3 = -\left[T_W(W) + E_W(W) + \mu 3 F_W(W) - \mu 3 Q_W(W, \theta) + \lambda 2 G_W(L, W)\right] \tag{21d}$$

$$\frac{\partial \lambda 2}{\partial t} - \delta \lambda 2 = -\left[D_L(L) + \lambda 2 G_L(L, W)\right] \tag{21e}$$

Assuming the steady state (where ∂μ3/∂t=∂λ2/∂t=0) and that we always have an interior solution for livestock the co-state equations yield

$$\mu 3\left[\delta - F_W(W) + Q_W(W, \theta)\right] = \left[T_W(W) + E_W(W) + p^L G_W(L, W)\right] \tag{21f}$$

$$p^L\left[\delta - G_L(L,W)\right] = D_L(L) \tag{21g}$$

$$\mu 3 = \frac{\left[T_W(W) + E_W(W) + p^L G_W(L,W)\right]}{\left[\delta - F_W(W) + Q_W(W,\theta)\right]} \geq p^W \tag{21h}$$

Conditions (21c) and (21h) imply that the anti-poaching effort of the social planner will be positive and even exceed the levels in the two market-based regimes i.e. $\theta^s > \theta^u > \theta^n$. *Ceteris paribus*, the greater anti-poaching activity under the social planner's unified resource management regime would bring about an increase in the stock of wildlife through its greater negative impact on poaching when compared to the pre-CAMPFIRE regime. Equations (21d) and (21e) yield the result that the marginal benefit of expanding each enterprise should be equated to its marginal cost. The ecological interaction of livestock and wildlife is being taken into account in the determination of the marginal benefits and costs. Thus ecological interaction implies economic interdependence. The necessary conditions for this maximisation reduce to (21a), (21b), (21c), (21f) and (21g). If the social planner does not discriminate between hunting and conservation then (21a) is satisfied with equality. From the necessary conditions we can solve for the social optimal θ^s, $\boldsymbol{L^s}$ and $\boldsymbol{W^s}$. The social optimal $\boldsymbol{h^{Ls}}$ and $\boldsymbol{h^{Ws}}$, if nonzero, can be obtained from the stock evolution equations (1) and (4) by assuming a steady state. Due to the competitive nature of livestock and wildlife it follows that sanctioning an increase in the stock of wildlife would entail a reduction in the stock of livestock.

The difference between the social planner's optimal solution and the pre-CAMPFIRE market solution is that the social planner's equation determining the optimal stock of wildlife has a greater anti-poaching effort and some extra terms, given in the bracket on the right hand side of (21f) i.e. ***[$E_W(W)+p^L G_W(L,W)$]***. These additional two terms are the marginal public good effect and the marginal value of livestock biomass lost due to grazing competition from wildlife. The magnitude of ***[$T_W(W)+E_W(W)+p^L G_W(L,W)$]*** relative to the impact of anti-poaching effort, θ^s, will determine whether or not social optimality yields a higher stock of wildlife and consequently a lower stock of livestock than the CAMPFIRE level.

The social planner needs some information about the magnitude of $[T_W(W)+E_W(W)+p^L G_W(L,W)]$ in order to make a ruling on whether a higher or lower stock of wildlife should be conserved compared to the two market based resource use regimes. A popular method used nowadays in the valuation of non-marketed goods such as $[T_W(W)+E_W(W)+p^L G_W(L,W)]$ is the contingent valuation method (CVM). The value of $[T_W(W)+E_W(W)+p^L G_W(L,W)]$ will depend on the constituency of the social planner. For instance, the value that is assigned by either the local community, or the citizens of Zimbabwe, or the international community, or combinations thereof will differ. The standard assumption is that $E_W(W)$ includes the three levels of society mentioned above. In that case, the 'preference effect' of wildlife would be greater than the 'nuisance effect' as shown by social optimality yielding more wildlife and less livestock than in the market regimes. Thus, by increasing the stock of wildlife, CAMPFIRE brought in its wake a financial reward that persuaded the local community to start exerting some anti-poaching effort, increment of the optimal stock of wildlife, greater wildlife-livestock conflicts but a social welfare gain.

6. Policy Recommendations

In this section we explore the policy responses that could stimulate wildlife conservation. Firstly, direct policy interventions concerning the size of stocks can adjust them in line with those in the social planner's optimal solution. Secondly, if we had that $E(W)=0$ then the changes that would have been required to bring the market solution in line with social optimality would have been to simply disallow hunting and give all profit from tourism to the local community i.e. $\alpha=0$ and $\beta=1$ would bring about social optimality. In practice hunting would need to be allowed only to the extent that the parks agency recoups costs i.e. $h^{W*}=R^C/p^W$ where R^C is the revenue requirement. Thirdly, with the current situation where $E(W)$ is positive, raising the value of tourism compared to hunting would help increase the optimal stock of wildlife. In particular, increasing the share of tourism profit going to the local community is expected to increase the stock of wildlife. Both the parks agency and the local community derive more incentives for conservation from the increase in the value of tourism. If hunting already makes parks agency earn more profit than the local community then hunting should be disallowed or allowed only to the extent that the parks agency recoups costs. Hunting revenues would have to be earmarked to subsidise anti-poaching enforcement. Fourthly, the fact that the need for a higher stock of wildlife is partly based on including the high option and existence values of the international community calls for the imposition of a 'Pigouvian subsidy' to the local community in the CAMPFIRE regime. CAMPFIRE requires external financial support in wildlife conservation. The external funding can be viewed as payment for the international portion of existence and option values of wildlife, $E_W(W)$. Also of interest in the policy realm is finding out how much value the social planner would assign to $[T_W(W)+E_W(W)+p^L G_W(L,W)]$ when working with the local community as his constituency. It is conceivable that $[T_W(W)+E_W(W)+p^L G_W(L,W)]$ might be substantially negative for, at least some members of, the local community given that some wildlife is considered a public bad, rather than a public good. For locals who bear most of the costs of living with wildlife it is more likely that local wildlife would be a public bad and even that $E(0)=0;\ E_W(W)<0$ and $E_{WW}(W)<0$. This is particularly true if the wildlife is not nationally endangered. Such a situation gives room to the possibility that the local community would reduce the stock of wildlife with devolution of wildlife user rights to the local community – thus go against social optimality. This strengthens the argument for the need to create extra incentives for the local community through external funding.

7. The Social Planner's Second Best Optimisation

We have so far assumed that the shares of profit from consumptive and non-consumptive wildlife uses going to the local community are determined exogenously. While the exogenous profit shares in our formulation may be justifiable from the fairness perspective they may not necessarily converge with the socially optimal levels. In this section we will therefore allow the social planner to determine the optimal levels of the profit shares, after the optimal stock sizes have already been decided. From (10) and (21f), as we 'force' the parks agency to act as the social planner would, we have that:

$$\frac{(1-\beta)}{(1-\alpha)p^W}T_W(W)=\frac{1}{p^W}\left[T_W(W)+E_W(W)+p^L G_W(L,W)\right] \qquad (22a)$$

$$\frac{(1-\beta)}{(1-\alpha)}=\frac{\left[T_W(W)+E_W(W)+p^L G_W(L,W)\right]}{T_W(W)} \qquad (22b)$$

Substituting for the left hand side ratio in (19), as we 'force' the local community to adopt the same valuation of wildlife as the social planner, we have that:

$$\mu 2=\frac{\left[\beta T_W(W)+p^L G_W(L,W)\right]}{\left[\delta-F_W(W)+Q_W(W,\theta)\right]}=\frac{\left[\beta T_W(W)+p^L G_W(L,W)\right]}{\left[\frac{(1-\beta)}{(1-\alpha)p^W}T_W(W)\right]}=p^W \qquad (22c)$$

$$p^W=\frac{\left[\beta T_W(W)+p^L G_W(L,W)\right]}{\left[\frac{\left[T_W(W)+E_W(W)+p^L G_W(L,W)\right]}{T_W(W)}\frac{T_W(W)}{p^W}\right]} \qquad (22d)$$

$$\beta=\frac{\left[T_W(W)+E_W(W)\right]}{T_W(W)}=1+\frac{E_W(W)}{T_W(W)} \qquad (22e)$$

(22e) shows that $\beta>1$ when $E_W(W)$ and $T_W(W)$ are both positive. Solving for α from (22b) by substituting the value of β from (22e)

$$\alpha=1+\frac{E_W(W)}{\left[T_W(W)+E_W(W)+p^L G_W(L,W)\right]} \qquad (22f)$$

(22f) shows that $\alpha>1$ if $E_W(W)+T_W(W)>-p^L G_W(L,W)$. The result that $\alpha>1$ and $\beta>1$ entails that the management of the parkland should be transferred to the local community and above that the local community should be given a subsidy equivalent to $(\alpha-1)p^W h^W+(\beta-1)T(W)$. In fact, the subsidy will be related to the amount of conservation carried out by the local community since both α and β are functions of **W***. We make four points with respect to the result in this section: Firstly, the result that $\alpha>1$ and $\beta>1$ also follows from the assumption of the lack of discrimination between the shadow value of wildlife and the price for hunting. In reality, it may be that the social planner values conservation more than harvesting hence the two uses will be weighted. Secondly, the result does not take into consideration the possible need by the social planner to raise revenues, partly to finance the subsidy. Bearing this in mind, the resulting optimal profit shares may each actually not exceed unity. Thirdly, if the profit shares α and β are in fact fixed by institutions then it will be difficult to raise them to the optimal levels where they exceed unity i.e. $\alpha>1$ and $\beta>1$. In such a case, optimality could be reached through taxing harvest or subsidising tourism. It should be noted that α will be higher with higher levels of $E_W(W)$. Lastly, an alternative interpretation of the result that α and β should exceed unity may be that the local community requires external financial support in wildlife conservation. The external funding can be viewed as payment for the international portion of existence and option values of wildlife, $E_W(W)$. The rationale for such external funding is that the global society must pay for its portion of existence and option values to raise the level of conservation towards globally desirable levels. External aid should be channelled directly to the producer community if it is to respond to this incentive emanating from increased demand for conservation as contributions towards α and β.

8. Comparative Statics of Social Optimality

The comparative statics can be analysed by appealing to the equations in the appendix. The negative sign of dW/dp^w implies that when the profit from harvesting wildlife increases the stock of wildlife will decrease. This decrease is effected through two mechanisms. Firstly, wildlife becomes relatively more profitable when hunted than alive thereby inducing the society to hunt down the stock. Secondly, the society would ordinarily want to have more of wildlife because it is a public good; but with increased hunting profitability it means that wildlife tourism is tolerated lesser than hunting hence a decrease in the stock of wildlife. Our result is different from that obtained by Skonhoft (1998) and Skonhoft and Solstad (1998) since they assumed a case where the nuisance from the roaming wildlife is strong. The difference comes from the fact that our addition of more non-consumptive wildlife values in our model changes the overall nature of wildlife from being a nuisance to the society to being an overall public good. Clark (1990), in a one-species harvesting model, found that the sign of dW/dp^W was negative. This was because of (i) the nature of the mechanisms in a one-species model where the wildlife-livestock tradeoffs are absent and (ii) the presence of a positive stock effect that originates from the assumption that harvesting costs are negatively related to the stock size. Skonhoft (1998) also concludes that the sign of dW/dp^W will be negative where the marginal tourism benefit dominates the marginal value of the livestock biomass lost due to grazing competition. If we introduce the assumption that harvesting costs are negatively related to the stock size, our result regarding dW/dp^W would continue to hold as long as the marginal effect of the imposed cost dominates the marginal effects originating from the nuisance term and the partial reduction of the stock of livestock.

The positive sign of dL/dp^W implies that an increase in hunting profitability of wildlife increases the stock of livestock. The intuition of this result follows from the above discussion since by preferring less wildlife the society must necessarily increase the optimal stock of livestock. In light of increased hunting profitability, the stock of livestock will inevitably increase because of decreased competition in grazing coming from a lower optimal stock of wildlife and the decreased opportunity cost of wildlife biomass suffered by keeping livestock.

The stock of livestock increases due to an increase in the price of livestock off-take i.e. dL/dp^L is positive. This is particularly true because of our assumption that the marginal non-consumptive effect of livestock, $D_L(L)$, exceeds the all time zero marginal value of wildlife biomass lost due to grazing competition from livestock, $F_L(W)=0$, since livestock does not pose a nuisance to wildlife. The sign of dW/dp^L is ambiguous but we could infer from the positive sign of dL/dp^L and the ecological interdependence of the two stocks that dW/dp^L will be negative. The negative sign of dW/dp^L would imply that an increase in the profitability of livestock would reduce the stock of wildlife. This is likely to be so because a more profitable livestock enterprise would lead to an increased optimal stock of livestock.

The signs of $dW/d\delta$ and $dL/d\delta$ are not clear. Thus we cannot ascertain, in this generalised case, the effects of a permanent increase in the rate of discount on the stocks of wildlife and livestock. We would suspect that initially there would be disinvestments in the biological capital as the rate of discount increases. This is because the opportunity costs of keeping stocks of wildlife and livestock would be high with higher discount rates. In subsequent periods the competition between the stocks of wildlife and livestock will be reduced. This will induce the stocks to once again grow. But the stocks cannot both grow indefinitely. Skonhoft and Solstad (1998) note that allowing larger stocks of both species when δ shifts up will contradict the second order condition for maximum. It must therefore be the case that the initial stock disinvestments effect must dominate for at least one of the stocks.

The change in the valuation of the non-consumptive livestock benefits was introduced by introducing a shift parameter, γ. Change in technology is one thing that could bring about such a change. For example, a new technology that makes nutritious livestock fodder from livestock dung would increase the value of the non-consumptive use of livestock. Dung would receive a higher valuation than it used to have when it only had traditional uses. An increase in the valuation of the non-consumptive livestock benefits would result in an increase in the optimal stock of livestock as implied by the positive sign of $dL/d\gamma$. The increase in the optimal stock of livestock will pave way for the decreased stock of wildlife as depicted by the sign of $dW/d\gamma$.

9. Conclusion

In this paper we formulated a bio-economic model with two agents and two land uses to analyse the wildlife-livestock conflict and welfare implications in a typical rural area in Zimbabwe, where a local community lives adjacent to a safari area. Competition takes place on the rangeland, as opposed to the parkland. The extent of the wildlife-livestock conflict in this setting is simply depicted by the stock of wildlife.

The effect of including poaching in the model is to reduce the optimal stock of wildlife while anti-poaching effort enhances the stock of wildlife. Thus the presence of poaching discourages wildlife conservation by the parks agency since it increases the opportunity cost of capital, δ, by decaying the stock of wildlife. Poaching has the effect to the local community that by acting as a disincentive for wildlife conservation, it allows for an increased stock of livestock.

The significance of the shares of profit from hunting and benign tourism going to the local community has been to directly partially compensate it for nuisance suffered from wildlife and induce it to exert anti-poaching effort. More importantly the profit shares weighted the consumptive and non-consumptive wildlife uses for the parks agency. Overall, the introduction of CAMPFIRE in Zimbabwe had the effect of enhancing wildlife conservation and consequently increasing wildlife-livestock conflict.

Four policies that could help enhance wildlife conservation further were suggested. They include increasing the share of tourism profit going to the local community and the need for external funding as payment for the international portion of existence and option values of wildlife. Relaxing the assumption of fixed and exogenous profit shares shows that optimal profit shares from hunting and tourism ought to exceed unity. Thus, devolution of wildlife conservation to the local community should be augmented by inflows of external funding.

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APPENDIX A

Table A1: Definition of symbols and functions

α	Local community's share of wildlife hunting profits
β	Local community's share of wildlife tourism profits
γ	Shifting parameter
δ	Discount rate
μ	Shadow price of wildlife
λ	Shadow price of livestock
η	Interaction coefficient showing effect of wildlife on livestock
Π^L	Profit of the local community
Π^W	Profit of the parks agency
D(L)	Non-consumptive use of livestock products
E(W)	Public good (bad) value of wildlife
F(W)	Natural growth function of wildlife
G(L,W)	Natural growth function of livestock
H^{CV}	Current value Hamiltonian
h^L	Rate of harvesting livestock
h^W	Rate of hunting wildlife
K^L	Carrying capacity of livestock in rangelands, in the absence of wildlife
K^W	Carrying capacity of wildlife in the absence of livestock
L	Biomass of stock of livestock
N	Vector reflecting additional factors affecting poaching
p^L	Fixed profit per unit of harvested livestock
PV^i	Present value of profits for agent specialising in activity i, i=W,L
p^W	Fixed profit per unit of hunted wildlife
Q(W,θ)	Loss of wildlife due to poaching
r^L	Maximum specific growth rate of livestock
r^W	Maximum specific growth rate of wildlife
T(W)	Revenue from wildlife tourism
W	Biomass of stock of wildlife
s(θ)	Cost of anti-poaching effort
θ	Anti-poaching effort exerted by the local community
n	Non-presence of local community's user rights on wildlife
u	Presence of local community's user rights on wildlife (profit shares)
s	Social optimality

APPENDIX B

B.1. Comparative Statics of the Social Optimality

Taking the total differential from equation (21f) where ***μ3=p^W***,

$$\left[F_{WW} - Q_{WW} - Q_{W\theta}\theta_W - Q_{\theta W}\theta_W - Q_{\theta\theta}\theta_W\theta_W - Q_{\theta}\theta_{WW}\right]dW + \frac{1}{p^W}\left[T_{WW} + E_{WW} + p^L G_{WW}\right]dW$$

$$+\frac{p^L}{p^W}G_{WL}dL + \frac{1}{p^W}G_W dp^L - \frac{1}{\left(p^W\right)^2}\left[T_W + E_W + p^L G_W\right]dp^W = d\delta \tag{23}$$

Adding a positive shift factor, γ, in (21g) which will enable us to evaluate the effect of a change in the valuation of the non-consumptive livestock benefits in section 8 we get,

$$G_L\left(L,W\right) + \frac{1}{p^L}\left[D_L\left(L\right)\right] + \frac{\gamma}{p^L} = \delta \tag{24}$$

Taking the total differential,

$$G_{LL}dL + G_{LW}dW + \frac{1}{p^L}D_{LL}dL - \frac{1}{\left(p^L\right)^2}D_L dp^L - \frac{\gamma}{\left(p^L\right)^2}dp^L + \frac{1}{p^L}d\gamma = d\delta \tag{25}$$

In matrix form equations (23) and (25) can thus be depicted as:

$$\begin{bmatrix} a_{11} & a_{12} \\ a_{21} & a_{22} \end{bmatrix}\begin{bmatrix} dW \\ dL \end{bmatrix} = \begin{bmatrix} b_{11} & b_{12} & b_{13} & b_{14} \\ b_{21} & b_{22} & b_{23} & b_{24} \end{bmatrix}\begin{bmatrix} dp^W \\ dp^L \\ d\delta \\ d\gamma \end{bmatrix}$$

$$a_{11} = \left[F_{WW} - Q_{WW} - Q_{W\theta}\theta_W - Q_{\theta W}\theta_W - Q_{\theta\theta}\theta_W\theta_W - Q_{\theta}\theta_{WW}\right] + \frac{1}{p^W}\left[T_{WW} + E_{WW} + p^L G_{WW}\right]$$

$$a_{12} = \frac{p^L}{p^W}G_{WL} \qquad a_{21} = G_{LW} + \qquad a_{22} = G_{LL} + \frac{1}{p^L}D_{LL}$$

$$b_{11} = \frac{1}{\left(p^W\right)^2}\left[T_W + E_W + p^L G_W\right] \qquad b_{12} = -\frac{1}{p^W}G_W \qquad b_{13} = 1 \qquad b_{14} = 0$$

$$b_{21} = 0 \qquad b_{22} = \left[\frac{1}{\left(p^L\right)^2}D_L + \frac{\gamma}{\left(p^L\right)^2}\right] \qquad b_{23} = 1 \qquad b_{24} = -\frac{1}{p^L}$$

The determinant of the left hand side,

$$D = a_{11}\left[G_{LL} + \frac{1}{p^L}D_{LL}\right] - \left(\frac{p^L}{p^W}G_{WL}\right)\left(G_{LW}\right)$$

is positive due to the second order conditions for a maximum i.e. $a_{11}<0$; $a_{22}<0$ and $D>0$.

$$\frac{dW}{dp^W} = \frac{\frac{1}{\left(p^W\right)^2}\left(T_W + E_W + p^L G_W\right)\left[G_{LL} + \frac{1}{p^L}D_{LL}\right]}{D} < 0$$

$$\frac{dL}{dp^W} = \frac{-\frac{1}{\left(p^W\right)^2}\left(T_W + E_W + p^L G_W\right)\left(G_{LW}\right)}{D} > 0$$

$$\frac{dW}{dp^L} = \frac{-\frac{1}{p^W}\left[G_{LL} + \frac{1}{p^L}D_{LL}\right]G_W - \left(\frac{p^L}{p^W}G_{WL}\right)\left[-\frac{1}{\left(p^L\right)^2}D_L - \frac{\gamma}{\left(p^L\right)^2}\right]}{D} ?$$

$$\frac{dL}{dp^L} = \frac{a_{11}\left[-\frac{1}{\left(p^L\right)^2}D_L - \frac{\gamma}{\left(p^L\right)^2}\right] + \frac{1}{p^W}G_W\left(G_{LW}\right)}{D} > 0$$

$$\frac{dW}{d\delta} = \frac{\left[G_{LL} + \frac{1}{P^L}D_{LL}\right] - \left(\frac{P^L}{P^W}G_{WL}\right)}{D} ?$$

$$\frac{dL}{d\delta} = \frac{a_{11} - \left(G_{LW}\right)}{D} ?$$

$$\frac{dW}{d\gamma} = \frac{\frac{1}{P^L}\left(\frac{P^L}{P^W}G_{WL}\right)}{D} < 0$$

$$\frac{dL}{d\gamma} = \frac{-\frac{1}{P^L}a_{11}}{D} > 0$$

CAN LOCAL COMMUNITIES IN ZIMBABWE BE TRUSTED WITH WILDLIFE MANAGEMENT? : APPLICATION OF CVM ON THE ELEPHANT IN MUDZI RURAL DISTRICT

Edwin Muchapondwa[25,26]

Abstract
If the local communities who live side by side with the elephant see it as valueless then they cannot be trusted to be its good stewards. To assess their valuation of it, a contingent valuation study was conducted for the case of one CAMPFIRE district, Mudzi, in Zimbabwe. An approach that can evaluate projects that generate both winners and losers is used. The study shows that the median willingness to pay for the preservation of an elephant population of 200 is ZW$300 (*US$5.45*) for the respondents who consider the elephant a public good while the same statistic is -ZW$98 (-*US$1.78*) for the respondents who consider the elephant a public bad. The preservation of an elephant population of 200 in Mudzi yields an annual net worth of ZW$123,771 (*US$2,250*) to the households living in CAMPFIRE wards. However, the majority of households do not support the preservation of the current elephant population since 62% of them would rather not have the elephant because they view it as a nuisance. This is one argument against devolution of elephant conservation to the local communities. The rural communities' perceptions of the elephant are generally useful for other species of wildlife since the elephant is considered a keystone species and, most importantly, an umbrella species in the African Savannas. Adequate economic incentives must be extended to the local communities if a majority of them is to be persuaded to partake in sound elephant conservation. External transfers constitute one way of providing additional economic incentives to encourage elephant conservation by local communities such as Mudzi. Co-management should be the preferred mode of communities' involvement in wildlife conservation.

JEL Classification: C25, H41, Q26
Keywords: CAMPFIRE, dichotomous choice, elephant nuisance, simple spike

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1. Introduction

Agriculture and wildlife conservation compete for the scarce land in rural Zimbabwe. Wildlife conservation must compete economically with agriculture, which is the prime source of rural communities' livelihood, if it is to be accepted as an alternative land-use. Thus the survival of wildlife depends on whether it is an asset or liability to the communities living adjacent to it. More often, even when the people in wildlife abundant areas are furnished with community development benefits from wildlife revenues, they still lose out in economic terms from the presence of wildlife (Emerton 2001). Of interest in the policy realm is assessing the economic value that the local communities living adjacent to national parks, game reserves and safari areas assign to wildlife, given that some people potentially consider it a public good while others consider it a public bad. If the economic value of wildlife relative to the communities' other economic activities were larger then it would imply that wildlife conservation might be enhanced through devolution of wildlife user rights to the local communities.

In this paper, we concentrate on the elephant as the representative wildlife since the elephant is the most important species to the local communities both in terms of the damage it causes to crops and the value it contributes to CAMPFIRE revenues. In general, the benefits from the elephant are (i) products that can be consumed directly, such as income from live sales, meat, hides, trophies, (ii) tourism, (iii) ecological and environmental services such as maintenance of the African savannas and biodiversity, (iv) possible future uses such as touristic, pharmaceutical, industrial and agricultural applications, and (v) intrinsic values such as religious, cultural, aesthetic, existence and bequest significance (Emerton 2001). The costs that the elephant imposes include (i) management costs such as costs of equipment, capital, wages, running costs, policing, etc, (ii) costs to other livelihood activities in the form of crop destruction, human injury, damage to structures, etc, and (iii) opportunity costs in the form of alternative land, money, time or resource uses. The benefits of the elephant potentially accrue at both global and local levels while the costs occur exclusively at the local level. The international component of elephant benefits gives room for international transfers to help locals with, and reward locals for, elephant conservation. This has been the motivation for international support to CAMPFIRE. However, external aid should be channelled directly to the producer communities if they are to respond to it as increased demand for conservation.

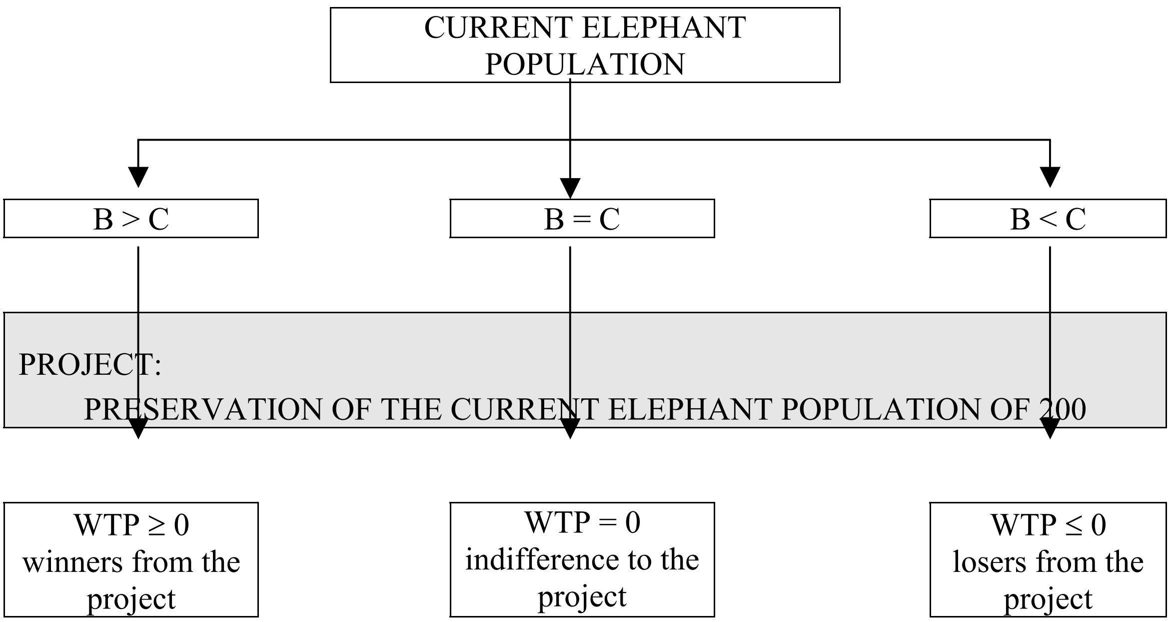
The objective of this paper is to estimate the willingness to pay (WTP) for the preservation of a designated sub-population of the African elephant (*Loxodonta africana*) in Zimbabwe by rural communities who live adjacent to a designated game reserve taking into account the reality that some people consider it a public good while others consider it a public bad. We do this by employing the contingent valuation method (CVM) for the case of one CAMPFIRE district, Mudzi. The rest of the paper is arranged as follows: Section 2 reviews literature on the economic valuation of non-marketed environmental goods (bads). Section 3 describes the survey area while Section 4 describes the survey, presents sample characteristics and gives economic rationale for the bid function. Section 5 presents estimation results while concluding remarks are given in Section 6.

2. Economic Valuation of Non-marketed Environmental Goods (Bads)

Many environmental projects generate both winners and losers. Some projects make it rather easy to pinpoint winners and losers while others might make such identifications much more difficult. In any case, even if a respondent is asked to state her WTP in an open-ended contingent valuation question, it might not occur to state a negative WTP if she dislikes the project (Kriström 1995). With the dichotomous choice question format, it is difficult to pose the question in such a way that potential losers are identified. The most commonly employed approach for capturing welfare losses in contingent valuation studies has been to make assumptions regarding the negative tail of the WTP distribution, after eliciting for WTP for a change in the provision of an environmental amenity in the traditional way. Nonetheless, this approach does not allow respondents themselves to bid negative amounts (Clinch and Murphy 1999). By restricting WTP to being non-negative, contingent valuation studies routinely ignore the fact that many environmental amenities manifest themselves as costs to some and benefits to others.

This paper uses an approach, also used by Clinch and Murphy (1999), where respondents are split into three sub-samples according to their stated preferences for the project and each sub-sample is analysed separately. A screening question is utilised to allocate respondents into their categories of preference for the project. In the valuation of the African elephant we identified the spectrum of preferences for the project by first asking the respondents to think about the costs and benefits their households assign to the current elephant population in a designated area and indicate which magnitude exceeds the other. There are three possible responses: (i) benefits exceed costs; (ii) benefits equal costs; and (iii) benefits are surpassed by costs. These responses are assumed to correspond to the following preferences for the project that entails the preservation of the current elephant population in a given area against the background of threats by the government to translocate the elephant out of that area: (i) winners from the project, (ii) those who are indifferent, and (iii) losers from the project. The figure below gives the taxonomy of the expected responses from the elicitation of WTP for the preservation of the elephant.

Figure 1: Taxonomy of responses on the WTP for the preservation of the elephant



In the stated preference survey conducted in Mudzi rural district, the respondent was reminded of the potential benefits and costs of the elephant. The first question asked the respondent how his/her household weighed the benefits and costs of living with the current elephant population of ***200***, by considering only those benefits and costs applicable to them. The district-wide elephant population was estimated at ***200*** and all respondents were asked about this population. The screening question allowed us to categorise the respondents into categories corresponding to their sign of WTP for the preservation of the current elephant population. Firstly, if the respondents indicated that the benefits of living with an elephant population of ***200*** exceeded the costs, B > C, then they would be expected to have a positive WTP for the preservation of the current elephant population. For some respondents in this category it could be that they say benefits exceed costs yet they will eventually report a zero WTP, possibly due to the eventual correction of initially misrepresented preferences in the wake of a demand for them to declare concrete evidence of their preferences by making a payment. To allow for such zero WTP we categorise those for whom benefits exceed costs as having a non-negative WTP for preserving the current elephant population, rather than positive WTP. Secondly, if the respondents indicated that the benefits of living with an elephant population of ***200*** equalled the costs, then they would be indifferent about having the elephant or not and subsequently we concluded that they would have a zero WTP for the preservation of the current elephant population. Thirdly, if the respondents indicated that the benefits of living with an elephant population of ***200*** were surpassed by costs, B < C, then they would be expected to have a negative WTP for the preservation of the current elephant population. When WTP for the preservation of the current elephant population is elicited as the cost of a project entailing the translocation of the elephant population it might occur that some respondents who claim membership in this category will report a zero WTP, possibly due to the eventual correction of initially misrepresented preferences in the wake of a demand for them to declare concrete evidence of their preferences by making a payment. To allow for such zero WTP we categorise those for whom benefits are surpassed by costs as having a non-positive WTP for preserving the current elephant population, rather than negative WTP.

No follow-up question was asked if the respondents said that the benefits of living with the elephant equalled costs. It was assumed that they have a zero WTP for the preservation of the current elephant population. Formally, they are indifferent about the preservation of the current elephant population hence they are not in the market for it. If the respondents indicated that the benefits exceeded costs, B > C, then they were presented with a proposal that the government was considering translocating the current elephant population of ***200*** from their district to other districts so that the people there could also benefit from the elephant since it is a national heritage. However, they could avoid the elephant translocation if their community could pay an annual 'translocation avoidance' tax to the government (*see questionnaire in Appendix for exact question formulation*). They were then asked whether their household was willing to pay an annual 'translocation avoidance' tax of ZW\$$\boldsymbol{x}$ for as long as these animals shall be in their area, given that all other households who do not find the elephant to be a nuisance will also pay the same amount, so that they could be allowed to continue living side by side with an elephant population of ***200***[27,28].

If the respondents indicated that the benefits were surpassed by costs, B < C, then they were presented with a similar proposal differing only in that they had to pay a 'translocation and compensation' tax in order for the government to be able to carry out the translocation of the current elephant population from their district to other districts, where the nuisance that they have been experiencing from these animals will consequently be transferred. They were then asked whether their household was willing to pay an annual 'translocation and compensation' tax of ZW\$$\boldsymbol{x}$ for as long as these animals shall live in the other area, given that all the other households who find the elephant to be a nuisance will also pay the same amount, so that the translocation and compensation exercise of an elephant population of ***200*** could be carried out[29].

Even though our approach enables us to classify respondents into distinct categories based on their assessment of elephant benefits and costs i.e. (i) benefits exceed costs,

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[27] The starting bids that were chosen on the basis of data collected in a pilot survey were $\boldsymbol{x} \in \{30, 190, 250, 330, 500\}$. We used the 20th, 35th, 50th, 65th and 80th percentiles.

[28] In fact, the double bounded dichotomous choice with open-ended follow-up question format was used here. The annual 'translocation avoidance' tax presented for the same project in the second question was (i) some lower amount ZW\$$\boldsymbol{x}_L$ if the respondent answered 'no' to ZW\$$\boldsymbol{x}$, or (ii) some higher amount ZW\$$\boldsymbol{x}^H$ if the respondent answered 'yes' to ZW\$$\boldsymbol{x}$. The respondents were eventually asked to state their household's maximum WTP for the project presented to them. This paper only analyses the single bounded responses.

[29] The double bounded dichotomous choice with open-ended follow-up question format was also used here.

(ii) benefits equal costs, and (iii) benefits are surpassed by costs, we have a challenge of disentangling and appropriately analysing potential zero WTP that could eventually be reported by some respondents in the categories where either elephant benefits exceed costs or benefits are surpassed by costs. Previous studies (for example, Kriström 1995, Clinch and Murphy 1999) first posed a question asking if the respondent would be willing to pay anything at all for the proposed project, followed by debriefing questions, ahead of a dichotomous choice question, to determine whether she had zero WTP or not. In our case, such information is obtained from an open-ended follow-up question and a debriefing question. All respondents who reported zero WTP despite their elephant benefits exceeding costs or elephant benefits being surpassed by costs attributed their zero WTP to the budget constraint. Despite earlier statements by the respondents that elephant benefits exceeded costs or elephant benefits were surpassed by costs, such zero WTP is interpreted as the 'true' WTP. The inconsistency in the two responses is resolved by arguing that either (i) the respondent initially misrepresented her preferences and eventually corrected that misrepresentation when concrete evidence of her preferences was sought or (ii) the respondent initially presented preferences that are consistent with a freely provided public good or freely disposable public bad and eventually revised them when it became apparent that she had to pay for the provision of the public good or disposal of the public bad. By indicating that they are not in a position to pay a positive amount for the proposed project, the respondents are essentially saying that their welfare is not affected by the project for as long as they have to contribute anything towards it. Thus in the sub-samples of winners and losers from the project we have respondents with 'true' zero WTP. In order to link the reported zero WTP to the particular projects that respondents faced we will analyse the data categories based on the respondents' assessment of elephant benefits and costs i.e. we make use of three sub-samples based on the responses that: (i) benefits exceed costs; (ii) benefits equal costs; and (iii) benefits are surpassed by costs.

Our interest lies in analysing the WTP elicited from the dichotomous choice question while making use of the additional information about respondents with zero WTP in the sub-samples of those for whom elephant benefits exceed costs or elephant benefits are surpassed by costs. The simple spike model provides one way to allow for zero WTP in the analysis of responses from the dichotomous choice question format. The sub-sample is essentially partitioned into respondents with zero WTP and positive WTP. In the simple spike-model it is assumed that the distribution function of WTP has the following form (Kriström 1995):

$$F_{WTP}(A) = \begin{cases} 0 & \text{if } A < 0 \\ p & \text{if } A = 0 \\ G_{WTP}(A) & \text{if } A > 0 \end{cases} \tag{1}$$

where $p \in (0,1)$ and $G_{WTP}(A)$ is a continuous and increasing function such that $G_{WTP}(0) = p$ and $lim_{A \to \infty} G_{WTP}(A)=1$. Thus, there is a jump-discontinuity – a spike – at zero. This model can be estimated with a variety of approaches (Kriström 1995). The simple spike model basically uses two valuation questions (i) whether or not the respondent has nonzero WTP, and (ii) whether or not the respondent would want to pay the price, *A*, suggested in the dichotomous choice question. For each respondent *i*, define an indicator that tells if the respondent has nonzero WTP or not.

$$S_i = 1 \text{ if WTP}>0 \text{ (0 otherwise)} \tag{2}$$

Similarly, let T_i indicate if the respondent is willing to pay the suggested price, A

$$T_i = 1 \text{ if WTP}>A \text{ (0 otherwise)} \tag{3}$$

The log-likelihood for the sub-sample is then given by:

$$l = \sum_{1}^{N} \left[S_i T_i \ln(1 - F_{WTP}(A)) + S_i (1 - T_i) \ln(F_{WTP}(A) - F_{WTP}(0)) + (1 - S_i) \ln(F_{WTP}(0)) \right] \tag{4}$$

The likelihood function (4) can be programmed in econometric packages such as LIMDEP. If WTP follows the logistic distribution and utility is linear-in-income, it can be shown that the mean WTP equals (1/b)ln(1+exp{a(s)}) and the median WTP equals [a(s)/b] if the spike occurs at a point less than 0.5, otherwise the median WTP would be zero; where ***s*** is a vector of social characteristics (Kriström 1995). The spike is defined as the value $F_{WTP}(A)=0$, i.e. the probability that WTP equals zero. In equation (1), setting A to zero gives the spike as $1/(1+\exp\{\alpha(s)\})$.

3. The Survey Area

Mudzi rural district lies at the north-eastern border of the country to Mozambique. The whole district's area of 4,222 square kilometres lies in agro-ecological regions IV and V, which are best suited for extensive cattle and wildlife ranching[30]. The district has a population of 23,995 households, 16 administrative wards, 7 chiefs, 9 headmen and 700 village heads. The Ministry of Public Service, Labour and Social Welfare (1997) reported that, according to the Food Poverty Line (FPL) and Total Consumption Poverty Line (TCPL), Mudzi was the fourth poorest district in Zimbabwe. Mudzi borders Nyatana game reserve and in 1992 Mudzi Rural District Council (RDC) got Appropriate Authority (AA) status to manage its share of wildlife in Nyatana alongside the adjacent Rushinga and Uzumba-Maramba-Pfungwe (UMP) RDCs[31]. In 1993, a safari operator called Zindere Safaris was contracted for 5 years (March 1993 to February 1998) to utilise the annual wildlife harvesting quotas issued by the Department of National Parks and Wildlife Management (DNPWLM) (*see the schedule of hunting quotas in table A4 in the appendix*). The safari operator paid an annual concession fee of ZW$15,000 (*US$2,300*) and 80% of the trophy fees. In 1994, 12 elephant bulls were translocated into Nyatana game reserve with the help of Zindere Safaris and an organization called Elephant for Africa. In the late 1990s, a buffer zone was established and the safari operator harshly enforced the buffer zone borders denying adjacent communities access to fodder, timber and grass. Due to enormous pressure from the communities the safari operator's contract was not renewed.

Since March 1998 a new safari operator called Umfurudzi Wilderness Safaris has been engaged to utilize the wildlife harvesting quotas on a 20-year contract. In 1999 eight

[30] Based primarily on the amount and reliability of rainfall, Zimbabwe is divided into five agro-ecological regions of generalised land use potential. For purposes of agriculture, region I is excellent land while region V is marginal land.
[31] The landscape of Nyatana is such that there could be long term movements of wildlife between and amongst the three rural districts' boundaries even though in the short run distinct clans of wildlife can be identified and ownership assigned to the three RDs.

elephant bulls were translocated into Nyatana game reserve with the help of Elephant for Africa and the new safari operator. Elephant translocations are resented by some communities who argue that the elephant is already locally overpopulated considering the high incidence of elephant intrusion. Currently there are plans to re-stock Nyatana game reserve with an elephant population of 100, among other species, with grants from Elephant for Africa and USAID.

Two wards, Mukota A and Chimukoko, with a population of 3,009 households and covering 1,009 square kilometres have been designated as CAMPFIRE wards. These are the only wards in Mudzi that border Nyatana. 70% of wildlife revenue is shared equally amongst 7 villages in these two wards while the remaining 30% is retained by the RDC for district development (15%) and administrative expenses (15%). The villages decide how to use their share of the proceeds from game without any interference from the RDC. The revenue has been used to undertake various community projects such as buying scotch-carts for ferrying local materials for construction of the local clinic, building a classroom block at the nearby school, construction of blair toilets, a pre-school and roofing teachers' houses, buying cattle for hiring out to members, maize trading, sinking boreholes and water trophies for livestock, buying some fence, textbooks, building materials for the local school and building a court-room for the traditional chief. The RDC audits the villages' books of accounts and adjudicates misuse cases.

4. The Survey

A questionnaire was administered in Mudzi rural district in December 2000 to 570 randomly selected households. Given the objective of valuing the elephant from the adjacent communities' perspective, the extent of the market was demarcated by proximity to Nyatana game reserve. Thus, our interest was in households that live in wards that have been designated as CAMPFIRE wards. The questionnaire sought basic household data, household participation in CAMPFIRE activities, data on human-wildlife conflict and compensation, stated preference survey and data reliability (*see questionnaire in appendix*). The focus groups and pilot study indicated that it was difficult to obtain time series data on incomes and production. The heads of the households or their interview representatives were the interviewees and in both cases responses can be interpreted as coming from the heads themselves. The age range of the heads of households was 17 to 90. The basic sample characteristics are shown in table 1 below.

Table 1: Basic sample and sub-sample characteristics

	Full sample N=570		B > C sub-sample N_1=197		B < C sub-sample N_2=352	
Characteristics	**Mean**	**Std Dev.**	Mean	**Std Dev.**	**Mean**	**Std Dev.**
Household Size	5.61	2.67	5.35	2.90	5.72	2.49
Sex of Household Head (M=1,F=0)	0.66	0.47	0.64	0.48	0.67	0.47
Age of Household Head (years)	41.84	14.87	37.23	12.53	44.53	15.35
Education-years of Household Head	5.25	4.29	6.85	3.96	4.34	4.17
Awareness of CAMPFIRE (Y=1,N=0)	0.55	0.50	0.65	0.48	0.49	0.50
Distance to Elephant Reserve (km)	10.23	10.32	14.74	12.52	7.46	7.53
Size of Intruding Elephant Herd	5.71	6.34	3.38	6.81	7.19	5.69
Existence of Mitigation Measures (Y=1,N=0)	0.36	0.48	0.21	0.41	0.45	0.50
Support Parks Driven Ele. Conservation (Y=1)	0.28	0.45	0.36	0.48	0.22	0.42
Agriculture as Main Activity (Y=1,N=0)	0.85	0.35	0.81	0.39	0.89	0.31
Labour-days Against Elephant Intrusion	39.17	45.40	14.46	34.06	53.89	44.99
Annual Household Income (ZW$)	19,488	42,493	26,748	55,948	15,763	33,520

4.1 Economic rationale for the bid function

The objective of stated preference surveys is to elicit respondents' valuation of the projects described to them in scenarios. The reliability of each survey is typically measured through the estimation of a bid function relating WTP responses to a variety of covariates collected in the survey. The goal is to assess the extent to which relevant expectations from (i) economic theory, (ii) prior intuition, and (iii) observed empirical regularities are fulfilled. The analysis of those variables in the design of the survey that can potentially affect WTP can shed light on the robustness of the survey design and implementation of the study. From a policy perspective, the reasons behind differences in WTP can therefore be better understood.

Since we are dealing with household-level data, household characteristics such as household size, sex of the household head, age of the household head, education of the household head and total annual household income are expected to be important in explaining the households' WTP for the preservation of the current elephant population.

It is reasonable to think that larger households would benefit more than proportionately from a public good than smaller households. As such household size is expected to have an effect on WTP for the preservation of the current elephant population. Male-heads are more in contact with nature and have more ways to cope with the elephant nuisance than their female counterparts hence male-headed and female-headed households are expected to have differences in WTP for the preservation of the current elephant population. Education helps people to appreciate more the value of the elephant, or lack of it, since they can easily comprehend its negative externality and passive uses as well, which are expected to carry more weight in its valuation. Households with higher annual incomes are expected to afford relatively higher WTP as they may have less income constraints. The extent to which young and old household heads observe the African culture and traditions differ. The elephant is an integral part of the African culture and could be valued differently by these two categories of household heads, each according to the extent to which they uphold the culture and traditions.

Attitudinal variables are also expected to be important in explaining the households' differences in WTP for the preservation of the current elephant population. In this broadly defined category we have variables such as (i) awareness of CAMPFIRE, and (ii) support of parks agency driven elephant conservation. Respondents were asked whether they were aware of the working of CAMPFIRE by having actively participated in its activities. Awareness of CAMPFIRE could be used to evaluate the success of the programme in cushioning the effects of the negative externality from the elephant. Respondents were also asked whether they would support parks agency driven elephant conservation rather than community driven elephant conservation. Wildlife revenue investment decisions are expected to differ between the parks agency and the communities. A plausible scenario is one where investment by the community increases the threat of elephant intrusion while perceived investment by the parks agency could reduce it. In community driven conservation, the community might use wildlife income to purchase communal livestock and declare some land adjacent to the wildlife reserve a buffer zone. For members with personal livestock this decision reduces grazing land, increases grazing competition, denies everyone access to other resources in the buffer zone and possibly does not reduce human-elephant conflict. Indeed, in many CAMPFIRE areas, in general, and Mudzi, in particular, wildlife incomes have been used for social infrastructure rather than intrusion preventive measures. In parks agency driven elephant conservation, the parks agency might be expected to use wildlife income to fence off the wildlife reserve. Households potentially support parks agency driven elephant conservation rather than community driven elephant conservation regardless of whether they consider the elephant a public good or nuisance. Thus, the attitude of households with respect to whether they support either parks agency driven elephant conservation or community driven elephant conservation has a bearing on the characterisation of the elephant as either a public good or public bad and its valuation.

The next set of variables that would be of interest are those which indicate the access of the elephant to the means of livelihood of households, and the risk of suffering elephant intrusion. The following variables are in this category: existence of elephant intrusion mitigation measures, distance to the elephant reserve, labour-days spent guarding against elephant intrusion, average size of intruding elephant herd, and having agriculture as a main livelihood activity. In this paragraph we will describe each variable’s individual impact. Those households who have cushioned themselves from elephant intrusion by installing elephant intrusion mitigation measures such as thorny shrub fences are expected to regard the elephant as being more valuable, all other things being equal. Distance to the elephant reserve captures a certain kind of risk of elephant intrusion. If agricultural activities of a household are closer to the elephant reserve there is a greater chance that they will be intruded, *ceteris paribus*. Those households who spend relatively more labour-days guarding against elephant intrusion, all other things remaining equal, are less likely to view the current elephant population as a public bad because they face a lower level of intrusion risk. This is particularly true where the opportunity cost of such labour is negligible, if not zero. If a larger herd of the elephant potentially has access to a household's assets then that household is expected to have a lower WTP for the preservation of the current elephant population compared to those households who face intrusion threats from a smaller herd. The larger the threat of elephant intrusion, the lower the public good nature of the elephant, *ceteris paribus*. Households whose major source of livelihood is agriculture would be expected to be more concerned about probable elephant intrusion than those whose livelihood is financed elsewhere, *ceteris paribus*. Thus farmer-households, who have a potential of suffering more at the feet of the elephant, would eagerly want to see the elephant being driven away fast.

5. Estimation Results

The survey questions used a project $z^1 \rightarrow z^0$ that entails the avoidance of the translocation of the current elephant population and a project $z^0 \rightarrow z^1$ that entails the translocation of the current elephant population to elicit WTP values. The WTP for avoiding the translocation of the current elephant population by those who consider it a public good constitutes the positive WTP for the preservation of the current elephant population. Those who find the benefits of the elephant to be surpassed by costs (B<C) potentially have negative WTP for the preservation of the current elephant population since they would rather receive compensation for bearing with the preservation of the current elephant population. This compensation is approximated by their WTP for the translocation of the current elephant population[32]. Thus, the WTP for the translocation

[32] The assumption of the linear-in-income utility function allows us to make this statement since income effects are zero. If it happened that the utility function was of a flexible form, which does not restrict the income effects to zero, then WTP for translocation would constitute the minimum compensation required for the preservation of the current elephant population by those for whom cost exceed benefits.

of the current elephant population by those who consider it a nuisance constitutes the negative WTP for the preservation of the current elephant population. We will report the absolute value of this potential negative WTP for the preservation of the current elephant population, which we label NWTP, so that it could be interpreted as the value of nuisance that negatively affected respondents bear from the preservation of the current elephant population. Consequently, our interpretations are going to be in terms of WTP/NWTP for the preservation of the current elephant population i.e. project $z^1 \rightarrow z^0$.

5.1 Determinants of the Characterisation of the Elephant as a Public Bad and Estimates from Spike Analysis of Answer to Single Bounded Bid

Some people consider the elephant a public good while others consider it a public bad. We make use of data for those households that indicate that the benefits of the elephant are surpassed by costs (N_2=352, dummy=1) and those that indicate that the benefits of the elephant exceed costs (N_1=197, dummy=0) to determine how people characterise the elephant with respect to the sign of its externality. Knowledge of the social characteristics of those households that are likely to view the elephant as a negative externality could help in (i) designing appropriate compensation schemes or (i) targeting devolution of elephant user rights to specific groups of people. Alongside household characteristics and other factors that are significant determinants of the characterisation of the elephant as a public bad, Table 2 below reports the spike model analysis of the answers to the single bounded bid. The presence of significant coefficients with expected signs means that the responses to the proposed bids were sensitive to the respondents' needs and ability to pay. Results are reported separately for the two sub-samples where the elephant is considered either a public good or a public bad.

Table 2: Determinants of the characterisation of the elephant as a public bad and estimates from spike analysis of answer to single bounded bid

	Probit Estimates of Prob(B < C)		Spike Estimates of Prob(YES) to the bid	
			B < C	B > C
Variable	Marginal Effect	Coefficient	Coefficient	Coefficient
Intercept	-0.1214 (-0.917)	-0.3467 (-0.914)	0.1751 (0.229)	3.7553*** (4.119)
Bid			-0.0112*** (-13.737)	-0.0095*** (-8.831)
Household Size	0.0125 (1.381)	0.0357 (1.376)	-0.0084 (-0.164)	0.0198 (0.287)
Sex of Household Head	-0.0354 (-0.693)	-0.1004 (-0.698)	-0.1111 (-0.435)	-0.4109 (-1.055)
Age of Household Head	-0.0062*** (-2.932)	-0.0176*** (-2.908)	-0.0158 (-1.639)	-0.0112 (-0.681)
Education of Household Head	0.0117* (1.691)	0.0334* (1.689)	0.0966*** (2.588)	0.0512 (0.973)
Awareness of CAMPFIRE	0.2040*** (4.547)	0.5971*** (4.375)	0.0187 (0.079)	-0.8493** (-2.088)
Distance to the Elephant Reserve	-0.0102*** (-4.353)	-0.0291*** (-4.396)	-0.0558*** (-3.180)	0.0105 (0.687)
Size of Intruding Elephant Herd	-0.0091** (-2.293)	-0.0259** (-2.297)	0.0108 (0.414)	-0.0345 (-1.155)
Existence of Mitigation Measures	0.0437 (0.720)	0.1237 (0.725)	0.3665 (1.343)	0.4777 (0.911)
Support Parks Driven Elephant Conservation	-0.0481 (-0.964)	-0.1354 (-0.977)	-1.3156*** (-3.918)	0.1410 (0.403)
Agriculture as Main Activity	0.0301 (0.452)	0.0874 (0.444)	0.7418* (1.879)	0.2691 (0.662)
Labour-days Against Intrusion	-0.0038*** (-5.576)	-0.0109*** (-5.449)	-0.0054* (-1.654)	-0.0091 (-1.551)
Annual Household Income	0.1E-05** (2.250)	0.4E-05** (2.249)		
N	549	549	352	197
Log Likelihood	-262.561	-262.561	311.237	139.387

The figures in parentheses are t-values. ***, ** and * indicate significance at the 1%, 5% and 10% levels, respectively.

Age of the household head influences how households characterize the elephant with respect to its sign of externality. Relatively young household heads tend to be the ones who are still actively engaged in larger scale agricultural activities hence potentially suffer a higher value of elephant intrusion threat. Also, younger household heads tend to undervalue the religious value of the elephant owing to the upheaval of the African culture and traditions following modernisation. Thus, households headed by younger heads have a higher probability of viewing the elephant as a nuisance.

Highly educated household heads seem to comprehend more the negative externality they suffer from the elephant and be more knowledgeable about elephant populations elsewhere hence would harshly penalise any elephant intrusion threat knowing that their decisions do not deprive them of the elephant resource since it is not endangered nationally. As such, a higher level of education increases the probability of viewing the elephant as a public bad.

The positive coefficient of the education of the household head, which is significant at a 1% level, implies that a higher level of education enhances the comprehension of the negative externality from the elephant and consequently increases the probability of a household accepting the proposed bid in the elicitation of NWTP. Perhaps the household heads with a low education have difficulties in quantifying damage inflicted during elephant intrusion since the valuation of some of the damage is not obvious. In some instances one has to extrapolate the trend of growth of a destroyed crop to assess the loss incurred from elephant intrusion. Such assessments even burden specialist agricultural officers and that is one of the reasons why schemes to compensate elephant intrusion victims have not taken off in many places. In short, we observe the expected result that education of the household head is positively related to the probability of acceptance of the proposed bid in the proposed translocation case.

CAMPFIRE was designed to compensate local communities who are living adjacent to wildlife reserves for bearing the costs of living with wildlife. In Mudzi, wildlife benefits are shared in the form of the provision of social infrastructure and thus rarely provide subsistence, income or secure livelihoods to the majority of community members. The presence of wildlife gives rise to costs by interfering with other components of community livelihood systems while the meagre wildlife revenue allocated to the communities is frequently not sufficient to put people in an economic position to forgo wildlife damage. In most areas, including Mudzi, the elephant is responsible for the major fraction of the value of crop damages. Those households who have actively participated in CAMPFIRE have known the limitations of the programme with respect to provision of adequate economic incentives and continued access to wildlife and its habitats. It is therefore to be expected that awareness of CAMPFIRE would increase the probability of households expressing negative sentiments towards the elephant.

CAMPFIRE has operated in Mudzi since 1992. The observed negative relationship between awareness of CAMPFIRE and acceptance of the proposed bid suggests that, for those households who view the elephant as a public good, participation in CAMPFIRE has decreased welfare. The result implies that the elephant yields less utility under the CAMPFIRE regime than otherwise. The policy implication of the same result is that CAMPFIRE as a resource use regime may have been successful in targeting the right population, the most vulnerable, but it has not been successful in providing adequate compensation. One of the reasons for the decrease in welfare emanating from the change in the resource use regime could be that the nature of use of the resource may have been altered. For example, prior to CAMPFIRE households could view and occasionally hunt the elephant, and utilise other resources in its habitat whereas with CAMPFIRE the resource and its habitat are reserved for the exclusive use of the rich foreign hunters who engage in trophy hunting. Reservation of the resource for trophy hunting purposes may entail alienating the local people from the resource, due to the fear that they may disturb the trophy hunting operations. A case in point is the establishment of buffer zones in Mudzi and the consequent denial of access to those areas for the local people. While the new use of the elephant resource yields benefit (revenue) for the local people, the benefit may not be equally large for individual households as the one it displaced. The result with respect to awareness of CAMPFIRE can therefore be rationalized by arguing that participants in CAMPFIRE activities are showing their disgruntlement with lack of adequate incentives from the programme by a higher probability of turning down proposed bids.

The farther away one is from the elephant the more the ‘bad’ attribute of the elephant is eliminated because the risk of elephant intrusion decreases with distance from the elephant reserve, *ceteris paribus*. The observed negative effect of distance to the elephant reserve on the probability that a household views the current elephant population as a nuisance is therefore reasonable. In the same manner, the negative sign of the coefficient of the distance to the elephant reserve in the spike model is consistent with expectations since it suggests that households living farther away suffer lesser damages than those living closer, all other things remaining equal. In this category of people who view the elephant as a public bad, households who are living farther away from the elephant have a lower probability of accepting proposed bids for the translocation of the elephant. Since they live farther they are less likely to suffer elephant intrusion hence their probability of acceptance of proposed bids would be depressed by their relatively lesser risk of facing damages. However, the result that those facing intrusion threat from a smaller size of elephant herd resent the elephant more than those facing intrusion threat from a larger size of elephant herd is unexpected.

The negative sign of the coefficient of supporting parks agency driven elephant conservation rather than community driven elephant conservation in the spike model is consistent with expectations since it suggests that parks agency driven elephant conservation is expected to reduce damage from the elephant. Thus, support for the parks agency reduces the probability of a household accepting the proposed bid of translocating the elephant. Parks agency supporters are more hesitant to have the elephant removed from their area even if they suffer some damages from it. In other words, they are willing to pay a premium towards the existence of the elephant through accepting some damage that it inflicts on them. This is a signal of the faith that such households have on the parks agency's capability to reduce human-elephant conflict and consequently a vote of no confidence in the community driven elephant conservation's role in eliminating human-elephant conflict.

Households whose major source of livelihood is agriculture would be expected to be more concerned about probable elephant damage than those whose livelihood is financed elsewhere. For those households for whom benefits from the elephant are surpassed by costs, if agricultural production is the main income generating activity of the household then the NWTP for the preservation of the current elephant population would be relatively higher compared to those whose livelihood is financed elsewhere, *ceteris paribus*. Farmer-households who have a potential of suffering more at the hands of the elephant would eagerly want to see the elephant being driven away fast hence a higher NWTP. This scenario gives room for the observed result that having agriculture as the main livelihood activity is positively related to acceptance of proposed bids.

Those households who spend relatively lesser labour-days guarding against elephant intrusion are more likely to view the current elephant population as a public bad because they face a higher level of intrusion risk. Use of relatively lesser labour-days could emanate from shortages of appropriate labour for guarding against elephant intrusion. As such these households are expected to regard the elephant as a nuisance.

In the spike model, the coefficient of labour-days spent guarding against elephant intrusion is significant at the 10% level. This entails that those households who have applied more effort in cushioning themselves from elephant intrusion have a lower probability of accepting proposed bids. This result could be explained by the fact that labour expended in guarding against elephant intrusion acts as some kind of insurance against elephant intrusion. Thus, those households who spend relatively more labour-days guarding against elephant intrusion face a lower level of intrusion risk and are less likely to accept proposed bids for the translocation of the current elephant population. As we argued earlier, labour spent guarding against elephant intrusion has negligible, if not zero, opportunity cost in Mudzi.

The result that households with high annual incomes are likely to view the current elephant population as a public bad could be due to the fact that such households are more likely to have more valuable assets or bigger fields that result in huge losses once they are targeted by the elephant. Having a negative WTP for the preservation of the current elephant population is seeking some kind of insurance against potential huge losses from elephant intrusion.

This sub-section has presented results that show that households who view the elephant as a public bad have the following characteristics: they typically have younger household heads, with high levels of education and awareness of CAMPFIRE. Furthermore, proximity to the elephant reserve, less labour-days spent guarding against elephant intrusion and high annual household income increase the probability for WTP for the translocation of the elephant. The results from the spike model analysis suggested that the responses to the proposed single bounded bids were sensitive to the respondents' needs and ability to pay.

The message from the results presented in this section is three-pronged: Firstly, appropriate compensation schemes should be designed so that they can adequately benefit households who view the elephant as a public bad. These households are not expected to support conservation measures until their view of the elephant as a negative externality has been eliminated. The appropriate compensation schemes should improve the incentive system at the local level rather than at the district level.

Secondly, where devolution of elephant user rights is targeted to specific groups of people, it is advisable not to target it to groups that are composed of a majority of households with the stated social characteristics. In areas where CAMPFIRE operates, decisions are taken on the basis of the majority-voting criterion, where the voting unit is the household. In the absence of sufficient incentives, dominance of households with the stated social characteristics in the group to which devolution is targeted is likely to lead to the failure of collective action in community-based elephant conservation.

Thirdly, the characteristics of the households with a negative attitude towards the elephant such as young and highly educated casts a bleak future for community-based elephant conservation in Mudzi. The group of households under discussion is the group that is expected to take over the current community-based elephant conservation initiatives in the future. If this group's perspective of the elephant as a public bad is not eliminated then they cannot be trusted to be good stewards of the elephant in the future. The good news is that their possession of a high level of education makes them appropriate targets of informational campaigns. It is much easier to convince literate people of the potential benefits that could come alongside improved incentive systems from elephant conservation.

5.2 Reliability of the CVM study

Particular CVM questionnaires ought to be unbiased and transparent instruments, which give respondents the best possible chance to deliberate about their preferences. One criterion upon which success is judged is through conducting reliability tests i.e. investigating whether the survey instrument can be relied upon to give the same values if repeatedly implemented under controlled conditions. The most basic test is that bids can be explained by variables proposed by economic reasoning (Köhlin 2001). We have seen many examples of this in the analysis of the bid function.

5.2.1 The method of splitting the respondents

The method of splitting the respondents into (i) winners from the project, (ii) indifference about the project, and (iii) losers from the project of preserving the current elephant population proved to be appropriate. Each of these categories has a positive membership. The splitting method ensured that respondents were confronted with the appropriate project matching their class of preferences. Blindly confronting all respondents in the main sample with the same project would have resulted in a lot of zero responses to the elicitation of WTP[33]. Instead the splitting method allowed us to obtain valuable information about the spectrum of preferences and the magnitude of the negative WTP for the preservation of the current elephant population in a designated area. Furthermore, the proportions of households in each category of preferences has helped us realise that the project of the preservation of the current elephant population is likely to be blocked if it is voted on by the local communities.

5.2.2 Strategic behaviour

The previous section presented results that show that households who view the elephant as a public bad typically have younger household heads, with high levels of education and awareness of CAMPFIRE. These characteristics fit well with the profile of households that would be more likely to act strategically, if there is an opportunity to do so. The introduction to the survey indicated to the respondents that our interest was in

[33] We find spikes of 0.05 and 0.28 for the two sub-samples. These are quite close to the observed fractions of respondents reporting zero WTP and NWTP.

investigating the economic value that the local communities living adjacent to wildlife placed on the elephant. It may have occurred that the households with the profile mentioned above intentionally sought to understate their valuation of the elephant with the hope that they would achieve two goals: (i) bait a larger proportion of CAMPFIRE income and external financial inflows in the future, and (ii) block the increment of the elephant population which has been proposed by Mudzi RDC. It is a well-known fact that CAMPFIRE, at least at the national level, has generated a significant amount of income and attracted huge sums of foreign financial aid. There is a potential that the households in question would want to portray a misleading picture that they have become destitute as a result of the negative externality from the elephant. This could be done to bait a larger proportion of CAMPFIRE income and external financial inflows, which could be mobilized as compensation to the vulnerable communities. Currently there are plans to re-stock Nyatana game reserve in Mudzi with an elephant population of 100, among other species, with grants from Elephant for Africa and USAID. The younger household heads with high levels of education and awareness of CAMPFIRE could have used the survey as a means to block the increment of the elephant population that has been proposed by Mudzi RDC. They could have interpreted the survey as an indirect elicitation of their views with regards to the desired course of action. The likelihood that the survey might have presented the households with the mentioned profile with the opportunity of realizing two plausible goals calls for caution on the interpretation of the results and design of incentive systems.

5.2.3 Yea-saying and nay-saying

The contingent valuation method could potentially suffer from either yea-saying or nay-saying i.e. acceptance or rejection of a bid that does not reflect true preferences. This would have been the case if either many respondents accepted the bids irrespective of their sizes or many respondents rejected the bids irrespective of their sizes. Out of 5 groups of respondents who consider the elephant a public good and confronting various bids, there is one group that had 100% acceptance of the starting bid (see Figure A1 in the appendix). The groups that confronted higher bids had their acceptance rates decreasing gradually. However, the group that confronted the lowest bid had a 60% acceptance rate. There is a possibility that this group could have been a victim of nay-saying. We believe that the lowest bid was far below the true average WTP. Out of 5 groups of respondents who consider the elephant a nuisance and confronting various bids, there is one group that seemed to have signs of yea-saying (see Figure A2 in the appendix). Despite the above evidence, the coefficients of the bid for the spike model for both sub-samples are significantly different from zero and have the expected negative sign. This result suggests that the bid acceptance rate generally decreases with the size of the bid. Nevertheless, the smallness of the coefficients leads us to continue holding suspicion for the presence of yes-saying and nay-saying.

5.2.4 The internal scope test

One issue in the CVM debate is whether WTP is sensitive to the scope of the good being valued i.e. whether the value of the good is significantly different from the value of a more inclusive good. For the CVM to yield theoretically consistent results it is crucial that WTP is sensitive to scope variations. In this study, respondents were also asked to evaluate the preservation of the top 50% of the current elephant population in a situation in which they already had the lower 50% of the current elephant population. This allowed us to conduct an internal scope test because the respondents indicated the values they assign to different levels of elephant populations i.e. ***y*** and ***0.5y***. We got the result that the welfare measures for the preservation of a higher elephant population are higher than for a lower population[34]. The *z*-test confirmed the difference between the two sets of welfare measures. We therefore concluded that respondents were sensitive to the scope of the environmental amenity they were confronted with.

5.2.5 Cash constraint[35]

In Zimbabwe, inhabitants in rural areas are largely subsistence farmers who occasionally sell surplus production to the modern sector. Consequently, the rural areas of Zimbabwe are less monetized than the urban areas. In such a scenario one may contemplate that monetary bids would be biased downwards. It is not obvious that in

[34] The mean and median WTP for the preservation of an elephant population of 100 for 198 respondents was ZW$199.28 (11.072) and ZW$195.42 (11.110) respectively while the mean and median NWTP for 345 respondents was ZW$62.35 (4.972) and ZW$34.46 (7.462) respectively. These measures should be compared with those given in Table 3 below.

[35] If we think of education as human capital then these people might also potentially face a kind of "capital constraint". We will not pursue this argument any further.

this study there may have been cash constraints due to the semi-subsistence economy in rural areas. The fact that Mudzi lies in the vicinity of the border point, which is quite monetized, absolves this study from the speculation of the presence of cash constraints. Other studies have provided for the presence of cash constraints by stating the bids in barter commodities such as rice (see for example, Shyamsundar and Kramer (1996)). If our questionnaire design had included opportunities of non-monetary bids we would have wanted to test the presence of cash constraints by giving those respondents who stated zero WTP (7.6% in the sub-sample where B > C and 28.1% in the sub-sample where B < C) an opportunity to restate their bids in terms of maize or labour-days.

5.3 Welfare Measures for the Preservation of an Elephant Population of 200

This sub-section reports the mean and median WTP/NWTP for the preservation of an elephant population of 200 for the two[36] sub-samples where the elephant is considered as either a public good (N_1=197) or a public bad (N_2=352). The welfare measures are calculated from the single bounded dichotomous choice responses. The mean and median WTP/NWTP are calculated for the simple spike model and linear-in-income utility function with covariates.

Table 3: Mean and Median WTP for the preservation of an elephant population of 200

'Public Good' Sub-Sample (B > C)		'Public Bad' Sub-Sample (B < C)	
Mean WTP	Median WTP	Mean NWTP	Median NWTP
ZW$305.95 (17.426) (N=197)	ZW$300.00 (17.729) (N=197)	ZW$123.90 (8.543) (N=352)	ZW$98.14 (10.186) (N=352)

The figures in parentheses are standard errors. The exchange rate is 1US$=ZW$55.

While the mean WTP for the preservation of the current elephant population for the sub-sample where B > C is ZW$306 (*1.57% of mean annual income or US$5.56*) the mean

[36] We do not report the welfare measures for the third sub-sample (N_3=21) of households that are indifferent to the elephant preservation project. In fact, the mean and median WTP for this sub-sample is zero.

NWTP for the preservation of the current elephant population for the sub-sample where B < C is ZW$124 (*0.64% of mean annual income or US$2.25*). The median WTP for the preservation of the current elephant population for the sub-sample where B > C is ZW$300 (*3.87% of median annual income or US$5.45*) while the median NWTP for the preservation of the current elephant population for the sub-sample where B < C is ZW$98 (*1.27% of median annual income or US$1.78*). The choice of any one of the two welfare measures implies a particular approach to the aggregation of welfare across the population (Hanemann and Kanninen 1999). The mean is equivalent to adopting the Kaldor-Hicks potential compensation principle while the median is equivalent to adopting the majority-voting principle. The Kaldor-Hicks criterion is commonly used but it can lead to logical inconsistencies and it has been severely criticised on ethical grounds (Little 1957, quoted in Hanemann and Kanninen 1999, p325). While the majority-voting criterion could be considered as ethically superior, it has been criticised for not satisfying even potential Pareto efficiency. Thus, the choice of welfare measure is subjective and should ideally conform to the decision rule dominant in the sampled population.

In areas where CAMPFIRE operates, democratic principles have been instilled to replace the paternalistic tendencies of traditional chiefs, headmen, and village heads. Every household is given an equal opportunity to determine the outcome of issues under consideration. Thus, decisions are taken on the basis of the majority-voting criterion, where the voting unit is the household. In appraising the valuation of the preservation of the current elephant population in Mudzi, the median WTP/NWTP should therefore be utilised, given that the project has already been sanctioned for adoption as attested to by the granting of AA status to Mudzi RDC. Considering the proportions of households who are (i) winners from the project (34%), (ii) indifferent about the project (4%), and (iii) losers from the project (62%) of preserving the current elephant population in Mudzi, the median WTP and NWTP show that the gainers from preservation of the current elephant population, in aggregate, benefit more than the losers. Table 4 depicts the benefit-cost analysis (BCA) of the preservation of the current elephant population.

Table 4: Benefit-cost analysis of the preservation of an elephant population of 200

Preference	N	Sub-Population	**Spike Median**	BCA
B > C	197	0.34 x 3009 = 1023	300.00	ZW$306,900
B < C	352	0.62 x 3009 = 1866	-98.14	- ZW$183,129
B = C	21	0.04 x 3009 = 120	0.00	ZW$ 0
	570	3009		ZW$123,771

The table shows that those households who view the current elephant population as a public good derive an annual value of ZW$306,900 (*US$5,580*) from its preservation while those who consider it a public bad suffer an annual cost of ZW$183,129 (*US$3,330*) from its preservation. An examination of the actual annual incomes from CAMPFIRE activities indicate that Mudzi has been generating an annual average of ZW$159,526 (*US$2,900*), which is lower than the costs suffered by losers from the preservation of the elephant (see Table A3 in the appendix). Those households who are indifferent to the preservation of the current elephant population put a zero valuation on it.

In principle, if a decision were adopted by the government for the local communities to continue preserving the current elephant population, as has been done by issuing AA status to Mudzi RDC, then that would benefit part of the population while harming others but, in aggregate, the beneficiaries would benefit more than the losers. The preservation of the current elephant population in Mudzi yields an annual net worth of ZW$123,771 (*US$2,250*) for the households in CAMPFIRE wards[37]. However, the majority of the households in the local communities would not support the decision since 62% of them would rather not have the elephant because they view it as a nuisance. Thus if it were left to the local communities to decide whether or not the project of the preservation of the current elephant population in Mudzi should be carried out then the project would be blocked. There is a fear that imposing the project on the basis that, in aggregate, the winners benefit more than the losers, would result in lack of

[37] While this figure assumes the feasibility of actual compensation of losers from the preservation of the current elephant population in Mudzi, in practice there has not been any compensation of victims of elephant damage, in particular, and wildlife, in general.

proactive cooperation from the majority and that could lead to the failure of collective action in community-based elephant conservation.

The realisation that a majority of households consider the elephant a public bad is one argument against devolution of elephant conservation to the local communities. Devolution entails the complete surrender of elephant conservation power. Full ownership of the elephant by the local communities would therefore imply the complete power to control the access and use of the elephant, and the capacity to hold the elephant for own use or to alienate or destroy it (Schlager and Ostrom 1993). The spectrum of preferences for the project of the preservation of the current elephant population shows that there have not been adequate incentives trickling down to the local communities to encourage a majority of them to change their perspective of the elephant as an agricultural liability. Devolution of elephant conservation to the local communities in Mudzi could be detrimental to its survival.

Adequate economic incentives must be extended to the local communities if a majority of them is to be persuaded to partake in sound elephant conservation. Given that studies generally show that a majority of people in countries that are not endowed with the African elephant have a positive WTP for its preservation (see for example Vredin, 1999), external transfers constitute one way of providing additional economic incentives to encourage elephant conservation by local communities such as Mudzi. However, external aid should be channelled directly to the producer communities if they are to respond to it as increased demand for conservation. Given that wildlife conservation potentially increases the aggregate welfare of the local communities and that decentralisation of elephant user rights has already been adopted with the inception of CAMPFIRE, co-management should be the preferred mode of communities' involvement in wildlife conservation. This gives room for checks and balances so that mistakes overlooked at the local communities' level can be rectified by other organisations. Co-management acknowledges the multiple jurisdictions that exist in the conservation of the elephant (Hasler 1999) and also takes advantage of the lower costs of provision, monitoring, enforcement, conflict resolution, etc that occurs at the local level. As will be pointed out in Chapter 4, what remains to be done to set co-management in motion is to increase the contestations of the sub-district local communities and establish adequate incentive schemes.

6. Conclusion

Wildlife conservation must compete economically with agriculture, which is the prime source of rural communities' livelihood, if it is to be accepted as an alternative land-use. If the economic value of wildlife relative to the communities' other economic activities were larger, then it would imply that wildlife conservation might be enhanced through devolution of wildlife conservation to the local communities. This paper, focused on the elephant as the representative wildlife since it is the most important species to the local communities both in terms of the damage it causes to crops and the value it contributes to CAMPFIRE revenues. The paper estimated the willingness to pay (WTP) for the preservation of a designated sub-population of the African elephant (*Loxodonta africana*) in Zimbabwe by rural communities who live adjacent to a designated game reserve taking into account the reality that some people consider it a public good while others consider it a public bad. The paper has used an approach that can evaluate a project that generates both winners and losers.

The paper showed that households who view the elephant as a public bad have the following characteristics: they typically have younger household heads, with high levels of education and awareness of CAMPFIRE. Furthermore, proximity to the elephant reserve, less labour-days spent guarding against elephant intrusion and high annual household income increase the probability for WTP for the translocation of the elephant. This knowledge could help in (i) designing appropriate compensation schemes or (ii) targeting devolution of user rights to specific groups of people or (iii) designing informational campaigns.

The median WTP and NWTP for the project of the preservation of the current elephant population is ZW$300 and ZW$98 respectively. Considering the proportions of households who are (i) winners from the project (34%), (ii) indifferent about the project (4%), and (iii) losers from the project (62%), in aggregate, the winners benefit more than the losers. However, the majority of households do not support the project hence imposing it could lead to the failure of collective action or if the local communities could decide whether or not to carry out the project then it would be blocked. Of importance to note also is the result that while it is generally believed that poor people are not willing and able to pay for sound environmental management, as many as 34% of the respondents are willing to pay for the preservation of the elephant in Mudzi.

The realisation that a majority of households consider the elephant a public bad is one argument against devolution of elephant conservation to the local communities in Mudzi. The rural communities' perceptions of the elephant are generally useful for other species of wildlife since the elephant is considered a keystone species and, most importantly, an umbrella species in the African Savannas. Adequate economic incentives must be extended to the local communities if a majority of them is to be persuaded to partake in sound elephant conservation. External transfers constitute one way of providing additional economic incentives to encourage elephant conservation by local communities. It is obvious from this study that CAMPFIRE has not been successful enough in doing this in Mudzi. Co-management should be the preferred mode of communities' involvement in wildlife conservation.

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APPENDIX A

Table A1: Summary statistics for the open-ended WTP question with 200 elephants

	In the Market		Out of the Market
	Positive WTP *(Costs>Benefits)*	Negative WTP *(Costs<Benefits)*	Zero WTP *(Costs=Benefits)*
Number of households (including those with zero WTP in the market)	352	197	21
H/holds with zero WTP in the market	99	15	Not applicable
Mean WTP including zero WTP in the market	110.43	389.54	Not applicable
Median WTP including zero WTP in the market	50.00	250.00	Not applicable
Mean WTP excluding zero WTP in the market	153.64	421.65	Not applicable
Median WTP excluding zero WTP in the market	100.00	250.00	Not applicable

Table A2: Summary statistics for the open-ended WTP question with 100 elephants

	In the Market		Out of the Market
	Positive WTP *(Costs>Benefits)*	Negative WTP *(Costs<Benefits)*	Zero WTP *(Costs=Benefits)*
Number of households (including those with zero WTP in the market)	345	198	27
H/holds with zero WTP in the market	133	14	Not applicable
Mean WTP including zero WTP in the market	74.29	258.36	Not applicable
Median WTP including zero WTP in the market	20.00	175.00	Not applicable
Mean WTP excluding zero WTP in the market	120.90	278.02	Not applicable
Median WTP excluding zero WTP in the market	100.00	175.00	Not applicable

Table A3: Mudzi Rural District's Annual Income from CAMPFIRE Activities (ZW$)

Year	GDP Deflator	Current (ZW$)*	Current (US$)*	Deflated (ZW$)
1994	19.356	28,000	3,410	144,656
1995	21.205	-	-	-
1996	26.676	44,488	5,958	166,767
1997	30.997	59,488	4,780	191,912
1998	40.059	50,000	2,051	124,813
1999	62.541	226,926	5,919	362,838
2000	100.000	125,695	2,817	125,695

Source: *WWF SARPO, HARARE & WORLD BANK

Figure A1: Proportions of ‘YES’ responses from the starting bid: B > C

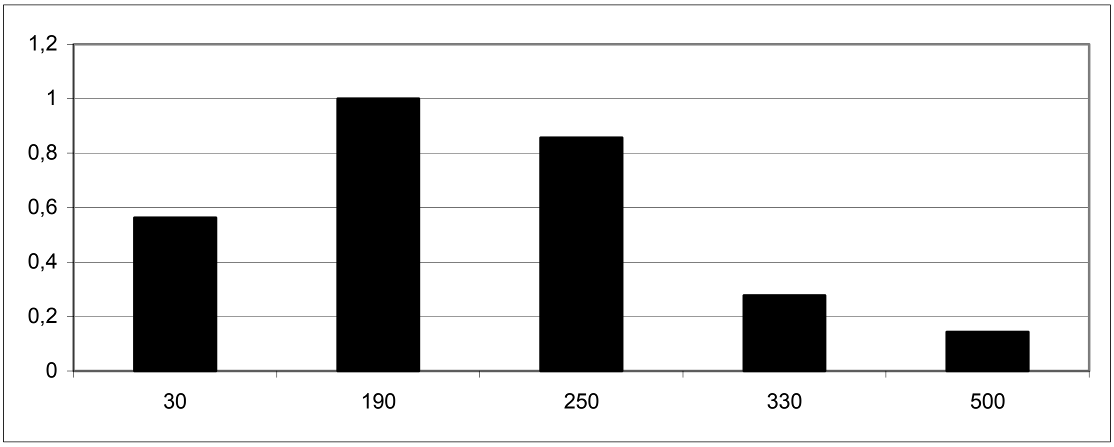

Figure A2: Proportions of ‘YES’ responses from the starting bid: B < C

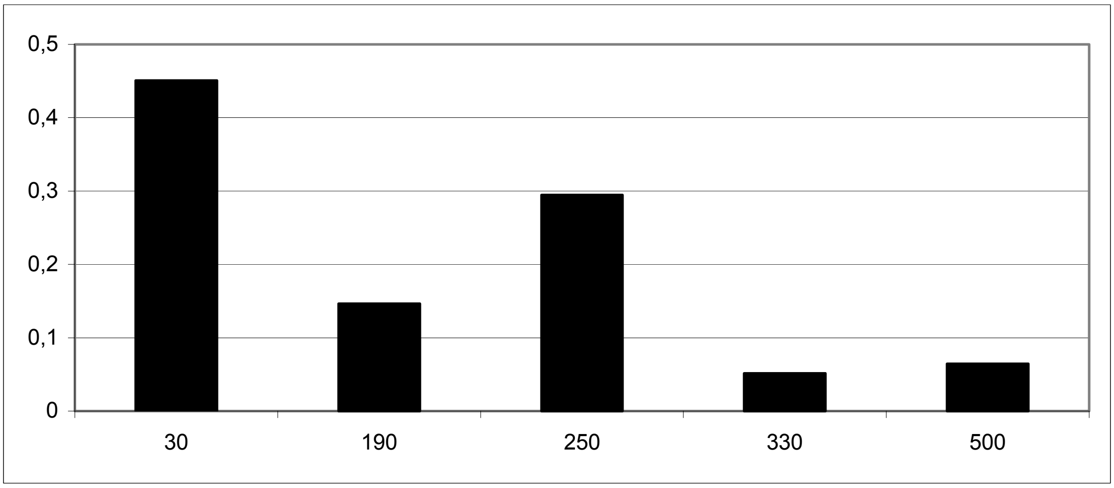

Table A4: Mudzi, Rushinga & UMP Zvataida Districts Combined Quotas

Species	1993	1994	1995	1996	1997	1998	1999	2000	2001
Baboon	10	2	10	10	10	10	10	10	
Buffalo bull							1	2	
Buffalo cow									
Bushbuck		1	1	1	1	2	2	3	
Bushpig	4	2	1	1	1	3	3	3	
Civet	1								
Crocodile	2	2	1	1	1	2	2	2	
Doves	10	100	50	50	50	100	100	100	
Ducks/Geese	10	5	10	10	10	10	10	10	
Duiker	2	2	2	2	2	3	3	3	
Eland	1								
Elephant bull	2	2	2	2	2	2	2	2	
Elephant cow									
Francolin	10	10	10	10	10	10	10	10	
Genet	1					2	2	2	
Grysbok	1	1	1	1	1	0	1	1	
Guinea fowl	10	10	10	10	10	10	10	10	
Hippopotamus	1	1	1	1	1	1	1	1	
Honey badger	1					1	1	1	
Hyena	1	1	1	1	1	2	2	2	
Impala female									
Impala male									
Jackal	2								
Klipspringer	2	1	1	1	1	2	2	2	
Kudu bull	1		1	1	1	1	2	2	
Kudu cow									
Leopard	2	2	2	2	2	3	3	3	
Lion				1	1				
Lioness									
Porcupine	2					2	3	3	
Reedbuck	1								
Sable	1								
Sandgroose	10	10	10	10	10	10	10	10	
Serval	1								
Spring hare						3	2	2	
Vervet monkey	10	2				2	2	2	
Warthog	5								
Waterbuck							1	1	
Wild cat	1					1	1	1	

Source: Mudzi Rural District Council

APPENDIX B

QUESTIONNAIRE FOR THE STUDY ON

THE RURAL COMMUNITIES' PERSPECTIVE OF THE ECONOMIC VALUE OF THE ELEPHANT:
MUDZI RURAL DISTRICT, ZIMBABWE

(translated from the Shona questionnaire)

DECEMBER 2000

Name of Ward :____
Name of Village :____
Name of Household Head :____
Household ID for this Survey :____
Name of Interviewer :____
Date of Interview :____

Introduction

I am a researcher attached to the University of Zimbabwe and the University of Gothenburg, Sweden. Your household has been chosen, through random sampling, to participate in a survey regarding the value that local communities living side by side with wildlife in Mudzi place on the elephant. Your answers will be voluntary and will be kept strictly confidential. Your answers will be put together with many other answers and will be used at a highly aggregated level so that no-one will be able to single out your responses. One can have different reasons for putting the value that they put on the elephant. Everybody has good reasons for thinking the way they do. When you answer these questions you should note that there may be other things in nature to which you would like to pay regard. Moreover, perhaps there are political, social and economic issues that you believe are more urgent. In some cases you might think that the questions are out of touch with reality. However, the questions are still important since it is only by posing such questions that I can gain an insight into the value of elephants from your community's perspective. Bearing this in mind I ask you to do your best to answer the questions.

Date Checked	Checker's Name	Status		Comments
		Ok	Return	

Part 1: Basic Household Information

1. How many members does the household have? :____

2. List the following details for the head of the household

ID Code	Name	Relation to head	Resident or not?* Y N	Sex* M F	Age (years)	Marital Status *(a)*	**Main work activity** *(b)*	Type of Education* U P S T	Grade completed *(c)*	Years of schooling **(compute)**	Household annual income
01	Head	Self	1 2	1 2				0 1 2 3			

***Y**=yes **N**=no **M**=male **F**=female **U**=uneducated **P**=primary **S**=secondary **T**=tertiary

Part 2: Household Participation in CAMPFIRE activities

3. Are you aware of the CAMPFIRE programme in your area (through having actively participated in it)? Y N (***if no go to question 8***)
4. What positions do/did members of the household hold in CAMPFIRE structures?

ID Code	Current Position	Current Committee	Period	Previous Position	Which Committee	Period	Is head aware of wildlife quota? Y N	Ever participated in quota determination? Y N	Ever received CAMPFIRE dividend? Y N	Ever been CAMPFIRE employee? Y N
01							1 2	1 2	1 2	1 2
02										

5. What are the most important benefits that one could possibly get from CAMPFIRE? (*put responses in the table below*)
6. What are the most severe problems facing CAMPFIRE? (*put responses in the table below*)

Benefit 1		Problem 1	
Benefit 2		Problem 2	
Benefit 3		Problem 3	

7. If at all it did, how much and when did the household ever benefit financially or otherwise from CAMPFIRE activities? :______________________

Part 3: Wildlife-human conflict and compensation

8. How far are you from the closest elephant reserve? :____
9. What is the average size of the local elephant herd that has access to your agricultural activities? :________________________
10. On average how many working days per year do you spend guarding fields from elephants? :________________________
11. Do you use any mitigation measures to deter elephant intrusion? Y N (***if no go to question 14***)
12. Specify what you have. :____
13. What was your household's attitude towards elephants before the establishment of CAMPFIRE? :________________________
14. What has been your household's attitude towards elephants since the establishment of CAMPFIRE? :________________________
15. Would you support elephant conservation even if all revenue generated by them would go to the DNPWLM? Y N
16. If at all you do, what do you do to help protect elephants?

Activity 1	Activity 2	Activity 3	Activity 4	Activity 5

Part 4: Agricultural Production and Income

17. Is farming the main household income generating activity? Y N
18. What other income generating activities besides farming does the household do? Rank in terms of importance in contribution to household income. Also indicate the extent to which the income generated from the activity is from CAMPFIRE.

Activity 1		Extent ascribed to CAMPFIRE	
Activity 2		Extent ascribed to CAMPFIRE	
Activity 3		Extent ascribed to CAMPFIRE	

In answering the questions in the next section may you note that in this rural district there is an estimated elephant population of *200*. In general, the benefits from elephants are (i) products that can be consumed directly, such as live sales, meat, hides, trophies, (ii) education, (iii) tourism, (iv) research opportunities, (v) ecological and environmental services such as maintenance of the African savannas and biodiversity, (vi) possible future uses such as touristic, pharmaceutical, industrial and agricultural applications, and (vii) intrinsic value such as religious, cultural, aesthetic, existence and bequest significance. The costs that elephants impose include (i) management costs such as costs of equipment, capital, wages, running costs, policing, etc, (ii) costs to other livelihood activities in the form of livestock losses, crop destruction, human injury, damage to structures, etc, and (iii) opportunity costs in the form of alternative land, money, time or resource uses and profits forgone, including unsustainable use. Remember that the elephant accounts for over 80 percent of all the wildlife perpetrated agricultural damage but it also accounts for over 65 percent of all CAMPFIRE revenues.

Part 5: Stated Preference Survey

19. The government is considering putting up an insurance scheme for your area. All other compensation channels that may have existed before will therefore cease to exist henceforth. This scheme will be such that you will be compensated for all crop damages, livestock and human injuries or losses you will suffer from elephants. Only those who are willing to pay a premium will be part of the scheme i.e. anyone who does not pay the premium will not be compensated for any elephant perpetrated damage or loss whatsoever. What maximum amount in yearly premiums would your household be willing to pay for it to participate in such a scheme? :____ [***Probe if zero***]

20. **a**. The district-wide elephant population is estimated to be 200. Considering those benefits and costs of elephants that are applicable to your household, how do you think the benefits of living with elephants compare with the associated costs?
    (i) benefits > costs (***go to question 20b***)
    (ii) benefits < costs (***go to question 20c***)
    (iii) benefits = costs(***go to question 20d***)

b. The government is considering translocating the current elephant population of ***200*** from your district to other districts so that the people there can also benefit from elephants since they are a national heritage. However, preliminary calculations show that it is possible to avoid the elephant translocation if your community can pay annual 'translocation avoidance' taxes to the government for as long as the animals shall be in your area. The revenue from this tax will then be distributed to the communities without elephants so that they can also benefit somehow from these animals. Would your household be willing to pay an annual 'translocation avoidance' tax of ZW$$\boldsymbol{x}$ for as long as the animals shall be in your area, given that all other households who do not find elephants to be a nuisance will also pay the same amount, so that you could be allowed to continue living side by side with the ***200***? Y [***go to (i)***] N [***go to (ii)***]

(i) Suppose it turned out that the true annual 'translocation avoidance' tax is ZW$$\boldsymbol{x}^H$, would your household be willing to pay it? Y [*skip(ii)*] N [*skip(ii)*]
(ii) Suppose it turned out that the true annual 'translocation avoidance' tax is ZW$$\boldsymbol{x}_L$, would your household be willing to pay it? Y N
(iii) What would be the maximum annual 'translocation avoidance' tax that your household would be willing to pay? :_______________[***Probe if zero***]

c. The government is considering translocating the current elephant population of ***200*** from your district to other districts so that your pain of living side by side with the elephants will be eased. It is expected that the people there will experience the nuisance that you have been experiencing from these animals. Nevertheless the government will insist that these people do live with these elephants and receive financial compensation annually. The government does not have the money to fund the translocation and annual compensation of the potential new neighbours of these elephants and preliminary calculations show that it is possible to translocate the elephants if your community can pay an annual 'translocation' tax that could then be used for this translocation and compensation exercise of the ***200***. Your community will be expected to continue paying this annual 'translocation' tax for as long as the animals shall live in the other area. Would your household be willing to pay an annual 'translocation' tax of ZW$$\boldsymbol{x}$ for as long as the animals shall be in the other area, given that all other households who find elephants to be a nuisance will also pay the same amount, so that you could be allowed to continue living free from these elephants? Y [***go to (i)***] N [***go to (ii)***]

(i) Suppose it turned out that the true annual 'translocation' tax is ZW$$\boldsymbol{x}_L$, would your household be willing to pay it? Y [*skip(ii)*] N [*skip(ii)*]
(ii) Suppose it turned out that the true annual 'translocation' tax is ZW$$\boldsymbol{x}^H$, would your household be willing to pay it? Y N
(iii) What would be the maximum annual 'translocation' tax that your household would be willing to pay? :_______________________ [***Probe if zero***]

d. Suppose that the government could reduce the elephant population in this district by ***50%*** so that the remaining elephant population would be ***100*** how do you think the benefits that are provided by the removal targeted ***100*** elephants compare with the costs they impose?

(iv) benefits > costs (***go to question 20e***)
(v) benefits < costs (***go to question 20f***)
(vi) benefits = costs

e. The government is considering translocating ***50%*** of the current elephant population of ***200*** from your district to other districts so that the people there can also benefit from elephants since they are a national heritage. However, preliminary calculations show that it is possible to avoid the elephant translocation if your community can pay annual 'translocation avoidance' taxes to the government for as long as these ***100*** animals shall be in your area. The revenue from this tax will then be distributed to the communities without elephants so that they can also benefit somehow from these animals. Would your household be willing to pay an annual 'translocation avoidance' tax of ZW$$\boldsymbol{v}$ for as long as these ***100*** animals shall be in your area, given that all other households who do not find these ***100*** elephants to be a nuisance will also pay the same amount, so that you could be allowed to continue living side by side with these taxable ***100*** elephants above the non-taxable ones? Y [***go to (i)***] N [***go to (ii)***]

(i) Suppose it turned out that the true annual 'translocation avoidance' tax is ZW$$\boldsymbol{v}^H$, would your household be willing to pay it? Y [*skip(ii)*] N [*skip(ii)*]
(ii) Suppose it turned out that the true annual 'translocation avoidance' tax is ZW$$\boldsymbol{v}_L$, would your household be willing to pay it? Y N
(iii) What would be the maximum annual 'translocation avoidance' tax that your household would be willing to pay? :_______________ [***Probe if zero***]

f. The government is considering translocating ***50%*** of the current elephant population of ***200*** from your district to other districts so that your pain of living side by side with the elephants will be eased. It is expected that the people there will experience the nuisance that you have been experiencing from these animals. Nevertheless the government will insist that these people do live with these elephants and receive financial compensation annually. The government does not have the money to fund the translocation and annual compensation of the potential new neighbours of these elephants and preliminary calculations show that it is possible to translocate the ***100*** elephants if your community can pay an annual 'translocation' tax that could then be used for this translocation and compensation exercise of the ***100*** elephants. Your community will be expected to continue paying this annual 'translocation' tax for as long as the ***100*** animals shall live in the other area. Would your household be willing to pay an annual 'translocation' tax of

ZW$$\boldsymbol{v}$ for as long as the ***100*** animals shall be in the other area, given that all other households who find these ***100*** elephants to be a nuisance will also pay the same amount, so that you could be allowed to continue living free from these ***100*** elephants? Y [***go to (i)***] N [***go to (ii)***]

(i) Suppose it turned out that the true annual 'translocation' tax is ZW$$\boldsymbol{v}_L$, would your household be willing to pay it? Y [*skip(ii)*] N [*skip(ii)*]
(ii) Suppose it turned out that the true annual 'translocation' tax is ZW$$\boldsymbol{v}^H$, would your household be willing to pay it? Y N
(iii) What would be the maximum annual 'translocation' tax that your household would be willing to pay? :________________________[***Probe if zero***]

g. If in (**a**) the respondent finds:
- (i) benefits > costs (***go to question 20h***)
- (ii) benefits < costs (***go to question 20i***)
- (iii) benefits = costs (***go to question 21***)

h. Elephants are more beneficial to some districts than to others. The government is considering a programme to move elephants from districts deriving lowest values to districts deriving highest values from elephants. The government realizes that districts benefiting from living with the elephants will be made worse off by the relocation, if elephants were relocated from them. The government may be willing to give some compensation to the negatively affected districts. In this exercise we are conducting, the government wants to record the values placed on elephants by districts by asking the individual households' valuation and then summing up those values to get the district level valuation. If you are in the district where the lowest aggregate value is recorded compared to other districts, your district will be compensated. The compensation will be in the form of a certain amount of money every year to each household in your district. Remember that the government will only relocate elephants from your district if the total amount claimed as compensation by households of your district is lowest when compared to that claimed by households of each and every other district with elephants. Given that the amount that you claim is what will be paid to every other household in your district, what minimum amount of money per year would your household claim as adequate compensation for the relocation of your district's elephant population of ***200***? :______________________________

i. Elephants are more beneficial to some districts than to others. The government is considering a programme to move elephants from districts deriving highest values to districts deriving lowest values from elephants. The government realizes that districts that are not benefiting from living with the elephants will be made worse off by the relocation, if elephants were relocated to them. The government may be willing to give some compensation to the negatively affected districts. In this exercise we are conducting, the government wants to record the values placed on elephants by districts, by asking the individual households' valuation and then summing up those values to get the district level valuation. If you are in the district where the lowest aggregate value is recorded compared to other districts, your district will be compensated. The compensation will be in the form of a certain amount of money every year to each household in your district. Remember that the government will only relocate elephants to your district if the total amount claimed as compensation by households of your district is lowest when compared to that claimed by households of each and every other district with elephants. Given that the amount that you claim is what will be paid to every other household in your district, what minimum amount of money per year would your household claim as adequate compensation for the relocation of an additional elephant population of ***200*** to your district? :__________

Part 6: Data reliability

[TO BE COMPLETED BY ALL INTERVIEWERS]

21. How well do you think the interviewee understood the questions? Rank in order of comprehension i.e. 1 means that the interviewee clearly understood while 5 means the interviewee did not understand at all.
    1.□ 2.□ 3.□ 4.□ 5.□
22. How do you rate the reliability of the responses given by this interviewee? Rank in order of reliability i.e. 1 means that the responses are quite reliable while 5 means that the responses are not at all reliable.
    1.□ 2.□ 3.□ 4.□ 5.□
23. Give reasons for your responses to question 21 and 22. :____
    ____
    ____
    ____

24. Any other comments about this interview (***or from this interview***)
    ____
    ____

DOES CAMPFIRE SATISFY THE DESIGN PRINCIPLES OF ROBUST INSTITUTIONS?[38]

Edwin Muchapondwa[39,40]

Abstract
Zimbabwe faces an increasing incidence of poverty with the poorest areas being wildlife-abundant rural districts where the sustainable use of wildlife and other natural resources could greatly reduce rural poverty. Despite significant gains that CAMPFIRE has recorded it has not significantly alleviated rural poverty because of the current low levels of monetary benefit and local participation, among other problems. With reforms, CAMPFIRE could enhance sustainable wildlife conservation and consequently reduce rural poverty. Our starting point in search for potentially beneficial reforms in CAMPFIRE is an investigation of the extent to which the design principles that are shared by the institutions of the world's long-enduring common pool resources are satisfied. Our investigation suggests that the large-scale and irreversible nature of wildlife ecologies require co-management for effective long-term sustainable resource management. Most importantly, increased local communities' contestations should be promoted. The potentially beneficial reforms needed in CAMPFIRE consist of specific actions that honour and encourage the formation of institutions satisfying the design principles such as: congruence between clearly defined resource and governance boundaries; congruence between appropriation and provision rules and local conditions; collective choice arrangements and localised monitoring.

JEL Classification: D71, H41
Keywords: CAMPFIRE, wildlife conservation, design principles, institutions, common property

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1. Introduction

Uncertainty clouds the interaction between humans and the environment through various systems of ownership. Such uncertainty is caused, among other things, by climate, political upheaval, health risks, or financial variability (Hanna and Munasinghe 1995). In human systems, uncertainty creates incentives for accelerated rates of use due to the lack of assurance that resources not used in the present will be available in the future (Bromley 1991; Hanna, Folke and Mäler 1995). The threat of the possibility of collapsing resource use decisions from the future to today as created by uncertainty in human systems create the need for institutions that constrain human actions (Hanna, Folke and Mäler 1995). Institutions refer to the rules, norms, and strategies adopted by individuals operating within and across organisations and exist in the minds of the participants and are sometimes shared as implicit knowledge rather than in an explicit and written form (Ostrom 1999). The knowledge of how institutions function in relation to humans and their use of the environment is thus critical to the design and implementation of effective environmental protection. Ultimately, economic development depends on institutions that can protect and maintain the environment's carrying capacity and resilience (Arrow *et al* 1995). Thus, sustainable use of the natural resources could enhance economic development and greatly reduce poverty.

During 1995/96, 61% of Zimbabwean households were officially classified as poor and this translates to 76% of the population being poor. Poverty is much more widespread in rural areas than in urban areas with 75% of the rural households being poor compared to 39% of urban households (CSO 1998). Measured by numbers of people, 86% of the rural and 53% of the urban population were viewed as poor. The majority of the Zimbabwean population lives in the rural areas (Child 1995) – 63% of the households live in rural areas. The poorest districts are wildlife-abundant areas, especially the poorest three districts namely Hwange, Binga and Nyaminyami (Ministry of Public Service, Labour and Social Welfare 1997). Of course this result is to be expected since these communities live in agriculturally less productive areas that are largely only suitable for extensive livestock production and wildlife conservation. This gives room for driving rural economic development by complementing the ongoing poverty-reduction strategies of land reform and centrally funded programmes to mitigate the social dimensions of adjustment through the sustainable use of the wildlife resource.

Significant gains have been recorded in the communal areas management programme for indigenous resources (CAMPFIRE), such as the increased share of land devoted to wildlife management, building up of institutional and administrative capacity at rural district[41] level, development of social infrastructure and influencing sensible regional wildlife policy reform. However, literature (for example Halser 1999, Patel 1998, Murombedzi 1992) indicates an array of problems that have emanated from or have not been resolved by CAMPFIRE. These include the paternalistic tendencies of Rural District Councils (RDCs) towards local villages and wards, elite capture by both traditional and democratically elected authorities, the failure of the programme to incorporate local knowledge and practices, the continued prohibition of local use of wildlife resources, failure of the programme to resolve human-wildlife conflict, and continued subsistence and commercial poaching.

Investigating many of the world's long enduring, self-governing resource systems and drawing from the works of scholars such as Fikret Berkes, Daniel Bromley, David Feeny, Margaret McKean, Robert Netting and David North, among others, Ostrom (1990) noted that their similarity was the perseverance of these resource systems and their institutions. The institutions, being the framework through which the resource system is managed, may have been responsible for the long endurance of the resource system (Hanna and Munasinghe 1995). Even though institutions do not have to be exactly the same in every resource system and over time, they have some common sustaining characteristics that enable them to yield sustainability in the resource systems (Shepsle 1989). Ostrom (1990) calls these sustaining characteristics design principles of robust institutions. Technically, a "design principle" is defined as a conception used either consciously or unconsciously by those constituting and reconstituting a continuing association of individuals about a general organising principle (Bromley

[41] The terms rural district, RDCs and local communities are not necessarily interchangeable. The term rural district is used to denote the territory of communal area inhabitants (10,000 to 50,000 households) while RDC is the communal area inhabitants' administrative body, which is made up of representatives elected from sub-district structures called wards. The RDC is a legal institution created by an Act of Parliament, the District Councils Act (1980), while local communities have no legal status at all.

1991, Ostrom 1995). Even though Ostrom (1990) does not claim the necessity nor sufficiency of these principles in ensuring that the institutions will yield a sustainable resource system, we believe that their satisfaction under CAMPFIRE will only help the institutions to make the resource system sustainable rather than harm it. To alleviate the CAMPFIRE problems highlighted earlier, the starting point in search for potentially beneficial reforms needed in CAMPFIRE would be an investigation of the extent to which the design principles of robust institutions are satisfied. Reform of institutions under CAMPFIRE could then be formulated with the intention of ensuring that the design principles for sustainability are satisfied. Our interest in the rest of this paper is to investigate the extent to which the institutions under CAMPFIRE satisfy the design principles of robust institutions and recommend broad reforms that are needed in CAMPFIRE.

2. Does CAMPFIRE satisfy the design principles of robust institutions?

In this section we investigate the extent to which the property rights regime being used to manage wildlife in Zimbabwe's rural areas satisfies the design principles of robust institutions. We will discuss the eight design principles as put forward by Ostrom (1990), indicating the extent to which each of them is satisfied in CAMPFIRE.

1. ***clearly defined boundaries*** – individuals or households with rights to withdraw resource units from the common pool resource and the boundaries of the resource itself must be clearly defined.

It is important to ensure that a property rights regime has clearly defined boundaries of the appropriators, i.e. individuals or households with rights to withdraw units from the common pool resource, and clearly defined boundaries of the resource to be managed. If either of the two boundaries remain uncertain then no-one knows what they are managing or for whom. Without clearly defining the resource boundaries and successfully excluding outsiders, there is the risk that any benefits produced by the local appropriators through their own efforts will be reaped by others who do not contribute to these efforts – the free riding problem. Depending on the extent of the free riding, those who invest in the resource may not receive as high a return as they expected or as would give them enough incentive to continue managing the common. At the worst, the actions of the free riders could bring about the so-called 'tragedy of the commons' (Hardin 1968). Apart from simply clearly defining the resource and governance boundaries it is important to ensure that, to the extent possible, those boundaries are consistent with each other. Boundary congruency would serve to bring the area of decision-making in line with areas of ecological interaction lest decisions taken by the appropriators have only a partial effect on the ecological system or be in conflict with decisions made elsewhere about the remaining parts of the ecological system (Hanna, Folke and Mäler 1995).

In principle, all the residents of a rural district qualify as appropriators of the common pool wildlife resource by virtue of the RDC holding the appropriate authority[42] (AA) status under CAMPFIRE. However, it is not necessarily the case that the group of individuals or households with rights to the resource is the same as the residents of the rural district hence each RDC designates the wards[43] and villages that should be regarded as appropriators of the common pool resource. Appropriation in the case of the common pool wildlife resource under CAMPFIRE entails the receipt of revenue from CAMPFIRE wildlife based activities since utilisation has mainly been limited to tourism and trophy hunting, a practical preserve for foreigners. The criterion for choosing the wards and villages has usually been proximity to wildlife routes and bases. The rationale is that wildlife has access to assets of those who are living close to it and hence if there is any destruction it is most likely that it is perpetrated against the wards and villages in the vicinity of wildlife. Indeed, the philosophy behind CAMPFIRE is to, at least partially, compensate those who bear the costs of living with wildlife. In most rural districts only a fraction of the total number of wards and villages has been designated as appropriators of the resource. For the 23 rural districts for which data

[42] "Appropriate Authority in effect grants [Rural] District Councils the same rights as commercial farmers enjoy on private land. Councils are empowered to enter into contracts with private organisations for the exploitation of their wildlife, receive all payments directly and carry out their own problem animal control. Equally well the onus is on them to carry out their own law enforcement and protect the resource" (DNPWLM 1989, p5, quoted in Murombedzi 1992, p13).

[43] A ward is a sub-district administrative unit that is made up of about six villages or at least 600 households. The current definitions of ward and village were established when the ward development committee (WADCO) and village development committee (VIDCO) were created in 1984 by a Prime Minister's directive, as opposed to an Act of Parliament, and as such do not have legal status.

exists 29.63% of the total wards have been designated as CAMPFIRE wards. Both in principle and practice, the boundaries of the individuals or households with rights to withdraw units from the common pool wildlife resource under CAMPFIRE is clearly defined since the RDCs, wards and villages do not overlap.

The accordance of AA status to the RDCs opened the resource to outsiders and thereby imposed a passive tax on the resource. The RDC, which is made up of representatives from all wards in the rural district, makes management and appropriation decisions about a common belonging to that fraction of the wards that have been designated as appropriators. This allows outsiders, those representatives from the non-appropriator wards, to make wildlife management and appropriation decisions about a common that does not belong to their constituency. The current problem in excluding non-appropriator wards from the common pool wildlife resource is that sub-district communities are not organised as corporate or legal bodies hence cannot legally own the common. In cases where the RDC retains some benefit from the common pool wildlife resource this retention can be viewed as constituting a benefit to outsiders. If the whole rural district should sufficiently benefit from the resource invested in by a few wards then "the closer the situation is to that of a one-shot dilemma where the dominant strategy of all participants is to overuse the resource" (Ostrom 1995, p36). The threat of overuse becomes more likely where the fraction of retention by the RDC is large. Revenue allocation data for the period 1989-2001 for all rural districts shows that only 45.88% of the revenue was distributed to the sub-district communities (*see table A2 in appendix*). Thus the non-appropriator wards could also benefit once these retained funds are used for general district administration.

One possible situation in which management of the common pool wildlife resource at the RDC level may not work is when people in neighbouring wards that each lie in adjacent rural districts are from the same ethnic group and give superiority to traditional ethnic institutions over modern political administrative institutions. The fact that traditional ethnic boundaries are not necessarily aligned with the modern political administrative boundaries means that when it comes to active management of resources these people would follow their traditions in doing so just like they follow traditions in dealing with daily problems. The resolutions passed by each of the RDCs independently to be carried out by their respective wards will most likely not both succeed if they contradicted one another since the traditional institutions for this ethnic group would decide what course of action to take thereby making at least one of the resolutions to fail. The incorporation of traditional chiefs as an interest group in the RDCs in the 1990s does not help this situation since they are always in the minority in RDCs which are purported to be governed by modern democratic institutions rather than traditional institutions. The good thing about this ethnic group scenario is that it will unconsciously bring about useful, though informal, coordination in the management of the common pool wildlife resource especially if the ethnic group adopts the pro-wildlife resolution. It is unlikely that this would happen where the extent of free riding is large.

It is difficult to define the boundary for migratory species that move across villages, wards and districts. Any attempt to define boundaries will therefore largely be along the habitat lines. Under CAMPFIRE, the resource boundaries have been defined so that they conform to the geographical boundaries of the rural districts. This has been necessitated by the need to align the resource boundaries with the governance boundaries created by the granting of the AA status to RDCs. It is not necessarily the case that the rural district boundaries are aligned with the ecological boundaries of the common pool wildlife resource. If anything the rural district boundaries are politically motivated constructs, in which the central government sought ways of getting representation at the grass roots level. Indeed, political administrative borders are completely arbitrary from the perspective of wildlife management in Zimbabwe. There is nothing to suggest that wildlife respects politically determined boundaries. As a result, it happens that at times some villages, wards and RDCs extract benefits not only from that wildlife that falls under their jurisdiction but also from that wildlife that falls under the jurisdiction of other villages, wards and districts. The resource ownership conflicts between villages, wards and districts have normally been resolved by requiring that the wildlife resource's residence be determined upon death i.e. it belongs to the village, ward and district on which it eventually dies regardless of where it has all along been living and causing destruction. Thus even though the resource boundaries may not have been the best possible, they have been clearly defined under CAMPFIRE by insisting on conformity with the geographical boundaries of the rural districts and post-mortem residence assignment for the resource.

We have alluded to the need for the synchronisation of governance boundaries with ecological boundaries as one condition for the effectiveness of a property rights regime. We reiterate that this is a useful way forward because wildlife, being fugitive, may have inter-village, inter-ward, inter-district or inter-regional dimensions that require coordination (Hanna, Folke and Mäler 1995). In some CAMPFIRE areas coordination amongst RDCs has been forthcoming especially where they share a significant wildlife reserve. For example, in the north-eastern border of the country three rural districts, namely Mudzi, Uzumba-Maramba-Pfungwe and Rushinga are jointly managing wildlife in the shared Nyatana Wildlife Reserve. Ecological boundary demarcation should take precedence over governance boundary demarcation so that management decisions have a complete effect on the ecological system and that they do not conflict. In the event that there is no readily available legal governance boundary such a boundary should be put in place. Of course, it is possible within this framework to subdivide responsibilities in resource management by assigning tasks to subdivisions of the governance. After all it has been shown from an analysis of experience that local users are effective managers of small-scale resource systems (Ostrom 1995). The crucial requirement that should be placed upon such subdivisions of governance is that they should be coordinated so that everyone knows what others are doing about a part of the larger ecological boundary. The difficulty in demarcating ecological boundaries for migratory species could be a lesson that in some cases the potential coordinating units may not necessarily fit into the borders of rural districts i.e. several villages and wards in various RDCs may be the units that need to be brought together to manage a certain common. Insisting on the guardianship of the wards and villages' parent RDCs could take away the advantages of managing commons at the level that is local to the resource. In this regard, without reinventing the boundaries of the appropriate authority, partnerships of adjoining rural districts could be encouraged and the configuration of partnerships being determined by the extent of habitats.

If governance boundaries are to be reinvented then this should be done taking many different considerations into account. Conservationists have pointed to the need to manage entire ecosystems by unified methods designed to save all their inhabitants at one time, thereby economising on tightening conservation budgets, achieving economies of scale and efficiency. However, the absence of a consensus definition for ecosystem management frustrates conservation efforts coupled with the lack of consensus about what constitutes a healthy ecosystem. Also, the fact that various ecosystem processes are maintained even as species disappear is but one aspect that works against focusing on conserving ecosystems rather than on conserving species. Since monitoring and managing all aspects of biodiversity that might interest us including species richness and composition, physical structure, and processes are so difficult, a variety of shortcuts have been proposed whereby attention is focussed on one or a few species (*see Simberloff (1998) for more details*). Umbrella species are those with such demanding habitat requirements and large area requirements that saving them will automatically save many other species. However, whether many other species will really fall under the umbrella is a matter of faith rather than research. Keystone species, at least in some ecosystems, have significant impacts on many others. Since a keystone species approach is focused squarely on an understanding of the mechanisms that underlie the function and structure of an ecosystem, it appears that it might suggest entirely new ways of managing a problem, rather than the successive-approximation approach that dominates adaptive management (Simberloff 1998). In Zimbabwe, "the elephant[44] population is probably the single greatest factor influencing ecosystem conservation in protected areas" (DNPWLM 1999). Perhaps this points towards that governance boundaries under CAMPFIRE should be along the lines of ranges of elephants (and other keystones) and take refuge in it being both a keystone and umbrella species.

[44] Elephant trophy hunting was largely unaffected by the 1989 CITES ban on trade in elephant products because the ivory and other elephant products are considered the personal property of the client. Furthermore, within the duration of the ban, Zimbabwe had an annual CITES quota for trophy hunted elephant of 500 animals (WWF SARPO 2000, p2.23).

2. ***congruence between appropriation and provision rules and local conditions*** – appropriation rules restricting time, place, technology, or quantity of resource units should be related to local conditions and to provision rules requiring labour, materials, or money.

African wildlife ecological systems are subject to great variation depending on drought and environmental factors (Hasler 1999). Establishing a link between appropriation rules and local conditions helps to institutionalise heeding the feedback effects from the ecosystem to enhance sustainability. If a community must benefit from its wildlife in the long run then the wildlife harvest quota must be sustainable. The number of key species in Zimbabwe has been relatively stable or increasing (Child, *et al* 1997) and habitat loss has been held back in those areas where CAMPFIRE exists. Within and around national parks, elephants exceed their carrying capacity of five individuals per square kilometre and woodlands are under severe pressure (Royal Netherlands Embassy 1998). Hasler (1999, p14) points out that, "hunting quotas in CAMPFIRE areas are considered to be conservative". This may have emanated from the following factors: (i) the DNPWLM often used "population and growth rate estimates [that] were inaccurate" (WWF SARPO 2000, p3.17), (ii) the DNPWLM did not take into account the number of animals that the communities wanted in their areas, and (iii) the "setting of quotas is primarily aimed at identifying annual 'sustainable' off-take for the safari [trophy hunting] industry" (Murombedzi 1992, p31), which are generally lower than quotas for non-selective hunting. However, there is increasing convergence between the quotas that the DNPWLM sets and what the RDCs expect.

It is important to note that if the local communities are to take an interest in managing the wildlife resource they must be able to get a reward for their conservation efforts. A direct link between reward and provision of conservation is established by aligning appropriation and provision rules. Wildlife is a unique resource that does not require the usual provisions. However, damage that people put up with, guarding fields from wildlife intrusion, protecting fields with thorny-bush fences, and looking out for poachers constitutes some kind of provision. Ideally those who render the highest proportion of provision should reap a greater proportion of the benefits. Over and above that, the local communities may need to be compensated for foregoing some opportunities for economically more rewarding uses of land within their territory. The benefits under CAMPFIRE consist primarily of the utilisation of the wildlife harvest quota, though increasingly other projects of a social and economic nature are being added to the benefit matrix. Under CAMPFIRE, the twelve[45] rural districts that are adjacent to major protected areas, and thus suffer more nuisance, have the potential to earn more income than those removed from these core biodiversity areas. However, within all rural districts, the benefit from wildlife utilisation at the household level has been highly variable with sparsely populated wards having the potential to earn more than those that are densely populated. "The average CAMPFIRE ward dividend benefit per household (excluding indirect benefits) was US$19.40 per household in 1989 but dropped in 1991 to US$5.97 and then to US$4.49 in 1996" (Hasler 1999, p12). The drop in this variable, which was US$3.05 in 1998, is largely due to the increasing number of households joining the programme in low wildlife potential areas.

One threat to the long-term sustainability of local institutions is the availability of large quantities of funds from external authorities that appear to be "easy money" (Ostrom 1995). These can undercut the capabilities of local institutions to be sustained over time. The problem of local units becoming dependent on external funding is not limited to the funding provided by international aid agencies (Ostrom 1995). The rationale for such external funding is that the larger Zimbabwean or global society must mobilise additional resources to raise the level of conservation efforts towards socially desirable levels. This is because conservation of wildlife resources includes retaining options for future economic use, or ethical or aesthetic grounds, and simply assuring access to villagers for immediate use would lead to socially sub-optimal levels of wildlife conservation (Gadgil and Rao 1995). Depending on the conduit through which vast amounts of external funds replace programme-generated resources, the connection between provision and appropriation is lost. Individuals using "other people's money" are rarely as prudent as when they are using funds derived from themselves and their neighbours (Ostrom 1995). Ideally, external aid should constitute additional demand for conservation by outsiders, over the locally determined levels, and the funds should be

[45] The major wildlife districts in Zimbabwe are Hwange, Tsholotsho, Chipinge, Binga, Gokwe North, Guruve, Beitbridge, Bulilimamangwe, Chiredzi, Nyaminyami, Hurungwe and Muzarabani.

channelled directly to the producer communities so that they respond to this incentive emanating from increased demand for conservation.

Processes that encourage looking to external sources of funding make it difficult to build upon indigenous knowledge and institutions (Gadgil, Berkes, and Flokes 1993). This is unfortunate especially if traditional institutions are more likely to lead to greater conservation than modern institutions. If those at a local level ask for funds repeatedly, those at a national or international level have an excuse to exert more influence over what is happening at a local level. As central officials begin to finance and take a more active role, those at a local level may pull back even further, thus accelerating a process toward central dominance (Ostrom 1995).

The effect of donor funds in CAMPFIRE[46] may be viewed as having been two-pronged but in both ways affecting the congruence between provision and appropriation. On the one hand, donor funds may have acted to stifle the formation of traditional institutions, which would have reduced the costs of running the programme (particularly the use of game scouts, communication radios, guns, cars, and generally the RDC CAMPFIRE office in monitoring) and thereby increase the financial benefits to communities hence positively affect stewardship practice. Murombedzi (1997, p16) concurs that, "external aid seems to have negative implications for the ability of CAMPFIRE to facilitate local community participation in decision-making". By providing the requisite operating capital and sponsoring skilled labour[47], external aid resulted in the development of sophisticated top-heavy management structures aimed at managing wildlife, carrying out problem animal control (PAC) and other crop protection measures and entering into wildlife exploitation joint ventures with safari operators. Such structures resulted in increased technical management capacity for the RDCs to manage the resource at the expense of the basic tenet of CAMPFIRE, namely local communities' participation in the management of the resource. In the presence of external aid, sub-district devolution

[46] CAMPFIRE was backed by at least US$33 million for a ten-year period from 1989-1999 in funds from the United States, European Union, United Kingdom, Norway, Netherlands, Germany and Japan (Patel 1998).

[47] For example Zimbabwe Trust, an NGO funded by the British Overseas Development Association (ODA) and blocked (frozen) funds of former Zimbabweans exiled in the United Kingdom and USAID, provided grants to RDCs for infrastructural and capital development, training, recruitment and funding of rural district level Institutions Officer in Guruve and Nyaminyami (Murombedzi 1997).

did not and might never take place leading to the persistence of the current scenario in which sub-district communities receive insignificant dividends without expending any conservation effort beyond the damages suffered from wildlife. Experiences from rural districts such as Nyaminyami, Guruve, Binga, Tsholotsho, Bulilimamangwe and Hwange show that local community participation was not enhanced by the presence of external aid (Murombedzi 1997). In light of the foregoing, and if the highly technical structures were indispensable, the publicised success of CAMPFIRE may have been very artificial since only the funds from donors kept the programme floating. Without donor funds one would envisage a near collapse of the programme because it has been spoiled by external funds inflows and has not learnt to be self-sufficient in the last decade.

On the other hand, donor funds may have helped CAMPFIRE to kick off without the problems of inadequate incentives that it could have faced in the absence of donor funds. For instance, in 1989 Zimbabwe Trust subsidized Nyaminyami Wildlife Management Trust – a sub-committee of the Nyaminyami RDC charged with managing the wildlife resource – to the tune of ZW$171,000 (approximately US$80,433) as well as services of an interim general manager thus freeing revenue to pay the communities, which would have been impossible and probably affected the continuance of the programme in Omay communal lands (Murombedzi 1997). Despite having been channelled indirectly through NGOs and other participating organisations removed from the local communities, the donor funds managed to provide some incentives by way of community development, applied research, regional communication, project management, project evaluation and wildlife conservation. These overheads would otherwise have been paid from CAMPFIRE revenues thereby reducing disbursements to producer communities and negatively affecting stewardship practice (*see table A1 in appendix for CAMPFIRE incomes*). Given that wildlife conservation largely depends on stewardship practice any of these effects actually realised would have affected the course of wildlife conservation and sustainability somehow. Murombedzi (1997) shows that two villages, Mahenye in Chipinge rural district and Chikwarakwara in Beitbridge rural district, managed to kick off without any external aid. Despite the absence of external aid in these villages, local community participation has been negligible also mainly because of the ‘paternalistic’ tendencies of the RDCs. We insist that incentives for conservation of a resource should come from that resource for sustainability given that donor funds are not long-term inflows. For the reason that they are short term and that they are obscured from the local communities who are the resource producers, donor funds mismatch provision and appropriation.

3. ***collective choice arrangements*** – most individuals affected by operational rules should participate in modifying operational rules.

A regime functions best when decision rules are consistent with ownership, for example, when collectively owned resources are managed through collective choice arrangements (Hanna, Folke and Mäler 1995). Collective choice arrangements allow resource institutions to tailor better their rules to local circumstances since the individuals who directly interact with one another and with the resource can modify the rules over time so as to fit them better to specific characteristics of their setting (Ostrom 1995). In CAMPFIRE there has not been much room for collective participation in the making or modification of operational rules for three reasons. Firstly, most operational rules were designed by the DNPWLM when it initiated the CAMPFIRE programme. These rules were to become CAMPFIRE “guidelines” and all RDCs are expected to follow them in as much as their situation permits. Secondly, the nature of the dominant wildlife resource utilisation strategy in Zimbabwe is such that it is reserved for an international trophy hunting market. Local communities have been inhibited from participating by the nature of the high skills (professional hunting and marketing) and capital (finance and equipment) and foreign clientele required. For instance, Guruve RDC experimented safari operations without much success. Thirdly, the fact that the RDC constitutes representatives democratically elected by the grass roots has been interpreted to mean that the RDC can act on behalf of the local communities and they do not need to participate in any other form except through their representative.

The market orientation of CAMPFIRE precludes the use of indigenous knowledge, customs and strategies of resource management thereby relegating the role of RDCs to that of providing services to the private safari enterprises (Murombedzi 1992). The

RDCs have had to ensure that a viable resource base exists for exploitation by the private safari operators by policing local insurrection such as poaching, haphazard expansion of arable agriculture, human settlement in wildlife habitats, livestock population expansion and non-acceptance of the status quo. The programme primarily seeks to produce a financial dividend and thereby curtails the ability of the local communities to define their own resource management objectives. In most CAMPFIRE areas, the communities are not in contact with the actual resource for as far as monitoring, marketing and harvesting is concerned. This reality does not give them an opportunity to contribute in the making and modifying of operational rules. Under CAMPFIRE, after receiving the quotas from the DNPWLM, the RDCs as the AA decides how many animals to put under trophy hunting, PAC, cropping, live animals sales, culling and local hunting – with most animals usually being put under trophy hunting because of the need to produce a financial dividend. The RDCs will then market and sell hunting concessions/leases to private non-local safari operators. The safari operators will find clients of their own so that they make profit on the hunts that they have claims to. The clients then carry out the actual hunting through the engagement of a Zimbabwean registered professional hunter. If communities were to harvest the resources in the concession area that would be illegal because the rights would have been surrendered to the safari operator through the lease agreement. The RDCs collect the trophy fees and concession/lease fees as the benefit from the resource. The local communities rarely get resource allocations for cropping and local hunting. At times they may get some meat if large animals such as elephants are hunted because the safari operator or client does not have use for it apart from the parts collected as trophy. The communities will get the benefit from the use of their resource when the RDC disburses revenue. Communities have always charged that resource utilisation is an RDC-safari operator affair and it leaves the communities out. Local communities have usually only been given a chance to participate in deciding how their share of wildlife revenues could be used. Child, *et al* (1997) gives an account of how Chikwarakwara village in Beitbridge rural district spent four days gathered under a baobab tree democratically deciding how to use their share (ZW$60,000 or US$28,222) of 1989 revenues (ZW$96,000 or US$45,155), which they finally decided to use on school infrastructure, household dividend and setting up a village grinding mill. Murombedzi (1997) witnessed the same village refusing to accept its share (ZW$19,000 or US$5,065) of 1991 revenues (ZW$142,170 or US$37,902) because it felt that the RDC wanted to impose its own decisions on them by suggesting that the village decision to invest in a grocery store was not viable. In general, RDCs have been accused of being too 'paternalistic' in that they usually ask the communities to identify viable projects/programmes in which they would want to invest their shares of wildlife revenues before the revenues are released. Even though the objective of local government in Zimbabwe is to provide accountable and democratic government for local communities, it is because of this possibility of lack of downward accountability (and presence of upward accountability) that the RDC and the communities could be thought of as different entities that optimise in different ways. It is therefore for this reason that the real owner of the wildlife is thought to be the communities, as opposed to the RDC, and as such communities should participate in the collective choice arrangements.

4. ***monitoring*** – monitors, who actively audit resource conditions and appropriator behaviour, should be accountable to the appropriators or are the appropriators themselves.

The challenge for the design and performance of property rights regimes is to ensure those making decisions have the appropriate incentives to take ecosystem equilibrium shifts into account and make the appropriate trade-off between the costs and benefits. This requires that decision makers do benefit from monitoring appropriation and feedback from the ecological system and ensuring that appropriation allows perturbations to enter the system at a scale that allows subsystem variability but does not challenge the underlying ecological and economic activity (Berkes and Folke 1994). Essentially, monitoring should be conducted with respect to resource condition (species diversity, wildlife populations, age class structures, cross boundary movements, problem animals, wildlife health, trophy quality, habitat condition, etc) and appropriator behaviour (settlement patterns, fire management, uncontrolled hunting, etc). Such monitoring is likely to be effective if done by the appropriators themselves or monitors who are accountable to the appropriators because that ensures that there would be an immediate reaction to collected data. Being localised such monitoring is likely to extract information about the resource and appropriator behaviour accurately and timely. Also all the necessary monitoring is likely to be conducted since it would be cheap to do so unlike if the monitor was external who could decide to forgo some monitoring routines to reduce costs. Thus monitoring by local communities constitutes one way to reduce costs and dependence on donor funds. The DNPWLM and the RDC need trucks, helicopters, skilled manpower, etc to carry out state-of-the-art monitoring hence the need for high capital. Communities could render monitoring cheaply since the costs of monitoring at a local level are lower as a result of the rules-in-use (informal rules). Rules-in-use stem from the traditional systems of beliefs and taboos, where the ancestral spirits are largely responsible for enforcement. Of course the costs and benefits of monitoring a set of rules are not independent of the particular set of rules adopted (Ostrom 1995). If the set of rules adopted are modern rather than traditional it does not imply negligible costs of monitoring them. It is likely that the local communities will take refuge in the traditional systems of resource management. Even though communities have been alienated from the wildlife resource for a long time the traditional resource management systems have not been completely destroyed since other resources continued to be under the guardianship of the communities.

The case of CAMPFIRE is such that the DNPWLM, with the help of the WWF that carries out aerial wildlife surveys for communal lands, has effectively been responsible for monitoring the resource condition. This has been necessary because the DNPWLM has to determine the wildlife harvest quota and it has the expertise. Quotas are set using a system called triangulation, which involves assessing information from three sources – (i) aerial surveys, (ii) ground counts, and (iii) trophy measurements as well as stakeholders' opinions. The DNPWLM has encouraged the RDCs to acquire the necessary skills so that they can take over as is required by their AA status while the DNPWLM would sit back and assume the role of regulator. Zimbabwe Trust, WWF and the Safari Club International have facilitated training workshops and rendered technical assistance, particularly for quota setting. Since 1995, in some areas such as Omay communal lands, the RDCs and communities started learning about quotas, counting wildlife and trophy quality assessment, and how to review information on wildlife in order to set quotas (WWF SARPO 2000). Most RDCs, for example Muzarabani, Guruve, Chipinge, Gokwe North, UMP Zvataida, Binga, Hwange and Nyaminyami, have employed game guards who have been trained and equipped to monitor the state of the resource, carry out problem animal control, carry out anti-poaching campaigns, and monitor the interaction between local communities, safari operators, safari clients and the resource. Some sub-district communities, particularly in areas where benefits have been high, have appointed voluntary resource monitors who are also tasked with monitoring appropriator behaviour in their areas. There is an increasing role for the game guards and voluntary resource monitors in the monitoring of the resource as RDCs are now asked to set and propose quotas for their areas. However, the DNPWLM still has to approve and adjust, where necessary, the proposed quotas. In this framework the appropriators (the RDC and the communities) have had to be accountable to the effective resource monitor (the DNPWLM). Sub-district communities submit their reports to the RDCs, which in turn submit their annual reports to the DNPWLM before the quota for the following year could be disbursed. As a matter of fact the monitoring conducted by the DNPWLM focuses on three areas namely, (a) setting of quotas, (b) ensuring that revenues are returned to producer communities as the incentive for sustainable management, and (c) the following of informal "guidelines" aimed at promoting economically sound and democratic wildlife management. The monitoring of appropriator behaviour has largely been relegated to the RDCs in line with the AA status. Since authority over use and benefits from wildlife ultimately belongs to them, the RDC and its CAMPFIRE wards constitute the appropriators even though safari operators and their clients carry out the actual hunting. In general, appropriators are expected to monitor each other's behaviour and the behaviour of poachers even though the scope of use of monitoring information provided by other groups is limited.

The role of RDCs and communities in resource monitoring could greatly be increased if they know that the information they provide will actually be used. Even in areas where local communities perform ground surveys it has tended to be the case that the DNPWLM makes 'big game' harvest quota decisions predominantly on the basis of aerial surveys it conducts in collaboration with the WWF, assigning less weights to ground surveys as they are usually thought to give population indices only rather than population estimates. Aerial surveys rely on estimating wildlife numbers from sample counts and use of indicators to ascertain whether the population is stable, increasing or decreasing. In general sample counting relies on animals being evenly distributed and if they are not then this can lead to inaccurate population estimates (WWF SARPO 2000). Local communities have always accused the DNPWLM of setting wildlife harvest quotas conservatively. However, with the increasing role accorded to RDCs in quota setting there is an increasing convergence between the quotas that the RDCs propose and what the DNPWLM eventually approves.

5. ***graduated sanctions*** – appropriators who violate operational rules should be likely to receive sanctions depending on the seriousness and context of the offence from other appropriators, from officials accountable to these appropriators, or from both.

Commitment to the observance of operational rules in many sustainable community-governed resources cannot be explained by external enforcement since external enforcers rarely travel to remote areas. Instead appropriators create their own internal enforcement to (a) deter those who are tempted to break rules, and thereby (b) assure quasi-voluntary compliers that others also comply (Ostrom 1995). In case there are rule infractions, sanctioning is largely carried out by the appropriators themselves or their appointees. Even though the RDC and communities in CAMPFIRE could be thought of as appropriators there are other two agents who have access to the common pool wildlife resource namely external poachers and safari operators. Violation of operational rules primarily constitutes the illegal harvesting of the wildlife resource. Thus, infraction of operational rules could potentially be committed by appropriator-poachers, external poachers and safari operators. Appropriator-poachers are usually involved in subsistence poaching while external poachers and safari operators could be engaged in commercial poaching. In subsistence poaching wildlife products are for consumption while commercial poachers primarily hunt trophy for sale at a market. Commercial poachers who are usually outsiders employed by dealers include carriers and professional hunters armed with automatic weapons and often hunt deep into the protected areas. Subsistence hunters hunt in small gangs in areas relatively close to their homes and use primitive firearms, spears, snares and dogs. Subsistence hunters have close historical, traditional and cultural ties to wildlife hunting; and subsistence hunting is a skill and profession that has an important social role, and the number of hunters in each generation is controlled by the community elders (Marks 1984, Skonhoft and Solstad 1996). Under CAMPFIRE individuals have the right to utilise wildlife as part of a community as sanctioned by the RDC hence wildlife utilisation by individuals is still illegal. Poaching by appropriators is indicative of two aspects namely (i) the local resistance to the exclusion from direct household utilisation of the resource that brings about costs, and (ii) the competing property claims to the resource between the RDC and communities. Local institutions cease to function to regulate such use and the tendency is towards the operation of open access (Murombedzi 1992). It therefore becomes difficult to come across instances of community regulation of this illegal activity. Murombedzi (1992) reports that knowledge of the existence of poaching activities is universal in the Nyaminyami communities he studied even though no one dared to punish these subsistence poachers.

According to Milner-Gulland and Leader-Williams (1992) illegal harvesting of wildlife of the subsistence type depends on factors such as the detection rate, the size of the penalty for being caught, the money income in alternative activities and the size of the stock of wildlife. A greater anti-poaching drive, higher penalties and a higher opportunity cost of poaching help to reduce poaching (Skonhoft and Solstad 1996). We maintain that both types of poaching will be high when there is a low level of community co-operation (monitoring) in wildlife management. This is mainly because external monitors such as the parks agency will not find it profitable to invest fully in insurance against poaching and also because the poachers are supported and tolerated by the local communities. Community co-operation itself is a positive function of the net benefits from wildlife. In other words the absence of net benefits from wildlife entail that there is high grazing competition and damage from wildlife and the local community will be angry about the existence of the wildlife resource hence the more they will support and tolerate poachers. Evidence from Zimbabwe shows that poaching was rampant in some areas prior to the introduction of CAMPFIRE but was drastically reduced afterwards as the neighbouring communities started reaping economic benefits from wildlife (Child *et al* 1997).

The problem of existence of sanctions is part of a broader weakness of the property rights regime under which wildlife is managed under CAMPFIRE. Common pool wildlife resource management is usually based on some form of legal and recognised utilisation, out of which the need to regulate or manage arises i.e. the appropriators have to develop the management institution out of direct and acceptable utilisation. Direct utilisation of the resource is reserved for outsiders under CAMPFIRE namely safari operators and their clients. Local resistance to the exclusion from the direct consumptive utilisation of the resource and the apparent competing property claims to the resource between the RDC and communities will therefore emanate in communities condoning subsistence poaching. In the experience of CAMPFIRE, only those communities that have benefited very much at a household level have opened their eyes to poaching and made reports to the relevant local wildlife committees if animals involved are large game on which the programme relies for income generation. Due to the fact that the zero hunting option is not historical, traditional or cultural, some subsistence hunting has been condoned by the society if it relates to the usually hunted smaller game such as guinea fowl, klipspringer, spring hare, buck and antelope and if it is strictly for home consumption. Generally the small game populations are large and can thrive despite subsistence hunting off-takes. This absence of scarcities perhaps helps to explain why there has not been any serious community regulation.

In the traditional African religion, sanctioning with respect to misuse of natural resources such as unwarranted hoarding and killing what one cannot consume comes from the ancestral spirits and community elders. The ancestral spirits that are always on guard would punish anyone violating rules regardless of whether they are physically caught or not. Individualistic sanctions meted out by the ancestral spirits could be in the form of destruction of one's crops by wild animals or bad luck. Some punishments are society-wide, for example poor rains, such that every member of the community has to refrain from infractions for the good of the society. There has been a high tolerance zone for infraction of operational rules by appropriators for as long as they are not breaching any traditional rules regardless of whether they are violating CAMPFIRE rules of zero household use. Members have been issued warnings by community elders after which they would get graduated sanctions through the modern criminal court system when they have started violating both traditional and CAMPFIRE rules.

The law that assists wildlife management in Zimbabwe allows game wardens to curb particularly commercial poaching even by shooting poachers. Child (1995) narrates a situation in which poaching was severe in Mahenye village in Chipinge district before the inception of CAMPFIRE. Hunting was a way of life for these people and they resented the Park for denying them rights to use the resources and for isolating them from others of their tribe. In one fortnight in 1982, through the efforts of the DNPWLM, there were about 80 convictions against people in the community, which did nothing to reduce their antagonism towards the Park (Child 1995). It seems that these people were hunting relatively big game, which would also have been unusual in the traditional African culture. A safari hunter and rancher brokered an agreement between the DNPWLM and the Mahenye people, whereby he could shoot a small quota of elephant, buffalo and nyala crossing out of the Park (Child 1995). The people would receive the meat and all the revenue in exchange for not poaching. As a result of these measures, poaching decreased sharply signaling that the motive for the initial poaching may have been commercial. In general, commercial poaching particularly of the elephant intensified since the 1989 CITES ban, apparently because law enforcement was curtailed due to reduced Treasury allocations, to which the loss of ivory revenue had contributed. From 1984 to January 1993, the DNPWLM's Operation Stronghold resulted in the deaths of 167 poachers and the wounding and capture of 137 others (Child *et al* 1997). In areas where communities have been compensated fairly, CAMPFIRE has allowed the commercial poaching levels to subside since communities are now also helping with enforcement. In most cases the sanctions imposed on external commercial poachers have not been graduated since death has been applied. For those that have been arrested they have been tried through the criminal court system, which imposes graduated sanctions.

Under CAMPFIRE a strict system has been put in place to discourage the safari operator from hunting illegally. Once a safari operator has been selected the RDC develops a contract, specifying key terms and conditions to be followed, which the safari operator and the RDC sign and the hunt return form, which is basically the permit to hunt. The contract, which is usually for five years, is binding in law and may take the form of a concession, lease, joint venture or any other arrangement that has been negotiated between the RDC and the safari operator. Some of the terms and conditions specified in the contract could be that the designated safari operator should complete a hunt return form in respect of each client to be returned to the RDC within 30 days after expiry of the permit. The holder should comply with the requirements of the Parks and Wildlife Act (1975, 1982) and regulations and with any relevant RDC by-laws issued with respect to access to the wildlife in the area. There could be conditions on disposal of carcasses or special conditions relating to hunting, for example, that no animals may be shot from a vehicle, no use of aircraft for spotting, no use of spotlights, no hunting at night, etc. Where the sex of the animal has been specified on the permit, the opposite sex of the same species should not be hunted instead. The hunt return form that is signed by the client and professional hunter conducting the hunt and the RDC records the following information for each animal shot: date shot, whether killed or wounded, trophy size, sex, number of hunt days, etc. Safari operators are also asked to provide a hunting schedule of their activities containing information on the names of client and professional hunter, the proposed bag of animals and the time of the hunt so that the RDC ensures that there are no omissions and that the safari operator does not exceed the quota. The RDC game guards have usually been tasked to monitor the trophy hunting activities of the safari operator. The infractions that the safari operators could potentially commit are (i) under-utilising the quota, (ii) hunting the wrong sex or species, (ii) over-harvests, and (iv) using bad harvesting techniques. The RDCs have always been encouraged to incorporate penalties to discourage these infractions in the contract. If infractions occur then the sanctions will be as per the penalties in the contract. The penalties have been usually graduated i.e. depended on the seriousness and context of the offence. Repeated violation of operational rules only dampens chances of renewing the contract in the future and earns the safari operator a bad reputation in the safari industry.

Sanctions can also be applied at the next level. The DNPWLM can reduce the next quota for the RDC that has exceeded its previous quota.

6. ***conflict resolution mechanisms*** – appropriators and their officials should have rapid access to low-cost, local arenas to resolve conflicts among appropriators or between appropriators and officials.

The potential conflicts in CAMPFIRE involve property rights over wildlife, designation of buffer zones for wildlife, the nature of acceptable use of wildlife versus compensation for damages, representation in wildlife committees, distribution of revenues and the nature of acceptable use of revenues. Just as we motivated in our discussion of the prevalence of poaching there are competing property claims to the wildlife resource between the RDC and communities. The RDC derives property claims to the wildlife resource from the AA status that it has been accorded by the Parks and Wildlife Act (1975, 1982). Citing instances of corruption and embezzlement of funds that have been engaged by officials, dissatisfaction about the transparency of the elected councillors' deliberations with local officials through the District Wildlife Committees (Hasler 1999) and exclusion from direct household management and utilisation of wildlife (Murombedzi 1992), communities emphasise the dichotomy between their RDC and themselves in wildlife conservation hence derive property claims to wildlife from traditional heritage, proximity to wildlife and suffering wildlife perpetrated damages. In most areas this is an unresolved conflict, whose only solution lies in the RDCs emulating the good gesture done by the central government and surrendering their AA status to the relevant sub-district communities by means of by-laws.

In some RDCs, for example Chipinge, Hurungwe, Mudzi and Nyaminyami, some land has been designated as unsettled buffer zones for wildlife, conservancy areas, etc. There have usually been conflicts as to who should decide the allocation of land for such and other uses. While the RDCs have usually designated some areas for the benefit of wildlife conservation some traditional leaders such as chiefs and headmen have counter-designated such areas for human settlement. A notable feature of communal lands in

Zimbabwe is that inhabitants do not possess titles to land. The land is communally owned and allocated to households for arable farming and settlement. Historically, allocation of land was the preserve for the chiefs. At independence in 1980, the traditional leader system that had dominated local government during the colonial era was not removed but in terms of the supposedly democratic District Councils Act (1980, 1981, 1982) the traditional leaders' powers of adjudication and land allocation were transferred to the District Councils because it was believed that they were puppets of colonialists having participated in the African Councils[48], almost the equivalent of the present RDCs. Since the passing of the Rural District Councils Act (1988), purported to end the dual system of local government in rural Zimbabwe through amalgamation of the Rural Councils (formerly representing large-scale commercial farming areas) and the District Councils (formerly representing the communal African farming areas) into 57 RDCs, the traditional leaders in the affairs of RDCs have the role of an interest group together with the commercial farmers. Interest groups participate fully (have power to vote and can be voted). In many areas there is conflict between RDCs and chiefs with regards to power over land allocation. Communities as well as modern sub-district institutions such as villages and wards have to a large extent continued to recognize the chiefs' authority over land and other local natural resources (Murombedzi 1992). The government has recognized the indispensability of traditional leaders and enacted the Traditional Leaders Act (2000) that seeks to give the traditional leaders incentives to work in unison with the RDCs.

It is no secret that many of the communities that have received tangible financial benefits from wildlife support the dominant use of wildlife in CAMPFIRE – trophy hunting. Trophy hunting has been carried out on large game such as elephant and buffalo, which are otherwise highly indivisible if household subsistence hunting were to be permitted on it. Besides subsistence hunting would not be sustainable since it would require large off-take if most households are to benefit given the unlikely scenario that hunters would share their hunt. In some RDCs communities used to and still benefit by

[48] They were initially called Native Councils and covered the communal African farming areas. They were subject to central control through a key official, the Native (later District) Commissioner, who was appointed by the central government to be ex-officio President of the Council. The number of African Councils grew over time to 242 by 1980 (Stewart *et al* 1994).

getting protein-rich meat from trophy hunted large game in their neighbourhood almost for free. Where the RDCs have decided to sell the meat at exorbitant prices and from locations removed from the producer communities there have been conflicts over use of the wildlife resource. This has been because that policy favours residents of RDCs who live closer to the business centres where RDC offices are situated and areas of which are the nucleus of urbanisation in the RDC. The residents who will have access and afford the meat in that setting would be those who are not living with wildlife i.e. those in non-CAMPFIRE wards. This conflict has deteriorated further in areas where RDCs do not allocate wildlife for cropping purposes. Communities have insisted that whatever uses of wildlife are approved by RDCs some wildlife should be reserved for direct household utilisation through hunting. RDCs have not been allocating much wildlife for cropping purposes because it earns little income at a time when RDCs are increasingly being called upon to be financially self-sufficient hence would want to generate adequate revenue for their communities, some of which they could retain to run the RDC activities. In fact, the government is requiring RDCs to be financially self-sufficient by decentralising activities without the requisite finance. It is therefore not surprising that most communities who have received CAMPFIRE revenues have been utilising it on social infrastructure such as schools, clinics, roads, bridges, water sources, etc – which should have been provided by the government at some level – centre or local. By allocating more wildlife to uses that generate the most financial rewards, RDCs remove the burden on themselves to provide residents of the producer communities with social infrastructure thereby somehow relax the tight budgets that they are supposed to work with. The low rates of devolution of revenues attest to stretching the incomes that the RDCs have at their disposal. Communities on the other hand would like to have some direct utilisation of wildlife out of necessity – wildlife destroys their crops and livestock and reduce livelihood and food security hence with permissible hunting they could have access to supplementary protein rich meat.

The distribution of revenue has generated a lot of interest at many levels. By virtue of holding the AA the RDCs have the right to sign contracts with and receive financial benefits from safari operators for wildlife utilisation. The RDC has the mandate under the AA to decide how to distribute these revenues to its population. The RDCs have always distributed the revenues to those sub-district units that they have designated as CAMPFIRE villages and wards. In some areas the set of these villages and wards has not been the same over time. Various communities have made representations explaining why they should benefit from the revenues. After the first five years the set of CAMPFIRE villages and wards was almost determined and closed. The RDCs have tended to distribute revenues either equally among all CAMPFIRE villages and wards or on the basis of the site of consumption (killing or capture in the case of translocations) of the animals. Once the revenues are at the village or ward the membership households would decide how to spend them. Expenditure of some revenues has been decided at village levels while some has been decided at the ward level through majority voting, by show of hands. As we alluded to earlier, most villages and wards have opted to use the revenues on community level infrastructure despite the keenness by most young members to receive household cash dividends which they would immediately derive satisfaction from through purchase of consumer goods as opposed to future satisfaction that would be derived from a school or clinic. The community infrastructure featured in most cases either because of influence from the RDC or the realisation that only negligible household cash dividends would be possible from the available revenues. Hasler (1999) notes a case where provincial and local governments in Matabeleland (one of the eight provinces in the country) favour the establishment of local development projects rather than the distribution of household dividends (common in the mid-Zambezi valley) thereby exerting pressure on local communities to vote against the household dividend.

Even though the presence of conflict resolution mechanisms is not a guarantee that appropriators will be able to maintain enduring institutions, it is difficult to imagine how complex systems of rules could be maintained over time without conflict resolution mechanisms (Ostrom 1995). Under CAMPFIRE, the structures are arranged in a hierarchy giving room for conflict resolution of lower structures' disputes by higher structures. The conflict resolution mechanisms are usually quite informal and those who are selected as leaders are implicitly tasked with resolving conflicts. Conflicts that involve the village are resolved by the village wildlife subcommittee while conflicts that involve the village subcommittees and wards are resolved by the ward wildlife subcommittee. The district wildlife subcommittee attends to conflicts involving ward subcommittees and the district at large. The RDC resolves conflicts involving the district subcommittee and the district at large. Inter-district conflicts are resolved by either the DNPWLM or the Department of Administration or provincial political leadership depending on whether they pertain to wildlife or administration or politics. It has been common in some areas to find that conflicts are resolved in any one of the three other channels that are parallel to the CAMPFIRE structures. The initial district administration makes use of VIDCOs at the village, the WADCOs at the ward and the district development committee. Traditionally, the kraal head adjudicates at the village, the headman at the ward and the chief at subsets of the district. Politically, the ruling ZANU PF political party, that has the rural areas as its stronghold, influences the development committees or where it fails to do so uses the cell leadership at the village, branch leadership at the wards and district leadership at the rural district. It has not been uncommon that further conflicts are created while trying to resolve others depending on the route that has been taken and perceived legitimacy of that route. Fortunately, recourse to the courts of law can be taken at any level. So arenas for resolving wildlife related conflicts exist within the realm of CAMPFIRE even though there are a lot of redundancies due to the existence of and close inter-linkage with other routes. Some members of the communities lack information regarding the appropriate route to use in resolving conflicts. Of importance to note is that the existence of many conflicting claims to authority to resolve CAMPFIRE conflicts creates confusion and considerably reduces the efficiency of the conflict resolution mechanisms.

7. ***minimal recognition of rights*** – the rights of appropriators to devise their own institutions should not be challenged by external government authorities.

It has been observed that appropriators in long enduring institutions devise their own rules that are rather informal from a governmental point of view. External government officials should give at least minimal recognition to the legitimacy of such rules if the appropriators are to enforce those rules and enhance sustainability. Ostrom (1995) notes that in many inshore fisheries, local fishermen devise extensive rules defining who can use a fishing ground and what kind of equipment can be used. In the presence of governmental recognition of the legitimacy of such rules, the fishermen enforce the rules themselves. The presumption by external government officials that only they have the authority to make rules makes it difficult for local appropriators to sustain a rule-governed resource over the long run since governmental rules are rarely foolproof and exhaustive. Thus, while large-scale governmental agencies such as the DNPWLM are an essential part of the mix of governance units, if these agencies come to dominate decision making through the imposition of force, be it legal, the effectiveness of local organisations is reduced substantially (Ostrom 1995).

Under CAMPFIRE, the RDC's right to devise rules with respect to the exploitation of the approved quota is recognised as long as they do not seek to overturn the rules already promulgated from the DNPWLM. The RDC has a right to decide how to allocate the approved quota under the different wildlife uses. AA status empowers the RDCs to enter into contracts with private organisations for the exploitation of their wildlife, receive all payments directly and carry out their own problem animal control. Equally well the onus is on them to devise rules that help to carry out their own law enforcement and protection of the resource. Despite the fact that most RDCs have chosen to manage and protect the resource by use of trained and armed game guards the option of enlisting members of the sub-district communities has also been an option available to them. The nature of the potential threat from external commercial poachers has necessitated the engagement of trained and armed game guards. Despite the existence of "guidelines" for disbursement of proceeds from wildlife activities drafted by the DNPWLM, the RDCs have significant breathing space. RDCs decide who benefits and to what extent. The "guidelines" say the RDCs can retain 5% for general council administration and development and 15% to manage CAMPFIRE in the area while a target of 80% of the revenue (at least 50% should be disbursed) should be disbursed to producer communities.

With respect to the resource, the rights of a structure at the upper level are recognised first before the rights of the structure at the lower level in the context of CAMPFIRE. Sub-district communities do not have rights to separately devise rules regarding the resource except through their membership in the RDC. In fact, most RDCs employ game guards who monitor the state of the resources, appropriator behaviour and engage in problem animal control. This has reduced the potentially huge role of communities in resource management to protectors of agricultural activities from wildlife intrusion, victims of wildlife-perpetrated damage, and informants about poaching activities. Communities have been excluded from devising rules regarding access to the resource, magnitude of resource off-take, resource population regulation, acceptable uses of the resource, and resource harvesting technology. However, to a large extent, the sub-district communities have recognised rights to devise rules regarding use of disbursed incomes. Once the RDCs distribute the CAMPFIRE revenue the sub-district communities decide what to do with the money in their respective localities. Whatever revenue disbursement rules they devise as a community will be respected by the RDC. In most RDCs, community decisions regarding use of income tend to be made at the ward as if it were the smallest producer community.

The organisation of CAMPFIRE does not fit the "minimum recognition of rights" criterion very well. The rights are granted to higher levels in the organisation that partially pass them on to the lower levels. Thus higher levels do not respect the independent rights of lower levels. To some extent this is a reflection of the fact that large game such as the elephant roams over very large areas and cannot easily be managed exclusively at the local level. However, it still seems that CAMPFIRE should be more decentralised.

8. ***nested enterprises*** – appropriation, provision, monitoring, enforcement, conflict resolution, and governance activities should be organised in multiple layers of nested enterprises.

Many biological processes occur at small, medium and large-scales such that their effective management require that governance systems are organised in multiple scales that are effectively linked (Ostrom 1995). Exclusive emphasis on simple large-scale institutional arrangements destroys arrangements at the smaller scales, where local knowledge and concerns about natural capital can be applied on a daily basis. Thus the governance system must be as complex as the biological process it is trying to manage.

It is not uncommon to find smaller scale organisations that are nested within larger ones, each with its own distinct set of rules. Wildlife ecosystems are not an exception since it is possible to delineate wildlife ecosystems into multiple scales relating to the territorial or habitat requirements of species. Including many semiautonomous local communities in the regulatory effort allows access and harvesting rules to be matched to local conditions than would a large-scale national organisation that seeks to apply uniform and detailed rules to the entire country which is characterised by immense diversity of local environmental conditions (rainfall, soil types, hydrology, temperature, elevation, scale of plant and animal ecology, etc) (Ostrom 1995). Some wildlife brings about nuisance and damage costs to the local communities. In fact wildlife has two other external effects: (i) the ecosystem effect is such that when you kill a leopard you may get, say, 20 impalas in the following year or when you reduce the number of elephants the grasslands will turn into forests in the next period thereby significantly affecting the ecosystem, and (ii) the stock effect is such that when you kill an animal there will be a lower density hence it could have a negative impact on reproduction or it could become difficult to hunt animals in the next period. The larger the nuisance effect and the smaller the ecosystem and stock effects the more you can give appropriate conservation rights to the local level. Also the inter-temporal benefits that local communities may obtain from sound management of the wildlife resource are potentially greater. Thus the romantic view that national problems should be solved nationally is no longer at the heart of sustainable resource management. Large-scale organisations alone are not the solution because if large-scale units destroy viability of the small-scale units, then organisational failure is likely to be on a much larger scale than organisational failure at a local level (Ostrom 1995). The CAMPFIRE initiative was propagated from the realisation that as long as natural resources, particularly wildlife, remained the property of the state through the DNPWLM then communal landholders would not invest in them as resources thereby threatening their existence.

The similarly romantic view that anything small-scale and local is to be preferred to anything organised at a national or larger level has also been refuted because local participants do not uniformly expend the effort needed to organise and manage these resources, even when given formal authority (Ostrom 1995). Some potential small-scale organisations never form at all and where they have managed to form they suffer from elite capture by both traditional and democratically elected authorities, i.e. they are often dominated by the elite who divert resources to achieve their own goals at the expense of the community. Cases of corruption have been reported in several CAMPFIRE districts. Small-scale local organisations by themselves are rarely the effective form of regulation of resources ranging over very large scales. One argument against devolution is the large-scale and irreversible nature of wildlife ecologies. Devolution gives full ownership of wildlife to the local communities thereby implying the complete power to control the access and use of a resource, and have the capacity to hold the resource for private use or to alienate or destroy the resource (Schlager and Ostrom 1993). Success and failure at a local level are not monitored and no compensatory actions are taken to offset failure at the local level in a devolved resource. Other local organisations possess inadequate scientific knowledge about a resource to complement their own indigenous knowledge (Ostrom 1995). While traditionally commons have been managed successfully, the same cannot be automatic today because the communities that are being tasked with managing the wildlife resources have been divorced from their resources for almost a century and they have not been freely and effectively learning the ecological systems of these resources. For most communities no modern or traditional knowledge exists because of the long-term alienation from the resources. This calls for collaboration between modern organisations with easily and quickly available information about the functioning of ecosystems and the small-scale local appropriators of the resources. This explains why organisations such as the Africa Resources Trust, WWF, Zimbabwe Trust, etc have been involved in CAMPFIRE. More so, the nature of the dominant use of wildlife – trophy hunting – is such that capital-intensive management is required, which communities by themselves would not be able to sustain.

The solution to effective long-term sustainable wildlife resource management under CAMPFIRE is co-management or nested enterprises, where appropriation, provision, monitoring, enforcement, conflict resolution, and governance activities should be organised in multiple layers. Co-management accentuates the different vested interests of stakeholders rather than just communities or just the DNPWLM and thereby acknowledges the multiple jurisdictions that exist in the management of the wildlife resource (Hasler 1999). This recognises that while small-scale and large-scale organisations are not independently sufficient, they each constitute the necessary part of the hierarchical governance needed in wildlife resource management under CAMPFIRE. While there has been much involvement of medium to large-scale organisations the small-scale sub-district local communities have largely been left out from active wildlife conservation under CAMPFIRE. There is a need to increase the contestations of sub-district local communities in wildlife conservation under CAMPFIRE. Involvement of communities naturally entails also taking refuge in the traditional systems of resource management. Diluting the modern local governance systems of resource management with traditional ones will not necessarily entail becoming primitive again. To go forward into the future that preserves high levels of resources, may require going back to traditional systems of resource management (Ostrom 1995). The layers that should be operational and nested according to Hasler (1999) are (i) village and ward, (ii) District, (iv) National, and (v) International. Ideally, the villages and wards should elect district and national representatives so that the power comes from the grassroots instead of from above.

Table 1: Layers and stakeholders in wildlife co-management

Level	**Stakeholders**
International	Donor agencies, CITES, International Wildlife Lobby Groups
National	CAMPFIRE Service Providers[49], Politicians, Civil Servants, Technocrats, Private Sector
District	Local Government, RDC Officials, RDC Committees, Technocrats
Village & Ward	Village and Ward Wildlife Committees, Chiefs, Councillors, VIDCOs, WADCOs, Households

[49] This group of organisations that is responsible for co-ordinating the various inputs, including policy, training, institution building, scientific and sociological research, monitoring and international advocacy comprises of CAMPFIRE Association, DNPWLM, MLGRUD, Zimbabwe Trust, Africa Resource Trust, WWF, ACTION, CASS. Organizations that recently joined the group are The Department of Natural Resources, The Southern Alliance for Indigenous Resources, The Forestry Commision, The Agricultural, Technical and Extension Services Department.

3. Conclusions and policy implications

Our investigation of the extent to which Ostrom's design principles are satisfied by CAMPFIRE shows that they are not wholly satisfied. Of course, CAMPFIRE cannot be expected to fit all the design principles perfectly since it is far from being a typical traditionally analysed common such as irrigation systems, inshore fisheries, mountain grazing lands and forests. Firstly, wildlife is very complex and the most valuable components such as the elephant have very large ranges and cannot be managed exclusively at the local level. Secondly, the most valuable use of the resource (safari hunting) is not directly by the local people but by big game hunters, which introduces several extra layers of complexity – which again cannot easily be handled exclusively by the local or traditional organisations. Even though Ostrom (1990) does not say the satisfaction of these design principles is necessary – a view that we share – we believe that their satisfaction in CAMPFIRE can only help the institutions to make the resource system endure rather than harm it. Our investigation concludes that the large-scale and irreversible nature of wildlife ecologies require co-management for effective long-term sustainable resource management. Co-management recognises that while small-scale and large-scale organisations are not independently sufficient, they each constitute the necessary part of the hierarchical governance needed in wildlife resource management under CAMPFIRE. Most importantly, increased local communities' contestations should be promoted. The potentially beneficial reforms needed in CAMPFIRE consist of specific actions that honour and encourage the formation of institutions satisfying the design principles such as: congruence between clearly defined resource and governance boundaries; congruence between appropriation and provision rules and local conditions; collective choice arrangements and localised monitoring.

Even though we started off by listing an array of problems that need to be resolved in CAMPFIRE, our investigation did not yield a set of specific actions that must be carried out for their resolution. CAMPFIRE is a large-scale programme operating in various areas where different local communities face different circumstances and as such no uniform specific solutions can be suggested. The task of propounding specific actions to resolve the problems of CAMPFIRE has been left for practitioners. The usefulness of this investigation has been to give them the broad guidelines against which they may judge their recommended specific actions. The recommended specific actions should honour the institutional design principles mentioned above. In fact, Ostrom (1995) warns against the proliferation of blueprints. The institutional design principles can only be taught as part of extension programmes with the hope that communities themselves will set in motion mechanisms for adapting them. If adapting the missing design principles helps to culminate into a successful property rights regime in wildlife conservation then sustainability could be yielded in three dimensions: economic, social, and ecological (Hanna and Munasinghe 1995).

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APPENDIX A

Table A1: Rural District Councils' Annual Income from CAMPFIRE Activities (US$)

Year	Sport Hunting	Tourism	PAC Hides & Ivory	Other	TOTAL
1989	326,798	28	5,294	17,690	349,811
1990	453,424	2,865	42,847	57,297	556,433
1991	638,153	15,904	20,859	101,105	776,021
1992	1,154,082	18,951	9,429	34,216	1,216,678
1993	1,394,060	21,095	14,988	53,730	1,483,873
1994	1,553,543	39,985	2,770	46,373	1,642,671
1995	1,476,812	54,866	11,685	48,204	1,591,567
1996	1,656,338	23,275	39,869	36,429	1,755,912
1997	1,708,234	71,258	44,331	13,615	1,837,438
1998	1,787,977	40,871	25,205	37,713	1,891,766
1999	1,940,366	78,709	720,440	14,442	2,753,958
2000	1,919,980	55,668	116,075	13,482	2,105,204
2001	2,142,306	41,439	111,914	32,793	2,328,452
TOTAL	**18,152,074**	**464,915**	**1,165,706**	**507,090**	**20,289,784**

Source: WWF SARPO, Harare

Notes:
1. Sport hunting - income earned from lease and trophy fees paid by safari operators
2. Tourism - income earned from the lease of wild areas for non-consumptive tourism
3. PAC Hides & Ivory - income from the sale of animal products primarily from problem animal control
4. Other - income from the sale of live animals, collection of ostrich eggs and crocodile eggs, etc
5. Mean annual exchange rate based on RBZ end of month exchange rates

Table A2: Allocation of Revenue from CAMPFIRE Activities by Year (US$)

Year	Disbursed to Communities	Wildlife Mgt.	Council Levy	Other	Not Detailed	TOTAL
1989	186,268	81,458	28,404	12,032	41,651	349,811
1990	206,308	121,485	52,530	22,501	153,609	556,433
1991	320,894	219,526	120,444	56,930	56,884	774,678
1992	601,385	207,291	115,398	17,837	274,767	1,216,678
1993	851,732	357,055	251,082	32,172	-14,216	1,477,824
1994	949,138	314,572	148,517	42,514	187,889	1,642,631
1995	946,777	353,772	193,080	26,214	71,723	1,591,565
1996	833,025	405,755	301,091	7,796	191,792	1,739,458
1997	858,357	29,661	26,746	12,415	915,884	1,843,063
1998	910,200	521,373	70,666	82,939	306,589	1,891,766
1999	1,341,853	608,678	253,252	29,477	520,698	2,753,958
2000	1,025,586	320,973	491,411	127,276	139,958	2,105,204
2001	858,869	538,596	454,265	210,388	278,156	2,340,274
TOTAL	**9,890,392**	**4,080,194**	**2,506,885**	**680,491**	**3,125,382**	**20,283,343**

Source: WWF SARPO, Harare

Notes:
1. Disbursed to communities - revenue allocated to sub-district CAMPFIRE institutions
2. Wildlife Management - revenue allocated for wildlife and programme management
3. Council Levy - revenue allocated to district council general account
4. Other - revenue invested in capital development projects and RDC levy to CAMPFIRE Association
5. Amount Not Detailed - revenue not allocated but retained by RDC for general account
6. Mean annual exchange rate based on RBZ end of month exchange rates

RISK MANAGEMENT THROUGH COMMUNITY-BASED WILDLIFE CONSERVATION AND WILDLIFE DAMAGE INSURANCE: THEORETICAL ARGUMENTS

Edwin Muchapondwa[50,51]

Abstract
This paper focuses on risk management in agricultural production. Risk faced by rural farmers in agricultural production could potentially be managed in two ways. Firstly, adding wildlife conservation as a land use in the framework of CAMPFIRE could diversify and consequently reduce risk, particularly where evidence suggests that wildlife conservation is a feasible hedge asset. Risk management through diversification into wildlife conservation could help farmers but it could also help efforts to conserve wildlife. Secondly, establishing a wildlife damage insurance programme would assist farmers, particularly those living in less marginal areas where the benefits of diversification into wildlife conservation are likely to be low. A complement to the insurance programme could be an investment in electric fences and buffer zones to reduce the likelihood and severity of loss. Without detailed empirical investigations we can only speculate that highly marginal and wildlife-abundant districts would benefit more from diversification into wildlife conservation as a risk management strategy while the remaining wildlife-endowed districts would benefit more from the wildlife damage insurance.

JEL Classification: D81, G11, Q29
Keywords: CAMPFIRE, diversification, insurance, risk management

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1. Introduction

A distinction is usually made between risk and uncertainty. Traditionally, the distinction has been that risk arises when the stochastic elements of a decision problem can be characterized in terms of numerical objective probabilities, whereas uncertainty refers to decision settings with random outcomes that lack such objective probabilities. In analyses regarding farmers, current economic analysis of risk is not based on this notion of objective risk, but rather on the decision maker's subjective belief about the occurrence of events (Ellis 1993, Moschini and Hennessy 2000).

It is widely recognized that a high level of uncertainty typifies the lives of rural farmers in developing countries (Ellis 1993). The presence of uncertainty means that more than one outcome is possible and typically not all possible outcomes are equally desirable. The outcome of uncertain events can often make the difference between survival and starvation (Ellis 1993). Uncertainty is more of a problem for agricultural production than for industrial production due to the influence of climate and other natural factors on output and the length of the production cycle. There are four main categories of uncertainty that are relevant from the point of view of agriculture – output, price, technological and policy uncertainties.

Output uncertainty is such that the quantity and quality of output that will result in agriculture from a given bundle of inputs is typically not known with certainty (Moschini and Hennessy 2000). This uncertainty is due to the fact that uncontrollable elements such as weather, pests, disease outbreaks, and wildlife intrusions play a fundamental role in agricultural production. Adverse climate and wildlife intrusions may affect the outcome of planting decisions at any stage from cultivation through to the final harvest. The capacity to combat pests and disease outbreaks may depend on the ability to purchase relevant cash inputs. The effects of these uncontrollable factors are heightened by the fact that time itself plays a particularly important role in agricultural production, because long production lags are dictated by the biological processes that underlie the production of crops and the growth of animals.

Price uncertainty is also a standard attribute of farming activities because the lengthy biological production lag between the decision to plant a crop or to start up a livestock enterprise and the achievement of an output means that market prices at the point of sale are unknown at the time decisions are made (Ellis 1993). Price uncertainty is more relevant to agriculture than to other sectors because of the inherent volatility of agricultural markets. Such volatility may be due to demand fluctuations, which are particularly important when a sizable portion of output is destined for the export market (Moschini and Hennessy 2000). Output uncertainty also contributes to price uncertainty because price needs to adjust to clear the market. In this process some typical features of agricultural markets such as the presence of a large number of competitive producers, relatively homogeneous output, and inelastic demand are responsible for generating considerable price volatility, even for moderate production shocks. On the other hand, these uncertainties tend to be negatively correlated with high prices when total output is low and vice versa. Price uncertainty is the major reason for government intervention in agricultural markets in many countries.

In the long run technological uncertainty, associated with the evolution of production techniques that may make quasi-fixed past investments obsolete, emerges as a marked feature of agricultural production (Moschini and Hennessy 2000). Clearly, the randomness of new knowledge development affects production technologies in all sectors. What makes it perhaps more relevant to agriculture, however, is the fact that technological innovations here are the product of research and development efforts carried out elsewhere, for instance, by firms supplying inputs to agriculture such that competitive farmers are captive players in the process (Moschini and Hennessy 2000).

The agricultural sector is susceptible to the unexpected and uncontrollable acts of state agencies that may change greatly from one moment to the next. This policy uncertainty plays an important role in agriculture. Again, general economic policies have impacts on all sectors through their effects on variables such as taxes, interest rates and exchange rates. Yet, because agriculture in many countries is characterized by an intricate system of government interventions, and because of the need for changing these policy interventions in recent times has remained strong, for example, the emerging concerns about the environmental impacts of agricultural production, this source of uncertainty creates considerable risk for agricultural investments (Moschini and Hennessy 2000).

In Zimbabwean agriculture, all the sources of uncertainty outlined above are well known. Most importantly, the farmers in the communal lands are often victims of drought and wildlife intrusions. While droughts do not occur every year, continuous wildlife intrusions are worrisome in some wildlife-abundant rural areas. For instance, large elephant populations are really very destructive. Outside the national parks, crop damage is a serious problem and local farmers are often killed or seriously injured trying to protect their crops from the marauders. For example, elephants killed 21 people in the Nyaminyami communal lands in 2001.[52] A farmer might wake up one morning to find his maize crop flattened and eaten, or his granary smashed and empty, his irrigation system destroyed, his harvest and his investment gone (WWF 1998). When there is known to be a risk, the farmer and family are likely to stay up all night on guard. They try to scare the animals off by banging pots and pans, lighting flares, and throwing missiles. This is not even always effective since some elephants quickly become used to noise and lights, particularly those bulls that have acquired the habit of raiding crops, just as some big wild cats develop a taste for livestock meat and human flesh (WWF 1998).

Jones (1994) reported a problem of massive livestock deaths in Binga rural district due to predators coming from the adjacent Hwange National Park. A compensation scheme that was put in place in one of the wards paid out for 106 animals in a period of six months in 1992. In the subsequent 16 months seven lions, three leopards and a hyaena were shot as part of problem animal control. Nationally, as much as 300 elephants used to be killed annually as part of problem animal control in Zimbabwe's communal lands (CAMPFIRE Association 2002). Even though data generally does not exist on the extent of the damage suffered from wildlife in Zimbabwe, Kenyan studies show that the typical Maasai Mara wildlife-perpetrated crop damage is between US$200-US$400

[52] The Sunday Mail, 18 August 2002. Most of the victims were killed while trying to chase animals from their fields. The families of the victims were paid Z$15,000 (US$273) from CAMPFIRE revenue as compensation.

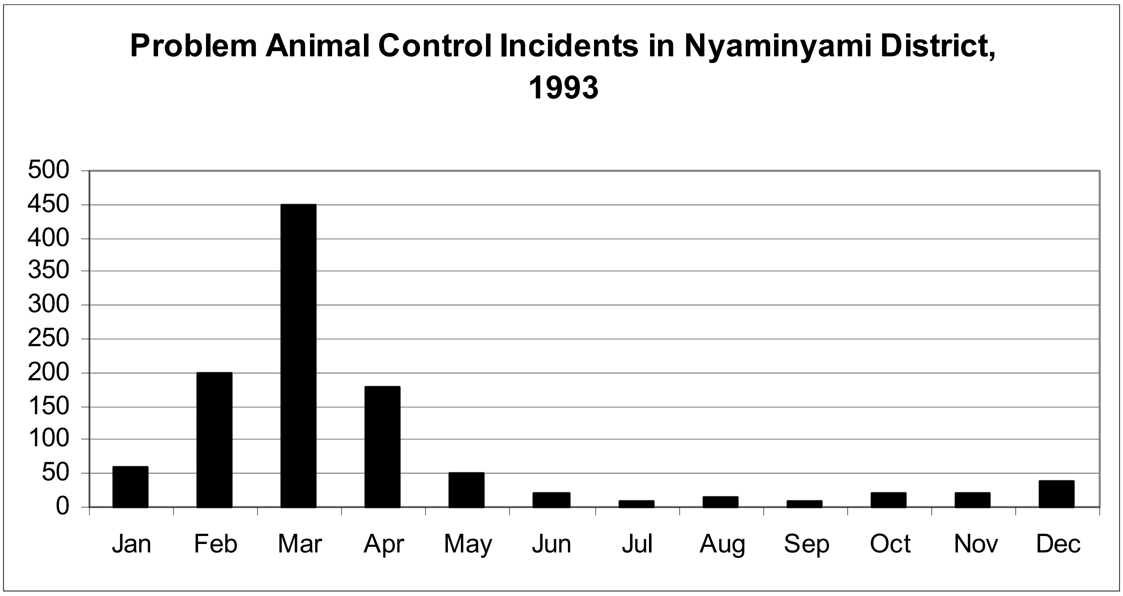
annually per household (Omondi 1994, quoted in Emerton 2001, p218) and Shimba Hills elephant crop damage is US$100 annually per household (PDS 1997, quoted in Emerton 2001, p218), while in Zambia the Mumbwa Game Management Area crop damage is US$122 annually per household (Siachoono 1995, quoted in Emerton 2001, p218). For people living on income of the order of US$1 per day, these are very sizeable losses. Figure 1, overleaf, gives an indication of the severity of wildlife intrusions in a typical Zimbabwean rural district, Nyaminyami. Against this difficult background characterised by serious crop damage, livestock injury and death, human injury and death, the threat to wildlife from indiscriminate killing and often horrible wounding, the communal areas management programme for indigenous resources (CAMPFIRE) was developed to (i) reduce the nuisance perpetrated on rural farmers by wildlife, (ii) give financial benefits to rural farmers through the commercialised use of wildlife, and (iii) protect wildlife through securing the support of the rural communities in wildlife conservation. Under the programme, trophy fees are charged to visiting hunters to hunt game in communal lands under very strict quotas[53]. These fees go directly to the local rural communities, through their administrative authorities called Rural District Councils (RDCs), to be used for social infrastructure and as a further source of income.

Although there are parallels in other production activities, it is fair to say that uncertainty is a typical feature of agricultural production. Some decision settings with random outcomes that traditionally could be classified as uncertain can now be classified as risky since subjective probabilities can be assigned to them and as such some traditional uncertainty can be managed through risk management techniques. Farmers need assistance to deal with risk. This paper sets out the theoretical foundations on how rural farmers in Zimbabwe could manage the risk that they face in agricultural production. We seek to appraise the management of risk by farmers through the use of two modes, (i) diversification into wildlife conservation, and (ii) the purchase of wildlife damage insurance. The rest of the paper is organised as follows: Section 2 appraises diversification into wildlife conservation as a risk management tool. Section 3

[53] It must be noted that to protect the habitat for all wildlife, an annual off-take of most species is necessary given the scarcity of land. The annual quota in the case of elephant, for which a very substantial fee is paid, is less than 0.3% of the population against the background of an annual growth of about 5%.

considers the management of risk through the use of wildlife damage insurance. Section 4 discusses the zoning of risk management strategies while Section 5 concludes.

Figure 1: Problem Animal Incidents in Nyaminyami District, 1993



Source: Jones (1994)

2. Diversification into wildlife conservation as a risk management tool

The purpose of risk management is to control the possible adverse consequences of uncertainty that may arise from production decisions. Risk management activities in general do not seek to increase profits *per se* but rather involve shifting profits from more favorable states of nature to less favorable ones, thus increasing the expected well-being of a risk-averse individual. Of course, agricultural inputs may have both self-insurance and self-protection attributes, where self-insurance arises when a decision alters the magnitude of a loss given that the loss occurs and self-protection takes place when a decision alters the probability that a loss will occur (Ehrlich and Becker 1972). For instance, fertilizer may reduce both the probability and conditional magnitude of a crop nutrient deficiency, and livestock buildings can operate in the same way upon weather-related losses.

Farmers, particularly in developed countries, may have access to other more direct risk management tools such as the possibility of diversifying their portfolio by purchasing hedge assets, with payoffs (i.e. rates of return) negatively correlated with the rate of return on agricultural production. While purchasing hedge assets is an effective way of dealing with risk, more often than not, it is not a feasible option for farmers in developing countries due to high informational and transaction costs. We contend that if farmers in rural Zimbabwe engaged in wildlife conservation that could potentially diversify and consequently reduce the risk they face in agricultural production. The beauty of diversification through wildlife conservation is that it brings about two good attributes: (i) the overall risk faced by rural farmers is reduced, and (ii) a greater area of land is made available for wildlife to allow wild populations to increase. Evidence presented elsewhere suggests that since the 1990s, by making wildlife another form of land use, wildlife has outstripped cattle and crops in terms of economic value in many areas in Zimbabwe, but most importantly in the ecologically fragile marginal lands (Child 1995).

There are five reasons why rural farmers could potentially use wildlife conservation to diversify and consequently reduce the risk they face in agricultural production.

Firstly, it is usually observed, particularly in relation to physical and ecological catastrophes such as drought, that wildlife copes relatively better than either crops or livestock in that wildlife is naturally more tolerant. Wildlife is more drought and disease tolerant. Wildlife is better at utilising local vegetation and therefore gives less erosion than, say, cattle. Child (1995) reports that wildlife makes a more efficient use of forage to produce income than cattle in medium rainfall areas. For a given level of profit, wildlife ventures retain better herbaceous cover, providing better financial and ecological resilience to droughts through increased plant production and reduced variability in available forage (Child 1989).

Secondly, there are ecological and other factors such as spatial heterogeneity that imply that some areas are best suited, or less risky, for wildlife than for livestock and crops. Most of the marginal areas on which most rural farmers practice agriculture are in fact suitable for extensive livestock production and intensive wildlife ranching rather than intensive crop production and livestock rearing which most farmers seem to implement.

Thirdly, there could be ecological interdependence between some species of wildlife and livestock that could reduce risk. Some species of wildlife are browsers, rather than grazers, and therefore do not directly compete with livestock for grazing. The distinction between grazers and browsers is that the former feed on grasses while the latter feed on leaves, stems, flowers, seeds and fruit of trees. Often, foraging by wild herbivores, which tend to be browsers, has only minimal influence on production of domestic livestock, which tend to be grazers. A good example is that of the giraffe, which is a browser and could therefore co-exist with livestock without grazing competition and predation. Ranchers in Africa have taken advantage of the natural partitioning between browsing and grazing herbivores of different sizes in range management and meat production through game ranching.

Fourthly, the uncertainty that affects crops and livestock from instances of wildlife intrusions affects wildlife conservation differently. The variability of rates of return on agricultural production observed as a result of wildlife damage is not observed with respect to wildlife conservation since wildlife is more resistant to damage by itself outside predation relations. Thus, wildlife populations can afford to grow despite some species of wildlife preying on other species. Predation may even have a positive role in wildlife conservation since it selectively removes the weakest individuals from the prey. Crops and livestock populations are seriously negatively affected if they fall prey to some species of wildlife. While it may be conceivable that introducing wildlife alongside agricultural activities may even increase the risk of their destruction it does not necessarily follow. Embarking on wildlife conservation could entail cutting back on agricultural activities and sparing some land to act as buffer zones between agriculture and wildlife, if they are conflict-ridden, thereby insulating agriculture from the risk of wildlife intrusions. It is expected that the benefit from adopting wildlife conservation would be greater than the benefit from the agricultural activities that it displaces.

Fifthly, even though wildlife income is associated with risks, in the sense of variation in income, these risks that emanate from sources such as hunter and tourist boycotts are unlikely to be positively correlated with agricultural pests, agricultural disease outbreaks, drought, price shocks etc, which are usual sources of risk to agricultural income. In cases where there are common sources of risk such as business cycles, inflation, interest rates and exchange rates it is likely that their impacts on the two enterprises are different, with agricultural production being more vulnerable since wildlife incomes depend on external factors given that safari hunters and most tourists are usually rich foreigners who cope relatively better with similar sources of risk in their own countries.

We shall briefly illustrate how the addition of wildlife conservation as an asset to the usual activities of agricultural production of rural farmers could be used to diversify and subsequently reduce risk faced by rural farmers with the help of the portfolio theory, which was propounded by Markowitz (1952). Portfolio theory looks at the performance of a portfolio of assets based on the combination of its components' risk and return. Its goal is to explore how investors, particularly risk-averse ones, construct portfolios in order to optimise market risk against expected returns. To formalise, a risk-averse investor is one who when faced with assets which promise to provide the same return will choose the asset with the lowest risk. Although some farmers can take more risk than others, we assume that farmers are in general risk-averse. We assume risk aversion as fundamental because these farmers are very poor. They will thus not adopt methods that increase average yield if they also increase risks significantly since one bad harvest will mean starvation. We believe that most farmers accept this view from simple introspection. Put in other words, portfolio theory seeks to quantify the benefits of diversification, particularly for risk-averse investors. The logic of diversification is intuitively obvious: "Don't put all your eggs in one basket".

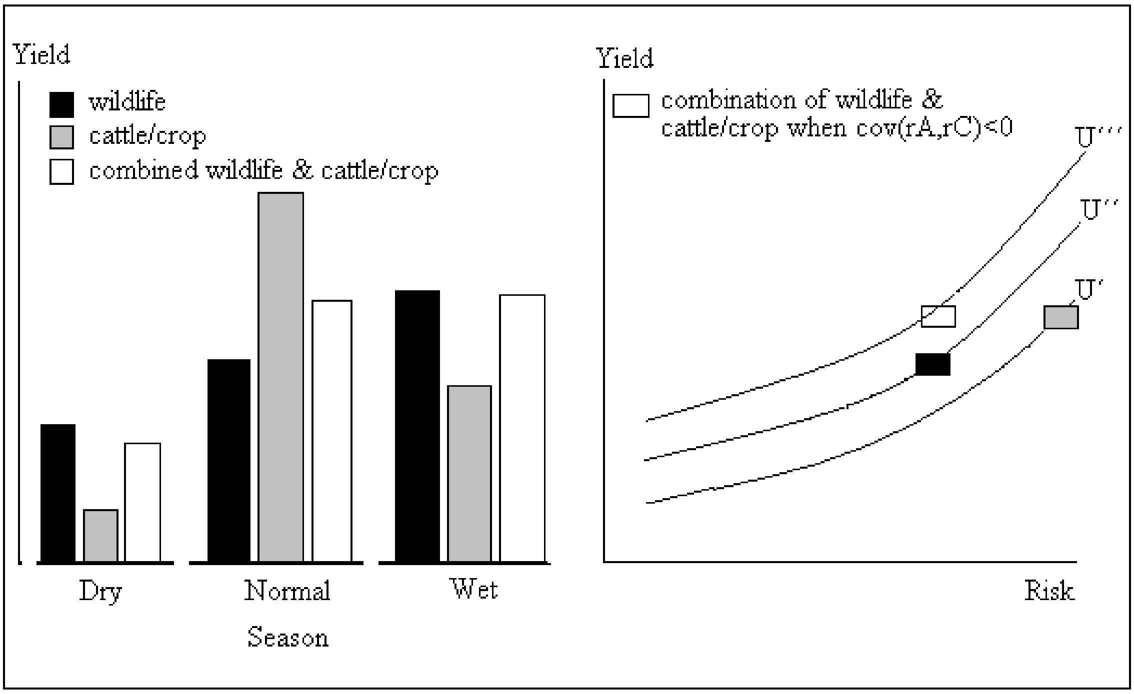
We put forward the contention that rural farmers already have an asset that we will term agricultural production, which for purposes of simplicity is made up of the aggregation of livestock rearing and crop production. Agricultural production does not yield a certain rate of return because of the risk and uncertainty characterised earlier. Under the CAMPFIRE philosophy, rural farmers have the opportunity to acquire wildlife conservation as an additional asset that gives them economic benefits. Like agricultural production, wildlife conservation is characterised by uncertainty but we assume, based on what is usually observed, that the sources of risk in wildlife conservation are not the same as those to which agricultural production is subjected. We suppose that a typical farmer has decided to invest in the two assets with the land fraction w_A in agricultural production, *A*, and the remainder w_C $(=1-w_A)$ in wildlife conservation, *C*. Each activity has a rate of return of r_i and an associated variance of σ^2_i where $i = A,C$. The covariance between the two rates of return is $Cov(r_A,r_C)$. The variance of the farmer's two risky assets portfolio, σ^2_P, would be (Markowitz 1959):

$$\sigma_P^2 = w_A^2\sigma_A^2 + w_C^2\sigma_C^2 + 2w_A w_C Cov(r_A, r_C) \qquad (1)$$

While the expected return of the portfolio would be the weighted average of the expected returns of the individual assets, it is clear that the portfolio variance, and thus the risk faced, would be less than the weighted average of the variances of the individual assets if the covariance were negative.[54] It would be possible for rural farmers to invest in wildlife conservation, and in some cases disinvest in agricultural production, as a way to offset exposure to risk in agricultural production without necessarily reducing the expected return if it could be shown that the rates of return of the two assets are negatively correlated. This is a potentially feasible result considering the earlier discussion of the five reasons why rural farmers could potentially use wildlife conservation to diversify and consequently reduce the risk they face in agricultural production. Figure 2 illustrates the characteristics of wildlife conservation and agricultural production that could reduce portfolio variance and bring about increased welfare for the rural farmers.

[54] Investors' risk preferences can be characterised by their preferences for the various moments of the distribution of the rate of return from an investment. However, the fundamental approximation theorem of Samuelson (1970) states that "when portfolios are revised often enough and prices are continuous the desirability of a portfolio can be measured by its mean and variance alone". By propounding the approximation theorem, Samuelson (1970) provided the justification for the mean-variance analysis by showing that (i) the importance of all moments beyond the variance is much smaller than that of the expected value and variance hence disregarding moments higher than the variance will not affect portfolio choice and (ii) that the variance is as important as the mean to investor welfare.

Figure 2: Rural farmers' benefits of diversification into wildlife conservation



Empirically, historical data on rates of return on agricultural production and wildlife conservation could help us show whether wildlife conservation is the asset for farmers to offset exposure to risk associated with agricultural production. The data that would be appropriate are for all communal lands that are participating in CAMPFIRE since these are the only communal areas with the potential of raising significant wildlife populations. The CAMPFIRE districts constitute about 75% of the districts in the communal lands. It should be noted that the limitations of the wildlife data are that wildlife is a unique resource that does not require the usual cash investment to acquire it. Foregoing opportunities for economically rewarding uses of land within a territory could be interpreted as the most important kind of investment. Damage that people put up with, guarding fields against wildlife intrusions, protecting fields with thorny-bush fences, and looking out for poachers also constitute a kind of investment. Ideally those who undertake a greater proportion of this kind of investment would expect to reap a higher proportion of the benefits. Another limitation of the wildlife data is that the benefits and costs emanating from wildlife are more public than private. Whereas a farmer could invest in agricultural production and the benefits be earned exclusively by him, the same is not true with wildlife. Investment in wildlife usually entails setting aside communal land and sharing the resultant wildlife benefits among all households in the community. This reality requires that the unit of analysis be the community, say the district, rather than the individual farmer.

The data that would be relevant to use for purposes of computing the covariance of rates of return on the two assets would be the rates of return on capital committed to wildlife conservation and agricultural production by each district. Unfortunately such data are not readily available for Zimbabwe's communal lands. Of importance to note is that communal lands entail non-ownership rights to land except usufruct rights. As a result there exists no market for land in these areas such that it is quite difficult to have the value of land invested in each of the enterprises. The division of land in Zimbabwe into five agro-ecological regions makes it difficult to infer the shadow value of communal land from commercial land because of differences in quality. Given that the communal land farmers are predominantly subsistence farmers who use household labour and only sell surpluses to the market, the estimates of profit that exist include returns to labour (salaries and wages). Reliance on existing data that understates capital and overstates profits would yield abnormally high rates of return[55]. Using such flawed data trivialises research on this important subject, which could eventually be a source of huge improvements in the welfare of rural farmers. We have carried out an analysis with the readily available data but refrain from reporting it since the data are not sufficiently reliable. We leave the investigation of the empirical feasibility of using wildlife conservation as a hedge asset to agricultural production as a suggestion for future research.

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[55] The abnormally high rates of return on wildlife conservation may even be depressed because it has not been possible to judge potential wildlife income due to the difficult political conditions in Zimbabwe. It is believed that wildlife income is very sensitive to marketing and political stability. However, another school of thought suggests that the current political uncertainty has not significantly affected wildlife income going to CAMPFIRE since political uncertainty has mostly affected non-hunter tourists while CAMPFIRE derives most of its income from hunter tourists, who are relatively risk tolerant. The potential adverse impacts of political uncertainty on CAMPFIRE have also been harnessed by the fact that most safari operators to whom RDCs sell their hunting quotas are white and they have continuously been able to scout for foreign hunters, who are also predominantly white. For as long as these white safari operators have still been in Zimbabwe, and had to survive on the safari hunting business, they have done their best to encourage foreign hunters to come to Zimbabwe despite the political climate, citing their own continued existence in such a climate as an assurance. The CAMPFIRE revenue for the period 1989-2001 shows that trophy hunting revenue has been increasing steadily throughout the period while tourism revenue has fluctuated (*see table 2 on page 21 in Chapter 1*).

3. Wildlife damage insurance as a risk management tool

3.1 The demand for wildlife damage insurance

If there existed a perfect capital market where farmers could have unfettered access to shares in any claim including wildlife conservation and effectively eliminate diversifiable risks then diversification, as we suggested above, would potentially be a risk management strategy for all rural farmers. However, it so happens that the more marginal areas have a natural comparative advantage in wildlife than the less marginal ones and as such it is expected that some rural areas will be in a position to invest in a substantial proportion of wildlife conservation, should it be economically viable, while the remaining areas might have ecological constraints in doing so. Better still, the more marginal areas could even benefit more if they reinforced risk reduction through wildlife conservation with yet another risk reduction strategy. Still, the problem remains that of trying to find ways in which rural farmers could reduce the risk that they face in agricultural production. Apart from eliminating diversifiable risks through diversification, insurance contracts constitute another way by which rural farmers could cushion themselves against the risk they face in agricultural production, particularly that emanating from wildlife intrusions. Specific insurance contracts are only demanded if diversification by purchase of other assets is costly. More specifically, the necessary condition for a specific insurance demand is that the costs of diversification must exceed the costs of hedging the risk with insurance (Horgby1997).

Even though there are several sources of risk affecting agricultural production, for example drought and wildlife intrusions, some kind of mitigation has traditionally been adopted for other factors. For instance, the propagation of drought resistant and short-season varieties has long been promoted to counter drought. There is climatic-oriented diversity in the agricultural production that farmers undertake, for example sorghum cultivation in the more arid areas. The stalks of the maize crop are usually saved to provide supplementary livestock fodder in the next season in times of drought. We contemplate that the only other major source of risk to agricultural production that has not been internalised and for which possibilities to do so exist in some areas is intrusions by wildlife. The less marginal areas do not have a natural comparative advantage in wildlife and might have ecological constraints in internalising wildlife intrusions risk in agricultural production through wildlife conservation. Indeed, wildlife conservation in such areas, along the lines of CAMPFIRE, could increase the threat of wildlife intrusions and consequently increase the risk faced by the farmers in agricultural production. However, insurance constitutes a potential strategy for managing wildlife intrusions risk in agricultural production in less marginal areas, and perhaps beyond. In this section we seek to appraise the management of risk by farmers through the use of wildlife damage insurance.

Although agricultural insurance markets have existed for a long time in some parts of the world such as the United States and Canada, little coverage against wildlife damage has been available through the private market. In fact, non-subsidized private insurance markets for agricultural risks have been confined mostly to single-peril insurance contracts, where the perils are something other than wildlife damage. Insurance against wildlife damage, as for some other perils, cannot be written for several reasons namely, (i) losses are a virtual certainty in some wildlife abundant areas implying that the costs of monitoring will be high due to incentives of moral hazard, (ii) wildlife damage suffered even by a few farmers can be catastrophic in nature denying the insurer even break-even opportunities, (iii) premiums usually have to be prohibitively high to cover the loss exposure, and (iv) insurers are unable to pool insureds with varying degrees of exposure to wildlife damage because lower risks will not purchase coverage at a pooled rate.

Several possible reasons for individuals' failure to purchase wildlife damage insurance include, (i) that individuals may underestimate the probability that they will suffer wildlife damage as a result of having little or no past experience with the peril (Kunreuther 1984), (ii) that other farmers are unaware that it is feasible to purchase wildlife damage insurance coverage, (iii) that the premium of wildlife damage insurance is too high, even if it might be subsidized, (iv) the limited liability provided by government programmes of disaster relief, (v) the charity hazard, which is the tendency of an individual at risk not to procure insurance as a result of reliance on expected charity from others (Browne and Hoyt 2000), and (vi) the unwillingness to accept the responsibility of financing compensation for damages caused by wildlife that is perceived to be state property – the state should accept such responsibility and pay the premium instead. Examples of state-run compensation schemes are the wolf damage compensation schemes in Croatia, Finland, France, Poland, and Slovania. In Norway and Sweden the counties are responsible for paying compensation for livestock deaths while the indigenous Saami Assembly pays compensation in the Swedish reindeer areas[56] from an annual state grant.

3.2 Wildlife damage insurance in the presence of symmetric information

Suppose a typical risk-averse farmer initially has wealth with a monetary value of Ω. The wealth of the farmer could be in the form of livestock and crops. If there is some probability p that he will lose part of the wealth valued at Z possibly due to the fact that his livestock and crops will be destroyed by wildlife then he would want to purchase insurance that will reimburse him by indemnity q dollars in the event that he incurs the loss. The loss Z is assumed to be lower than the wealth Ω. The amount of money that he has to pay for q dollars of insurance is rq where r is the premium per dollar of coverage.

If a risk-averse farmer cannot influence the probability of loss then he will completely insure himself against the loss Z. If the farmer's actions do affect the probability of loss, a risk neutral insurer may only want to offer partial coverage so that the farmer will still have an incentive to be careful. The farmer would be considered to be careful if he fences his fields and cattle pens or takes some such other preventive measures that reduce the risk of wildlife intruding into his livestock or crop activities. There are basically two kinds of problems that could hinder the full insurance coverage result: (i) adverse selection, and (ii) moral hazard. These problems centre on the different levels of access to information or asymmetric information that the insurer and the insured might have.

[56] For more details, see http://www.wolf.org/wolves/learn/scientific/symposium/status.asp

3.3 The adverse selection problem in wildlife damage insurance

When considering a risk, insurance companies may observe certain parameters of the decision environment such as geographic location, soil type, and yield history. It is often infeasible to observe all relevant facts, however, and even if observable it may be impossible to write an insurance contract based upon these observations. When contracts based upon relevant environmental parameters are infeasible then adverse selection problems may arise. Adverse selection may be either spatial or temporal in nature. In a market characterized by adverse selection there are two types of insureds with different probabilities of loss, high risks and low risks. On the one hand, the high risks estimate that their probability of loss exceeds the insurance company's estimate. On the other hand, the low risks perceive that their probability of loss is less than that estimated for them by the insurance company. Low risks will purchase less insurance in a market with adverse selection than in a market free of adverse selection. The reason why the insureds and insurers are arriving at different risk cohorts is that they are not equally informed about the nature of the risk being insured. Thus when, unlike the producer, the insurer is not completely informed about the nature of the risk being insured, then the insurer faces the problem of adverse selection. We will use the example given by Moschini and Hennessy (2000) to illustrate the adverse selection problem.

Suppose a risk-neutral insurer has categorized three farms of equal size, say one acre, owned by different farmers into the same risk cohort. We denote the farms by A, B and C. Suppose further that all farms realize two outcomes, each with the probability 0.5, but the realizations for farm A are $\{y^*-10, y^*+10\}$, those for B are $\{y^*-20, y^*+20\}$, and those for C are $\{y^*-30, y^*+30\}$. Despite the associated yield distributions differing if the only information available to the insurer was the average yield $\$y^*$, the insurer would not observe any difference among these three farms since they have a common average yield. Suppose the insurer compensates farmers with the shortfall from the average yield in a 'poor' harvest and nothing in the 'good' harvest then the expected payouts would be \$5, \$10 and \$15 respectively for farms A, B and C. To clarify, farmer A expects to get a compensation of \$10 in a 'bad' year with probability 0.5 while he expects to get \$0 in a 'good' year with the same probability - this results in an expected payout of

0.5(10)+0.5(0)=5. Assuming full participation, the actuarially fair premium for a contract covering all three risks would be \$10 per farm (or acre). The degree of homogeneity required to sustain the contract depends upon, among other things, the degree of risk aversion expressed by farmers. The more risk averse the farmers, the more tolerant they will be of actuarially unfair rates. In the presence of an actuarially fair premium of \$10, if farmer A is insufficiently risk averse, then he may conclude that the loss ratio for farm A, at 5/10=0.5, is too low and may not insure the farm. If the insurer continues to charge \$10 per acre when covering only farms B and C, then an average loss of \$2.50 per acre is incurred. On the other hand, if the premium is raised to \$12.50 per acre so that a loss is avoided, then acre B may not be insured. Thus, the market may disentangle in stages.

In the case of elephant damages, adverse selection could potentially result from farmers possessing local knowledge about their fields' proximity to the migration routes of the elephant. Even though the insurance agent may have knowledge about the proximity of these farmers' fields to an elephant reserve she may not observe proximity to migration routes, which may also be of crucial importance in the determination of the risk of elephant intrusion. Avoiding adverse selection may require the successful crop insurance program to identify, acquire, and skillfully use data that discriminate among different risks. Insurers have developed mechanisms such as risk categorization, experience rating and partial insurance coverage in order to reduce adverse selection (Horgby 1997). Categorization can be achieved by using many verifiable variables and using information related to past experience of the insured motivates them to reveal their true risk. In the case of wildlife damage insurance there is no likelihood of adverse selection because damage depends mainly on wildlife density, which is observable, and not on the individual's unobservable characteristics.

3.4 The moral hazard problem in wildlife damage insurance

Moral hazard expresses an insurance-induced change in the policy-holders' behaviour and has been subject of intense discussions in insurance theory (Horgby 1997). The moral hazard being that purchasers of insurance will not take an appropriate level of care. Any decision made after an insurance contract is signed that affects the probability or size of the insured's loss is subject to inefficiencies due to moral hazard. This type of principal-agent problem is also known in literature as the hidden action incentive problem, since the action of the agent is not perfectly observed by the principal. Wildlife damages usually take place before harvesting. In some cases there is a chance of recovery of the damaged crop or injured livestock. Effective damage assessment would ideally be conducted at harvest time. Farmers could avert the impact of wildlife intrusions by supporting the damaged crop and livestock to ensure recovery. Instead they may not opt to do so hence in wildlife damage insurance four cases of moral hazard can be singled out: (i) not taking additional *ad hoc* measures such as burning fires, beating drums, etc to prevent wildlife intrusions, (ii) not chasing away the wildlife when it has already intruded, (iii) not supporting the damaged agricultural produce to ensure recovery, and (iv) choosing crops that are attractive to wildlife.

To illustrate the consequences of an insurance market characterized by moral hazard, suppose that there are many identical farmers who are contemplating buying insurance against destruction of agricultural activities due to wildlife intrusions. A typical farmer's income is basically composed of incomes from agricultural production and community based wildlife conservation. Here we are thinking of those rural areas that are implementing the CAMPFIRE philosophy, where local communities are being conferred, through their RDCs, (a) greater control over formerly public wildlife in communal areas in defined territories, (b) enhanced capacities to add value to local wildlife, and (c) specific financial rewards likened to alleged conservation value of wildlife within their territories (Gadgil and Rao 1995). As pointed out earlier, local communities sell hunting licences to visiting hunters to hunt game in their territories and collect fees from tourists viewing game. The fees go directly to the local communities, who themselves make decisions about how to distribute and share the incomes. The farmer's total income depends on the share of wildlife income reinvested into conservation efforts, ω, and the allocation of labour between agricultural activities (L^A) and mitigation of wildlife intrusions (L^M), given that the total available labour is fully utilized between the two alternatives i.e. $L^A + L^M = \mathrm{L}$. The biomass of wildlife, W, is directly related to the share of wildlife income reinvested into conservation efforts, ω.

If a typical farmer's wealth, which is in the form of agricultural production, is destroyed he suffers a loss of Z dollars. We assume that the probability of wealth destruction depends on the farmer's actions, where the farmer can take mitigatory measures against wildlife intrusions by use of labour in fencing his fields and cattle pens with thorny-bushes or taking other preventive measures like burning fires, beating drums, etc that reduce the risk of wildlife intruding into his livestock or crop activities. Thus, among other things, the use of labour in wildlife intrusions mitigation activities reduces the probability of loss of the farmer's wealth. The farmer has to decide how much labour to allocate between agricultural activities and mitigation of wildlife intrusions. Due to the perfect substitutability of the two types of labour, it suffices for the farmer to decide how much labour to put in one use and the other use would utilise the residual labour. On the one hand, agricultural income is positively related to the labour allocated to agricultural activities, L^A, but, on the other hand, negatively affected by higher biomass of wildlife and lower mitigation activities. The community decides the share of wildlife income to reinvest in wildlife conservation, ω. As pointed out above, the biomass of wildlife responds positively to ω while more biomass of wildlife increases the probability of loss of the farmer's wealth. To recap, the probability of loss of the farmer's wealth is a function of labour allocated in mitigation activities, L^M, and the biomass of wildlife, W.

When the farmer has already signed an insurance contract he has incentives to cheat in deciding the optimal level of the decision variable, L^M, given ω. The farmer would prefer to allocate low levels of labour to mitigation activities, thereby increasing the probability of loss, since: (i) there is a cost of taking the preventive measures against wildlife intrusions, (ii) he can make more use of labour in agricultural production given the substitutability and trade-offs in labour, (iii) it is generally dangerous to guard against wildlife intrusions given that some species of wild animals are rogue, (iv) most of the wildlife intrusions are likely to take place during the night, and (v) those who are not efficient in agricultural production would not suffer a huge loss anyway.

In the presence of moral hazard, the insurer will be faced with the challenge of maximising achievable profits while creating adequate incentives for the insureds to take preventive measures. Among other things, the insurer will be governed in the selection of premiums by the need for the farmer to receive at least his reservation utility from another possible opportunity available to him since one possible action is not to participate in the insurance market – this is the participation constraint. Equally important is that the farmer should find it optimal to choose to take preventive measures against wildlife intrusions – the incentive compatibility constraint. Standard results show that if there is no incentive problem so that the probability of the destruction occurring is independent of the actions of the farmer and if competition in the insurance industry forces expected profits to zero, the optimal solution will be such that the insurer will fully compensate the farmer such that his wealth stays the same regardless of whether or not the destruction occurs (for example, see textbooks such as Varian (1992) and Mas-Colell, *et al* (1995)).

If on the other hand there is an incentive problem so that the probability of the destruction occurring is dependent on the actions of the farmer, there will no longer be full insurance coverage. Thus, when the probability of loss depends on the actions of the farmer, as in a situation characterized by moral hazard, full insurance will no longer be optimal. In general, the principal wants to make the agent's consumption depend on the choices he himself makes so as to leave him the incentive to take proper care. In this case the farmer's demand for insurance will be rationed. The farmer would like to buy more insurance at actuarially fair rates, but the industry will not offer such contracts since that would induce the consumer to take inadequate level of care. Thus moral hazard causes the failure of full insurance coverage. Under moral hazard it is not optimal for a risk-neutral insurer to assume all risk. Some residual risk must be borne by the risk-averse farmer. In some markets, moral hazard can actually cause the market to disappear. One solution to support the creation of insurance in the presence of moral hazard would be to have an insurance policy with a deductible i.e. the farmer pays part of the loss, particularly the initial D after which the insurer will pay the rest up to the loss value, Z. Deductibles encourage the farmer to self-protect (Chambers 1989). If some Z exists for which indemnity is zero, farmers using poor wildlife damage preventive practices bear all the risk in these loss states. Meyer and Ormiston (1999) have shown that there exists an optimal level of deductible, D^*, that maximises the utility of farmers. Thus implementing the insurance scheme with a deductible potentially solves the moral hazard problem.

3.5 Which way forward with the Wildlife Damage Insurance?

We have stated the need for wildlife damage insurance and mentioned that it may be theoretically feasible to set up such insurance in a competitive setting, with the moral hazard problem being guarded against by the deductible. The natural extension of this work would be an investigation of the empirical extent of the wildlife damage insurance market. On the demand side, this could potentially be done with the help of contingent valuation data on willingness to pay for insurance covering specified damage[57]. The mean willingness to pay for insurance would enable the assessment of whether there is sufficient demand for private market wildlife damage insurance at a given cost of insurance. On the supply side, the expected cost of insurance could be estimated from data on the probabilities of households suffering wildlife damage and the average value of damage from wildlife. Such data have hardly been collated in Zimbabwe. The comparison of the mean willingness to pay and the expected cost of insurance would indicate whether or not a private insurance scheme could potentially be feasible. Basically, if the willingness to pay for a good is lower than its cost, economics would predict that there would be no exchange. If there existed some justification for it, a subsidy would be required from the government for a private insurance scheme to operate. A subsidy could be justified for moral or conservation arguments. Wildlife is a public good and it generates benefits for the whole country but the poor people living adjacent to it bear the costs of living with it – compensation might be in order. The negative externality to the poor might be an argument for a solidarity or collective element in the insurance system. The extent of the subsidy would be the difference in the cost of insurance and the mean willingness to pay. An investigation of the

[57] A typical question to elicit willingness to pay for specified insurance coverage is: The government is considering putting up a wildlife damage insurance scheme for your area. All other compensation channels that may have existed before with respect to wildlife intrusions will therefore cease to exist. The wildlife damage insurance scheme will be such that your household will be compensated for all crop and livestock losses it will suffer from wildlife. Only those households that are willing to pay a premium will be part of the scheme i.e. any household that does not pay the premium will not be compensated for any wildlife perpetrated damage or loss. What maximum amount in yearly premiums would your household be willing to pay for it to participate in such a scheme? For what coverage?

determinants of the willingness to pay values will help the government to discriminate in the distribution of insurance subsidies.

The existence of agricultural insurance markets has depended crucially on government support and governments have often subsidized or even run these insurance markets. The apparent failure of the private insurance market has been the rationale for government sponsored insurance programmes. Given the susceptibility of agricultural production to wildlife damage there is obviously a latent need for wildlife damage insurance. Given the large number of rural farmers suffering wildlife perpetrated damages, which are external effects from government sponsored conservation efforts, and the importance that has tended to be attributed to rural farmers, in general, by the Zimbabwean government in the development of the country there is a need to assist their situation. Private insurance may fail to form due to lack of knowledge or organization. If some low-cost organization could be formed then it might become optimal in the long run. It might even be profitable when it passes a certain threshold. The state could help start such a small-scale insurance organization with the hope of reproducing successful results similar to those posted by the informal and poor people oriented Grameen Bank in Bangladesh (Grameen Communications 1998, quoted in Sterner (2003), p401)[58]. However, if the private insurance market never forms, government-sponsored wildlife damage insurance could be established to provide farmers subsidized insurance and to reduce the exposure to wildlife damage through land use limits and other control measures.

We note here that the private insurance companies still have a role to play in insuring wildlife damages. The private insurance companies – formal and informal – could insure wildlife damages perpetrated against rural farmers while it is the government that pays the premium, or part of it, for the rural farmers. Wildlife damage insurance would be made available if the community agrees to adopt and enforce damage mitigation and land use measures. These measures could include traditional fencing of agricultural

[58] Grameen Bank emerged in 1976 out of an action research project on the design of a credit delivery system to provide banking services to the rural poor. After seven years of operation the project culminated into an independent bank, which is mainly owned by the borrowers (90% of the shares) and the government (10% of the shares). The bank has provided loans to more than 2 million people and operates in 35,000 villages countrywide, where it actively seeks to do transactions with beggars, illiterates, widows, and other most deprived people and still succeeds to attain a 99% repayment rate. Operating without offices, phones or faxes but villages meetings, peer pressure and peer support has effectively replaced collateral.

activities, planting agricultural crops that are not vulnerable to damage from certain species of wildlife, implementation of effective problem animal control, creation of agriculture and wildlife buffer zones, and the maintenance of designated wildlife composition. As happens in the US with the flood insurance programme (Browne and Hoyt 2000), wildlife damage insurance could be divided into two phases, emergency and regular. Under the emergency phase, a wildlife abundance map could be provided and farmers would be allowed to purchase limited amounts of insurance at subsidized rates. Once the map has been drawn that divides communities into specific zones with the probability of being inflicted by wildlife damage determined for each zone, and the community has agreed to adopt more stringent mitigation and land use measures, it would be allowed to enter the regular phase of the programme (Rejda 1998, quoted in Browne and Hoyt 2000). The subsidized insurance would only be made available to those communities that adopt permanent land use measures and wildlife damage control programmes. It is also important that such subsidized insurance is not extended to unsanctioned new settlements. The foregoing would be an alternative way to guard against the moral hazard problem we discussed before, if a deductible cannot easily be incorporated into the insurance.

A complement to insurance is an investment in reducing the likelihood or severity of loss. In limited cases, electric fences could be erected to separate agricultural production and wildlife activities. This is not always possible to do given the haphazard nature of settlements in some rural areas and due to the concern by some wildlife conservationists who say that fences adversely affect wildlife emotionally and physically. There have been cases of wildlife dying along the fences as their traditional routes had been blocked. Other species do not learn from the electrocuting experiences of others and they continuously try to make their way through the fences resulting in frequent breakages and need for huge maintenance expenses.

4. Zoning of risk management strategies

Without detailed empirical investigations we can only speculate that highly marginal and wildlife-abundant districts such as Binga, Nyaminyami, Guruve, Hurungwe, Gokwe North, Hwange, Tsholotsho, Chipinge, Beitbridge, Bulilimamangwe, Chiredzi, and Muzarabani (*see map on page 24 in Chapter 1*) would benefit more from diversification into wildlife conservation as a risk management strategy while the remaining wildlife-endowed districts would benefit more from the wildlife damage insurance.

Figure 3: Zoning of risk management strategies

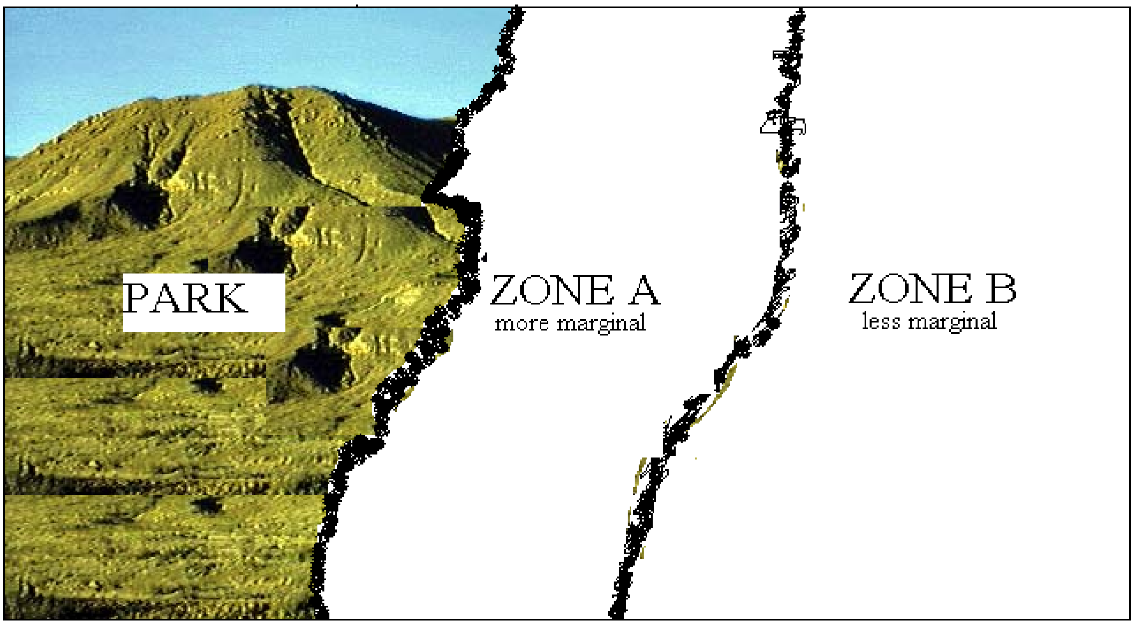

Figure 3 above illustrates our reasoning. In the more marginal Zone A, risks of drought and wildlife damage are both high. Damage is frequent and thus hard to monitor because the high incidence of damage creates incentives for moral hazard by the inhabitants. Individual insurance may not be viable but CAMPFIRE income is large. CAMPFIRE income serves to reduce total portfolio risk since there are more opportunities for giving up some land for wildlife conservation. It is most likely that it is more attractive to give up land for wildlife conservation since it is likely to be more profitable than marginal agriculture, given the low land quality, high risk of drought and high risk of intrusion. While it may be conceivable that introducing wildlife alongside agricultural activities may even increase the risk of their destruction it does not necessarily follow. As we said earlier, embarking on wildlife conservation could entail cutting back on agricultural activities and sparing some land to act as buffer zones between agriculture and wildlife, if they are conflict-ridden, thereby insulating agriculture from the risk of wildlife intrusions. In the next paragraph we motivate the argument for the wildlife damage insurance in Zone A in cases where introducing wildlife alongside agricultural activities may even increase the risk of their destruction, possibly because the buffer zones between agriculture and wildlife, if they are conflict-ridden, are unsuitable or insufficient. In Zone B, CAMPFIRE income is low owing to the relatively lower density of wildlife, which nevertheless perpetrates damage. It would not be of much use if CAMPFIRE income is divided evenly. Instead it could be used to support wildlife damage insurance. Damage is very rare and could for that reason be monitored more easily. Thus the wildlife damage insurance, rather than diversification through wildlife conservation, is likely to be the feasible risk management strategy for Zone B.

If adequate damage monitoring mechanisms existed in Zone A then the wildlife damage insurance would be yet another risk reduction strategy at its disposal. This is particularly true where introducing wildlife alongside agricultural activities may even increase the risk of their destruction, possibly because the buffer zones between agriculture and wildlife, if they are conflict-ridden, are unsuitable or insufficient. Even though wildlife may increase the risk of intrusions, it may also continue to be a hedge asset against drought and consequently reduce risk somewhat. Thus the combination of diversification through wildlife conservation and the wildlife damage insurance could benefit Zone A even more.

Figure 4: Risks associated with agriculture, wildlife conservation and wildlife insurance

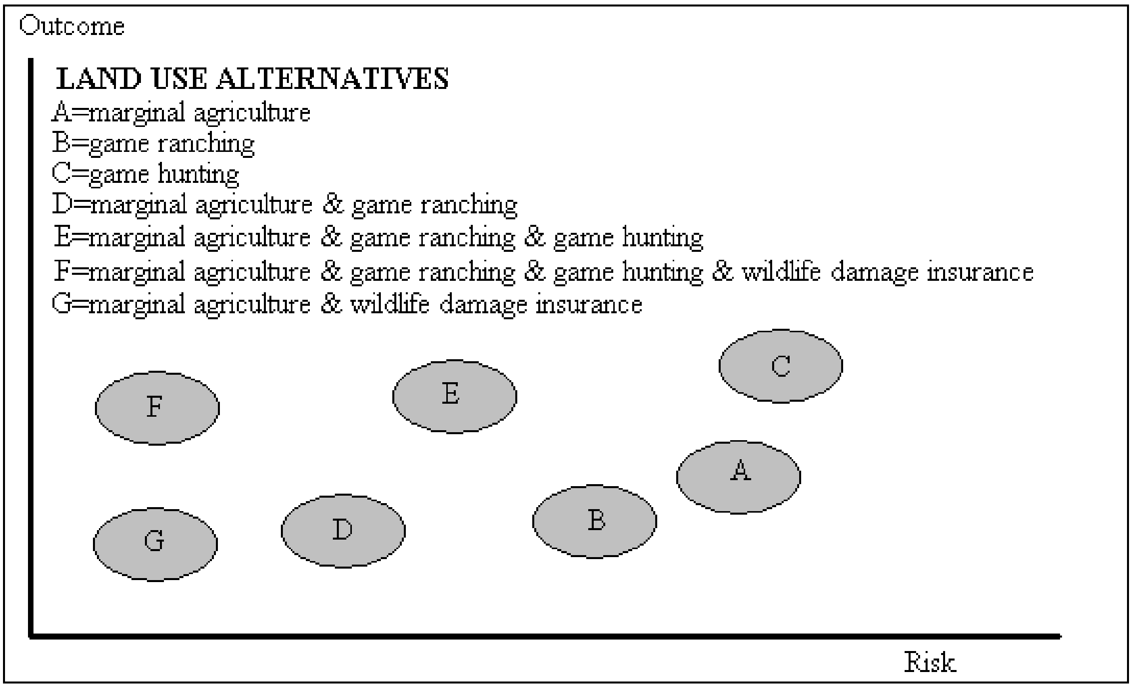

Figure 4 above shows that Zone A may initially benefit from diversification through wildlife conservation by moving from land-use alternative A to E and subsequently benefit from the wildlife damage insurance by moving from land-use alternative E to F. Zone B benefits from the wildlife damage insurance by moving from land-use alternative A to G. Land-use alternatives E, F and G depict lower levels of risk than the starting land-use alternative A.

5. Conclusion

This paper focused on risk in agricultural production. Rural farmers' production activities are characterised by uncertainty due to unpredictable climatic conditions and wildlife intrusions into agricultural production, which is particularly serious in areas with high wildlife populations. Risk faced by rural farmers in agricultural production could potentially be managed in two ways. Firstly, adding wildlife conservation as a land-use in the framework of CAMPFIRE could diversify and consequently reduce risk, particularly where evidence suggests that wildlife conservation is a feasible hedge asset. Risk management through diversification into wildlife conservation could help farmers deal with risks such as the risk of drought but it could also help efforts to conserve wildlife. Naturally this strategy does nothing to reduce the risk of wildlife damage, which is something the communities living adjacent to the game reserves have to learn to live with. Secondly, establishing a wildlife damage insurance programme would assist farmers, particularly those living in less marginal areas where the benefits of diversification into wildlife conservation could be low, to cushion themselves against losses from wildlife intrusions that are in any case believed to be a significant source of risk. To guard against the potential problem of moral hazard, an insurance scheme with a deductible could be introduced to entice farmers to adopt reasonable and effective mitigation measures. Alternatively, wildlife damage insurance could be divided into two phases, emergency and regular. Once the map has been drawn that divides communities into specific zones with the probability of being inflicted by wildlife damage determined for each zone, and the community has agreed to adopt more stringent mitigation and land use measures, it would be allowed to enter the regular phase of the programme. A complement to the insurance programme could be an investment in electric fences and buffer zones to reduce the likelihood and severity of loss. Without detailed empirical investigations we can only speculate that highly marginal and wildlife-abundant districts such as Binga, Nyaminyami, Guruve, Hurungwe, Gokwe North, Hwange, Tsholotsho, Chipinge, Beitbridge, Bulilimamangwe, Chiredzi, and Muzarabani would benefit more from diversification into wildlife conservation as a risk management strategy while the remaining wildlife-endowed districts would benefit more from the wildlife damage insurance.

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