International Journal of the Commons
Vol. 12, no 1 2018, pp. 134–169
Publisher: Uopen Journals
URL:http://www.thecommonsjournal.org
DOI: 10.18352/ijc.803

ISSN: 1875-0281

The role of institutions in community wildlife conservation in Zimbabwe

Herbert Ntuli
School of Economics, University of Cape Town, South Africa
ntlher001@myuct.ac.za

Edwin Muchapondwa
School of Economics, University of Cape Town, South Africa; Department of Business Administration, Technology and Social Sciences, Luleå University of Technology, Sweden
edwin.muchapondwa@uct.ac.za

Abstract: Institutions play a significant role in stabilising large-scale cooperation in common pool resource management. Without restrictions to govern human behaviour, most natural resources are vulnerable to overexploitation. This study used a sample size of 336 households and community-level data from 30 communities around Gonarezhou National Park in Zimbabwe, to analyse the relationship between institutions and biodiversity outcomes in community-based wildlife conservation. Our results suggest a much stronger effect of institutions on biodiversity outcomes via the intermediacy of cooperation. Overall, the performance of most communities was below the desired level of institutional attributes that matter for conservation. Good institutions are an important ingredient for cooperation in the respective communities. Disaggregating the metric measure of institutions into its components shows that governance, monitoring and enforcement are more important for increased cooperation, while fairness of institutions seems to work against cooperation. Cooperation increases with trust and group size, and is also higher in communities that have endogenised punishment as opposed to communities that still rely on external enforcement of rules and regulations. Cooperation declines as we move from communal areas into the resettlement schemes and with increasing size of the resource system. A very strong positive relationship exists between cooperation and biodiversity outcomes implying that communities with elevated levels of cooperation are associated with a healthy wildlife population. Biodiversity outcomes are more successful in communities that either received wildlife management training, share information or those that are located far away from urban areas and are not very close to the boundary of the game park. Erecting an electric fence, the household head's age, the number of years in school and number of years living in the area negatively affect biodiversity outcomes. One policy implication of this study is to increase autonomy in CAMPFIRE communities so that they are able to invest in good institutions, which allows them to self-organise and to manage wildlife sustainably.

Keywords: Common pool resources, conservation, institutions, self-organisation, wildlife

Acknowledgements: We thank the CAMPFIRE communities for their willingness to participate in the surveys. Funding from ERSA (Economic Research Southern Africa), AERC (African Economic Research Consortium), FORMAS through its COMMONS program, Sida (Swedish International Development Cooperation Agency) through its Environment for Development program and People Programme (Marie Curie Actions) of the European Union's Seventh Framework Programme FP7/2007-2013/under REA grant agreement number 609642 is gratefully acknowledged. Last but not least, we thank anonymous referees for their useful comments. An earlier version of this manuscript is published as an EfD working paper.

1. Introduction

Wildlife conservation has become popular as a vehicle for rural development with policymakers in Southern Africa. A considerable proportion of wildlife outside protected areas interacts with adjacent local communities. Experience shows that rural households are not given full authority (ownership and responsibility) over wildlife on communal land or in the neighbouring protected areas (Murphree 1991; Jones and Murphree 2001; Taylor 2009). As a result, local communities lack ownership and adequate incentives to manage wildlife sustainably, except when attached to a conservation or tourism contract, e.g. Kunene and Caprivi regions of Namibia (Ashley 2000), Makuleke in South Africa and a few existing CAMPFIRE[1] communities such as Mahenye in Zimbabwe (Murphree 2001; Frost and Bond 2008). Lack of ownership, responsibility and incentives affects

[1] The Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) is a benefit-sharing scheme that involves local communities that live in the vicinity of national game parks and suffer wildlife intrusions (Murphree 1991; Murombedzi 1999). The programme was instituted by the government of Zimbabwe during the mid-1980s in order to create incentives to conserve wildlife by directly transferring benefits from conservation to the local communities (Balint and Mashinya 2006). However, poaching still remains the most threat to wildlife conservation in the country. Since its inception, CAMPFIRE enjoyed limited success as poaching initially subsided as beneficiaries started receiving wildlife income and increased thereafter (Muchapondwa 2003). Many of the challenges faced by CAMPFIRE during the 1980s continue even up to today.

the likelihood of households deciding to collaborate and develop rules for managing the resource. The situation is made even worse by the fugitive character of the resource, which makes it difficult to assign property rights to wildlife (Muir-Leresche and Nelson 2000). For example, it is difficult to claim ownership to wildlife because it freely moves across boundaries of farms, communities or geographical areas with different jurisdictions. Furthermore, wildlife is quite different to other common pool resources (CPRs) found in the region such as water, forests and rangelands because it is often not a resource that individual households heavily depend upon, whereas the others often are.

Wildlife management challenges in developing countries have resulted in a search for policy options in an effort to make the human-environment or social-ecological systems (SESs) sustainable over time (Murombedzi 2000). By definition, SESs are nested, multilevel systems that provide essential services to society such as supply of food, fibre, energy, and drinking water (Berkes and Folke 1998). Following the publication of Hardin (1968), both state control and private ownership were embraced by colonial governments in the region as panaceas for all environmental problems. While such policies have benefited minority groups and persisted even after independence, the colonial legacy resulted in tension between wildlife authorities and local communities living adjacent to national parks and overexploitation of wildlife resources (Murombedzi 1999; Songorwa 1999). This realisation led policymakers in some countries such as Namibia, Zambia and Zimbabwe to initiate a transition from the conventional top-down governance systems to people-oriented approaches such as Communal Conservancies, Administrative Management Design (ADMADE) and CAMPFIRE. In Zimbabwe, the achievements of CAMPFIRE dwindled due to political and economic factors from early 2000 and the fact that benefits were never directly transferred to local communities, but to the Rural District Councils (RDCs) (Balint and Mashinya 2008; Taylor 2009). Elsewhere in Southern Africa, policymakers have not moved away from the colonial exclusionist models, while contractual parks in South Africa are just paying lip service to the expression “people-oriented approaches” (Roe et al. 2009).

Our study builds on an earlier proposition in the literature, which suggest that the impact of most integrated conservation and development projects (ICDPs) is limited by possible dilemmas in the actual design of the scheme or trade-offs inherent in linking development and conservation objectives (Frost and Bond 2008; Sandker et al. 2009; Ntuli and Muchapondwa 2017a). ICDPs are biodiversity conservation projects with rural development components (Hughes and Flintan 2001). The key debate around ICDPs such as CAMPFIRE is centered on state failure to devolve full authority over natural resources to local communities, which translates into non-alignment of incentives (Murombedzi 2000; Balint and Mashinya 2008; DeGeorges and Reilly 2009). The second most important element after devolution for a community to manage natural resources sustainably is its capacity to self-organise, which in turn presumably depends on the quality of community institutions. Devolving property rights to communities shifts resource governance, responsibility and benefit appropriately to the local level (Child and Barnes 2010).

Existing institutions have failed to protect wildlife in the region because of inappropriate and poor policy designs (Taylor 2009), hence the need for more ideas to feed into future policy reforms. Policies and legislation can enable or discourage community attempts to self-organise, but at the same time, if the need to manage resources sustainably arises (e.g. due to increasing scarcity), communities may start to self-organise in order to create new institutions (Ostrom 1990; Agrawal 2001). The institutional literature distinguishes between different types of establishments, i.e. political, social rules, organisations and processes, interactions, policies and so on that provide institutions with mandates (North 1990; Hodgson 2006). Viewed this way, the idea of institutions is a very broad concept. In this study, we narrow down and look at local CPR institutions such as social rules, organisations and interactions, which matter for wildlife conservation and are within the community's control. Thus, we define institutions as systems of established and prevalent social rules that structure social interactions, including community-level organisations.

The literature is not very clear on the pathways through which institutions affect biodiversity, i.e. whether institutions affect biodiversity directly or indirectly through cooperation. In as much as there are many studies linking institutions directly to the success of biodiversity outcomes[2] (Murphree 1994; Jones 2001), there are also studies that link collective action to sound natural resource management (Ostrom et al. 2007). The concept of biodiversity outcomes is central in our study because it measures an important dimension of SESs. We are also concerned about biodiversity outcomes because most CAMPFIRE communities are a home to endangered wildlife species such as pangolins, wild dogs and cheetahs. As a result, this paper contributes to our knowledge on the role of institutions in community wildlife conservation in Zimbabwe and the CPR literature in general by collecting primary data and applying a framework for analysing complex SESs developed by Ostrom (2007). Specifically, we address an important gap in the literature pertaining to the indirect link between institutions and biodiversity outcomes by considering two relationships that are part of a bigger picture and as such should not be considered in isolation, i.e. (i) the link between institutions and cooperation, and (ii) cooperation and biodiversity outcomes.

Ostrom's framework has been used extensively in areas such as forestry, fisheries, rangelands and water resources management, while little has been done in the wildlife sector, particularly in Southern Africa. Unlike in other resource contexts where institutions also matter, community-based wildlife conservation occurs in an environment where communities do not directly utilise wildlife, but

[2] Biodiversity outcomes are measured in terms of two main dimensions, namely the actual improvements in biodiversity (e.g. species richness, evenness and rarity) and benefits accruing to the community (Mouillot and Lepretre 1999; Nagendra 2002). The success of biodiversity outcome is achieved if the community is able to balance these key attributes of biodiversity.

benefit from its utilisation by others to whom they have assigned appropriation rights after payment. Furthermore, community wildlife conservation occurs in contexts where the CPR is fugitive, a nuisance and largely has marketable value as opposed to consumptive value. Following the discussion above, two important questions arise. Under what conditions will the users of common pool wildlife self-organise? What attributes of resource units, resource users and local institutions are consistent with sound biodiversity outcomes? By answering these important questions, the paper sheds light on the complex processes governing the human-environment systems (i.e. how we can relate institutions, collective action and biodiversity outcomes) and provides results comparable to other studies. Our research findings allow policymakers to question their wildlife management strategies and policies, while at the same time identifying areas that need to be improved.

The rest of the paper is organised as follows. Section 2 presents the conceptual framework and a review of the theory of collective action, while Section 3 gives an outline of the research methods, i.e. the analytical framework, empirical model specifications and data issues. We then proceed to present and discuss the results in Section 4 and wind up with conclusions and policy implications in Section 5.

2. Conceptual framework

Complex environmental problems have been constantly increasing in relevance from both a scientific and policy perspective (Vitousek et al. 1997; Young et al. 2006; Ostrom 2009). The experiences of various scholars have led to the insight that these complex problems cannot be analysed with disciplinary approaches alone, but they have to be dealt with in an integrative, interdisciplinary way that considers the interaction between social and ecological systems (Newell et al. 2005; Folke 2006; Ostrom 2007; Binder et al. 2013). We acknowledge a number of frameworks in the literature that are used to analyse SESs by a wider community of researchers and development practitioners, e.g. the Ecosystem Services (ES) framework (Bouman et al. 2000; de Groot et al. 2002), the Earth Systems Analysis (ESA) framework (Schellnhuber et al. 2005), the Human-Environment System (HES) framework (Scholz and Binder 2004; Scholz et al. 2011), the Material and Energy Flow Analysis (MEFA) framework (Haberl et al. 2004; Brunner and Rechberger 2005), the SES framework (Ostrom 2007, 2009) and the Sustainable Livelihood Approach (Scoones 1998; Ashley and Carney 1999).

These frameworks emerged from the need for concepts that permit structured, interdisciplinary reasoning about complex problems in social-ecological systems. However, they differ significantly with respect to contextual and structural criteria, such as conceptualisation of the ecological and social systems and their interrelation. Binder et al. (2013) identified three main criteria sufficient to produce a classification of frameworks that may be used to make decisions when choosing a

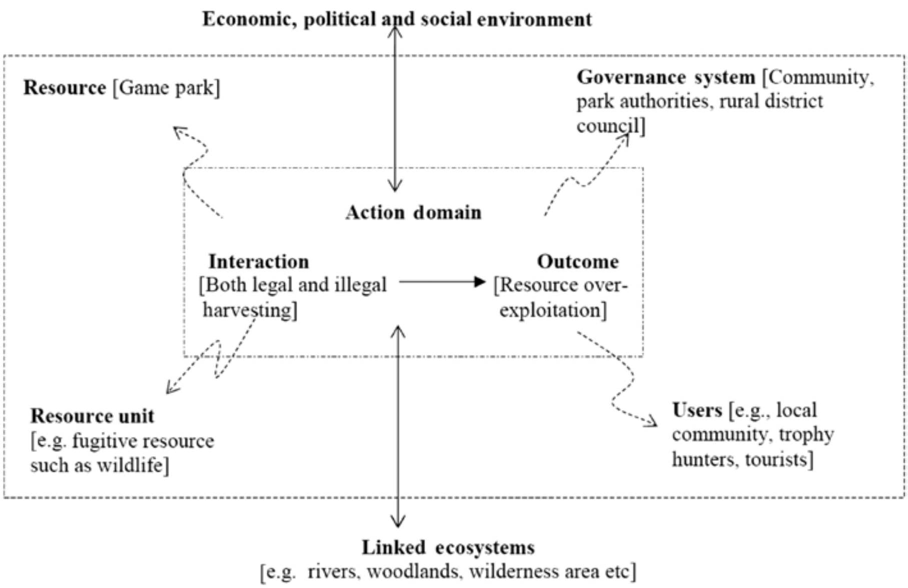

Figure 1: The core subsystems in a framework for analysing social-ecological systems.
Source: Adapted from Ostrom (2007, 15182).

framework for analysis. These criteria are (i) whether a framework conceptualises the relationship between the social and ecological systems as being uni- or bidirectional; (ii) whether it takes an anthropocentric or an eco-centric perspective on the ecological system; and (iii) whether it is an action-oriented or an analysis-oriented framework. For a detailed discussion of these frameworks, we refer the reader to Binder et al. (2013).

This study employs Ostrom's general framework for analysing SESs because it captures the bidirectional relationship between SESs, it takes an anthropocentric perspective on the ecological system and it is action oriented in addition to other features of a SES such as complexity, interdependence, resilience, responsiveness, vulnerability and adaptive capacity (Ostrom 2009). Furthermore, it has been used extensively in the literature in areas such as forestry (Agrawal 2001; Ostrom 2009), pastures (Baur 2014), fisheries (Basurto and Coleman 2010; Basurto et al. 2013), water (Delgado-Serrano and Ramos 2015) and irrigation systems (Cox 2014). This framework helped to identify the key variables that are relevant in studying community wildlife conservation, designing the research instrument and analysing findings about the sustainability of a SES. Figure 1 shows eight core subsystems that have been observed to affect both collective action and sustainability of a resource system (Ostrom 2003, 2007). In empirical work, many variables have been observed to affect the patterns of interaction and outcomes. In an earlier study, Agrawal (2001) identified more than 30 variables that had been

Table 1: Second tier variables used in this paper.

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

Source: Adapted from Ostrom (2007).

posited in major theoretical work to affect incentives, actors and outcomes related to sustainable governance of a resource system.

By applying Ostrom's framework, it is possible to analyse different environmental problems under different scenarios and predict the associated outcomes. Table 1 shows how we can adapt this framework in the context of the wildlife sector in Southern Africa. For example, this study deals with a fugitive and finite renewable resource which is either harvested legally through trophy hunting activities by engaging a safari operator or illegally by poachers, who maximise their individual short-term gains. Subsistence hunting outside the park by local communities is not allowed. Trophy hunting is the only means of generating revenue from conservation. The community applies for a hunting quota from the state and generates income by selling the quota to safari operators, who will in turn obtain clients through organised international events and sell the quota at a premium. The client or trophy hunter utilises the quota. The most important institutions involved in community wildlife conservation are the wildlife management committee, park authorities, the RDC and NGOs.

The interaction between the community and the resource system sometimes produces undesirable outcomes, for instance, poachers cause harvesting levels to exceed the quota. In setting the quota, the state uses the precautionary approach in order to improve trophies and because the maximum sustainable yield is usually not known in advance. Therefore, if the communities in question are without the means of controlling extraction, then the end result is resource overexploitation and eventually total collapse of the social-ecological system. The prediction of resource collapse usually comes true in a very large and highly valuable resource system under open access conditions when users are diverse, do not communicate and have failed to develop institutions for managing resources (Agrawal 2001; Ostrom 2009).

Sometimes resource users come together and form strong bonds (social capital) in order to exploit or conserve a resource. Social capital is a multi-dimensional concept consisting of non-material values such as belonging to social networks, social connections and social norms which influence individual behaviour and interaction among people and make it possible to form a community (Pretty and Smith 2004). It provides the necessary means for extraction in the form of knowledge sharing, equipment and security for illegal harvesters. Social capital can also have positive benefits to SESs when a certain group of people share knowledge to conserve a particular resource (Pretty and Ward 2001). The link between social capital and exploitation of natural resources has received considerable attention in the literature (Ostrom 1990; Pretty and Smith 2004; Ostrom and Ahn 2009). From a theoretical perspective, social capital is viewed as the fundamental motivation for collective action (Ostrom and Ahn 2009). To manage common pool wildlife sustainably requires cooperation among all resource users and this is achievable when communities invest in good institutions that are capable of fostering collection action. Furthermore, substantial welfare benefits from wildlife flow to both individuals (Ntuli and Muchapondwa 2017b) and society (Ashley 2000; Sebele 2010) when people work together to conserve the resource, while lack of collective action dissipates these benefits.

3. The collective action problem

Even though in our lives we constantly engage in collective action, i.e. quite frequently we produce and consume collective goods, our understanding of how people organise to produce such goods is far from complete (Ostrom 2000). As it turns out, in spite of our daily collective behaviour, we have a clearer understanding of why individuals fail to act collectively than why they succeed. Although the collective action problem was initially defined for public goods by earlier scholars, its application has increased to include other research areas such as CPRs, social movements and development challenges. Both local and large-scale collective action is increasingly becoming important in the maintenance of trans-boundary resources such as common pool wildlife, water and forest shared by different communities and countries (Ostrom 1990). In Zimbabwe, the CAMPFIRE programme was initiated to foster cooperation among communities involved in wildlife conservation so that poaching is abolished in the wildlife buffer zone (Gandiwa 2011).

The theory of collective action has matured tremendously since the publication of Olson (1965), entitled 'The Logic of Collective Action'. He defined collective action as any action taken together by a group of people whose goal is to enhance or achieve a common objective. As observed by Meinzen-Dick et al. (2004), the more specific and varied definitions which have been added later have in common the following features: the involvement of a group of people, shared interests and common and voluntary actions to pursue those shared interests. The collective action problem relates to the group's lack of capacity (except under certain conditions) to solve what is referred to as the '*collective action problem'*. For instance, in the maintenance of a CPR, everyone has an incentive to cheat (free ride) on the effort of others (Ostrom 2003). Moreover, larger groups are more likely to experience free rider problems than relatively smaller ones (Ostrom 1998). Collective action is not assumed to flow automatically from common interest, but there are other ingredients that go into it. Resource economists and theorists ask questions about the conditions under which those who face the 'tragedy of the commons' are able to organise a system of rules by which the tragedy is averted (Wade 1987).

Of particular interest to this study is the literature that focuses on creating incentives for collective action through designing sound CPR institutions. Olson (1965) argues that selective incentives (private benefits given to people on the basis of whether they have contributed) are necessary for collective action as inducements. Types of selective incentives include material, monetary, solidary and purposive or moral incentives. This is a very useful way of thinking about the problem of motivating group action. Agrawal (2001) emphasises the differences between a self-organised community and externally imposed collective action in terms of rule enforcement and sanctioning. Studies reveal that communities benefit when institutions are endogenised by the community, compared to a situation when rules are externally enforced by the government (Murphy and Cardenas 2004; Akpalu and Martinsson 2011).

It is certainly inadequate to study collective action without recognising the important concept of participatory development and the contributions made by Chambers (1983) and Cernea (1985). Chambers (1983) defines participatory development as a process through which stakeholders can influence, share control over development initiatives and make important decisions over resources that affect their livelihoods. Olson (1965) argues that people do not participate in social events and development initiatives out of rational self-interest, but out of altruism. The goal of Integrated Conservation and Development Projects is to engage local populations, balance development and conservation objectives and to set in motion a process of self-reliant and sustainable development. Self-reliant development means building the endogenous mechanisms of society that will enable local communities to manage their natural resources to their own benefit. According to the Brundtland report (1987), sustainable development implies that a community is consuming on a stable growth pattern, which is in harmony with the environment. This means that a resource system is harvested in such a way that it provides the same goods and services today, tomorrow and well into the future. It is thus imperative for policymakers to design policies that engage local communities in resource management, utilisation and decision-making at both local and higher levels.

4. Research methods

4.1. Data sources and sample size

This study collected primary data between May and July 2013 from a sample of 30 local communities in 13 wards around Gonarezhou National Park (GNP) in Zimbabwe that are participating in wildlife conservation. GNP is located in south-eastern Zimbabwe (coordinates 21° 40′ S 31° 40′ E) and covers about 5053 km^2. It forms part of the Great Limpopo Trans-frontier Park that links Gonarezhou

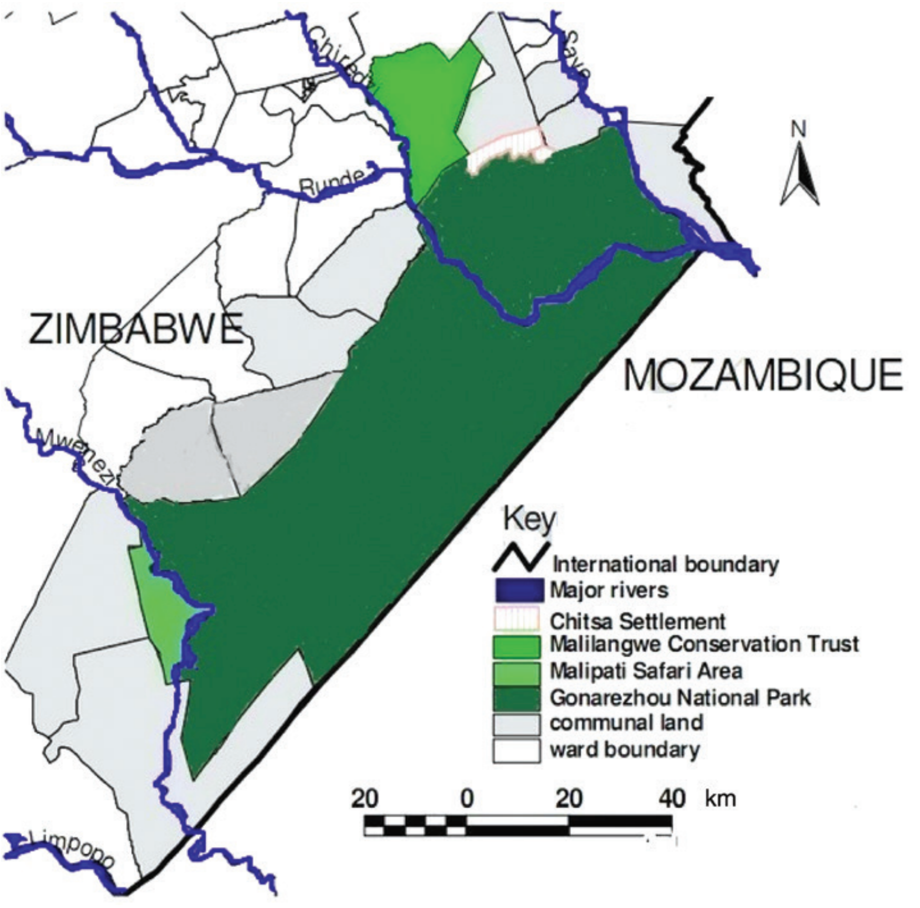

Figure 2: Map of Gonarezhou National Park and the local communities.

with Kruger National Park in South Africa and Limpopo National Park in Mozambique. Owing to its vast size, rugged terrain and location away from the main tourist routes, large tracks of Gonarezhou still remain pristine wilderness. Figure 2 shows the map of GNP (in dark green) and the communal areas (in grey) bordering the national park.

The national park is located in natural region five, which is very dry with very low agricultural potential. The mean annual rainfall for the area is about 499 mm, with a standard deviation of about 195 mm. The average maximum monthly temperature ranges from less than 25.9°C in winter to over 36°C in summer, while the average monthly minimum ranges from 9°C to 24°C in winter and summer, respectively (Gandiwa 2011). The vegetation of the Gonarezhou ecosystem is typical semi-arid savanna, dominated by *Colophospermum mopane* woodlands (Gandiwa and Kativhu 2009). The area under study is located approximately 100km away from the nearest Chiredzi town, relatively sparsely populated and predominantly occupied by the Shangani people; other ethnic groups such as the Venda, Ndau, Shona and Ndebele people are also found in the area. Due to its proximity to the South African border and lack of employment in the area, most school children hardly proceed beyond primary level of education. Although peasant farmers in the area engage in a diversified portfolio of economic activities including wildlife conservation, the most dominant livelihood activities are livestock production and crop cultivation, e.g. maize, sorghum, millet, groundnuts and cotton. The mode of production is primarily subsistence in nature.

Through household surveys and key informant interviews, the study collected data from a sample of 336 households and 30 key informants. Out of the 30 communities visited, 25 are CAMPFIRE projects while the remainder are resettlement[3] schemes. Very few interviews were conducted in resettlement schemes because they constitute less than 1% of the communities involved in wildlife conservation in the study area. Resettlement schemes differ from communal areas in that land tenure is governed by lease agreement under the former, while the latter have communal tenure.

The household questionnaire collected information about the household's socio-economic characteristics, such as demographics, agricultural activities, assets, consumption, income and expenditure, and household involvement in community wildlife activities, while the key informant questionnaire collected information at the community level about the community's involvement in wildlife activities, e.g. existence of a wildlife management committee, selection process of committee members, existence of a constitution, type and number of rules and regulations that are in place, clarity of rules, whether rules are recognised by the community members, monitoring (patrols), enforcement (punishment), number of meetings, attendance, communication, type of information shared,

[3] These are the old resettlement schemes (commonly refered to as the1982 type) that were created before land invasions. As a result, we would expect communities in the old resettlement schemes to display cooperative behaviour and less conflict with conservation objectives.

Table 2: Number of households by ward.

Ward	Interviews		Household involved in CWM		Ward population	
	Number	Percent	Number	Percent	Number	Percent
Ward 10	20	5.95	820	7.82	1300	7.46
Ward 12	7	2.08	43	0.41	43	0.25
Ward 13	15	4.46	861	8.21	1000	5.74
Ward 14	9	2.68	492	4.69	1750	10.04
Ward 15	64	19.05	955	9.11	2000	11.47
Ward 22	10	2.98	81	0.77	81	0.46
Ward 23	13	3.87	181	1.73	181	1.04
Ward 29	19	5.65	2000	19.08	2000	11.47
Ward 30	29	8.63	960	9.16	1700	9.75
Ward 4	10	2.98	350	3.34	1850	10.61
Ward 5	32	9.52	1589	15.16	2100	12.04
Ward 8	84	25.00	1235	11.78	1470	8.43
Ward 9	24	7.14	917	8.79	1960	11.24
Total	336	100.00	10,484	100.0	17,435	100,00

Source: Survey data Aug 2013.

bookkeeping, problem animal control, animal counting and provision of watering points among others. To complement the primary data, we used secondary data on conservation activities obtained from the RDC. A brief description about the quality of the data, how it was collected and used is found in the next section.

The household survey employed a simple random sampling procedure (without replacement) using the list of beneficiaries from each community as the sampling frame.[4] Using the list of households from each community, we selected every 10th household starting with a randomly chosen household from the list. The selected household was then removed from the list and this process continued until the sample size was achieved. These households represent peasant farmers in rural communities located adjacent to protected areas.

Table 2 shows the number of households interviewed, by ward, against the total number of households participating in wildlife conservation in each ward, and ward population expressed in terms of households.[5] Since the study purposely targeted CAMPFIRE communities, a complete enumeration of all CAMPFIRE projects in each ward was carried out. Ward 8 and Ward 15 were over-sampled

[4] Initially, sampling was done using information gathered at district level from the RDC. There was a huge disparity between the information supplied at the district level and what was found on the ground in terms of the number of communities and number of households participating in wildlife conservation, as the RDC does not update its records regularly. As a result, information gathered from the chairperson of the respective community groups was used to update the initial calculation of the sample size.

[5] The sample size was further compromised by the fact that some enumerators did not meet their targets, particularly in areas that were sparsely populated, while some questionnaires were not usable due to non-response or lack of critical information.

because these two wards had a higher number of CAMPFIRE projects than any other ward. We do not expect this to affect our results since the idea was to sample households in CAMPFIRE communities, over and under-sampling is therefore taking place at project level and not community level.

5. Empirical strategy

This study makes use of econometric modelling to establish the relationship between (a) cooperation and other institutional variables identified by Ostrom (2007) as affecting the likelihood of cooperation, and (b) cooperation (as well as other variables identified in Ostrom's framework as affecting the sustainability of SESs) and biodiversity outcomes. These two relationships are embedded in a system of equations explained below. Therefore, we are not looking at the role of institutions in maintaining the integrity of the ecological system in isolation, but we analyse this relationship via the intermediacy of collective actions.

5.1. Model 1: participation in wildlife conservation

A variable measuring ability to self-organise is used as a dependent variable in our first model. This could be cooperation of individuals in community activities including wildlife. From this point onwards, we will think of cooperation as refraining from illegal harvesting of wildlife. Following McCarthy et al. (2002) and Pennings and Leuthold (2000), cooperation is assumed to be a latent variable. Factor analysis of variables thought to be associated with cooperative capacity is employed in order to recover the latent variable. Indicators of cooperation are drawn from two main categories, namely, networks and organisational performance variables. Network indicators include the density of organisations and density of household participation, while organisational performance indicators include the number of rules, regulations, activities and meetings. Mathematically, we write Model 1:

$$C_1 = \beta_0 + \beta_1 Institutions + \beta_2 Groupsize + \beta_3 Trust + \beta_4 Ethinicity + \beta_5 Wealth$$
$$+ \beta_6 Projectyear + \beta_7 Punishment + \beta_8 Resourcesize + \beta_9 Stakeholders \ldots\ldots + \varepsilon_i$$

To measure the quality of institutions in a community, we made use of Ostrom's (2010) design principles for stable local common pool resource institutions. A complete enumeration of CPR institutions in each community provided vital information for calculating an index of the quality of institutions. The index captured mechanisms of conflict resolution, monitoring, enforcement, operational rules, collective choice rules, organisations, clarity and acceptance of rules and boundaries. We then used this index plus other explanatory variables to explain cooperation, as described in Model 1 above.

5.2. Model 2: success of biodiversity outcomes

To measure the success of biodiversity outcomes across communities, a measure of relative abundance of animal species was used. The relative abundance of an animal species in a community is defined as the proportion of individual organisms in the community that belong to that species. Let P_j for j=1, 2, …, N be the relative abundance of animal species j and N the number of species. We can define the Shannon index as:

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

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

To calculate the Shannon index, we used information about commercial animal species counted by the RDC and the respective communities.[6] Each year, communities gather information by type of species about the number of wild animals traversing their conservation area.[7] The records are kept at the RDC offices and this information is then used by the community as justification when applying for a hunting quota. Wild animal counting is done at the community level by a team of people that includes members from local communities, the RDC and sometimes park authorities. This data is likely to be reliable since the collection process involves all major stakeholders, i.e. representatives from the local community, RDC, park authorities and sometimes non-governmental organisations. So, algebraically, we have the following equation linking biodiversity and cooperation, Model 2:

$$\begin{aligned} S_j = {} & \beta_0 + \beta_1 Cooperation + \beta_2 Training + \beta_3 Benefits + \beta_4 Market + \beta_5 Distfence \\ & + \beta_5 Socialcapital + \beta_7 Age + \beta_8 School + \beta_9 Yearsliving + \beta_{10} Newfence \\ & + \beta_{10} Info \ldots\ldots + \varepsilon_i \end{aligned}$$

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

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

For each index computed in this analysis (i.e. institutions, cooperation, biodiversity, social capital and wealth), we asked a variety of questions to measure specific attributes of each variable (the questionnaire is available upon request) and factor analysis was then used to recover the latent variables. Table 3 provides a description and expected signs of the explanatory variables that are used in both models. The institutional variables came from Ostrom's design principles, while the other explanatory variables came from a review of literature. From an empirical and theoretical perspective, we expect our measure of institutions, trust and number of stakeholders in the community (in our first model), and our measure of cooperation, benefits from wildlife conservation, distance to the market, educational level of the head of the household, number of years living in the area, whether community members received training, whether community members share information (in our second model) to carry a positive sign (Wade 1987; Ostrom 1990; Baland and Platteau 1996; Agrawal 2001). We expected group size and ethnic diversity in the first model, to carry a negative sign (e.g. Olson 1965; Ostrom 1990, 2007; Baland and Platteau 1996). Social capital could have both

Table 3: Nature of variables and expected signs.

Variable	Nature of variable	Expected sign
Variables for the first model		
Cooperation index	Dependent variable	
Institutions	Institutional index	+
Clarity	Clarity of institutions	
Fairness	Fairness of institutions	
Governance	Governance	
Monitoring	Monitoring and enforcement	
Groupsize	Group size	±
Trust	Measured on a scale from 0 to 10	+
Ethnicity	Number of ethnic groups	−
Wealth	Wealth index	±
Projectyear	Number of years in operation of the project	+
Punishment	Punishment [0=exogenous 1=endogenised]	+
Resource size	Resource size (ha)	±
Stakeholders	Number of stakeholders	Undetermined
Variables for the second model		
Biodiversity index	Dependent variable	
Cooperation index	Cooperation in a community	
Training	Number of training courses	+
Benefits	Benefits from wildlife conservation	+
Market	Distance to the nearest town (km)	+
Distfence	Distance to the fence	Undetermined
Socialcapital	Social capital index	±
Age	Average age of household head	±
School	Average number of years in school	±
Yearsliving	Average number of years living in the area	+
Newfence	New electric fence	Undetermined
Info	Information sharing index	+

positive and negative effects on biodiversity outcomes depending on the basis of its formation, i.e. sometimes people form strong coalitions and share information to exploit a particular resource. However, the expected signs of some variables could not be determined theoretically or from the empirical literature.

We suspect the problem of endogeneity, particularly in the relationship between cooperation and biodiversity. This is because the theory posits that there is reverse causality between biodiversity outcomes and cooperation, i.e. less biodiversity translates into more cooperation in order to avoid the tragedy of the commons, and vice versa. As a result, some scholars argue that the incentive to self-organise does not always hold, especially when resources occur in abundance (Ostrom 1990). However, this is not the case in Zimbabwe because wildlife resources have declined tremendously since the turn of the 21st century. To put this into perspective, the wildlife management policy in the country speaks to the scarcity of resources, which necessitated the need for conservation and sustainable development. This is also one of the reasons that the government institutionalised the CAMPFIRE programme in order to balance development and conservation objectives (Martin 1986; Murphree 1991, 1993; Murombedzi 1999).

We first estimate the models using ordinary least squares (OLS), ignoring any issues of endogeneity. Because of endogeneity issues discussed above, we then employ instrumental variables estimation with heteroscedasticity-based instruments, which methodologically deals with the problem of endogeneity, and compare the results. Following Lewbel (2012), this method estimates an instrumental variables regression model providing the option to generate instruments and allows the identification of structural parameters in regression models with endogeneity in the absence of traditional identification information such as external instruments.

Identification is achieved in this context by having explanatory variables that are uncorrelated with the product of heteroscedastic errors (Baum et al. 2013). According to Lewbel (2012), instruments may be constructed as simple functions of the model's data. As a result, the approach may be applied in cases where no external instruments are available or used to supplement weak external instruments in order to improve the efficiency of the instrumental variables estimator. Thus, Lewbel's approach can be a good substitute of the standard IV approach in terms of addressing the problem of endogeneity. The choice one uses depends on the availability of sound external instruments. If good external instruments are available, then the standard IV approach is superior. If external instruments are either weak or not available, then the method of heteroscedasticity-based instruments is superior to the conventional IV approach.

This method is gaining popularity and it is being used widely in many studies (e.g. Emran et al. 2012; Banerjee et al. 2013; Mishra and Smyth 2015). Using two data sets from China to compare the identification strategy which utilises a heteroscedastic covariance restriction to construct an internal IV and the standard IV, Mishra and Smyth (2015) found that the Lewbel's method provides plausible estimates in datasets in which conventional IVs are not available. One of the major drawbacks of Lewbel's approach is that identification in Lewbel's approach relies upon higher moments, and is likely to be less reliable than identification based on coefficient zero restrictions. For a detailed description and mathematics behind the method for constructing instruments as simple functions of the model's data, please refer to annexes.

Because we are considering about 30 observations (30 communities), there is a worry that we might be dealing with too few observations. One way to deal with that is bootstrapping.[8] We applied the bootstrapping procedure to check for robustness of the standard errors. We also checked for multicollinearity, under-identification, weak-identification and over-identification of instruments using the VIF test, The Kleibergen-Paap test, the Cragg-Donald Wald F-statistic and the Hansen J statistic before proceeding with heteroscedasticity-based instruments in both models.

6. Results and discussion

6.1. Community characterisation

Table 4 presents the descriptive statistics of the key variables used in econometric modelling. Our results show significant variation in terms of household, community and institutional characteristics.

Table 5 shows that the performance of most communities in the study area is well below the desirable level in terms of the institutional attributes that matter for conservation. On a scale from 0 to 100, where 0 signifies the complete absence and 100 the complete presence of an attribute, all community attributes in Table 5 fall below half (50.0), except for information sharing, which has a mean of about 60.56. Most communities share vital information such as financial matters, past actions and knowledge of the SES, mainly through village meetings. The mean level of cooperation is about 39.26, while the mean for the overall institutional index is 34.06.

Disaggregating the institutional index into four attributes, namely, clarity of institutions, fairness, governance (including participation and democracy) and monitoring and enforcement (including formal punishment and social sanctioning), we observe that the mean of each attribute is low, especially for the governance, participation and democracy index. This seems to suggest that, in most communities, the quality of local institutions is poor since the index falls below half (50.0). The situation is further exacerbated by the overall enabling environment which prevents institutions from operating effectively (Balint and Mashinya 2008). This would have implications for cooperation in a community.

[8] Bootstrapping is a resampling method that allows us to compute standard errors from a sampling distribution without any knowledge of the parent distribution from which our samples are drawn (Efron 1979; Horowitz 2001, 2003).

Table 4: Descriptive statistics of the variables used in the econometric model.

Variable	Obs	Mean	Std. Dev.	Min	Max
Group size	336	451.6	461.4	6	2000
Trust [scale from 0 to 10]	336	4.896	2.192	1	10
Number of ethnic groups	336	2.631	0.901	1	5
Number of years in operation of the project	336	12.99	7.157	4	27
Nature of punishment [0=Endo, 1=Exo]	336	0.506	0.501	0	1
Resource size (ha)	336	14,186	7614	0	26,000
Training (0=no 1=yes)	336	0.628	0.484	0	1
Benefits	336	20,047	17,290	0	68,880
Distance to market (km)	336	65.45	25.88	33	133
Distance to the fence (km)	336	9.843	16.77	0.10	80
Number of poaching incidents	336	7.955	6.192	0	22
Average age of household head	336	48.88	13.62	22	89
Average number of years in school	336	5.524	4.257	0	15
Average number of years living in the area	336	36.64	13.57	6	73
Average number of stakeholders	336	4.280	1.698	2	9
Number of WMC meetings	336	3.58	1.62	0	12
Number of WMC activities	336	0.860	1.141	0	8
Number of WMC major rules	336	13.89	10.90	0	32
Number of WMC minor rules	336	33.04	27.44	0	78
Total number of WMC rules and regulations	336	46.87	38.09	0	110
Number of other NRM rules	336	6.846	5.304	0	19.25
Number of NRM meetings	336	2.48	1.50	0	5.25
Number of NRM activities	336	3.24	3.16	0	12.5
Number of non-NRM rules	336	14.34	4.57	7.5	25
Number of non-NRM meetings	336	4.00	1.45	2	7
Number of non-NRM activities	336	8.59	3.83	2	15
Extent to which rules are recognised by society	336	3.51	2.75	0	8
Extent to which rules are recognised by authorities	336	6.10	4.32	0	10
Number of poaching incidents	336	7.955	6.192	0	22

Source: Survey data Aug 2013.

Table 5: Summary of indices.

Variable	Obs	Mean	Std. Dev.	Min	Max
Cooperation index	336	39.26	28.47	0	100
Institutional index	336	34.06	22.05	0	100
Clarity index*	336	40.97	23.62	0	100
Fairness index*	336	40.81	20.85	0	100
Governance index*	336	34.96	20.82	0	100
Monitoring index*	336	37.67	24.40	0	100
Wealth index	336	31.84	24.18	0	100
Social capital index	336	18.18	14.49	0	100
Information index	336	60.56	22.95	0	100
Biodiversity index**	336	1.520	0.950	0.06	3.14

Source: Survey data Aug 2013.
*Indices for institutional attributes.
**The Shannon index for biodiversity.

6.2. Results of the regression models

This paper argues that institutions directly affect cooperation and indirectly influence the success of biodiversity outcomes through cooperation. We use regression analysis techniques to analyse these relationships. As stated earlier, we are aware of the endogeneity problem and it has to be corrected. The endogeneity issue is discussed in the research methods section. First, we use OLS regression analysis, and then we use instrumental variables estimation with heteroscedasticity-based instruments, which methodologically deals with the endogeneity problem. Following standard procedures in regression analysis, various tests were conducted. The Durbin-Wu-Hausman test for endogeneity suggest that OLS yields better results in Model 1, while the instrumental variables estimation yields superior results in Model 2.

6.2.1. Relationship between cooperation and institutions

The results in Table 6 show that both the OLS regression model and the instrumental variables estimation are highly significant and explain over 80.0% of the variation in the dependent variable. We consider two models, a model with the overall institutional index and another with disaggregated institutional indices.[9] Replacing the overall institutional index with disaggregated institutional indices does not affect the signs of the explanatory variables in the model and so the interpretation remains the same. Comparing OLS and the IV results, we observe that most variables are highly significant, except for group size, ethnicity, resource size and wealth. The variable ethnicity measures heterogeneity in a community, but the variable is insignificant in both models, suggesting that ethnic diversity is not an important factor in explaining cooperation in the area. Ethnicity is important only when you have mixed ethnic groups working together (Posner 2005). The overall results show that socio-economic factors are drivers of cooperation. However, with disaggregated institutional indices, group size, resource size and wealth become significant under both OLS and IV estimation, implying that cooperation is explained better with the latter model than with the former.

The results show that the institutional variable is positive and significant at the 1% level of significance. This seems to suggest that an improvement in the quality of institutions increases cooperation in the respective communities. This result is consistent with the literature and confirms our hypothesis that institutions matter for self-organisation. Information gathered through qualitative interviews show that all communities in the study area have managed to develop some form of institutions, although these institutions differ in terms of their attributes as we move from one community to another. This is in line with the theory on

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

Table 6: Relationship between ability of a community to self-organise and institutions.

Cooperation	Model with overall institutional variable		Model with institutional attributes	
	OLS	IV	OLS	IV
Number of obs	336	336	336	336
Prob>F	0.0000	0.0000	0.0000	0.0000
R-squared	0.8542	–	0.8854	–
F (10, 325)	–	223.22	–	320.2
Centred R2	–	0.8191	–	0.844
Uncentred R2	–	0.9378	–	0.946
Overall institutional index	0.129***	0.557***		
	(0.0487)	(0.177)		
Clarity index			0.117*	0.195 (0.137)
			(0.0696)	
Fairness index			−0.160***	−0.515**
			(0.0537)	(0.150)
Governance index			0.192***	0.603***
			(0.0732)	(0.168)
Monitoring and enforcement index			0.360***	0.245***
			(0.0512)	(0.103)
Group size	0.00121	0.00472*	0.00309*	0.00205**
	(0.00181)	(0.00197)	(0.00161)	(0.00152)
Trust	3.120***	1.151***	1.800***	1.496***
	(0.574)	(0.893)	(0.554)	(0.756)
Ethnicity	−1.172	0.231 (1.135)	−0.626	−0.594
	(0.767)		(0.714)	(0.960)
Wealth index	0.0673**	0.0254	0.0555**	0.0642**
	(0.0268)	(0.0342)	(0.0239)	(0.0278)
Years in operation of project	1.672***	1.181***	1.255***	1.166***
	(0.202)	(0.328)	(0.187)	(0.267)
Punishment [0=Exo, 1=Endo]	12.99***	11.23***	7.195***	9.815***
	(1.843)	(2.170)	(1.831)	(2.528)
Resource size	−0.000150	−0.000287***	−0.000265**	−0.000366***
	(9.22e-05)	(9.84e-05)	(0.000111)	(0.000157)
Number of stakeholders	1.552***	1.741**	0.912*	0.557***
	(0.564)	(0.702)	(0.533)	(0.596)
Tenure	−8.657***	−7.465***	−6.617**	−12.17***
	(2.883)	(2.153)	(3.271)	(4.855)
Cons	−12.04***	−12.62**	−9.085***	−3.016
	(3.635)	(5.121)	(3.360)	(4.072)
Under identification test (Kleibergen-Paap rk LM statistic):		72.208		54.450
	$\chi^2(10)$ P-val	=0.0000		=0.0000
Weak identification test (Cragg-Donald Wald F statistic):		19.373		16.281
Hansen J statistic (overidentification test of all instruments):		10.574		9.5841
	$\chi^2(9)$ P-val	=0.2308		=0.2623
Breusch-Pagan test for heteroscedasticity $\chi^2(1)$		=18.543		=29.172
	Prob>χ^2	=0.0000		=0.0000

Source: Survey data Aug 2013.
NB: Standard errors shown in brackets.
*Significant at 10%, **significant at 5%, ***significant at 1%.

endogenous institutional development where community members develop operational and collective choice rules to govern or control access to resources (Ostrom et al. 1994). Based on our finding, we can rule out the possibility of an open access regime where members of the community can access resources at any time without restraint. Ostrom (2007) and Agrawal (2001) argue that most CPRs in third world countries that were previously thought to be open access in earlier studies are in fact managed as common property.

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

However, to our surprise, the fairness index is highly significant and carries a negative sign. People perceive fairness in different ways based on their personal understanding and experience and a system of equality is not necessarily an equitable system. Because we collected qualitative data, a possible explanation for this result could be that beneficiaries closer to the park fence frequently suffer from wildlife intrusion and hence they feel that they should be treated differently from those farther away from the fence. This also explains why group size is very unstable at the ward level and why communities frequently divide into smaller groups in order to ensure that members who suffer more from wildlife intrusion benefit more than those farther away. The correlation between distance from the park boundary to biodiversity conservation is well researched and gives rise to the concept of "producer community". Our results are comparable to previous studies that looked at distance from the park boundary in relation to producer communities (Murphree 2005; Rihoy et al. 2007; Taylor 2009).

The coefficients of the variables measuring community-level trust, year of establishment, punishment and number of stakeholders have positive and significant impacts on cooperation. This confirms that cooperation is better in communities where members trust each other than in communities where trust is lacking. From a global perspective, this finding supports other studies demonstrating the importance of trust in human relations at various levels, and in various contexts (Ostrom 2000; Fischbacher et al. 2001; Boyd and Richerson 2002; Gintis 2004; Andersson and Wengström 2011; Useche 2013; Saunders 2014). Cooperation is higher in communities that joined wildlife conservation earlier than in communities that joined later. This makes sense because the longer a community is involved in wildlife conservation, the more likely it is for that community to develop robust institutions that are adapted to local conditions. Cooperation is also higher in communities that have endogenised punishment as opposed to communities that still rely on external force in order to enforce adherence to rules and regulations. Our methodology confirms other research conclusions that policy-makers should seriously consider institutional reforms that convey greater control of natural resources through devolution and decentralisation of managerial functions, decision making and authority into the hands of local communities, while the state maintains regulatory functions. Ostrom et al. (2007) argue that it is cheaper for local communities to engage in monitoring and enforcement activities than it is for the state apparatus to do so, due to budgetary and information constraints. Usually, members of the community have better information about what other members are doing since they live together, which greatly reduces monitoring costs. (Child and Barne 2010). Dore (2001) and Mohamed-Katerere (2001) agree that traditional authorities can administer effective punishment to offenders at a lower cost than the state.

Our results also reveal that cooperation increases with an increase in the number of stakeholders. This is true in the study area because there are a number of NGOs working with local communities in wildlife conservation, particularly in providing training or capacity building. The tenure variable is negative and highly significant (at 1%) under both models, implying a negative relationship between tenure and cooperation. This suggests that cooperation declines as we move from communal areas into the resettlement schemes. This is true for two main reasons. Firstly, the resettlement schemes are comprised of lease holders operating on individual plots, who are thus maximising individual objectives at the farm level but not as a community. Given the nature of resettlement schemes, it is difficult for park authorities to monitor and enforce rules on private property unless farm owners come together and act as a community in order to achieve a common objective of conservation. Secondly, unlike in communal areas where institutions bring people together, in resettlement schemes such institutions do not exist to the extent that farmers act independently (Cousins 1993; Chamunorwa 2010; Marimira 2010).

Furthermore, with the introduction of disaggregated institutional indices, group size, the size of the resource system and wealth become significant under both models. Our results show that there is a positive and significant relationship between group size and cooperation. In general, theory posits that the effect of group size on self-organisation tends to be negative, given the higher transaction costs associated with getting people together and agreeing on important issues (Wade 1994; Ostrom 2009). This hypothesis is based on the argument that in a smaller group, each member of the group would get a substantial share of the benefits from the resource, which may exceed their marginal costs of contribution and therefore, the possibility of free riding is reduced (Olson 1965). However, if the tasks of managing a resource system are very costly (e.g. monitoring an extensive resource system), larger groups are in a better position to mobilise the necessary resources required for such undertakings. Hence, the size of the group is always relevant, but its influence on the ability of a community to self-organise is contingent on other variables of the SES and the type of management activities in question (Ostrom 2009).

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

The wealth index becomes significant and still carries a positive sign under both models when the disaggregated institutional attributes are introduced, suggesting more cooperation in wealthier communities than in poor communities. The conventional wisdom from field experiments suggests that, at the group level, both average group wealth and variance in the distribution of wealth decrease the level of cooperation or social efficiency achieved by the group (Ostrom 1994; Cardenas 2003; Useche 2013). Baland and Platteau (2001) point out that the effects of wealth heterogeneity on collective action are ambiguous because inequality is not unilateral. The authors argue that different dimensions of inequality may have opposing effects on collective action and therefore when such dimensions are combined, the effect on collective action cannot be determined clearly.

6.3. Relationship between biodiversity and cooperation

Both OLS and IV estimation models are significant at the 1% level and explain over 72.2% of the variation in our dependent variable (please refer to Table 7). All variables in both models are significant and do not vary much in terms of their coefficients, standard errors and the level of significance for the two models.

The results in Table 7 show that cooperation is positive and highly significant. This implies that cooperation is an important variable explaining biodiversity outcomes, as suggested in the CPR literature. Hence, we expect to find sound biodiversity outcomes in those communities with higher levels of cooperation and strong institutions than in communities with less cooperation and weak institutions. In areas where the level of cooperation is low and poaching activities are rife, wildlife is either quickly decimated due to overharvesting or may respond to higher levels of poaching by retreating back into the park. We maintain that institutions affect biodiversity outcomes indirectly, through the ability to self-organise, or cooperation. We tested this hypothesis by regressing the institutional variable against biodiversity and found the relationship to be insignificant, but almost approaches significance as we add more explanatory variables.

Table 7: Model 2 – Relationship between biodiversity and cooperation.

Biodiversity	OLS	IV
Number of obs	336	336
Prob>F	0.0000	0.0000
R-squared	0.7223	–
F (10, 325)	–	283.38
Centred R2	–	0.807
Uncentred R2	–	0.9224
Cooperation	0.00595***	0.00758***
	(0.00246)	0.00215
Training [0=yes, 1=no]	0.215***	0.208***
	0.0745	0.0762
Benefits	1.51e-05***	1.25e-05***
	3.56e-06	3.37e-06
Distance to nearest urban centre	0.0122***	0.0126***
	0.00151	0.00130
Distance to the fence	0.0148***	0.0131***
	0.00212	0.00205
Social capital index	0.00704***	0.00710***
	0.00200	0.00202
Average age of household head	–0.357***	–0.381***
	0.123	0.135
Average number of years in school	–0.0926***	–0.0892***
	0.0227	0.0234
Average number of years living in the area	–0.00538**	–0.00597**
	0.00218	0.00247
Fence	–0.00345***	–0.00490***
	0.00129	0.00182
Information sharing index	0.242***	0.210***
	0.0429	0.0455
Cons	9.631***	10.34***
	3.025	3.347
Under identification test (Kleibergen-Paap rk LM statistic):		139.319
	$\chi^2(10)$ P-val	=0.0000
Weak identification test (Cragg-Donald Wald F statistic):		25.420
Hansen J statistic (overidentification test of all instruments):		0.0430
	$\chi^2(9)$ P-val	=0.8352
Breusch-Pagan test for heteroscedasticity	$\chi^2(1)$	=21.94
	Prob>χ^2	=0.0000

Source: Survey data Aug 2013.
NB: Standard errors shown in brackets.
*Significant at 10%, **significant at 5%, ***significant at 1%.

As expected, wildlife management training is positive and highly significant, implying that this variable is an important factor explaining the success of biodiversity outcomes. This implies that communities that received wildlife management training are better off in terms of managing and conserving wildlife than are communities where wildlife management training has not been offered. However, not all communities have received training relevant to wildlife management. The number of households and communities involved in wildlife conservation is growing, signalling the need for more training in the study area. The benefits from wildlife conservation significantly and positively affect biodiversity outcomes in a community. If the resource system is very important in the eyes of the users and generates a substantial flow of benefits, then users attach high value to sustainability of the resource (Berkes and Folke 1998; Chhatre and Agrawal 2008); otherwise, the cost of organising and maintaining a self-organised system may not be worth the effort (National Research Council 2002). Communities from the study area have come to realise that using the proceeds from wildlife conservation to invest in public goods such as schools, clinics, water, grinding mills and electricity is much more beneficial than getting dividends at the household level. Viewed from this angle, the benefits from wildlife conservation are tangible in the eyes of the community and therefore the incentives to conserve wildlife increase with benefits.

Market integration and global market trends are viewed worldwide as potential threats to wildlife conservation in developing countries. In this paper, we used distance to the nearest urban centre as a proxy for market integration. The variable distance to the nearest urban centre is positive and highly significant, suggesting that biodiversity outcomes improve as the distance to the market increases. We argue that the incentives for poaching are much higher for those communities that are located closer to urban centres or main routes linking rural communities to urban areas because animals and game fetch higher prices in wider markets. On the other hand, the variable measuring distance to the fence is positive and highly significant. This means that communities that are located closer to the fences are less likely to conserve biodiversity because they suffer more from wildlife intrusion, interact with wildlife quite frequently and hence have greater access to wildlife than those communities located further away. Our results seem to suggest that biodiversity outcomes are more successful for those communities that are located far away from urban centres or routes connecting rural communities to urban centres but are not very close to the boundary of the game park. This result seems to contradict other community experiences in Zimbabwe such as the Masoka community in the Zambezi valley. In real life, there are exceptional cases and our study might be an example of those few cases.

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

The average age, number of years in school and number of years living in the area for the household head are significant and negatively related to biodiversity outcomes. This means that the status of biodiversity deteriorates as the age, number of years in school and number of years living in the area for the head of the household increases. It is possible that the education of the household head could be related to knowledge of the markets for illegally hunted trophies (including bushmeat) both inside and outside the community, while number of years living in the area is related to ecological knowledge about wildlife movements and experience of the hunter. Viewed from this angle, both variables might be responsible for deriving the commercial poaching behaviour, which in turn lead to resource overexploitation and suboptimal biodiversity outcomes.

An interesting development in the study area is the idea of putting an electric fence around Gonarezhou National Park in order to conserve wildlife and reduce human-wildlife conflict. Most communities have lost part of their conservation land to the national park and the number of communities is growing with an increase in the area of the park covered by the electric fence. Beneficiaries of conservation payments view this development as a potential threat to the CAMPFIRE project. Our results show that the fence reduces biodiversity outcomes in the community's conservation area. This is true for two main reasons. Firstly, the fence greatly reduces the number of wild animals moving into the community's conservation area from the park. Secondly, the electric fence greatly reduces the benefits from conservation, thereby eroding the incentives to conserve wildlife resources. As a result, the community might fight back by increasing its poaching effort, leading to resource overexploitation and finally exhaustion of all resident species and those that can make it across the fence.

Recent studies demonstrated that fencing protected areas has unintended and ironic *effects* on the animals they are built to protect (Ferguson and Hanks 2012; Durant et al. 2015; Trouwborst et al. 2016). Fencing national parks could significantly and permanently alter entire processes and landscapes by keeping ecosystem engineers such as wildebeest and elephants from their regular migrations, while others are accidentally killed by the fence if they attempt to cross it. The effects of the fence would certainly trickle down to the communities by reducing the amount of benefits they get from conservation since wildlife movement is severely restricted. CAMPFIRE communities benefit through legal trophy hunting of wildlife roaming in their area of jurisdiction. For instance, the community gets money and meat when an elephant is killed in their territory. Because of the electric fence, most animals will not be able to cross to the community's side resulting in loss of revenue and game meat, which happened to be a significant source of protein for most households. Qualitative interviews with key informants revealed that local communities get approximately USD 11,000 per elephant killed and between 20 and 50 kgs of game meat per household per year.

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

7. Conclusions and policy implications

Institutions play an important role in stabilising large-scale cooperation in common pool resource management. Without restrictions to govern human behaviour, most natural resources are vulnerable to overexploitation. Wildlife in Southern Africa is one good example of a common pool resource that is threatened by the very communities that are supposed to manage it because of either the existence of poor institutions or lack of institutions.

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

Our results show that the performance of most communities in the study area is well below the desired level in terms of the characteristics that matter for conservation. Sound institutions are indeed an important ingredient for cooperation. Our results suggest a much stronger indirect relationship between institutions and biodiversity outcomes via the intermediacy of cooperation, which is not clearly defined in theoretical studies. Improvements in institutional attributes such as governance (participation and democracy), monitoring and enforcement might lead to increased cooperation, while fairness of institutions seems to work against cooperation. No evidence was found in favour of the role of clarity of institutions on cooperation. Because we collected qualitative data, a possible explanation for this result could be that beneficiaries closer to the park fence frequently suffer from wildlife intrusion and hence they feel that they should be treated differently from those farther away from the fence.

Cooperation is better in communities where members trust each other than in communities where trust is lacking. Cooperation is also higher in communities that have endogenised punishment as opposed to communities that still rely on external force in order to enforce adherence to rules and regulations. Cooperation declines as we move from communal areas into the resettlement schemes. With the introduction of disaggregated institutional indices, group size, the size of the resource system and wealth become significant. The results show that the level of cooperation in a community increases with group size and declines as the size of a resource system increases.

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

From a policy perspective, our results show that external enforcement of rules and regulations does not necessarily translate into sound biodiversity outcomes; rather, better outcomes are attainable when punishment is endogenised by local communities. This seems to suggest that communities should be supported in a way that promotes the emergence of robust institutions that are tailor-made to suit local needs; this will, in turn, facilitate good environmental husbandry. Future policy reforms should also consider the possibility of increasing the devolution of authority to local communities so that they can monitor each other and internalise enforcement of rules and regulations. The importance of endogenous punishment and increased autonomy falls within the context of Ostrom's design principles for successful CPR institutions. This result supports a large body of work that has already reached the same conclusions regarding community autonomy.

State authorities should reconsider the way in which they engage with farmers under resettlement schemes, because biodiversity suffers more under this type of arrangement than in communal areas. There is a need for appropriate institutional reforms that allow park authorities to work closely with resettlement schemes, while at the same time giving incentives for plot holders to work together for the improvement of the common pool resource. For example, farmers in resettlement schemes could set aside land for conservation by pooling resources instead of operating at plot level, which works against conservation efforts.

Wildlife management training, benefits and fencing could have important implications for policy formulation and design. Capacity building efforts of government agencies, NGOs and other stakeholders should complement each other to ensure that the necessary resources are mobilised and all communities receive the necessary training and resources. Both the extent to which communities benefit from wildlife conservation and the extent to which they are allowed to make important decisions about how benefits are distributed and used by the community affect incentives to conserve wildlife. Fencing the national park has a detrimental effect on wildlife, the CAMPFIRE project and the lives of people whose livelihoods depend on wildlife conservation. Recent studies demonstrated that fencing protected areas has unintended and ironic *effects* on the animals they are built to protect. Fencing national parks could significantly and permanently alter entire processes and landscapes by keeping ecosystem engineers such as wildebeest and elephants from their regular migrations, while others are accidentally killed by the fence if they attempt to cross it. The effects of the fence would certainly trickle down to the communities by reducing the amount of benefits they get from conservation since wildlife movement is severely restricted.

The implication of the study on our understanding of the role of institutions on wildlife conservation is threefold. First, local communities should be given more freedom so that they are able to invest in common pool resource institutions that are tailor made to suit their own needs. Viewed from this angle, our results support the idea of increased devolution of natural resource management functions from the RDC to CAMPFIRE projects. This also means that more benefits will flow directly to the community and thereby generate the necessary incentives to conserve wildlife. Second, local communities should not be viewed as bystanders or mere beneficiaries in wildlife conservation but as equal partners with an interest in conservation. Finally, more resources should be channelled towards the CAMPFIRE programme in order to equip projects with the necessary resources and capacity so that they can be self-reliant.

Literature cited

Agrawal, A. 2001. Common Property Institutions and Sustainable Governance of Resources. *World Development* 29(10):1649–1672.

Akpalu, W. and P. Martinsson. 2011. Ostracism and Common Pool Resource Management in a Developing Country – Young Fishers in the Laboratory. *Journal of African Economies* 21(2):266–306.

Andersson, O. and E. Wengström. 2011. Credible Communication and Cooperation: Experimental Evidence from Multi-Stage Games. *IFN Working Paper No. 883*.

Ashley, C. 2000. The Impacts of Tourism on Rural Livelihoods: Namibia's Experience. Overseas Development Institute (ODI). *Working Paper 128*.

Ashley, C. and D. Carney. 1999. *Sustainable Livelihoods: Lessons from Early Experience*. Department for International Development, Volume 7(1). London, UK: Russel Press, Nottingham.

Baland, J. M. and J. P. Platteau. 1996. *Halting Degradation of Natural Resources – Is there a Role for Rural Communities?* Rome, Italy: The Food and Agriculture Organisation (FAO) of the United Nations.

Baland, J. M. and J. P. Platteau. 2001. *Collective Action on the Commons: The Role of Inequality*. Belgium: CRED, University of Namur.

Balint, P. J. and J. Mashinya. 2006. The Decline of a Model Community-Based Conservation Project – Governance, Capacity, and Devolution in Mahenye, Zimbabwe. *Geoforum* 37(5):805–815.

Balint, P. J. and J. Mashinya. 2008. CAMPFIRE During Zimbabwe's National Crisis: Local Impacts and Broader Implications for Community-Based Wildlife Management. *Society & Natural Resources* 21:783–796.

Banerjee, S., P. Chatterji, and K. Lahiri. 2013. Effects of Psychiatric Disorders on Labor Market Outcomes – A Latent Variable Approach Using Multiple Clinical Indicators. *CESifo Working Paper Series No.* 4260.

Basurto, X. and E. Coleman. 2010. Institutional and Ecological Interplay for Successful Self-Governance of Community-Based Fisheries. *Ecological Economics* 69(5):1094–1103.

Basurto, X., S. Gelcich, and E. Ostrom. 2013. The Social-Ecological System Framework as a Knowledge Classificatory System for Benthic Small-Scale Fisheries. *Global Environmental Change* 23(6):1366–1380.

Baum, C. F., A. Lewbel, M. E. Schaffer, and O. Talavera. 2013. Instrumental Variables Estimation Using Heteroskedasticity-Based Instruments. United Kingdom Stata User's Group Meetings 2012 07, Stata Users Group.

Baur, I. 2014. *Analyzing and Modeling the use of Common Property Pastures in Grindelwald, Switzerland*. Dissertation, der Ludwig-Maximilians-Universität München.

Berkes, F. and C. Folke, eds. 1998. *Linking Social and Ecological Systems – Management Practices and Social Mechanisms for Building Resilience*. New York: Cambridge University Press.

Binder, C. R., J. Hinkel, P. W. G. Bots, and C. Pahl-Wostl. 2013. Comparison of Frameworks for Analysing Social-Ecological Systems. *Ecology and Society* 18(4):26.

Bouman, M., R. Heijungs, E. van der Voet, J. C. J. M. van den Bergh, and G. Huppes. 2000. Material Flows and Economic Models: An Analytical Comparison of SFA, LCA and Partial Equilibrium Models. *Ecological Economics* 32:195–216.

Boyd, R. and P. J. Richerson. 2002. Group Beneficial Norms can Spread Rapidly in a Structured Population. *Journal of theoretical biology* 215(3):287–296.

Brundtland, G. H., M. Khalid, S. Agnelli, S. Al-Athel, B. Chidzero, L. Fadika, V. Hauff, I. Lang, M. Shijun, M. M. de Botero, M. Singh, S. Okita, S. Ramphal, W. D. Ruckelshaus, M. Sahnoun, E. Salim, B. Shaib, V. Sokolom, J. Stanounik, and M. Strong. 1987. Our Common Future. Report of the World Commission on Environment and Development, United Nations.

Brunner, P. H. and H. Rechberger. 2005. *Practical Handbook of Material Flow Analysis*. New York, New York, USA: Lewis.

Cardenas, J. C. 2003. Real Wealth and Experimental Cooperation – Experiments in the Field Lab. *Journal of Development Economics* 70:263–289.

Cernea, M. ed. 1985. *Putting People First: Sociological Variable in Rural Development*. New York: Oxford University Press.

Chambers, R. 1983. *Rural Development: Putting the Last First*. Harlow: Longman.

Chamunorwa, A. 2010. Comparative Analysis of Agricultural Productivity Between Newly Resettled Farmers and Communal Farmers in Mashonaland East Province. Livelihoods after Land Reform in Zimbabwe, *PLAAS/IDS Working Paper 8*.

Chhatre, A. and A. Agrawal. 2008. Forest Commons and Local Enforcement. *Proceedings of the National Academy of Sciences of the United States of America* 105:36.

Child, B. and G. Barnes. 2010. The Conceptual Evolution and Practice of Community-Based Natural Resource Management in Southern Africa: Past, Present and Future. *Environmental Conservation* 37(3):283–295.

Cousins, B. 1993. Debating Communal Tenure in Zimbabwe. *Journal of Contemporary African Studies* 12(1):29–39.

Cox, M. 2014. Applying a Social-Ecological System Framework to the Study of the Taos Valley Irrigation System. *Human Ecology* 42:311–324.

DeGeorges, P. A. and B. K. Reilly, 2009. The Realities of Community Based Natural Resource Management and Biodiversity Conservation in Sub-Saharan Africa. *Sustainability* 1:734–788.

de Groot, R. S., M. A. Wilson, and R. M. J. Boumans. 2002. A Typology for the Classification, Description and Valuation of Ecosystem Functions, Goods and Services. *Ecological Economics* 41:393–408.

Delgado-Serrano, M. M. and P. A. Ramos, 2015. Making Ostrom's Framework Applicable to Characterise Social Ecological Systems at the Local Level. *International Journal of the Commons* 9(2):808–830.

Dore, D. 2001. Transforming Traditional Institutions for Sustainable Natural Resource Management: History, Narratives and Evidence from Zimbabwe's Communal Areas. *African Studies Quarterly* 5(3):1–18.

Durant, S. M., M. S. Becker, S. Creel, S. Bashir, A. J. Dickman, R. C. Beudels-Jamar, L. Lichtenfeld, R. Hilborn, J. Wall, G. Wittemyer, L. Badamjav, S. Blake, L. Boitani, C. Breitenmoser, F. Broekhuis, D. Christianson, G. Cozzi, T. R. B. Davenport, J. Deutsch, P. Devillers, L. Dollar, S. Dolrenry, I. Douglas-Hamilton, E. Dröge, E. FitzHerbert, C. Foley, L. Hazzah, J. G. C. Hopcraft, D. Ikanda, A. Jacobson, D. Joubert, M. J. Kelly, J. Milanzi, N. Mitchell, J. M'Soka, M. Msuha, T. Mweetwa, J. Nyahongo, E. Rosenblatt, P. Schuette, C. Sillero-Zubiri, A. R. E. Sinclair, M. R. S. Price, A. Zimmerman, and N. Pettorelli. 2015. Developing Fencing Policies for Dryland Ecosystems. *Journal of Applied Ecology* 52:544–551.

Efron, B. 1979. Bootstrap Methods: Another look at the Jackknife. *Annals of Statistics* 7:1–26.

Emran, M. S., V. Robano, and S. C. Smith. 2012. Assessing the Frontiers of Ultra-Poverty Reduction: Evidence from Targeting the Ultra-Poor (CFPR/TUP) program in Bangladesh. *IIEP Working Paper IIEP-WP*-2009-*06*.

Ferguson, K. and J. Hanks. 2012. The Effects of Protected Area and Veterinary Fencing on Wildlife Conservation in Southern Africa. *Parks* 18(1):49–60.

Fischbacher, U., S. Gächter, and E. Fehr. 2001. Are People Conditionally Cooperative? Evidence from a Public Goods Experiment. *Economics Letters* 71:397–404.

Folke, C. 2006. Resilience: The Emergence of a Perspective for Social-Ecological Systems Analyses. *Global Environmental Change* 16:253–267.

Frost, P. G. H. and I. Bond. 2008. The CAMPFIRE Programme in Zimbabwe: Payments for Wildlife Services. *Ecological Economics* 65:776–787.

Gandiwa, E. 2011. Preliminary Assessment of Illegal Hunting by Communities Adjacent to the Northern Gonarezhou National Park, Zimbabwe. *Tropical Conservation Science* 4(4):445–467 (Mongabay.com Open Access Journal).

Gandiwa, E. and S. Kativu. 2009. Influence of Fire Frequency on *Colophospermum Mopane* and *Combretum Apiculatum* Woodland Structure and Composition in Northern Gonarezhou National Park, Zimbabwe. *Koedoe* 51(1): Article #685, 13 p, DOI: 10.4102/koedoe, v51i1.685.

Gintis, H. 2004. Modeling Cooperation Among Self-Interested Agents – A Critique. *Journal of Socio-Economics* 33:697–717.

Haberl, H., M. Fischer-Kowalski, F. Krausmann, H. Weisz, and V. Winiwarter. 2004. Progress Towards Sustainability? What the Conceptual Framework of Material and Energy Flow Accounting (MEFA) Can Offer. *Land Use Policy* 21:199–213.

Hardin, G. 1968. The Tragedy of the Commons. *Science* 162:1243–1248.

Hodgson, G. M. 2006. What Are Institutions? *Journal of Economic Issues* 60(1):1–25.

Horowitz, J. L. 2001. The Bootstrap, Ch. 52. In *Handbook of Econometrics*, Vol. 5, eds. J. J. Heckman and E. E. Leamer. North-Holland: Amsterdam.

Horowitz, J. L. 2003. The Bootstrap in Econometrics. *Statistical Science* 18:211–218.

Hughes, R. and F. Flintan. 2001. Integrating Conservation and Development Experience: A Review and Bibliography of the ICDP Literature. In *Biodiversity and Livelihoods*, Issues No. 3, London: International Institute for Environment and Development.

Jones, B. 2001. The Evolution of a Community-Based Approach to Wildlife Management at Kunene, Namibia. In *African Wildlife and Livelihoods: The Promise and Performance of Community Conservation*. eds. D. Hulme and M. Murphree. Oxford: James Currey Ltd.

Jones, B. and M. Murphree. 2001. The Evolution of Policy on Community Conservation in Namibia and Zimbabwe. In *African Wildlife and Livelihoods: The Promise and Performance of Community Conservation*, eds. D. Hulme, and M. Murphree, 38–58. Oxford, UK: James Currey Ltd.

Lewbel, A. 2012. Using Heteroskedasticity to Identify and Estimate Mismeasured and Endogenous Regressor Models. *Journal of Business and Economic Statistics* 30(1):67–80.

Marimira, C. S. 2010. Institutions, Leadership and Service Delivery in New Resettlement Areas of Zimbabwe. Livelihoods after Land Reform in Zimbabwe. *PLAAS/IDS Working Paper 6*.

Martin, R. B. 1986. *Communal Areas Management Programme for Indigenous Resources (April* 1986*, revised edition). Branch of Terrestrial Ecology*. Harare, Zimbabwe, (mimeo): Department of National Parks and Wild Life Management.

McCarthy, N., C. Dutilly-Diane, and B. Drabo. 2002. Cooperation, Collective Action and Natural Resource Management in Burkina Faso – A Methodological Note. *CAPRi Working Paper No. 27*, CGIAR.

Meinzen-Dick, R., M. Di Gregorio, and N. McCarthy. 2004. Methods for Studying Collective Action in Rural Development. *Agricultural Systems* 82(3):197–214.

Mishra, V. and R. Smyth. 2015. Returns to Schooling in Urban China: New Evidence From Two Datasets Using Conventional and Heteroskedasticity-Based Instruments. *Economic Modelling* 47:166–173.

Mohamed-Katerere, J. 2001. Participatory Natural Resource Management in the Communal Lands of Zimbabwe: What Role for Customary Law? *African Studies Quarterly* 5(3).

Mouillot, D. and A. Lepretre. 1999. A Comparison of Species Diversity Estimators. *The Society of Population Ecology and Springer* 41:203–215.

Muchapondwa, E. 2003. The Economics of Community-Based Wildlife Conservation in Zimbabwe. PhD dissertation, Sweden: Göteborg University.

Muir-Leresche, K. and R. H. Nelson. 2000. *Private Property Rights to Wildlife – The Southern Africa Experience*. Torino, Italy: University of Zimbabwe, University of Maryland and the International Centre for Economic Research (ICER).

Murombedzi, J. C. 1999. Devolution and Stewardship in Zimbabwe's CAMPFIRE Programme. *Journal of International Development* 11(2):287–293.

Murombedzi, J. 2000. Committees, Rights, Costs and Benefits: The Question of Natural Resource Stewardship and Community Benefits in Zimbabwe's CAMPFIRE Programme. In *African Wildlife and Livelihoods: The Promise and Performance of Community Conservation*. eds. D. Hulme and M. Murphree. Oxford: James Currey.

Murphree, M. W. 1991. *Communities as Institutions for Resource Management. CASS Occasional Paper Series, Centre for Applied Social Sciences*. Harare, Zimbabwe, (mimeo): University of Zimbabwe.

Murphree, M. 1993. *Communal Land Wildlife Resources and Rural District Council Revenues. Occasional Paper Series, Centre for Applied Social Sciences*. Harare, Zimbabwe, (mimeo): University of Zimbabwe.

Murphree, M. W. 1994. The Role of Institutions in Community-Based Conservation. In *Natural Connections: Perspectives in Community-Based Conservation*. eds. D. Western, R. M. Wright, and S. C. Strum. Washington: Island Press.

Murphree, M. W. 2001. Community, Council and Client: A case study in Ecotourism Development from Mahenye, Zimbabwe. In *African Wildlife and Livelihoods. The Promise and Performance of Community Conservation*. eds. D. Hume and M. W. Murphree. London: James Currey.

Murphree, M. W. 2005. Congruent Objectives, Competing Interests, and Strategic Compromise. In *Communities and Conservation*. eds. J. P. Brosius, A. L. Tsing, C. Zerner. Walnut Creek, CA: AltaMira Press.

Murphy, J. J. and J. C. Cardenas. 2004. An Experiment on Enforcement Strategies for Managing a Local Environmental Resource. *Journal of Economic Education* 31(1):47.

Nagendra, H. 2002. Opposite Trends in Response for the Shannon and Simpson Indices of Landscape Diversity. *Applied Geography* 22:175–186.

National Research Council. 2002. The Drama of the Commons. Committee on the Human Dimensions of Global Change. In *Division of Behavioral and Social Sciences and Education*. eds E. Ostrom, T. Dietz, N. Dolsˇak, P. C. Stern, S. Stovich, and E. U. Weber. Washington, DC: National Academy Press.

Newell, B., C. L. Crumley, N. Hassan, E. F. Lambin, C. Pahl-Wostl, A. Underdalf, and R. Wasson. 2005. A Conceptual Template for Integrative Human–Environment Research. *Global Environmental Change* 15:299–307.

North, D. C. 1990. *Institutions, Institutional Change, and Economic Performance*. Cambridge: Cambridge University Press.

Ntuli, H. and E. Muchapondwa. 2017a. A Bioeconomic Analysis of Community Wildlife Conservation in Zimbabwe. *Journal for Nature Conservation* 37:106–121.

Ntuli, H. and E. Muchapondwa. 2017b. Effects of Wildlife Resources on Community Welfare in Southern Africa. *Ecological Economics* 131:572–583.

Olson, M. 1965. *The Logic of Collective Action – Public Goods and the Theory of Groups*. Harvard Economic Studies Volume CXXIV, Cambridge, MA, London: Harvard University Press.

Ostrom, E. 1990. *Governing the Commons: The Evolution of Institutions for Collective Action – The Political Economy of Institutions and Decisions*. Cambridge University Press.

Ostrom, E. 1994. Institutional Analysis, Design Principles and Threats to Sustainable Community Governance and Management of Commons. In *Community Management and Common Property of Coastal Fisheries in Asia and the Pacific: Concepts, Methods and Experiences*, ed. R. S. Pomeroy. Manila, Philippines: International Center for Living Aquatic Resources Management.

Ostrom, E. 1998. A Behavioural Approach to the Rational Choice Theory of Collective Action: Presidential Address, American Political Science Association. *American Political Science Review* 92(1):1–22.

Ostrom, E. 2000. Collective Action and the Evolution of Norms. *Journal of Economic Perspectives* 14(3):137–158.

Ostrom, E. 2003. How Types of Goods and Property Rights Jointly Affect Collective Action. *Journal of Theoretical Politics* 15(3):239–270.

Ostrom, E. 2007. A Diagnostic Approach for Going beyond Panaceas. Centre for the Study of Institutions, Population & Environmental Change. *Proceedings of the National Academy of Sciences of the United States of America* 104(39):15181–15187.

Ostrom, E. 2009. A General Framework for Analysing Sustainability of Social-Ecological Systems. *Science* 325:419–422.

Ostrom, E. 2010. Analysing Collective Action. *Agricultural Economics, The Journal of the International Association of Agricultural Economists (IAAE)* 41(1):155–166.

Ostrom, E. and T. K. Ahn. 2009. The Meaning of Social Capital and its Link to Collective Action. In *Handbook of Social Capital: The Troika of Sociology, Political Science and Economics*, eds. G. T. Svendsen and G. L. H. Svendsen, 17–35. Northhampton, USA: Edward Elgra.

Ostrom, E., R. Gardner, and J. Walker. 1994. *Rules, Games, and Common-Pool Resources*. Ann Arbor: University of Michigan Press.

Ostrom, E., M. A. Jansses, and J. M. Anderies. 2007. Going Beyond Panaceas. *Proceedings of the National Academy of Sciences of the United States of America* 104(39):15176–15178.

Pennings, J. M. E. and R. M. Leuthold. 2000. The Role of Farmer's Behavioural Attitudes and Heterogeneity in Futures Contracts Usage. *American Journal of Agricultural Economics* 4(82):908–919.

Posner, E. 2005. Sources of Institutional Change: The Supranational Origins of Europe's New Stock Markets. *World Politics* 58:1–40.

Pretty, J. and D. Smith. 2004. Social Capital in Biodiversity Conservation and Management. *Conservation Biology* 18(3):631–686.

Pretty, J. and H. Ward. 2001. Social Capital and the Environment. *World Development* 29(2):209–227.

Rihoy, E., C. Chirozva, and S. Anstey. 2007. 'People are not Happy' – Speaking up for Adaptive Natural Resource Governance in Mahenye. An occasional paper series No. 31, PLAAS.

Roe, D., F. Nelson, and C. Sandbrook, eds. 2009. *Community Management of Natural Resources in Africa: Impacts, Experiences and Future Directions, Natural Resource Issues No. 18*. London, UK: International Institute for Environment and Development.

Sandker, M., M. B. Campbell, Z. Nzooh, T. Sunderland, V. Amougou, L. Defo, and J. Sayer. 2009. Exploring the Effectiveness of Integrated Conservation and Development Interventions in a Central African Forest Landscape. *Biodiversity Conservation* 18(11):2875–2892. DOI 10.1007/s10531-009-9613-7.

Saunders, P. F. 2014. The Promise of Common Pool Resource Theory and the Reality of Commons Projects. *International Journal of the Commons* 8(2):636–656.

Schellnhuber, H.-J., P. J. Crutzen, W. C. Clark, and J. Hunt. 2005. Earth System Analysis for Sustainability. *Environment: Science and Policy for Sustainable Development* 47(8):10–25.
Scholz, R. W. and C. R. Binder. 2004. Principles of Human Environment Systems Research. In *iEMSs* 2004 *International Congress: Complexity and Integrated Resources Management. International Environmental Modelling and Software Society*, eds. C. Pahl, S. Schmidt, and T. Jakeman, 791–796. Germany: Osnabrueck.
Scholz, R. W., C. R. Binder, and D. J. Lang. 2011. The HESFramework. In *Environmental Literacy in Science and Society: From Knowledge to Decisions*, ed. R. W. Scholz, 453–462. Cambridge, UK: Cambridge University Press.
Scoones, I. 1998. *Sustainable Rural Livelihoods: A Framework for Analysis*. IDS Working Paper 72. Brighton, UK: Institute of Development Studies, University of Sussex.
Sebele, L. S. 2010. Community-Based Tourism Ventures, Benefits and Challenges: Khama Rhino Sanctuary Trust, Central District, Botswana. *Tourism Management* 31(2010):136–146.
Songorwa, A. N. 1999. Community-Based Wildlife Management (CWM) in Tanzania. Are the Communities Interested? *World Development* 27(12):2061–2079.
Taylor, R. 2009. Community Based Natural Resource Management in Zimbabwe: the Experience of CAMPFIRE. *Biodiversity Conservation* 18:2563–2583.
Trouwborst, A., F. Fleurke, and J. Dubrulle. 2016. Border Fences and their Impacts on Large Carnivores, Large Herbivores and Biodiversity: An International Wildlife Law Perspective. *Review of European, Comparative and International Environmental Law* 25(3):291–306.
Useche, P. 2013. Collective Action in Common Pool Resource Management, Including Heterogeneity of Opportunities and Exit Options. *Natural Resources* 4:483–489.
Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of Earth's ecosystems. *Science* 277:494–499.
Wade, R. 1987. The Management of Common Property Resources: Collective Action as an Alternative to Privatization and State Regulation. *Cambridge Journal of Economics* 11:95–106.
Wade, R. 1994. *Village Republics – Economic Conditions for Collective Action in South India*. San Francisco, California: Institute for Contemporary Studies.
Young, O. R., F. Berkhout, G. C. Gallopín, M. A. Janssen, E. Ostrom, and S. van der Leeuw. 2006. The Globalization of Socio-Ecological Systems: An Agenda for Scientific Research. *Global Environmental Change* 16:304–316.