Using Economic Incentives to encourage Conservation in Bioregions in South Africa

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Abstract

This paper starts from the now widely-held premise that biodiversity conservation ought to take place both inside and outside protected areas if biodiversity targets are to be met. Given the potential inter-linkages of areas inside and outside protected areas in ecosystems, the ultimate structure of biodiversity conservation should be bioregional landscape management. A framework for studying the factors affecting biodiversity conservation in bioregions is suggested. While many factors might affect biodiversity conservation, the use of economic incentives is argued to be potentially one of the most effective mechanisms for mainstreaming biodiversity conservation in bioregions. Institutions are singled out as one important class of socio-economic arrangements directly associated with economic incentives. Institutions are thus likely to be a major determinant of the vulnerability or success of biodiversity conservation. The paper uses South African examples, and concludes by outlining the research issues important in understanding the role of economic incentives in that context.

Keywords: biodiversity conservation, bioregions, economic incentives, institutions, mainstreaming

1 Introduction

Biodiversity conservation[1] has traditionally been practised in protected areas. However, authors such as Hansen and DeFries (2007) observe that many protected areas are not successfully conserving biodiversity, often despite adequate management within their borders. They suggest that the major reason for this could be the expansion and intensification of land use in the areas adjoining the protected areas. Changes in land use outside protected areas can alter ecological function inside protected areas and result in biodiversity loss given that protected areas are almost always parts of larger ecosystems.

The concept of ecosystem management originated from the goal of managing regional landscapes to maintain the ecological integrity of the protected areas contained therein. Since the early 1970s, UNESCO has advocated for the management of the lands around protected areas along a gradient of decreasing intensity of use towards the protected area boundaries. However, the abovementioned

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[1]We use the term "biodiversity conservation" in the same way as Blignaut and Aronson's (2008) use of the term "biodiversity maintenance" which incorporates preservation, restoration and rehabilitation of biodiversity.

conservation approach assumes that protected areas alone are capable of meeting biodiversity targets[2] and therefore merely guaranteeing their persistence is sufficient to meet biodiversity targets.[3] Indeed, there are many conservation priority areas outside of, often even far from, protected areas. Also, the total size of area under conservation-friendly land use of one or another kind may be functionally crucial. Thus, for successful biodiversity conservation, there is a need for bioregional management, where the region in question encompasses the protected ecosystem or the unprotected ecosystem or portions of both. For the purposes of this paper, the bioregion is simply either an agglomeration of smaller areas, for instance, districts at a sub-national scale, or an agglomeration of bigger areas at an adjacent, international scale, with some reason for being considered together.[4]

Good management inside the protected area may be relatively easy to achieve, but managing outside the protected area requires ways to tackle heightened complexity. Merely applying protectionist strategies outside protected areas, as inside the protected areas, will most likely not guarantee the achievement of biodiversity goals and targets, due to the presence of other competing land uses. Indeed, the biodiversity research community has responded to the challenge of biodiversity conservation outside protected areas by calling for mainstreaming biodiversity conservation in society generally (Cowling, 2005). Mainstreaming entails changing the behaviour of individuals and organisations towards the adoption of norms, values, and practices which promote biodiversity whenever they make decisions that are likely to affect it, particularly outside of protected areas. The widely cited mechanism of mainstreaming is effective communication of the issues to key stakeholders (*ibid*).

The most effective mechanism of mainstreaming may well be the tweaking of the economic conditions of individuals and organisations, since biodiversity conservation can be seen as an economic issue for two basic reasons. Firstly, biodiversity is a component of natural capital, which provides a flow of valuable ecosystem goods and services and therefore a source of economic wellbeing (Millenium Ecosystem Assessment, 2005; Blignaut and Aronson, 2008). Secondly, the extent of biodiversity conservation outside protected areas is largely guided by the norms, values and practices of market economies which promote the pursuit of economic optimisation objectives.[5]

Thus, a potentially effective mechanism for mainstreaming biodiversity conservation might be the use of economic incentives.[6] Economic incentives are potentially capable of binding actors to support norms, values, and practices that promote biodiversity persistence. This paper seeks to unravel the ways through which economic incentives can be used to enhance biodiversity conservation in bioregions. It does so by presenting what is believed to be a widely-applicable generic framework, but set in a predominantly South African context.

The structure of the rest of the paper is as follows. Section 2 suggests a conceptual framework for studying the factors affecting biodiversity conservation in bioregions. Section 3 looks at the role of economic incentives in bioregions. In section 4, institutions are singled out as one important class of socio-economic arrangements directly associated with economic incentives and likely to be a

[2] Biodiversity targets in this instance are quantitative targets that tell us how much of a biodiversity feature needs to be conserved in order to conserve a representative sample of biodiversity pattern and key ecological and evolutionary processes. Targets are expressed as, for example, numbers of hectares of a land class. Biodiversity targets represent thresholds or tipping points beyond which irreversible loss of ecosystem functioning or of species is likely to occur.

[3] This strategy might not assist biodiversity conservation very much because protected areas are not necessarily in the right places, or not large enough, for conserving biodiversity. Initial protected areas placement was not necessarily meant to achieve currently envisaged biodiversity targets. Furthermore, there may be critical biodiversity areas far from the protected areas that are relegated to high intensity use in the above approach.

[4] It should be noted here that the South African Department of Environmental Affairs and Tourism's (DEAT) bioregion, defined in South Africa's National Environmental Management: Biodiversity Act (Act 10 of 2004), contains "whole or several nested ecosystems and is characterised by its landforms, vegetation cover, human culture and history". Furthermore, there is already a host of different bioregion-type formulations: UNESCO biosphere, World Heritage sites, Conservation International Hotspots, AWF Heartlands, Transfrontier Conservation Areas. The principles developed in this paper generally apply to any chosen bioregion definition.

[5] Biodiversity conservation is only one land use in a suite of alternative land uses forming the land use mosaic.

[6] Economic incentives refer to mechanisms that change the behaviour of actors with respect to economic choices by altering their economic conditions. Positive economic incentives reward actors for complying with required actions while negative economic incentives punish actors for non-compliance.

major determinant of the vulnerability or success of biodiversity conservation in bioregions. Section 5 concludes by raising the key research issues that are important in understanding the role of economic incentives in bioregions in South Africa.

2 The conceptual framework for studying the factors affecting biodiversity in bioregions

Biodiversity conservation requires land. Thus, the extent of biodiversity conservation in a bioregion will depend on the configuration of the land use mosaic in that bioregion. A number of different land uses including 'pure' conservation, crop agriculture, livestock farming, natural resource harvesting, rural settlements, urban centres, logging, mining, other commercial industries, etc., can comprise the land use mosaic. The land use mosaic reflects people's value systems and management choices in response to a range of social, technological, ecological, economic and political driving factors (Biggs *et al.*, 2004; DeFries *et al.*, 2007). Thus, ultimately the extent of biodiversity conservation in a bioregion depends on a myriad of factors affecting land use decisions by a collection of stakeholders. In this section, we present a conceptual framework for exploring the factors that influence the configuration of the land use mosaic in a bioregion and therefore biodiversity conservation in that bioregion (see Figure 1).[7]

As indicated earlier, the configuration of a land use mosaic (Figure 1, Box 1.1) and therefore biodiversity conservation in that mosaic is shaped by a range of factors. Within the localised socio-ecological-economic system, land use dynamics are determined by (i) people's assets in the form of the different types of capital they have at their disposal at any given time (Figure 1, Box 1.2), (ii) the various systems of tenure which underpin the land use mosaic (Figure 1, Box 1.3), (iii) the extent of regional development (Figure 1, Box 1.4), and (iv) the economic approaches used in decision-making (Figure 1, Box 1.5).

People's asset registry consists of ecological, financial, physical, human and social capital (Scoones, 1998; DFID, 1999), as well as cultural capital (Figure 1, Box 1.2). Ecological capital refers to the ecological system and nature, which provides a range of ecosystem goods and services to support human livelihoods and well-being. Direct, tangible benefits from nature include the use of timber, firewood, medicines, food, water, honey, grazing land etc., for a variety of different landowners and user groups (Millennium Ecosystem Assessment, 2003; Kumar, 2005). Indirect benefits are nutrient cycling, water provision, groundwater recharge and bioremediation. In addition, intangible benefits include reviving or strengthening people's spiritual, cultural, traditional and aesthetic connections with the ecological system and nature. Hence, the natural potential of the land plays an important role in determining which land use options are open to the landowner or user (Aalders, 2008).

Financial capital includes cash, savings and investments, and underpins both formal and informal trade in ecosystem goods and services (Kumar, 2005).[8] Physical capital comprises infrastructure, telecommunications and transport networks, where having the appropriate infrastructure and equipment can help reduce labour costs and improve productivity, and thus, facilitate trade and industry (Jansen *et al.,* 2006). Human capital refers to people's skills and expertise, where being able to access current knowledge and innovations enables the use of environmentally sound practices, and promotes sustainable resource use and conservation (Kumar, 2005). Social capital constitutes social networks and relationships, which enable the sharing of resources (including knowledge) in kinship and across

[7]Mathematically, this framework can be represented at the aggregate level as a system of the following two simultaneous equations: *Biodiversity conservation=f(land use mosaic); Land use mosaic=g(values, social factors, technological factors, ecological factors, economic factors, political factors| economic approaches used by economic agents)* and the following reduced form equation: *Biodiversity conservation=h(values, social factors, technological factors, ecological factors, economic factors, political factors| economic approaches used by economic agents).*

[8]The use of ecosystem goods and services by local communities also serves a safety net function in times of hardship, and also plays a cost saving role, as the money saved by not having to pay for basic requirements can be used for other household purposes (Emerton, 2001; Shackleton and Shackleton, 2004).

business interests, diversification of risk across activities, access to new or more diverse markets, and facilitates the marketing and promotion of products (*ibid*). Cultural capital includes people's belief systems, customs and traditions, which influence their natural resource use and management, with particular relevance to traditional or indigenous communities.

However, it is having an adequate combination of these capitals that can unlock development opportunities for the landowner or user. For example, membership in a social group (social capital) may be necessary for access to certain land tenure rights (natural capital) that is necessary for access to credit (financial capital), which in turn, is needed to purchase inputs to take advantage of a new market (Meinzen-Dick and Adato, 2001). Hence, the different types of capital which people possess determine resource restrictions on the nature of development opportunities available to them (e.g. type of development, size and scope of the development, etc.).

Moreover, all development opportunities are subject to institutional constraints in terms of formal laws of access and use rights to land and resources, as well as informal rules, taboos or codes of conduct, particularly with regard to traditional leadership and natural resource management practices (Oström, 1990; Grundy and Michell, 2004). Although direct evidence of the relationship between landowners' decision-making and land use (including land cover) is difficult to obtain due to confidentiality issues, there is some evidence that land ownership represents the unit for decision-making, as forests have been actively maintained along straight-lined boundaries as opposed to the diffuse boundaries of natural forest patterns (Jansen *et al.,* 2006). In this instance, it is probable that secure tenure and use rights over forest management created an incentive for people to conserve or protect this vegetation type.

Hence, this framework recognises four major types of tenure which span rural, urban and peri-urban settings, all of which may occur within a particular bioregion. They are state-owned land, communal land, private land and commons (Figure 1, Box 1.3). State-owned land is managed by the state with controlled access to individuals and communities for various resource use purposes (Grundy and Michell, 2004). Communal land refers to a type of tenure arrangement whereby a community or 'contiguous group of people with like skills, socio-economic background and attitude' (Fabricius, 2004) are assigned rights of ownership to land by the state, and collectively make decisions about its use (Grundy and Michell, 2004; Reid and Turner, 2004). Private land refers to a type of tenure arrangement whereby an individual or private enterprise is assigned rights of ownership and use to land (Grundy and Michell, 2004). Commons constitute state-owned land, usually along the periphery of urban centres, e.g. towns or cities, which can be hired out for use by the public but which are often used by all people as 'open access' areas due to a lack of excludability.

It is important to understand how resources flow between these various types of tenure arrangements within a bioregional land use mosaic to promote its sustainable development (Oström, 1990; World Resources Institute, 2005). Joint venture partnerships between the state, local communities and the private sector, encourages a cross-flow of resources between different stakeholders and across different tenure arrangements (i.e. state, communal and private) to promote biodiversity conservation, local economic empowerment and capacity building. For example, new legislation on forest management in South Africa, namely the National Forests Act (No. 84 of 1998), makes allowances for the management of state forests, as well as indigenous forests on communal land, and underpins the Participatory Forestry Management (PFM) programme, championed by the Department of Water Affairs and Forestry (Grundy and Michell, 2004; Willis, 2004). This necessitates granting local communities access to state forests for the harvesting of ecosystem goods and services to reduce poaching, as well as the state providing an extension service to local communities to empower them to use and manage their own resources in a better way. However, for local communities to begin to conserve resources, it is important that they identify the value in doing so for subsistence, traditional or customary and commercial purposes. With regard to the latter, private sector partners often play an important role in the training and capacity-building of local communities, and help to connect them to formal product markets. The partnership works because local communities grow their social and financial capital while their private sector partners can cut management and operational costs.

People's asset registry can also interact with the land tenure system to influence land ownership and land use practices (Gupta, 1999; Lebel *et al.,* 2006). For example, local communities with strong cultural capital (i.e. good local knowledge, strong collective memory and a tendency to value a traditional lifestyle), may be more likely to engage in conservation and/or nature-based tourism activities that build on these assets (e.g. cultural villages) than more 'western' tourism initiatives. In contrast, the Makuleke land claim on part of the Kruger National Park witnessed the local community invest in a luxury game lodge and wildlife safaris, arguably at the expense of their local culture (Reid and Turner, 2004). Thus, the extent to which local communities engage in conservation and protect their local culture is an area that still requires further research.

The extent of regional development, measured by indicators such as income, employment, health and food consumption (Figure 1, Box 1.4), impacts on the land use mosaic (Biggs *et al.,* 2004), and feeds back to strengthening or weakening the asset base and making people more or less vulnerable to external factors (Meinzen-Dick and Adato, 2001). The challenge remains that of providing incentives for land uses complementary to conservation, and disincentives for land uses conflicting with conservation within the bioregion. For example, along the southern Cape coast of South Africa, the coastal forests of Tsitsikamma National Park and Wilderness National Park are managed for multifunctional sustainability. This means that the forests are managed for biodiversity conservation, small scale timber logging (where trees displaying early signs of mortality, across their age and size class distributions, are selected, logged and sold under auction, predominately to the furniture industry), as well as for the sustainable use of non-timber forest products by local communities (e.g. harvesting of seven week ferns for ornamental purposes, collection of tree seedlings for propagation in community nurseries etc.). Taken on its own, the small scale timber logging activities operate at a net loss of R3.5 million/annum. However, the downstream benefits justify timber logging as a complementary land use to biodiversity conservation, and promote this type of land use option above others such as mining, which would potentially provide greater short- to medium-term economic returns but which would conflict with conservation because of the destructive nature of the mining process.[9]

The framework also assumes that there is a set of economic approaches that landowners and users consult as rules of thumb to guide their land use decisions (Figure 1, Box 1.5). The first rule of thumb is that the benefits of an intervention, land use or development initiative must outweigh the costs, and the second rule is that any type of intervention needs to be cost-effective in achieving the desired goal. Borrowing from conventional consumer economic theory, approaches such as cost benefit analysis and the assessment of the cost effectiveness of institutions that underpin people's land use decisions can help to identify such incentive and disincentive mechanisms. Environmentally adjusted cost benefit analysis quantifies, as far as is possible, the total benefits (i.e. ecological, financial, physical, human, social and cultural benefits) from practicing a particular land use, measured in terms of the satisfaction gained by the landowner or user. However, as not all benefits can be maximised concurrently (e.g. conservation and mining), the use of land and/or the ecosystem goods and services it provides for a particular purpose comes at the cost of using the same land for another purpose (More *et al.,* 1996). Environmentally adjusted cost benefit analysis also quantifies, as far as is possible, the total costs (i.e. ecological, financial, physical, human, social and cultural costs) incurred by the landowner or user to utilise land or other ecosystem goods and services for a particular purpose (Pearce and Turner, 1990; Field, 2002). But it is the trade-off between the costs and benefits that determines people's land use decisions, where the option that yields 'good enough' net benefits is favoured over others (Sent, 2004).

Consequently, this framework draws on the theory of bounded rationality, and assumes that landowners and users refer to a stable set of rules (i.e. rules of thumb) by which to make their land use decisions, acting with incomplete knowledge (i.e. not knowing all past and future possibilities),

[9] The downstream benefits from timber logging (only) include R5.8 million/annum in taxes to the state, a contribution of R20 million/annum to the regional Gross Domestic Product and 640 jobs, which ultimately contributes to poverty alleviation, improved incomes and increased food security (Armin Seydack, personal communication).

towards meeting a specific target (near but not necessarily the optimal choice) (Gans, 1996; Sent, 2004). Therefore, the user's learning involves changing their choices on the basis of past outcomes to move nearer towards optimality (Sent, 2004). If the rules work well, they are retained and refined by the user whereas if the rules work poorly, they are used less and less, and eventually abandoned (Aumann, 1997).

It must also be acknowledged that other external factors can impact on biodiversity conservation in the bioregion (Biggs *et al.,* 2004). External ecological factors (Figure 1, Box 2) including natural disasters such as droughts, floods, biological invasions and climate change can have direct impacts on land use choices and biodiversity conservation outcomes. External socio-economic factors (Figure 1, Box 3) including political change (e.g. change in the ruling party), economic change (e.g. change in market prices) and institutional change (e.g. emergence of new institutions and power players) can influence tenure arrangements, development opportunities and land use options.

Other key elements of the framework are that it is dynamic through space and time (Gunderson and Holling, 2002), inclusive of formal and informal conservation (e.g. conservation ranging from formal protected areas to local taboos over protected or sacred forests) and embraces the principles of adaptive management (Biggs and Rogers, 2003) acknowledging multiple sustainable development options and strategies for a particular region. On the scale issue, the relative importance of the ecological, social and economic components may differ across scales. For example, at the scale of a park, ecological objectives such as conserving ecosystem functions, or in some cases, restoring ecosystem functions and processes and promoting habitat heterogeneity may take precedence over social objectives such as maximising the tourist experience, or economic objectives such as ensuring that the park is financially self-sustaining (Du Toit, Rogers and Biggs, 2003). However, a park forms part of a wider network, whereby it is possible to cross-subsidise the economically less productive parks, thus justifying their existence despite operating at a loss, and influencing the importance of different ecological, social and economic factors with scale.

3 The role of economic incentives in biodiversity conservation in a bioregion

As shown in figure 1, the decision about land use is made using particular economic approaches which assess the economic incentives or disincentives generated by a range of factors. Thus, what makes biodiversity conservation a preferred land use is the associated structure and magnitude of economic incentives in the range of factors that landowners and users consider in land use choice. As such, the way to affect the land use choices would be to tweak the economic incentives to which landowners and users experience respond. As will be argued later, it should be possible for landowners and users to experience new economic incentives from the existing range of driving factors or a modified range of driving factors such that they can be influenced to make land use choices which favour biodiversity conservation. This section unravels the range of economic incentives that can be used to influence the land use choices in favour of biodiversity conservation.

Biodiversity conservation plays an important role in creating opportunities and challenges that ultimately impact the sustainable development of a bioregion. This is because sustainable development implies the use and management of resources to provide benefits for the current generation without compromising the options available to future generations (Kumar 2005). Central to the concept of sustainable development is conservation, as it is through the act of valuing resources that people may begin to understand the need to conserve them for future generations (Duffy, Corson and Grant, 2001).

Moreover, the ecological system provides a number of ecosystem goods and services to society, either directly or indirectly, upon which economies and development are dependent. Yet despite this seemingly mutualistic relationship, many ecosystems have been unsustainably used and managed (Millennium Ecosystem Assessment, 2003; Biggs *et al.,* 2004; Kumar, 2005). Biodiversity conservation has tended to be unsuccessful because (i) it usually has "public good" characteristics of non-excludability and indivisibility, which makes its provision prone to free-riding behaviour, (ii) economic agents interact with biodiversity through various systems of tenure such as "open access", which usually creates incentives for accelerated rates of use due to the lack of assurance that biodiversity resources saved in the present will be available in the future, and (iii) biodiversity might not, in an economic sense, successfully compete against other land uses which offer better livelihood options.

It is also believed that poor use and management of ecosystems has resulted because conventional economic approaches have been unable to capture their 'true' value, and are often limited to situations where ecosystems are relatively intact and functioning properly (Emerton 2001a; Fabricius *et al.,* 2004).[10] While that may be true, it could be the case that weak incentive inputs into conventional economic approaches of decision-making result in the unsustainable use and management of ecosystems. For example, one of the reasons communal landowners and users invest in agriculture or livestock farming rather than biodiversity conservation is the difference in scale and distribution of the costs and benefits of the alternative land uses.[11] For communal landowners and users, the benefits from agriculture or livestock farming are high (e.g. they may generate a diversity of products such as crops, meat and milk for subsistence and commercial use) while the costs are low and may consist of labour and transport expenses (Shackleton, Shackleton and Cousins, 2001). The net benefits are positive albeit at the local scale often because of poor infrastructure and the lack of access to markets (Ashley, 1996). In comparison, the benefits from biodiversity conservation are low because the associated capital costs and management expenses of supporting big value, highly mobile wildlife species exceeds that of agriculture or livestock farming, and generally requires greater inputs in terms of resources (e.g. land, fencing, game guards etc.) in order to turn a profit. Thus, the net benefits are negative (Ashley, 1996; Tisdell, 2004).

The threat of the possibility of (i) free riding behaviour, (ii) open access tenure, (iii) failure of biodiversity to provide livelihoods, and (iv) the use of non-comprehensive economic approaches, together create the need to constrain or synergise human and enterprise actions with regards to their interaction with biodiversity. One could do this by use of (i) command-and-control system or (ii) economic incentives. Each of these options is likely to be better in particular circumstances – it mainly depends on the nature of the threat to biodiversity. However, while a command-and-control system imposes certain restrictions on people's access to, and use of resources, using a top-down approach to conservation-orientated regional management, economic incentives work with human behaviour to influence people's land use choices, using a more subtle bottom-up approach. Consequently, the former requires greater enforcement and policing than the latter. Thus, the more biodiversity is conserved through economic incentives, the more resources become available to conserve economically unviable and institutionally unprotected ecosystems.

Thus, economic incentives are required which increase the benefits of biodiversity conservation to local residents and increase the costs of alternative land uses (e.g. agriculture or livestock farming) in order to level the playing field. Economic incentives need to overcome the sources of biodiversity conservation market failures. A number of different economic incentives have been applied in different contexts, which have either direct or indirect linkages to conservation. Economic incentives such as (i) the promotion of value-adding activities (e.g. trophy hunting and up-market tourism), (ii) providing appropriate and adequate compensation for damages incurred by damage causing animals and (iii) revenue-sharing by tourist hunters (e.g. tourist levies paid to communal landowners on whose land hunting concessions are set up) provide direct and strong linkages to conservation (Ashley, 1996; Secretariat of the Convention on Biological Diversity, 2001; Emerton, 2001b; Kumar, 2005).

[10] The latter is of particular significance to developing countries, where significant trade-offs exist between conservation or restoration of ecosystem functions and processes and economic development, and decisions often go the way of big corporations (Kumar, 2005).

[11] Of course, in many cases, livestock farming can be compatible with biodiversity conservation (e.g. in succulent karoo and grasslands, there are efforts to encourage appropriately managed grazing as a biodiversity compatible land use).

Indirect economic incentives may include (i) training and capacity development from joint venture partnerships with government and the private sector, (ii) stimulating underdeveloped or inaccessible markets, and (iii) government support or grants from NGOs and donors for local development and infrastructure to compensate for biodiversity conservation's positive externalities not reflected in the market (Ashley, 1996; Secretariat of the Convention on Biological Diversity, 2001; Emerton, 2001b; Kumar, 2005).[12]

Similarly, a range of economic incentives has been employed to encourage private landowners to support conservation and complementary land uses in different contexts. These include (i) conservation levies, (ii) subsidies for the cost of habitat restoration, protection and conservation, (iii) subsidies on marketing certain species (where their value is too low to make their conservation economically viable for a private landowner), (iv) tax concessions for conservation activities on private land (at least as generous as those allowed for alternate productive land uses), (v) adequate compensation for damages incurred by animals, (vi) voluntary contributions for conservation efforts on private land (Ashley, 1996; Emerton, 2001b; Tisdell, 2004), (vii) the elimination of perverse subsidies that negate biodiversity conservation (and, where possible, transfer of these subsidies to payments for non-marketed biodiversity conservation), (viii) taxes or user fees for activities with "external" costs to biodiversity, (ix) creation of markets to reward biodiversity conservation, (x) payment for ecosystem services, and (xi) biodiversity offsets (Millenium Ecosystem Assessment, 2005).

In compensating for ecosystem goods and services, there are mainly two types of arrangements; public fiscal payment and market-based instruments. It is believed that fiscal arrangements are prone to a number of shortcomings, such as high transaction costs, low efficiency in fund use and ambiguity in target beneficiaries, e.g. funds for community-based natural resource management (CBNRM) projects are sometimes used for local economic empowerment and capacity-building initiatives rather than to promote strong linkages between sustainable resource use, community benefits and conservation, which is the central principle of CBNRM. Likewise, market-based instruments do not function without challenges. They require clearly defined tenure for ecosystem goods and services, measurable benefits and low transaction costs, which implies a relatively well-developed market infrastructure. For developing countries, such conditions are often illusive (Kumar, 2005).

Another challenge is deciding whether indirect or direct payments for ecosystem services would be more effective. A price premium on environmentally sustainable and eco-friendly products is an example of an indirect payment while a transaction between downstream and upstream water users for water allocations is an example of a direct payment. Both types of arrangements for ecosystem services require institutions that are proactive and effective. However, where indirect payment mechanisms are often plagued by their ambiguous impact on conservation incentives, complex requirements for implementation, and the mismatch between the scale of management and scale of impact on the ecosystem, direct payment mechanisms can be carefully designed to cater to the requirement for protecting entire ecosystems or specific species, with diverse institutional arrangements set up between multiple stakeholders (*ibid*).

It is difficult to predict the spin-off effects and feedbacks of a particular incentive mechanism. For example, where landowners were assigned conditional use rights over certain wildlife species in Australia in order to promote their conservation, and the conservation of their habitat, this incentive mechanism in some cases resulted in a negative feedback effect. While wildlife species of high economic value (e.g. crocodiles) were conserved and their numbers enhanced, species of lesser economic value were targeted for removal because the costs of their conservation were too high (particularly in the case of highly mobile species and species with large home ranges), and they competed with the economically favoured species for resources (Tisdell, 2004). This highlights the

[12] By providing communal landowners and users with a well-functioning social grant system to support communal landowners and users that (hopefully) practice complementary land uses to conservation, the pressure on ecosystem goods and services to supplement rural incomes, and perform a traditional safety net function can be somewhat alleviated, and in so doing, contribute to biodiversity conservation inside and outside of formal protected areas (Wolmer, 2003; Shackleton and Shackleton, 2004; Hansen and DeFries, 2007).

importance of monitoring and evaluation conservation interventions, and of utilising the principles of adaptive management.

Thus, it is necessary for the economic approaches used to capture, as far as is possible, the 'true' value of the various land uses within a region, and to be able to assess how cost-effective or efficient various alternative economic incentives will be in achieving the desired sustainable development goals. It is important to recognise that biodiversity contributes to the production of ecosystem services. Most resource management decisions are strongly influenced by the need for trade in ecosystem services in formal markets such that the non-marketed benefits are not lost or degraded (Millenium Ecosystem Assessment, 2005). These non-marketed benefits are often high and sometimes more valuable than the marketed ones. In order to enhance biodiversity conservation, it is imperative that decisions be improved by considering the total economic value of alternative land uses. Similarly, it is imperative that the relative values of biodiversity conservation experienced by decision-making units be enhanced. Economic incentives constitute the means to enhance such values for decision-making units. Biodiversity usually compares more favourably to alternative land uses after incorporating the value of ecosystem services.

Towards this purpose, a combination of direct preference methods, which study behavioural changes in simulated or hypothetical markets, and indirect preference methods such as travel cost models and hedonic valuation, which infer values from data on behavioural changes in actual markets, may be used to capture direct and indirect, tangible and intangible values of particular land uses and resources (Kumar, 2005). Even if biodiversity does not fare well in a cost-benefit analysis that is not necessarily a cause for concern in programme implementation – it simply signals the kind of intervention that needs to be used, if it is decided that biodiversity conservation ought to take place.

4 The importance of institutions in enhancing biodiversity conservation in bioregions

As outlined in the previous section, there is a need to unravel the range of economic incentives that affect biodiversity conservation in bioregions. In promoting biodiversity conservation, one requirement is that the programme of biodiversity conservation should be successful. Another requirement is that, in achieving some set biodiversity targets, the programme must cost as little as possible to implement, given that there are always budget constraints. Use of economic incentives can potentially help to satisfy these requirements.

The workability of the programme can be investigated using at least two approaches: cost benefit analysis and the institutions approaches. Cost benefit analysis as a way to probe the workability of biodiversity conservation programmes is generally being mainstreamed, even though decisions can be improved if they are informed by the total economic value of alternative land uses. On the other hand, one class of socio-economic arrangements called "institutions", directly associated with economic incentives, remains unexplored.

In this context, "institutions" refer to formal and informal laws, rules, codes of conduct, norms and strategies adopted by individuals operating within and across organisations; they exist in the minds of the participants, and are sometimes shared as implicit knowledge rather than in an explicit and written form (Ostrom, 1990). Thus, in a very general sense, institutional arrangements can be defined as the rules that govern the interaction with natural resources (see e.g., Hertzler, 2007); such rules confer a spectrum of rights over natural resources.[13] Institutions affect human livelihoods through influencing people's access to assets, livelihood strategies, vulnerability to external factors, and terms of exchange (Meinzen-Dick and Adato, 2001). They operate across multiple spatial and temporal scales and within all spheres (i.e. public and private) to significantly influence conditions

[13] Agrawal and Ostrom (2001) provide a useful framework for distinguishing between different types of rights. They suggest that different bundles of rights can be identified in terms of rights of access, withdrawal, management, exclusion and alienation.

that promote sustainable development (i.e. income, employment, health and food consumption). Institutions determine various biodiversity conservation outcomes. There is a need to ensure that the institutional arrangements generate enough economic incentives for sound biodiversity conservation. Investigating the workability of a biodiversity conservation programme under the institutions approach requires an analysis of the institutions in a particular jurisdiction and judging whether they are conducive for the workability of the programme or whether institutional change ought to be recommended. Thus, the knowledge of how institutions function in relation to humans/enterprises and their interaction with biodiversity is critical to the design and implementation of effective biodiversity conservation. The rest of this section looks specifically at how institutions influence the decisions for land use in any particular region and consequently, how those decisions affect biodiversity conservation under bioregions.

Institutions influence the decisions for land use, investment, natural resource use and therefore biodiversity conservation. With strong, appropriate institutions, investment in biodiversity conservation improves as the financial value of biodiversity approaches its total economic value. In the literature, the investigation of how institutions influence the decisions for land use and biodiversity conservation has focussed on the engagement of local communities in biodiversity conservation by protected area management. The ultimate goal has been for protected area managers to be able to influence land use decisions outside protected area boundaries. This has been necessitated by the fact that the land use decisions that the protected area managers have often/sometimes wanted to influence outside protected areas have been made mainly by local communities, as opposed to big commercial interests such as commercial agriculture, mining and property developers.

Further, the literature suggests that the centrepiece of this focus on the engagement of local communities in biodiversity conservation by protected area management, where the main pressure on the ecological integrity of protected areas, particularly in terrestrial and freshwater ecosystems, has tended to come from resource use by surrounding local communities, can be described as the various initiatives aimed at decentralizing decision-making with respect to the management of biodiversity (e.g., see Shyamsundar *et al.,* 2005). Knox & Meinzen-Dick in Shyamsundar *et al.* (2005) discuss decentralisation as part of a group of policies that are inter-related. The different policies include:

- Deconcentration—the transfer of decision-making authority to lower level units of government;
- Decentralisation—the transfer of decision-making and payment responsibility to lower levels of government;
- Privatisation—the transfer of public sector functions to the private sector or individuals; and
- Devolution—the transfer of rights and responsibilities to user groups at the local level.

The key decentralisation strategies that have been used in protected areas and biodiversity conservation since the 1980s are the integrated conservation and development projects (ICDPs) and CBNRM, and many studies have been done to date on the successes and impacts these strategies have had on protected areas and biodiversity (see Shyamsundar *et al.,* 2005). In a very informative and recent review of the ICDP/CBNRM literature, Shyamsundar (2005) investigated the following key questions:

- What do we understand about the impacts of devolution, in terms of poverty reduction, biodiversity conservation, and financial implications for governments and local agencies?
- What are some of the conditions that contribute to success?
- What does the future hold for decentralised biodiversity conservation; that is, what are some emerging challenges?

On the first question, the empirical evidence on the impacts of devolution, in terms of poverty reduction, biodiversity conservation, and financial implications for governments and local agencies can be summarised, point-wise as follows:

- The general consensus within the literature is that CBNRM contributes positively to poverty reduction, especially if one is willing to accept that enhanced community level benefits (contrasted from household level benefits) make a significant contribution to poverty reduction.
- The evidence suggests that where the opportunity costs and transaction costs of CBNRM outweigh their benefits, CBNRM programmes do not result in enhanced biodiversity conservation.
- The evidence suggests that CBNRM programmes create public goods but they do not specifically create incentives for those who conserve, nor do they punish households that engage in activities inconsistent with the objectives of biodiversity conservation (e.g. poaching and other illegal activities). There is thus an emerging concern that communities do not necessarily link these tangible benefits to the decision to conserve. It appears that whether CBNRM programmes are having an impact on biodiversity conservation could be influenced by the household's perception of the distinction between community and household benefits, and how these relate to the biodiversity supplying these benefits.
- That said, there is however some strong evidence showing that CBNRM has positive impacts in terms of reducing poaching, improving perceptions, stronger rights and reducing conflicts with the greater ecosystem surrounding protected areas.
- Finally, the financial implications for governments and local agencies depends on whether the level of decentralisation is to a lower level of the central government bureaucracy, to a local authority or to a community (tribal) level conservation institution.

On the second question, the existing empirical evidence suggests that the following factors can contribute to the improved performance of CBNRM: congruence between clearly defined resource and governance boundaries, congruence between appropriation and provision rules and local conditions, collective choice arrangements, localised monitoring, graduated sanctions, rapid access to low cost conflict resolution mechanisms, minimum recognition of rights by government authorities, and governance activities being organised in multiple layers of nested enterprises in synchrony with resource complexity (Ostrom, 1990).

On the final question, the challenges that need to be addressed by decentralized biodiversity conservation are summarised as follows:

- **Communities versus households**—the basic argument is that the focus on communities relative to households is a problem for biodiversity conservation (Emerton, 2001, Gibson, 1999). The benefits based approach to biodiversity conservation as it currently exists is having a positive effect on household preferences and attitudes toward biodiversity. However, there is reason to believe that this may not be sufficient to meet biodiversity targets. Individual household benefits tend to be small relative to total household income.
- **Heterogeneity within and between communities**—communities engaged in biodiversity conservation are rarely homogeneous entities that harmoniously agree to undertake biodiversity conservation. Rather *"conflicting values and resource priorities—rather than shared beliefs and interest—pervade social life"* (Leach *et al.,* 1999:230). Communities are characterised by heterogeneity of endowments and interests. Further, whether a group of households can be characterised as a community depends on the scale of analysis (*ibid*). These differences can lead to unequal costs as a result of institutional change; differing stakeholder needs can contribute to conflict and impact conservation efforts; compensation schemes might be needed to redistribute costs and benefits.

- **Competition/complementarity of institutions**—decentralised biodiversity conservation generally involves either the creation of new institutions or the assigning of new powers to an existing institution. In either case, competition among institutions is inevitable. In addition, any changes made affect both formal and informal institutions.

- **Tenure over land and resources**—biodiversity conservation programmes confer only usufruct rights to local stakeholders, while ownership rights remain with the state. There are many reasons why this is done. There are also reasons to be concerned about whether the lack of ownership will weaken conservation efforts in the long-run.

- **Mismatch between ecological and social scales of management**—decentralisation usually creates the problem of a mismatch between ecological and governance boundaries. This problem can potentially be addressed by working with appropriate scales of regional management.

- **Financial sustainability**—biodiversity conservation programmes are generally not self-sustained. Support for these programmes usually comes from tourism and from international donors. Tourism revenues vary depending on economic and political circumstances, while development assistance is often a function of agendas that are far beyond the control of local stakeholders. Thus, financial sustainability is an issue that each conservation area will need to confront. Looking beyond the financial sustainability of decentralised biodiversity programmes, one needs to investigate the question of whether these programmes promote future investments in biodiversity conservation.

- **Institutions to address new threats**—the traditional institutional analysis has concerned itself with wanting protected area managers to influence land use outside protected areas by local communities as opposed to big commercial interests such as commercial agriculture, mining, property developers, etc. The major pressures on biodiversity outside protected areas, and in some cases inside them as well, also includes outright loss of natural habitat (from cultivation, mining, urban expansion, coastal development etc), flow modification (includes over-abstraction of water, also altering e.g. seasonal flow of rivers) and invasive alien species. In the South African context, these could actually be more significant than subsistence use of natural resources by local communities. The challenge then would be to investigate which institutions influence/regulate these major pressures.

A range of institutions-based incentives have been employed to encourage private landowners to support conservation and complementary land uses in different contexts. These include (i) providing secure tenure (e.g. private rights to forest use and management), (ii) defining clear use rights (e.g. conditional use rights to use, hunt and sell wildlife on private land), (iii) improved tenure and use rights through the creation of communal conservancies, wildlife management areas and contractual parks etc. (Fabricius *et al.,* 2004), (iv) promoting multiple use and best practice (e.g. use of eco-friendly methods) and (v) the promotion of joint venture partnerships (Ashley, 1996; Emerton, 2001b; Tisdell, 2004).

At the national scale, in South Africa, institutions-based incentives have focused on (i) establishing appropriate institutions and lines of administration to manage finances such that funds are channeled where they are most needed and not 'trapped' in a national treasury, which is far removed from the management context, or reallocated for other purposes, (ii) providing secure tenure and use rights, (iii) capturing positive externalities from government conservation efforts, and (iv) promoting diverse livelihood options for the single landowner or user, as well as across regions (Ashley, 1996; Emerton, 2001b). Hence, government policies on land tenure may influence biodiversity conservation to a greater extent than biodiversity targets at the national scale. The land claims process in South Africa is such an example, and has been an effective mechanism by which local communities have re-acquired land formerly belonging to them, but from which they were forcibly removed during the

Apartheid era. Where land claims have been awarded in national parks, this has often resulted in the establishment of contractual parks – a kind of joint venture partnership between government and local communities, whereby the management authority leases the land from the community (usually on a 99-year lease) and it remains within the national conservation estate, a joint management board is established with decision-making authority, and economic benefits generated from conservation and tourism-based activities on the land, as well as from conservation or tourism levies (in some instances), and channeled back into the community (Reid and Turner, 2004).

5 Conclusion and suggestions for further research

This paper accepted that biodiversity conservation in South Africa ought to take place both inside and outside protected areas using a bioregional approach if biodiversity targets are to be met. A framework for studying the factors affecting biodiversity conservation in bioregions was proposed and it suggested that a range of social, technological, ecological, economic and political factors drive biodiversity conservation in bioregions. The use of economic incentives was argued to be a potentially more effective mechanism for mainstreaming biodiversity conservation in bioregions, given that landowners are likely to respond to certain amenable economic approaches in making land use decisions, than command-and-control systems. In many cases, a change of institutions is all that is needed in order to provide economic incentives. Accordingly, institutions were singled out as one important type of socioeconomic arrangement directly associated with economic incentives which is likely to be a major determinant of the vulnerability or success of biodiversity conservation.

The foregoing discussion suggests several questions that should be addressed for an adequate understanding and appropriate use of economic incentives in bioregions in South Africa. The ecology side of bioregions has been studied widely and is relatively better understood. However, there is a dearth of research on the socio-economic side of bioregions. Based on the analysis presented in this paper, there are three themes dealing with socioeconomic issues, particularly institutional arrangements, which could be placed on the research agenda, and which if answered, will enable the crafting of well-functioning bioregions in South Africa. These themes are identified below as a suggestion for further research:

1. How does one make a case for biodiversity conservation to policy makers from municipal to national level, especially when the short-term opportunity cost of conservation is high? (i.e. the public benefits are far outweighed by the private benefits)

2. Which institutions are most important in curbing the key drivers of biodiversity loss such as habitat loss, flow modification, invasive species and overharvesting in specific regions?

    - What is the role of different tenure systems? How does one deal with tenure changes under claims for land restitution without decreasing biodiversity conservation outcomes?
    - Where institutional change can enhance biodiversity conservation under bioregions, what is the best way of implementing/facilitating institutional change to reduce the potential for conflict between current institutions and desired institutions?
    - What institutional mechanisms does one need to protect critical biodiversity areas from unsustainable land and resource use, especially by global capital?
    - What advice would economics give to bioregions about what incentive mechanisms work best at what spatial scale, and in which biome?
    - How does one know when there is a case in which influencing behaviour in favour of biodiversity conservation can only be achieved through command-and-control, vs. a case in which behaviour can be influenced through incentives? In other words, when is it appropriate to regulate rather than to use incentives?

3. What synergies exist or can be created between economics and spatial biodiversity planning?
    - How can one achieve biodiversity conservation in spatial areas where one has less control, unlike in a protected area where the level of control is much greater?
    - What can economics do to corroborate the results from the systematic conservation planning framework?
    - How does one integrate economic costs and benefits into systematic conservation planning through the appropriate valuation of ecosystem services?
    - What is the significance of payment for ecosystem services for biodiversity planning?
    - How can stakeholder heterogeneity be factored into the design and implementation of systematic biodiversity conservation plans in bioregions?

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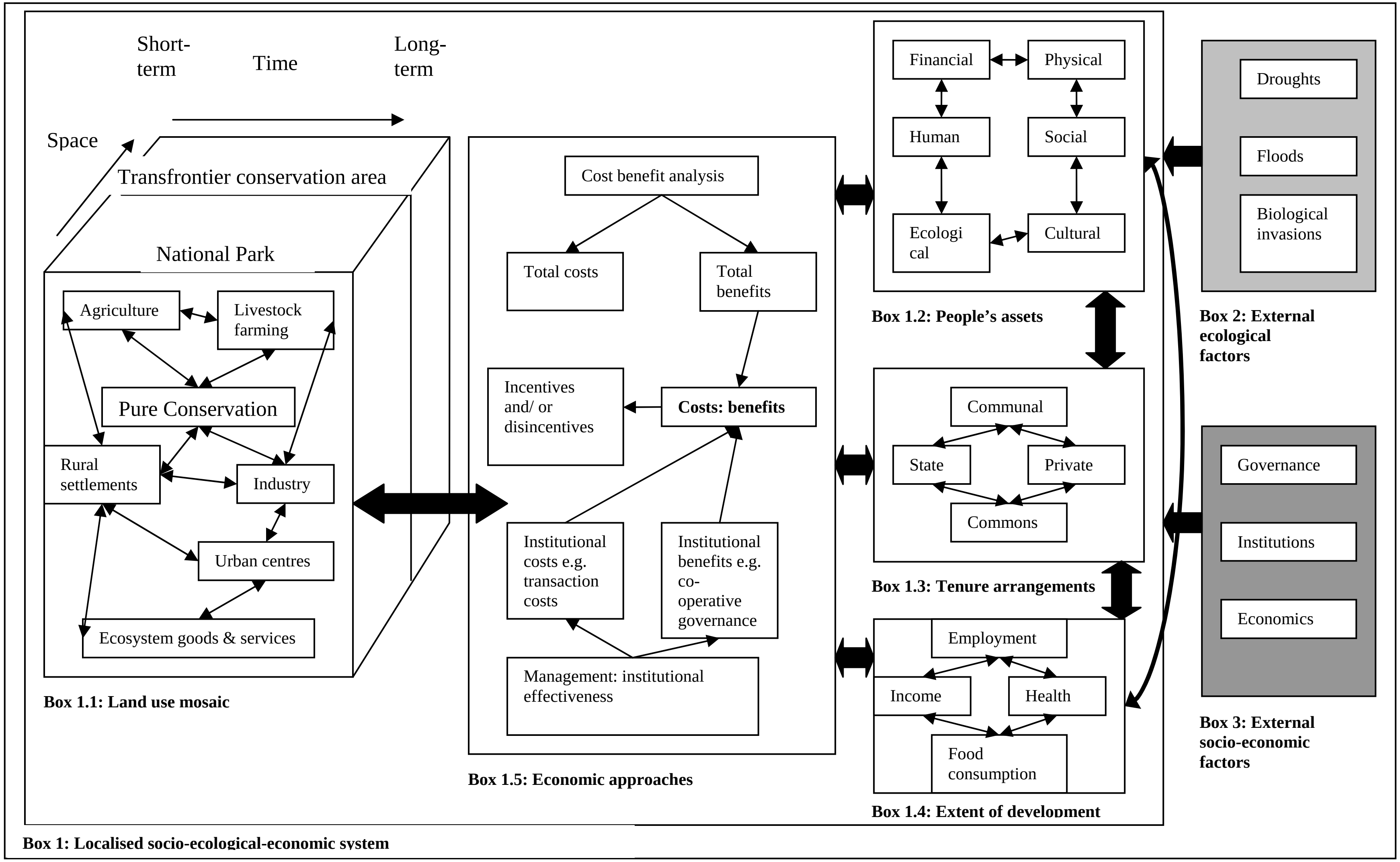

Figure 1: Conceptual framework for understanding the economics of bioregional land use mosaics