A COMPARISON OF ECOSYSTEM HEALTH AND SERVICES PROVIDED BY SUBTROPICAL THICKET IN AND AROUND THE BATHURST COMMONAGE

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ABSTRACT

Municipal commonage in South Africa offers previously disadvantaged, landless residents access to both direct ecosystem goods and services (EGS) that provide additional income options and indirect social and cultural services. Given that EGS production is a function of ecosystem health, it is imperative that commonage land be managed to maximize current local benefit streams while ensuring future options through the maintenance of natural ecosystem functions. The payments for ecosystem services (PES) model potentially offers an opportunity for contributing to local economic development while providing fiscal incentives for environmentally sustainable natural resource management. PES depends on the demonstration of quantifiable changes in EGS delivery due to improvement in or maintenance of high ecosystem health that are a verifiable result of modifications in management behavior. This thesis therefore compared spatial variations in (i) ecosystem health and (ii) nine direct and indirect EGS values derived from natural resources on the Bathurst municipal commonage and neighboring Waters Meeting Nature Reserve (NR) to explore how different land use intensities affect ecosystem health and the resulting provision of EGS. The results indicate that the total economic value of annually produced EGS on the study site is nearly R 9.8 million (US$ 1.2 million), with a standing stock of natural capital worth some R 28 million (US$ 3.4 million). Nearly 45 % of the total annual production is attributed to Waters Meeting NR, with roughly 34 % from the low use zone of the commonage and the remaining 22 % from the high use zone. Of the total annual production value on the study site, roughly 59 % is derived from indirect (non-consumptive) uses of wildlife for the study site as a whole, though this proportion varies from 25 % in the high use zone of the commonage to 94 % on Waters Meeting NR. The two largest annual production values on the study site derive from ecotourism (R 3.5 million, US$ 0.4 million) and livestock production (R 2.6 million, US$ 0.3 million), suggesting that while increased production of indirect EGS could generate significant additional revenues, especially on Waters Meeting NR and in the low use zone of the commonage, direct (consumptive) EGS will likely remain an important component of land use on the commonage. A PES project to support the adoption of silvo-pastoral practices could provide positive incentives for improved land use practices on the commonage and potentially be financed by conservation-friendly residents of the Kowie River catchment and/or increased ecotourism revenues from Waters Meeting NR. Allowing carefully designed and monitored local access to natural resources within Waters Meeting NR could also reduce pressure on commonage resources. Together, these approaches could lead to a more sustainable subtropical thicket landscape and ensure that critical natural resources remain available to support local livelihoods in the long-term.

Keywords: Payments for ecosystem services, ecosystem health, land use planning, municipal commonage

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Glossary

ABG	Above Ground
ANC	African National Congress
AR	Afforestation and Reforestation
ARC	Agricultural Research Council
BAU	Business as Usual
CBSA	Cumulative Basal Stem Area
CDM	Clean Development Mechanism
CEC	Cation Exchange Capacity
CMC	Commonage Management Committee
CONAFOR	National Forest Commission in Mexico
COP	Conference of Parties
CPI	Consumer Price Index
CPR	Common Property Resource
CR	Compensated Reduction
CV	Contingent Valuation
DEAT	Department of Environmental Affairs and Tourism
DLA	Department of Land Affairs
DTI	Department of Trade and Industry
DWAF	Department of Water Affairs and Forestry
EGS	Ecosystem Goods and Services
EPA	United States Environmental Protection Agency
ESI	Environmental Services Index
ESVs	Ecosystem Service Values
FONAFIFO	National Fund for Forest Financing in Costa Rica
GDP	Gross Domestic Product
GHG	Greenhouse Gas
HDI	Human Development Index
IAPs	Invasive Alien Plants
IDP	Integrated Development Plan
INR	Institute of Natural Resources
LAU	Large Animal Unit
LED	Local Economic Development

LM	Local Municipality
LRAD	Land Redistribution for Agricultural Development
MC	Municipal Commonage
MDTP	Maloti-Drakensberg Transfrontier Project
NOAA	United States National Oceanic and Atmospheric Administration
NR	Nature Reserve
NTFPs	Non-Timber Forest Products
PCA	Principal Components Analysis
PES	Payments for Ecosystem Services
PGRs	Private Game Reserves
PHPA	Per Household Per Annum
PPPA	Per Person Per Annum
REDD	Reduced Emissions from Deforestation and Forest Degradation
RISEM	Regional Integrated Silvopastoral Ecosystem Management
SAB	South African Breweries Limited
SADC	Southern African Development Community
SANBI	South African National Institute for Biodiversity
SLAG	Settlement and Land Acquisition Grant
SMP	Strategic Management Plan
SOC	Soil Organic Carbon
SOCP	Soil Organic Carbon Percent
STEP	Sub-Tropical Thicket Ecosystem Planning
STRP	Subtropical Thicket Rehabilitation Project
UNFCCC	United Nations Framework Convention on Climate Change
USAID	United States Agency for International Development
WfW	Working for Water
WfWet	Working for Wetlands
WfWood	Working for Woodlands
WTP	Willingness to Pay
WWF	World Wide Fund for Nature

1 Chapter One

Introduction

1.1 Land tenure and land reform in South Africa

It is well recognized that access to assets in general and secure land tenure in particular can play a critical role in economic development (Deininger 1999; De Soto 2000). Secure access to land can decrease fears over loss of land and increase access to credit markets, thereby encouraging investments that can enhance productivity and potentially lead to higher and more reliable incomes (see Bardhan, *et al.* 2000 for references; also Deininger, *et al.* 2008). Unfortunately, like so many of the socio-political challenges facing South Africa, the issue of land tenure is inextricably linked to the unequal distribution of resources between different racial groups (Walker 2005). Thus, land tenure is not only rooted in the racial segregation that is a product of the country's colonial and recent history, but also intertwined with its liberation movement and transition to a majority-rule democracy in 1994.

1.1.1 A history of forced segregation

Forced segregation along racial boundaries dates back almost to the first encounters between European settlers and the native Khoi peoples in the Cape (Thwala 2003). Nonetheless, it was under the leadership of the white-minority Nationalist Party, which came to power in 1948, that many of the discriminatory policies that had been incubated during British rule were codified into a powerful and self-destructive platform of policies, propaganda, and social norms known as Apartheid (Giliomee 2003). Over the course of the next five decades, the Nationalist government, which only represented the South African population classified as "white," ($\pm$ 20 %) decisively crafted a rigid racial hierarchy that deliberately discriminated against all non-whites, albeit to varying degrees.

Apartheid legislation assigned each South African to one of several supposedly exclusive racial categories, including black or "native" peoples of African descent, "Coloured" peoples of mixed ancestry, and "Asian" peoples, mostly descendent from Indian and Chinese immigrants who first came to South Africa as indentured servants (Seidman 1999). Far from being a benign label, racial status determined "most legal and political rights" and influenced almost all facets of life, from residency and marriage to education and occupation (Seidman 1999). Although racial segregation generally discriminated against all non-whites, black

South Africans, in particular, faced arguably the greatest obstacles to exercising their limited rights.

The deliberate segregation of black South Africans into "native reserves" began in the early nineteenth century and was made law by the Native Land Act in 1913 (Wotshela 2004). According to this legislation, the land reserved for habitation by black South Africans, referred to as "natives" or simply "Africans," was restricted to just 10 % of South Africa's land mass. Moreover, the Act repealed black South Africans' ability to own and lease their land, thereby limiting their participation in the agriculture sector to tenant labour. The 1936 Natives Trust and Land Act furthered the goal of "consolidating, as far as possible, contiguous areas of land occupied largely by African people" by relocating black South Africans to rural "homelands" that were supposedly self-governing administrative units within Apartheid South Africa (Wotshela 2004). Racial segregation policy was extended to urban areas with the passage of the Group Areas Act of 1950, but in practice segregation in cities began in the early 1920s (Thwala 2003).

During the era of Apartheid alone, more than 3.5 million people were forcibly relocated from their rural and urban homes, mostly to the nominally autonomous rural "homelands" according to their ethnic group (Meadows & Hoffman 2002; Surplus People Project 1983). This process exacerbated existing inequalities between racial groups and intensified the spatial dislocation of non-white communities from goods, services, and other racial groups. Furthermore, the cumulative effect of these policies was the appropriation of 87 % of South Africa's land for the exclusive use of the minority white citizens, while the majority non-white citizens were forced onto the remaining 13 % (Walker 2005).

1.1.2 The 'dual system' of agriculture

Among the most devastating and lasting impacts of this history of segregation is the gradual decline of black subsistence agriculture over the past century and the resulting widespread impoverishment of rural blacks (Walker 2005). By 1994, the Apartheid regime had essentially "created a dual system of agriculture…[with roughly] 55,000 highly-skilled, white commercial farmers and thousands of small subsistence producers growing mainly for household consumption and survival on very small allotments in the communal areas" (Andrews 2007). Whereas commercial farmers enjoy private land rights and occupy fully 67.5 % of South Africa's landmass, black farmers have largely been relegated to state-owned communal lands administered by local chiefdoms that cover only about 13 % of the country's area (Walker 2007).

Furthermore, in contrast to white-owned commercial farms, which benefited from government support in the form of subsidies and conservation programs, black subsistence agriculture was severely undermined by the concentration of population created by the government's large-scale forced relocation of blacks and its deliberate neglect of these so-called "autonomous" communal homelands (Meadows & Hoffman 2002). It is true that high population density does not in and of itself preclude agricultural productivity (Tiffen, *et al.* 1994). Nonetheless, the combination of high population concentration, poor governance, insecure tenure, and the threat of potential relocation at any time in South Africa's communal homelands led to overgrazing, deforestation, and declining field sizes, trends that ultimately resulted in the decline of black subsistence agriculture and oftentimes in the concomitant collapse of rural black economies (Andrews 2007; Meadows & Hoffman 2002).

This history of dispossession has come to define "the social and political identity of black people as a group" (Walker 2005). The national narrative of dispossession and forced removals was a driving force in the struggle to end Apartheid and continues to play an important role in efforts to redress the racial inequalities caused by 350 years of segregation. Still, it is important to note that the implementation of forced removals and racial segregation was carried out on a local scale and affected different individuals and communities to different degrees (Walker 2005).

1.1.3 Land reform in South Africa

During the struggle to end Apartheid, the reversal of this divisive history of dispossession and the restoration or redistribution of rural land to black South Africans was considered central to the resolution of racial inequalities (Walker 2005).

> "Although the liberation struggle in South Africa was not overtly fought around the land question, as was the case in Zimbabwe for example, there was always the expectation that unraveling centuries of land dispossession and oppression would be among the priorities of a democratic South Africa". (Ntsebeza 2007)

Soon after all South Africans were allowed to vote in their first truly democratic elections in 1994, the newly-elected democratic government of South Africa set out to address the unequal distribution of land in primarily rural but also urban areas. The centrality of land reform to overcoming South Africa's history of racial segregation was highlighted by direct references to the government's responsibility to enact land reform in both the new South

African Constitution (Act 108 of 1996, Section 25) and the 1997 White Paper on Land Policy (Hall, *et al.* 2007). The newly elected democratic government, led by the African National Congress (ANC), delineated the three major components of its land reform plan: land redistribution, land restitution, and tenure reform (DLA 1997).

Land *redistribution* was designed to ensure more equal access to land for South Africa's black majority primarily through the transfer of private commercial farmland to previously disadvantaged people living in the communal homelands. On the other hand, land *restitution* was designed to return (or provide compensation for) lands unjustly taken from black individuals and communities under racially discriminatory laws and practices since 1913 (DLA 1997). In conjunction with these two land transfer processes, the Department of Land Affairs (DLA) was also charged with securing the property rights of people living in tenuous arrangements on land owned by others, including both state and private landholders (Hall, *et al.* 2003). For the purpose of this thesis, land redistribution is most relevant.

Land redistribution aims to simultaneously reduce pressure on resources in the communal areas while also diversifying the ownership structure of commercial farmland (Hall 2007). Initially, households earning less than R 1,500 per month were eligible to receive state grants of up to R 16,000 per household to facilitate the land purchase and provide some start-up capital. However, the combination of high land prices, which resulted in groups of individuals pooling their grants to purchase a single farm, and the lack of support for efficient agricultural production post purchase limited the impact on beneficiaries' livelihoods (May & Roberts 2000; Turner 1997). To address these shortcomings, a new policy known as Land Redistribution for Agricultural Development (LRAD) began in 2001. Its primary objective was the promotion of black commercial farmers by coupling land acquisition with support for new aspiring commercial farmers (MALA 2001). Unlike the initial redistribution program, there is no income ceiling; instead, grants are offered on a sliding scale from R 20,000 to R 100,000 based on the level of cash or loans applicants can contribute (Hall 2007).

The focus of South Africa's land redistribution program has primarily been on transferring private ownership of white-held agricultural land; hence, there is a need to meet the needs of those who are not able to invest in, or sustain the risks associated with, commercial agriculture, as well as those in need of land for residential or other non-agricultural purposes (Hall 2007). While there have been programs to transfer non-agricultural land, including the Settlement and Land Acquisition Grant (SLAG), there has been little application of these programs and their future remains uncertain (Hall, *et al*. 2003). In contrast to these transfers of primarily private land, another form of land redistribution involves the expansion and enhanced management of government-owned “municipal commonage” land (Hall 2007). This land reform program has played a key role in achieving not only the DLA’s mandate to transfer land to black ownership, but also in enhancing poor peoples’ access to common resources and thereby providing opportunities for alternative livelihoods (Andrew, *et al.* 2003).

1.2 Municipal commonage in South Africa

In addition to private, freehold land (e.g. commercial farms) and land held in trust by the state for a community (e.g. the communal homelands), the DLA also recognizes a special form of public land termed “municipal commonage.” This land has traditionally been managed as communal grazing area owned by a municipality or local authority for the benefit of local residents (DLA 1997). In most cases, land for municipal commonage was given by state grants or the church to the local governing authority, typically at the time of formal town incorporation in the nineteenth century (Anderson & Pienaar 2003; Ingle 2006).

Although municipal commonages represent a sizeable area around many of South Africa’s rural towns, exact figures are lacking. This land tenure form is common in the Western Cape, Eastern Cape, and Northern Cape, but is also found in the Free State (Atkinson 2007a, b). A 2003 survey (Buso 2003) estimated that commonage occupies at least 112,795 ha of land in the Free State, varying from 83 ha to 29,701 ha per town. Meanwhile, Benseler (2003) estimated that municipal commonage covers a total of 1,641,433 ha in the Northern Cape (Pienaar & May 2003). Unfortunately, official records on the exact area of commonages in the Western and Eastern Capes are not available. Still, the importance of commonage to local livelihoods merits further study even in the absence of complete data on total area.

1.2.1 The contribution of commonage to local livelihoods

Commonages were originally set aside for use by the urban, predominantly white, poor to promote livestock grazing and other natural resource collection (Anderson & Pienaar 2003; Ingle 2006). Access for people of other non-white race groups was largely restricted throughout the 20th century even as urban whites diversified away from agricultural livelihoods. During the 1950s and 1960s most commonage areas were reclassified as land to be leased from the municipality for use by white commercial farmers (Atkinson & Benseler 2004). This provided an important source of revenue for local municipalities but continued the practice of discriminating against black and Coloured residents (DLA 1997). Unleased areas did provide subsistence resources to the urban poor, but were informal and policed.

It was not until after South Africa's transition in 1994 to a democratic government elected by the majority of all its citizens that previously disadvantaged, non-white residents gained formal access to commonage resources as municipalities terminated their leases with white commercial farmers. Grants for the purchase of additional commonage land have played a notable role in achieving the government's land distribution agenda. In fact, in the first five years, municipal commonage represented fully a third of all land transfers through the DLA's land distribution program (Hall 2007).

In contrast to historical commonage, new commonage, which is typically acquired through the purchase of land from commercial farmers, may only be established to benefit previously disadvantaged (i.e. non-white) people (Anderson & Pienaar 2003). To expand the size of some municipal commonage, the DLA purchased land on behalf of municipalities and transferred it to them free of charge. The primary goal of this new commonage has been to support the transition of so-called "emergent" farmers into viable commercial farmers (Buso 2003). Historically, livestock husbandry has been the predominant land use on municipal commonage. However, the DLA land reform program has also provided funding to allow municipalities to facilitate a variety of subsistence livelihood options for previously disadvantaged individuals on commonage land (Atkinson 2005, 2007b; Atkinson & Benseler 2004; DLA 2002).

In addition to the diversity of commonage uses, the users of commonage land have also been shown to espouse a variety of livelihood strategies, from strictly subsistence objectives to "proto-capitalist farmers" (Table 1.1) (Atkinson & Buscher 2006; Cartwright, *et al.* 2002). While municipal commonage policy has often been focused on helping emerging farmers to successfully progress to commercial agriculture, in practice it is clear that commonage land plays a crucial role in maintaining subsistence livelihoods (Anderson & Pienaar 2003; Atkinson & Benseler 2004; Buso 2003; Davenport 2008a).

Overall, municipalities are supposed to have a development-centered mandate for commonage management, and there is growing pressure from township residents to support pro-poor development projects on commonage land (Atkinson 2005; DLA 1997). It is well recognized that common property resources often represent an important source of supplemental income opportunities for impoverished communities (Saruchera 2004;

Shackleton 2000). As such, municipal commonage represents an important resource for pro-poor development primarily because it is usually sited within close proximity of poor residents, it "is often the only natural resource available" to the residents of small urban townships, and it offers varied livelihood options to promote local economic development (Atkinson & Benseler 2004).

Table 1.1: Commonage user classification

User type	Denotation	Connotation
Survivalists/subsistence	Households who have few alternatives	The majority rely on social grants and/or pensions. Keep small amounts of animals to supplement income. Not interested in expanding herds.
Micro-farmers	Households who supplement income through farming	Keep a limited number of livestock to either supplement other forms of income, or for cultural purposes.
Emerging farmers	Farmers who show signs of commercialisation	Have acquired some livestock and show signs of commercialisation; may have bank accounts and want to expand stock to start farming for a profit. May still be reliant on non-agricultural forms of income.
Proto-capitalist farmers	Farmers who have enough stock but need more land	Have built up large numbers of stock and are in need of additional land. May have other livelihoods, but want to start farming commercially on a full time basis. Ideal candidates for a "step-up" land reform strategy, thereby making more space for other farmers on the commonage.

Source: Davenport 2008a adapted from Atkinson & Buscher 2006; Cartwright, *et al*. 2002

Poor, landless residents can access numerous goods and services on commonage land to provide additional income options, including fodder for livestock production, fuel wood collection and building material, and supplemental food from vegetable production and the harvesting of wild fruits and vegetables (Anderson & Pienaar 2003; Andrew, *et al.* 2003; Cartwright, *et al.* 2002; Ingle 2006; Millennium Assessment 2005; Shackleton, *et al.* 2001). Commonage land also supports other social and cultural services, such as medicinal plants collection, recreation, housing, and ablution and refuse disposal that contribute to local livelihoods (Anderson & Pienaar 2003; Cartwright, *et al.* 2002; Ingle 2006). Of course, the availability and contribution of each of these resources to local livelihoods varies across commonages and is ultimately dependent on the local context (Buso 2003). For example, research by Buso (2003) showed that municipal commonage in the Free State is most often used for grazing stock and some limited crop farming, but vegetable garden projects and poultry farming were also important enterprises on the land nearest to the urban settlements

Thanks to this diversity of livelihood strategies on commonage land, it is clear that the resources provided by municipal commonage constitute an important strategy for mitigating the effects of inadequate or nonexistent cash incomes for the urban poor (Anderson & Pienaar 2003; Davenport 2008a). The valuation of commonage goods and services is often complicated by their primarily subsistence nature, which means that they are often not adequately addressed by South Africa's market-based economy (Cavendish 2002; Shackleton, *et al.* 2000; Smit & Wiseman 2001). Nonetheless, recent research on commonage-using households suggests that these resources contribute crucial income-generating opportunities for many impoverished urban and peri-urban residents with limited off-commonage livelihood options (Davenport 2008a).

In a survey of households living near three municipal commonages in the Eastern Cape, Davenport (2008a) found that 65 % of households used between two and five commonage resources, with the largest proportions of households utilizing fuel wood (on average 86 % of user households across the three towns), wild fruit (37 %), wild herbs (33 %), fencing poles (29 %), and livestock (28 %). Although the individual contribution of any single livelihood strategy may appear insignificant, the cumulative effect of multiple strategies can represent a significant proportion of total user incomes. Thus, while wild fruit and herbs contributed on average just R 200 $\pm$ 489 and R 68 $\pm$ 176 per annum to user households across the three towns, livestock contributed R 964 $\pm$ 4,399 per annum. As the most commonly utilized resource, it is perhaps unsurprising that fuel wood also contributed the largest value (R 1,833 $\pm$ 2,643) to annual household incomes.

Furthermore, Davenport's (2008a) household surveys demonstrated the significant contribution to overall household income in the absence of adequate opportunities for off-commonage income generation. The single largest contributor to total livelihoods was social grants (e.g. pension, unemployment, and child welfare grants), which accounted for nearly half (44.8 % $\pm$ 33.5) of commonage users' incomes, while employment contributed roughly one third (32.0 % $\pm$ 33.1) of total income. For comparison, household incomes derived from all commonage resources represented on average between 14 % and 20 % of total on- and off-commonage incomes across the three towns. This underlines not only the limited opportunities for earning off-commonage wages in these impoverished communities, but also the important role that commonage resources play in potentially mitigating poverty through alternative livelihood options.

Therefore, it is clear that municipal commonage is a crucial source of a variety of common resources that can be harnessed to provide multiple livelihood opportunities for different users, from subsistence to emerging and proto-capitalist farmers (Atkinson & Benseler 2004; Atkinson & Buscher 2006; Cartwright, *et al.* 2002; Davenport 2008a). Supporting this broad range of uses and users demands that municipalities design and implement complex and well-executed management policies to equitably distribute commonage resources among the different user groups. Unfortunately, good commonage management is often beyond the reach of under-resourced and over-stretched municipalities faced with meeting the needs of a growing number of beneficiaries (Atkinson 2005; Atkinson & Benseler 2004; Buso 2003).

1.2.2 Challenges facing commonage management

As the legal owner and management authority, the local municipality is responsible for making by-laws and regulations governing the supported uses and users of both historical and new commonage land (Atkinson & Benseler 2004). Atkinson (2005) notes that commonage management affects numerous prerequisites for development, including food security, local economic development (LED), and sustainable natural resource use. Unfortunately, commonage management in South Africa faces a number of challenges, especially limited management capacity, increasing numbers of livestock due in part to migration from rural agricultural areas to urban or peri-urban townships, and a concomitant slide towards inappropriate grazing practices (Atkinson 2005; Atkinson & Benseler 2004; Buso 2003).

Municipalities that have acquired new commonage under the DLA's land reform program often lack adequate funding to maintain land, infrastructure, management and policing functions, or training programs for farmers using commonage land (Buso 2003). In the absence of outside financial support, municipalities are forced to either divert funds from other budget items to manage commonage land or simply underfund commonage programs. Neither option is sustainable and typically the latter prevails, leaving the promise of commonage-centered development "seriously constrained" (Atkinson 2005; Wisborg 2002). In fact, according to a recent review of nine commonages in South Africa (Atkinson & Benseler 2004), many commonages lack a comprehensive land use plan or development framework to support the municipality's overall spatial development goals typified by some municipalities not knowing even the area of commonage land they have, or the precise boundaries. As a result, planning on commonage land has typically been limited to the project level, leaving significant room for improvement in achieving broader development goals.

At the same time, only a handful of municipalities have guidelines for the selection of beneficiaries, which in practice has resulted in an increasing number of commonage users as urbanization and concomitant demand for land increases (Atkinson & Benseler 2004). Due to complex political and economic pressures facing rural (white) commercial farmers, including their fear of land tenure reform in the post-apartheid South Africa and increasing competition from global markets, many farm workers are being evicted (Atkinson 2005; Atkinson & Buscher 2006; Simbi & Aliber 2000). As the agricultural sector has shed jobs, South Africa's towns and cities have seen a rapid urbanization over the past fifteen years, also driven by normal urban migration from the rural homelands (Atkinson 2005; Atkinson & Buscher 2006). This influx of rural farm workers has led to increased demand for local resources, including commonage land, and often leads to very high grazing pressures (Atkinson 2005; Atkinson & Benseler 2004; Palmer 2005).

Moreover, despite the favored status of subsistence and emerging (pre-commercial) farmers, it is not clear that farmers who move beyond these categories to become small-scale commercial farmers are accessing available DLA grants to acquire alternative grazing land (Fabricius, *et al.* 2006). In theory, limiting old commonage grazing to subsistence users should relieve grazing pressure by shifting emerging farmers, with larger herds, to new commonage land (Benseler 2003; Buso 2003). In practice, though, subsistence farmers may choose to continue increasing their herds beyond their immediate needs to satisfy socio-cultural or financial goals (Benseler 2003). This leads to an unrelenting increase in the number of animals grazed on the same commonage, which, combined with limited infrastructure maintenance and livestock management by the municipality, all too often leads to overstocking and other unsustainable grazing practices (Atkinson & Benseler 2004), with potential impacts on ecosystem health and productive capacity.

Although commonage land has historically been used primarily for livestock production, local residents also accessed a variety of other resources, including medicinal plants, wild fruits and vegetables, fuel wood, water, thatch grass and spiritual and recreational sites located on commonages (Andrew, *et al.* 2003; Millennium Assessment 2005). There is some concern that livestock grazing may be precluding other users from freely accessing commonage resources, such as by trampling or heavy grazing which negatively impacts on woody plant seedlings, medicinal plants and thatching grass (Fabricius, *et al.* 2006). As such, there is growing support for the creation of comprehensive land use plans for commonages and their inclusion within local integrated development planning (Atkinson & Benseler 2004).

The need for adequate land use planning in municipal commonage areas is underscored by the South African experience of human-induced land degradation, often due to unsustainable natural resource utilization as a result of insufficient management and policy enforcement (MDTP 2008; Meadows & Hoffman 2002). Meadows & Hoffman (2002) concluded that the impact of insecure tenure and higher population densities on demand for natural resources in communal areas, especially fuel wood and fodder for cattle, made these lands more vulnerable to environmental degradation than private commercial farms. In fact, communal areas were considered to have a rate of soil erosion nearly three times higher than commercial farms, especially in regions where grazing was the predominant land use (Meadows & Hoffman 2002).

Meadows & Hoffman (2002) demonstrated that the degree of environmental degradation on a given piece of land is correlated with the degree of poverty in the surrounding area as measured by the percent of unemployed residents, average number of dependents per household, and economic production per capita. Unlike the former homelands, in many cases black South Africans were excluded from areas now managed as municipal commonage (Atkinson & Benseler 2004). Still, the recent influx of poor, often landless farm workers into South Africa's rural towns and resulting increased demand for common natural resources have raised concerns that municipal commonage may face a similar fate without adequate management (Atkinson 2005; Atkinson & Benseler 2004).

While land degradation in South Africa tends to be associated with human actions, climate may play an exacerbating or predisposing role (Meadows & Hoffman 2002). For example, Mason & Jury (1997) found that large areas of southern Africa have been faced with decreasing precipitation for the past thirty years. It is true that much of South Africa's climate can be described as semi-arid with unreliable reliable rainfall (Binns, *et al.* 2001; Meadows & Hoffman 2002). In fact, roughly 70 % of the country receives less than 600 mm of rain annually, and 20 % receives less than 200 mm a year, well below the US threshold of 250 mm for defining arid environments (Binns, *et al.* 2001). Moreover, rainfall variability poses serious challenges for water management in South Africa. In parts of the northwest, rainfall differs by up to 40 % from the annual average, and it often varies by as much as 50 % above or below the average in the lower Orange River valley (Lester, *et al.* 2000).

1.3 Indicators of ecosystem health and the provision of ecosystem services

Although the term "health" has traditionally been applied to the status of individual humans or animals, over time it has evolved to include references to human or animal populations and, more recently, entire ecosystems (Bertollo 1997; Leopold, *et al.* 1999; Rapport 1995a, b). Rapport, *et al.* (1998) define a "healthy" ecosystem as a "stable and sustainable" system that can maintain its "organization and autonomy over time" and is resilient to stress. In fact, the study of ecosystem health builds on the field of stress ecology, which examines how ecosystems respond to issues, anthropogenic or otherwise, that lead to "biotic impoverishment, impaired productivity, altered biotic composition to favor opportunistic species, reduced resilience, increased disease prevalence, decreased economic opportunity, and risks to human and animal health" (Rapport, *et al.* 1998).

In contrast to assessing the integrity of an ecological system based on the degree of "naturalness" (i.e. lack of human influence), ecosystem health is primarily concerned with the system's ability to maintain and renew itself (Bertollo 1997). As such, it is "highly applicable to governed [i.e. human-influenced] landscapes because the 'natural' attributes of an area don't have to be in a pristine state (minimum human disturbance) to be judged as healthy" (Bertollo 1997). In fact, the application of the concept of "health" to ecosystems was a reaction to the growing evidence that ecosystems dominated by humans have become "highly dysfunctional," which has reduced their ability to provide critical "ecosystem services" (Rapport, *et al.* 1998).

1.3.1 Healthy ecosystems provide ecosystem services

In fact, the concepts of human and ecosystem health are interrelated, since human life is dependent on diverse natural resources, also known as ecosystem "goods," and the "services," or "functions recognized as satisfying human needs," supplied by ecosystems (Rapport, *et al.* 1998; Zurlini & Girardin 2008). For simplicity, ecosystem goods, such as fuel wood or food, and services, such as air and water purification, will be referred to here collectively as ecosystem services. The term "ecosystem functions" encompasses all the habitat, biological, or system properties or processes of an ecosystem, including those without *direct* benefits to human welfare such as nutrient formation and carbon sequestration (Costanza, *et al.* 1997). However, to the extent that these functions are crucial for providing ecosystem services to humans, such as fertile soils for agricultural production and reduced atmospheric carbon to prevent climate change, they are nonetheless critical to sustaining human health (Aylward, *et al.* 2005).

Ecosystem services include a diverse array of natural and biological processes that can be grouped into four broad categories: (i) provisioning services create material benefits or products, such as fuel wood, fodder, wild plant and animal foods, and honey; (ii) regulating services support ecosystem processes, including air and water quality regulation, erosion control, and climate regulation (via carbon sequestration); (iii) supporting services include processes that underlie the production of other services, such as soil and nutrient formation, water cycling, and photosynthesis; finally, (iv) cultural services refers to non-tangible benefits provided by ecosystems, including aesthetic and spiritual values, social and cultural heritage, and recreation and tourism (Aylward, *et al.* 2005). Based on their production of goods and services easily utilized by humans, provisioning and cultural services will be referred to here as "direct" ecosystem services; regulating and supporting services, on the other hand, will be considered "indirect" services.

From a human perspective, ecosystem services and the stocks of natural resources, also known as "natural capital," that produce them are crucial for facilitating the earth's ability to support life (Rapport, *et al.* 1998). Thus, while biophysical changes in an ecosystem would appear vital to an assessment of ecosystem health from a strictly ecological perspective, it is, in fact, the impact of these changes on the ecosystem's ability to provide services that determines their significance.

> "Linking ecosystem health to the provision of ecosystem services and determining how ecosystem [health] relates to these services are major challenges at the interface of the health, social, and natural sciences". (Rapport, *et al.* 1998)

This thesis will not attempt to conclusively prove the causal links between ecosystem health indicators and ecosystem service provision. Instead, it will consider these indicators as proxies for the ecosystem's ability to provide specific services that are of direct or indirect value to humans. In so doing, this work will attempt to demonstrate relationships between ecosystem health indicators and the value of services they support.

1.3.2 Indicators of ecosystem health

To understand ecosystem changes that may affect the future provision of natural resources or services, it is imperative that ecosystem health be assessed and monitored over time

(Petrosillo, *et al*. 2007). There are myriad ways of measuring ecosystem health with the aid of numerous different indicators that identify important aspects of ecosystem health from the tremendous number of signals at different spatial and temporal scales (e.g. Gray & Azuma 2005; Herrick, *et al.* 2006*;* Imeson & Prinsen 2004; Pyke, *et al.* 2002; Schulze, *et al.* 2009). It is important to recognize that the cumulative information provided by a set of indicators can never capture the complex, interrelated processes of the whole ecosystem. Nonetheless, indicators can play an important role in communicating information about the status and functioning of an ecosystem and the factors that influence these signals of ecosystem health (Zurlini & Girardin 2008).

1.3.2.1 Categories of ecosystem health indicators

Various authors have proposed different categories of indicators for assessing ecosystem health (Bertollo 1997; Cairns, *et al.* 1993; Munn 1993; Rapport, *et al.* 1998). Thus, Munn (1993) grouped "general indicators" together that deal primarily with the state of ecosystem health and resilience (e.g. primary productivity, efficiency of nutrient cycling, biodiversity. In addition, Munn (1993) proposes a number of indicators of threats (and reduced threats) from human influences on ecosystems, including threats such as increasing population and consumption and the rates of depletion of renewable and non-renewable resources. Indications of reduced threats to ecosystem health include increasing output of production per unit of natural resource consumed, conservation of scarce or highly valued resources (Munn 1993). Whereas, Bertollo (1997) names numerous "descriptors," such as various characteristics of flora and fauna (e.g. species types, composition, and diversity) and the abundance and distribution of invasive species. In addition to these biotic and abiotic descriptors, indicators of ecosystem distress, such as water quality decline and the impairment of ecosystem renewal processes, contribute to assessing the health of a particular terrestrial, aquatic, or marine ecosystem.

Rapport, *et al.* (1998) also emphasize the importance of measuring indicators of ecosystem distress, such as shifts in the community composition to favor smaller life forms and reduced symbiotic relationships amongst biota. They also identify three categories of positive indicators, including resilience, vitality, and organization (Haskell, *et al.* 1992; Rapport, *et al.* 1998). *Resilience*, also known as biotic integrity, is the ability of the ecosystem to support characteristic functional and structural communities and preserve these elements when faced with stress (Pyke, *et al.* 2002). Related to this concept is *vitality* or vigor, which captures "observed activity, metabolism or primary productivity" (Rapport, *et al.* 1998). Finally, *organization* is indicated by the number and diversity of functional groups of organisms (e.g. vascular plants, mosses, etc.), individual species, and their interactions with each other (Rapport, *et al.* 1998).

In recognition of the crucial inter-relationships between humans and ecosystems, the study of ecosystem health also encompasses the measurement of socio-economic indicators (Bertollo 1997; Rapport 2007). Rapport (2007) extends the concept of resilience to livelihoods by suggesting that the ability of communities to mitigate changes in economic conditions within their ecosystem is equally important to the assessment of ecosystem health. Similarly, Maffi (2001) highlights the potential impact of changes in ecosystems on the transmission of traditional knowledge between generations, an example of cultural resilience. Resilience is also relevant to public health, via humans' ability to cope with endemic diseases, and to good (bad) governance, which can contribute to (or be jeopardized by) ecosystem resilience (or the lack thereof) (McMichael 1993, 1996; Ullsten 2003).

Finally, the indicators appropriate for assessing the health of a given ecosystem vary according to the specific human-ecological system that characterizes the site (Bertollo 1997). Therefore, appropriate indicators for assessing forest ecosystems might include productivity, biotic composition, age structure, prevalence of insects and disease, fire regime, harvesting and management, physical restructuring, pollution, and climate. On the other hand, an assessment of the health of agroecosystems (ecosystems influenced by agricultural activities) would measure the impact of interactions between agricultural and "natural" processes on, or example, soil quality, land use change, cultural practices, human health, economic factors, and climate conditions (Bird & Rapport 1986).

This thesis will investigate the ecosystem health of a municipal commonage used for, among other activities, livestock grazing. Because a detailed assessment of socio-economic indicators of the site was completed immediately prior to this study (Davenport 2008a), the work will focus on the assessment of ecological variables, both biotic and abiotic.

1.3.2.2 Status vs. functional indicators

Regardless of which categories are most appropriate for assessing a given ecosystem, it is important to distinguish between indicators of the status of ecosystem health and those that measure its functionality. Status indicators, such as the degree of soil compaction at a given point or the concentration of organic matter within a particular area, give a snapshot of the current status of ecosystem health. In contrast, functional indicators, such as resilience, vigor, and organization, help assess long-term sustainability (Pyke, *et al.* 2002; Rapport, *et al.* 1998).

The assessment of ecosystem health at the field or landscape level based on status indicators is complicated by the spatial variation of soil, vegetation, and microclimates, and the impact of previous land uses on ecosystem health (Blackmore, *et al.* 1990; Schlesinger & Pilmanis 1998). The resulting "patchiness" of these variables at higher resolutions, such as field or landscape, has been demonstrated for a number of indicators, including soil nutrients (De Soyza, *et al.* 1998; Mazzarino, *et al.* 1996) and organic matter (Herman, *et al.* 1995; Mazzarino, *et al.* 1996). Thus, soil compaction measurements collected systematically from points along human or animal pathways could indicate a higher degree of compaction at the field level than those collected randomly but may also be influenced by soil morphology. Therefore, status indicators tend to be of only limited applicability to long-term monitoring (Pyke, *et al.* 2002). Table 1.2 presents 17 status indicators designed to capture data on the current ecosystem situation and trends to assess three functional indicators of the long-term sustainability of rangeland health (Pyke, *et al.* 2002).

However, status indicators can contribute to the measurement of long-term functional indicators. With respect to rangelands, important functional indicators of ecosystem health include soil stability—"the capacity of the site to limit redistribution and loss of soil resources (including nutrients and organic matter) by wind or water"—and hydrologic function—"the capacity of the site to capture, store, and safely release water from rainfall, run-on and snowmelt (where relevant), to resist a reduction in this capacity and to recover this capacity following degradation" (Pyke *et al.* 2002).

Still, observations of either status or functional indicators made at a given spatial or temporal scale are "at best" an indication of the trends and processes relevant to that particular scale (Zurlini & Girardin 2008). Unfortunately, the spatial and temporal scales at which ecosystem processes can be most easily measured are seldom those at which these processes are most relevant to either ecosystem processes or human interactions (Pyke, *et al.* 2002; Zurlini & Girardin 2008). Keeping this in mind, this thesis will attempt to capture spatially explicit ecological indicators of ecosystem health to quantify the impact of ecosystem changes on the magnitude and value of services provided.

Table 1.2: Rangeland status indicators and functional indicators to which they apply

Status indicators	Measurements	Functional indicators		
		Soil & site stability	Hydrologic function	Biotic integrity
Rills	Frequency & distribution of erosional rivulets	X	X	
Water flow patterns	Amount and distribution of overland flow paths identified by litter distribution and visual evidence of soil and gravel movt.	X	X	
Pedestals &/or terracettes	Frequency and distribution of rocks or plants where soil has been eroded from their base (pedestals) or areas of soil deposition behind obstacles (terracettes)	X	X	
Bare ground	Size & connectivity among areas of soil not protected by vegetation, biological soil crusts, litter, dead vegetation, gravel/rocks	X	X	
Gullies	Amount of channels cut into soil and amount & distribution of veg. in channel	X	X	
Wind scoured, blowouts, &/or deposition areas	Frequency of areas where soil is removed from under soil crust/around vegetation OR freq. of soil accumulation areas assoc. with large structural objects (e.g. woody plants)	X		
Litter movement	Frequency and size of litter displaced by wind and overland flow of water		X	
Soil surface loss or degradation	Frequency and size of areas missing all or portions of upper soil horizons that normally contain majority of organic matter	X	X	X
Plant community	Community composition or distribution of species that restrict the infiltration of water		X	
Compaction layer	Thickness and distribution of the structure of the soil near the soil surface ($\leq$15 cm)	X	X	X
Plant mortality	Frequency of dead or dying plants			X
Litter amount	Deviation in the amount of litter		X	X
Invasive plants	Abundance and distribution of invasive plants regardless of whether they are noxious weeds, exotic or native species			X
Functional or structural groups	Number of groups, number of species within groups, the rank order of dominance of groups and interactions among levels			X
Soil resistance to erosion	Ability of soils to resist erosion thru the incorporation of organic material into soil	X	X	X
Reproductive capability of perennial plants	Evidence of inflorescences or of vegetative tiller production relative to potential based on climatic conditions			X

Source: Adapted from Pyke, *et al*. 2002

1.4 <u>Human-ecosystem interactions in arid landscapes</u>

1.4.1 Ecosystem resilience in arid landscapes

The concept of ecosystem resilience is particularly relevant to the arid landscapes that often support livestock production. "Resilient" ecosystems (or landscapes) are so called for their ability to withstand moderate degradation processes thanks to "soils with high rates of infiltration, together with other edaphic [soil-influenced] and topographic features" that minimize soil losses and allow greater rainfall retention within the ecosystem (Holm, *et al*. 2005; Ludwig & Tongway 1995; Rietkerk, *et al.* 1997). On the other hand, "fragile" or "non-resilient" ecosystems (Holling, *et al.* 1995) are generally more susceptible to "catastrophic losses of soil and nutrients from the landscape, accelerated runoff and significant decreases in plant productivity and rainfall-use efficiency" due, for example, to heavy grazing by livestock or wildlife or to intensive crop cultivation (Holm, *et al.* 2005; Le Houérou 1984; Mabbutt & Fanning 1987; Snyman & Fouche 1991). Therefore, grazing management becomes much more important to both primary (plant) and secondary (animal) production in fragile landscapes (Holm, *et al.* 2005).

Furthermore, Holm, *et al.* (2005) note that nutrients in arid landscapes are often "concentrated…under patches vegetated with either shrubs or perennial grass" (e.g. Charley 1972) and that these vegetated patches may play an important role in mitigating the effects of wind and water on the circulation of plant nutrients, organic matter, and litter throughout these landscapes (Shachak & Pickett 1997; Tongway & Ludwig 1994). Thus, excessive grazing or other impacts that result in the loss of perennial plants and the concomitant destruction of these nutrient-rich patches can lead to ecological degradation (Holm, *et al.* 2005).

1.4.2 Models of rangeland ecology

Historically, rangeland management was based on a core model of "equilibrium" between animal populations and forage resources due to relatively predictable weather conditions with limited variation (Illius & O'Connor 1999; Scoones 1999). However, modeling of arid and semi-arid rangelands indicates that their ecology is often typified by high annual, seasonal and intra-seasonal variation in rainfall. In fact, research on arid and semi-arid rangelands throughout Africa has demonstrated that in areas with highly variable climates, *rainfall*, rather than animal population density, is the predominant factor in determining primary (plant) production (Behnke & Scoones 1993; Ellis & Swift 1988; Hahn, *et al.* 2005; Scoones 1994).

This body of research, based in part on mathematical modeling, suggests that extended droughts in arid and semi-arid regions (or severe winter weather in cold, dry climates) cause periodic severe mortality that keeps livestock densities below equilibrium values for much, if not all, of the time (Begzsuren, *et al.* 2004; Richardson, *et al.* 2005). Thus, Scoones (1994) concludes, "livestock do not have a long-term negative effect on range resources…[and] grazing therefore has a limited effect on long-term grass productivity" (Illius & O'Connor 1999).

This realization has led researchers to question the relevance of the equilibrium model for describing plant and animal production in arid and semi-arid rangeland ecosystems. Instead, these models of rangeland ecology suggest that they are in fact subject to a "non-equilibrium" state characterized by the decoupling of animal and plant dynamics (Illius & O'Connor 1999). In other words, Illius & O'Connor (1999) define non-equilibrium as a system where animal production is only dependent on the availability of essential resources in the dry season, and not to wet season grazing (Richardson, *et al.* 2005). Alternative definitions of non-equilibrium include the "threshold" model, characterized by boundaries dividing different equilibrium states, such as woody plant invasions of grassland; and the "state-and-transition" model, which highlights the importance of event-driven vegetation dynamics (Briske, *et al.* 2003).

Despite some earlier criticism of the applicability of this "non-equilibrium" model to arid grazing systems (Cowling 2000; Fynn & O'Connor 2000; Illius & O'Connor 1999, Roques, *et al.* 2001), most experts now agree with Sullivan & Rohde (2002) that "non-equilibrium theory provides a powerful explanatory model of pastoral eco- and social-system dynamics" (Richardson, *et al.* 2005). Nonetheless, recent simulation modeling (Richardson, *et al.* 2005) and long-term observations (Hahn, *et al.* 2005) of a semi-arid pastoral system in Namaqualand, South Africa suggest "that the system has more complexity than can be explained by either of these two paradigms" (Richardson, *et al.* 2005). Therefore, it is imperative that individual rangeland environments be studied as unique human-ecological systems (Richardson, *et al.* 2005).

1.4.3 Limitations of the carrying capacity concept

It is also worth noting that there is considerable debate surrounding the concept of ecosystem degradation due to livestock exceeding the calculated carrying capacity of communal rangelands (e.g. Blaikie & Brookfield 1987; Shackleton 1993a, b; Forsyth 2003). A growing body of research suggests that this concept is based on incorrect assumptions about the lack of technical expertise among communal herders and may be wholly inappropriate for managing arid and semi-arid rangelands (e.g. Allsopp, *et al.* 2007a; Benjaminsen, *et al.* 2006; Ellis & Swift 1988). In fact, research on communal rangelands in the arid Namaqualand region of South Africa suggests that rather than being a "fixed variable," the carrying capacity of a given area is instead "a variable dependent on rainfall" (Allsopp, *et al.* 2007a).

Commercial livestock grazing practices are often heralded as the most economically- and environmentally-viable business model, particularly among agricultural extensionists and bureaucrats in southern Africa. Conventional management in a commercial system requires that livestock numbers be maintained below a particular carrying capacity that is estimated based on local ecological characteristics. It is often assumed that communal livestock management is not similarly regulated and therefore often leads to the deterioration of ecosystem health. Nonetheless, evidence from Africa and beyond has shown that herders in communal areas do actually rely at least in part on "tacit and formalized knowledge" in their management decisions (Allsopp, *et al*. 2007a; Benjaminsen, *et al.* 2006).

Furthermore, evidence suggests that even fragile ecosystems, such as the Namaqualand range in South Africa, can support much higher livestock densities than those recommended by official estimations of carrying capacity (Allsopp, *et al*. 2007a; Benjaminsen, *et al.* 2006). Of course,

> "differences in vegetation cover and species composition would be expected between different grazing systems, but such differences do not prove any land degradation. Hence, there are no clear-cut answers to what 'degradation' and 'overstocking' actually imply". (Benjaminsen, *et al*. 2006)

Instead, the definitions of these emotive terms should be related to the ability of different grazing systems to achieve unique management objectives whilst retaining basic ecosystem functions (Allsopp, *et al*. 2007a; Benjaminsen, *et al.* 2006).

In light of these shortcomings to the carrying capacity model, this thesis will not attempt to identify the maximum livestock production potential of the commonage. Instead, the emphasis will be on comparing the relative level of various indicators of ecological condition, such as vegetative cover, soil loss, and soil compaction, according to the relative intensity of livestock grazing as identified by herders in a recent community survey (Fabricius, *et al.* 2006).

By evaluating the impact of current management practices on the health of a semi-arid ecosystem, this thesis aims to contribute to understandings of the resilience of arid landscapes. Moreover, it will also compare the flow of direct and indirect ecosystem services produced by natural resources under contrasting management regimes on a rangeland and an adjacent nature reserve. In so doing, this thesis hopes to reveal the implicit management objectives of herders and other users of local natural resources and suggest policies to maximize benefits to all users without compromising ecosystem health.

1.5 Payments for Ecosystem Services

The payments for ecosystem services (PES) model is based on the assumption that environmental goods and services, such as water yield and carbon sequestration, can be traded in the marketplace in a similar fashion to any other good or service (Edwards & Abivardi 1997; Farrow, *et al.* 2000).

> "More and more, the complementary factor in short supply (limiting factor) is remaining natural capital, not manmade capital as it used to be. For example, populations of fish, not fishing boats, limit fish catch worldwide. Economic logic says to invest in the limiting factor. That logic has not changed, but the identity of the limiting factor has". (Daly, pers. comm. quoted in MDTP 2008)

PES can be described as "voluntary payments for well-defined ecosystem services (or land uses that are likely to secure those services) that are conditional on service delivery" (Turpie, *et al.* 2008; Wunder 2005). While there can be multiple buyers and sellers of ecosystem services, a PES transaction involves at least one buyer and one service provider, and these actors can be individuals, companies, non-governmental organizations, or the state (Turpie, *et al.* 2008; Wunder 2005). Although the participation of local communities is in many cases essential to the success of PES projects, these programs are often based on the efficient functioning of a procedural framework that is driven by private investors, non-governmental organizations, governments and resource managers (Corbera & Brown 2008; Kosoy, *et al.* 2008; Landell-Mills & Porras 2002).

PES projects to date have been focused on a handful of ecosystem services for which clear markets have been developed, such as hydrological functions, carbon sequestration, and biodiversity conservation, all services that can be translated into tradable goods (Kosoy, *et al.* 2008). For example, payments for watershed conservation can facilitate decentralized water delivery and thereby reduce costs incurred by public water and sewage services (Kosoy, *et al* 2008; Rosa, *et al.* 2003). Similarly, the prevention of deforestation and promotion of afforestation (forest renewal through tree planting) sequesters carbon that can be traded on the global market as credits through the Clean Development Mechanism (CDM) established by the Kyoto Protocol under the United Nations Framework Convention on Climate Change (UNEP-Risoe 2008). Another common example of PES involves the development of activities that both promote and support wildlife conservation through income generated by ecotourism[1] (Wilkie, *et al*. 2001).

1.5.1 Rewarding behavioral changes that enhance ecosystem services

In contrast to many previous models of conservation, PES provides fiscal incentives for environmentally sustainable behaviors and can (at least partially) finance itself through payments to resource users and managers for the provision of ecosystem services (Kosoy, *et al.* 2008; Pagiola *et al.* 2002; Pagiola & Platais 2007). In each of the examples described above, the land owner(s), resource user(s), or resource manager(s), whether private individuals, companies, or communities, are eligible to receive payments for *verifiable ecosystem services* provided or enhanced by *behavioral changes* on the part of the land owner/user (Kosoy, *et al.* 2008). Thus, the PES model posits not only that ecosystem services have a quantifiable economic value to society, but also that individuals and organizations that preserve or augment these services should be compensated for their contributions to environmental stewardship.

The implementation of a PES scheme therefore depends on the demonstration of quantifiable changes in service delivery due to changes in ecosystem health that are a verifiable result of changes in management behavior (Turpie, *et al.* 2008). As such, it is imperative to first establish the degree to which current and future management practices affect ecosystem health in order to measure changes in service delivery under alternative practices (MDTP 2008; Turpie, *et al.* 2008). For example, a team of scientists and natural resource economists in South Africa recently designed an implementation model for improving the supply of water and carbon services provided by the Drakensberg mountain catchment system through a voluntary PES mechanism (MDTP 2008). Their model focuses on three readily quantifiable services, namely, enhanced river baseflows in the winter months, sediment reduction, and

[1] The World Conservation Union (Ceballos-Lascuráin 1996) defines ecotourism as "environmentally responsible travel to natural areas, in order to enjoy and appreciate nature (and accompanying cultural features, both past and present) that promote conservation, have low visitor impact and provide for [the] beneficially active socio-economic involvement of local peoples."

carbon sequestration, provided by mountain grasslands and demonstrates how changes in the management of these areas leads directly to improved water services. They note,

> "[c]ritical to this approach was the underlying fact that plant cover or basal cover of plants on the land plays a fundamental role in enhancing hydrological benefits in catchments. Management action that maintains and restores a robust basal cover within the grasslands will result in greater supplies of water retention or storage services, greater storm flow reduction, greater erosion prevention and greater soil carbon accumulation". (MDTP 2008)

Thus, it is important to establish causal relationships between changes in land use, their impact on ecosystem health, and the resulting provision of ecosystem services. It is important to note, however, that these relationships may very well be interdependent and overlapping. While in some ecosystems, a single ecosystem service is the product of two or more functions provided by a healthy ecosystem, in other cases it may be that a single ecosystem function leads to two or more different services (Costanza, *et al.* 1997).

Moreover, a single change in land use behavior can lead to one or more improvements in ecosystem health indicators that in turn influence one or more ecosystem services. For example, one change in land use, such as reduced grazing, has a positive impact on basal cover, which in turn enhances multiple services, including water retention, storm flow reduction, erosion prevention, and soil carbon accumulation (MDTP 2008). In many cases, though, it is likely that changes in multiple land use behaviors will be necessary to ensure sustainable improvements in ecosystem health and services. In the case of the Maloti-Drakensberg watershed, increased water abstractions by agricultural or other users downstream from the highland grazing areas could neutralize the impact of reduced grazing on water flows throughout the catchment.

Typically, the analysis of these complex and interdependent relationships among land use, ecosystem health, and ecosystem service provision is based on observed (or modeled) differences in ecosystem health and service provision over time at a particular site under different management systems (MDTP 2008). Rather than measuring temporal changes in ecosystem services according to different management regimes, however, this thesis will instead compare spatial variations in ecosystem health against the predominant land use behaviors across the study site to understand how these different behaviors affect ecosystem health and the resulting provision of ecosystem services. Once these relationships have been established, the value of these services can be estimated using a variety of methods. It is worth noting that most PES schemes and ecosystem service valuations operate at a large scale, such as the level of a catchment or even an entire biome (e.g. Hassan 2003; Powell, *et al.* 2006; Turpie 2003; Turpie, *et al.* 2008). In contrast, this thesis will attempt to quantify ecosystem health and services at a small scale (< 5,000 ha) and will consider the challenges involved in this.

1.5.2 Valuing ecosystem services

It has been argued that valuing ecosystem services is "impossible" or "unwise" in light of, for example, the large uncertainties inherent in valuing goods and services not traded in markets (Costanza, *et al.* 1997). Nonetheless, the decisions taken every day governing human use of natural resources necessarily rely on implicit value judgments. The degree to which these decisions are based on implicit or explicit value judgments depends on whether ecosystem services are measured in monetary terms (Costanza, *et al.* 1997). Thus, a decision to locate major infrastructure projects at a safe distance from sensitive ecosystems, such as wetlands, is based on the recognition that wetlands provide humans with valuable ecosystem services, such as water and waste cycling, or recreation, which would be costly to replace.

At the same time, natural resource valuation has also been accused of undermining the argument that ecosystems should be protected for moral or aesthetic reasons. However, Costanza, *et al.* (1997) point out that there are "equally compelling moral arguments" for resource use decisions that are in direct conflict with conservation, such as the eradication of hunger. Therefore, placing a monetary value on ecosystem goods and services can facilitate better decision-making by explicitly taking into account the implicit values assigned to different and sometimes competing choices about natural resource uses (Alyward & Barbier 1992).

It is true that the calculation of a cumulative value for all ecosystem services provided to humanity is made impossible by the fact that human life could not exist without ecosystem goods and services. Human-made creations can, to a certain extent, mimic or replace some of these services; thus water purification plants can be constructed to replace the loss of natural purification services provided by wetlands. However, many ecosystem services are technically irreplaceable, or certainly at a large scale, and efforts to recreate self-sufficient life-supporting systems by artificial means in space and on earth (e.g. Biosphere II) have proven expensive and extremely complicated. As such, the total value of ecosystem services is infinite (Costanza, *et al.* 1997).

However, it is meaningful to measure "how changes in the quantity or quality of…ecosystem services may have an impact on human welfare" (Costanza, *et al.* 1997). Changes that are small but occur at a large scale, such as a change in oceanic acidity, and those that take place at a small scale but are of large magnitude, such as the elimination of an important species from a particular ecosystem, can have equally substantial impacts on human welfare. Thus, relatively minor changes in the composition of atmospheric gases can have catastrophic effects on the global climate, and the elimination (or introduction) of elephants can have far-reaching impacts on vegetation structure, micro-climate, and even small mammals in a given ecosystem (e.g. Costanza, *et al.* 1997; Lombard, *et al.* 2001). This thesis will aim to quantify how changes in a small ecosystem impact the provision of ecosystem services by the system.

Numerous different methods have been employed to place monetary values on ecosystem services (e.g. Aylward & Barbier 1992; Bellassen & Gitz 2008; Glenday 2008; Mitchell & Carson 1989). While methods for valuing provisioning and regulating services in financial terms are fairly straightforward, placing monetary values on supporting and cultural services, or other non-use categories, can be more contentious (Edwards & Abivardi 1998). Typically, provisioning and regulating services can be valued by estimating the cost of replacing those services; e.g. the cost of purchasing fuel wood or an alternative fuel in the absence of freely collected fuel wood, or the cost of commercial water treatment in the absence of a functioning riverine or wetland ecosystem (Hassan 2003). In contrast, cultural and other non-use services are often difficult, if not impossible, to replicate. Instead, it is possible to estimate their value through "contingent valuation" (CV) methods that ask users to reveal their willingness to pay for the preservation of biodiversity.

There is considerable debate over the accuracy of CV methods due primarily to the hypothetical nature of the questionnaires used to gather information about an individual's willingness to pay for a particular ecosystem service (e.g. Shultz, *et al.* 1998). Since an individual's explicit value judgment is based on their implicit knowledge and biases, CV of an ecosystem service is sensitive to limited information about ecological systems and the inadequate incorporation of social fairness, ecological sustainability, and other public goals. Additionally, the actual monetary payment individuals are willing to pay is strongly influenced by their own economic position. As a result, CV methods likely tend to underestimate the full value of natural resources (Costanza, *et al.* 1997). However, various modifications to the questionnaire design can reduce the sampling error caused by these limitations, and CV continues to be the favored tool for estimating hypothetical and non-use values (e.g. Bostedt, *et al.* 2008; Broberg & Brannland 2008; Heinzen & Bridges 2007; Richardson & Loomis 2009).

It is also important to note that the nature and value of services provided by an ecosystem varies across different spatial scales (Brown 1998; Hein, *et al.* 2006;). For example, reed cutting and fisheries are important services provided by wetlands at the municipal scale, while recreation is important at both the municipal and provincial scales, and nature conservation is most significant at the national and international scales (Hein, *et al.* 2006). Although direct use values of provisioning ecosystem services are often calculated at the municipal or even household level, most valuations of indirect regulating and supporting services, in particular, have thus far been conducted at the landscape scale or even at the national or regional level (e.g. Hassan 2003; Powell, *et al.* 2006; Turpie 2003; Turpie, *et al.* 2008).

This thesis will instead focus on the provision of direct and indirect ecosystem services at a micro-scale of less than 5,000 ha. In so doing, it aims to contextualize the macro-scale values measured over areas of hundreds of thousands or even millions of hectares into a local scale appropriate for informing local resource management institutions. This exercise will incorporate spatial heterogeneity into the valuation of different services by examining the contribution of direct ecosystem services, including fodder and fuel wood production, to the livelihoods of nearby resource users, as well as the value generated by locally-produced indirect services, such as the conservation of endangered species, to users at the municipal or catchment scale.

1.6 <u>PES models in South Africa</u>

South Africa contains a rich variety of ecological resources, including several biodiversity hotspots characterized by high levels of endemism (Myers 1990; Turpie 2003). However, only approximately six percent of South Africa's land area is contained within formal, state-protected areas. This leaves many critical ecosystems under increasing pressure, including from impoverished communities in search of readily accessible resources (Turpie, *et al.* 2008).

As discussed, the poor rely heavily on environmental goods and services at the local scale, such as wood for fuel and construction, fodder for cattle, and water and nutrient cycling to augment their livelihoods (Anderson & Pienaar 2003; Cartwright, *et al.* 2002; Ingle 2006; Shackleton, *et al.* 2008). Wealthy citizens also use environmental goods and services, but often from outside their immediate environment. Affluent people often utilize products derived from natural resources produced elsewhere, such as irrigation and drinking water (sometimes diverted over great distances), food grown using soil fertility of far-away arable land, furniture built from exotic forests thousands of kilometers away, and decorative natural crafts transported from overseas. Although evidence shows that human use of ecosystems does not by definition lead to degradation (e.g. Hahn, *et al.* 2005; Scoones 1994), it is also clear that healthy ecosystems are fundamental for producing the goods and services upon which human communities depend (e.g. Petrosillo, *et al.* 2007; Rapport, *et al.* 1998). By addressing the need to carefully manage ecosystems for optimal (human and ecosystem) health, the payments for ecosystem services (PES) model potentially offers an opportunity for contributing to local economic development while encouraging sustainable natural resource management (Turpie, *et al.* 2008).

Especially in the context of developing countries, where spending on short-term economic growth and social development often trumps conservation planning, placing a monetary value on ecosystem services can be "the only way" to ensure rational decision-making about natural resource use (Aylward & Barbier 1992). Depending on the definition of poverty employed, it is estimated that between 26 % and 57 % of South African citizens live in poverty (Leibbrandt & Woolard 1999, 2001). As elsewhere in the developing world, "conservation in South Africa has historically been perceived as a luxury", with poverty relief programs given priority access to limited land and resources (Turpie, *et al.* 2008). At the same time, given the demonstrated importance of ecosystem goods and services to the livelihoods of poor, it is imperative that investments are made to ensure their continued availability (Anderson & Pienaar 2003; Davenport 2008a). The payments for ecosystem services (PES) model presents developing countries with a compelling mechanism for potentially meeting conservation and poverty relief goals simultaneously. Examples in South Africa include several "Working for" programs, such as Working for Water, Working for Wetlands, and Working for Woodlands, as well as the World Wildlife Fund's Water Neutral program.

1.6.1 Working for Water

Although theoretically the exploitable volume of water flowing in South Africa's rivers is sufficient to meet its water demands, in practice spatial and temporal variations can cause severe shortfalls in supply. This is the result of a number of challenges, including unreliable and inadequate rainfall, increasing demands for water from agriculture, industry, and fast-growing urban centers, and the impact of invasive alien plants on South Africa's constrained water supply (Binns, *et al.* 2001).

In fact, Lester, *et al.* (2000) suggest that "throughout its history the provision of an adequate water supply has been one of the key limiting factors in the economic development of South Africa." It is true that formidable water transfer schemes and dam projects have played a key role in South African policy and politics during the Apartheid era and beyond (ANC 1994; Binns, *et al.* 2001). By the 1990s, however, it had become clear that the construction of additional water supply schemes could not keep pace with future demands from a growing population and rapidly increasing urban centers in a cost-effective manner (van Wilgen, *et al.* 1998). As such, the government turned its attention to the augmentation of water supplies through the eradication of invasive alien plants (IAPs) (Binns, *et al.* 2001; van Wilgen, *et al.* 1998).

Although South Africa has a long history of introducing non-native, or alien, plants from around the world, the introduction of a handful of species from Australia, Europe, and the Americas has led to a significant reduction in both water supply and biodiversity in the affected areas. Since the first European settlement was established in 1652, 744 tree species and 8,000 shrub and herbaceous species have been introduced into South Africa. Of these, only 153 species are considered "invasive," meaning they tend to out-compete local (indigenous) species (WFWP 1998).

By 1998, however, it was estimated that these IAPs covered a total area of roughly 10 million hectares, or 8 % of South Africa's surface, and resulted in the loss of 3,300 million cubic meters of freshwater each year, a figure equal to roughly 7 % of the annual flow of South Africa's rivers (Asmal, Foreward, in Versveld, *et al.* 1998). In addition to reducing water supply, IAPs also have a significant and negative impact on biodiversity, reducing "species-rich vegetation to single-species stands of trees" (van Wilgen, *et al*. 1998). Moreover, perhaps 90 % of the damage was caused by just 15 species from Australia, Europe and America in the *Acacia, Euculyptus, Hakea, Pinus,* and *Prosopis* genera (van Wilgen, *et al*. 1998).

The Working for Water (WfW) program was established by the government shortly after its transition to majority-rule in 1995 to concurrently address the negative impact of IAPs on the nation's limited water supplies and the urgent demand for job creation and economic development in post-Apartheid South Africa (Binns, *et al.* 2001; Turpie, *et al.* 2008; van Wilgen, *et al.* 2001). WfW hires teams of previously disadvantaged individuals to remove

IAPs from affected watersheds around the country. In so doing, it seeks to achieve four goals:

(i) "Enhance water security and promote equity, efficiency and sustainability in the supply and use of water;

(ii) Improve ecological integrity and counteract abnormal fires, erosion, flooding, scouring, siltation, and protect biodiversity;

(iii) Restore the productivity potential of the land and promote the sustainable use of natural resources, and develop economic benefits from land, water, wood and people;

(iv) Invest in the most marginalized sectors of South African society and optimize social benefits in a public works program". (Binns, et al. 2001)

The WfW model is unique in the global PES context because its funding is sourced primarily through poverty relief public works programs. With an annual budget of over R400 million, compared to R728 million for all national and provincial parks for the same year (RSA 2003), WfW "is the largest single natural resource based poverty relief and public works expenditure in the country" (Turpie, *et al.* 2008). Most of this funding comes from the Expanded Public Works Program, although the Department of Water Affairs and Forestry (DWAF) also contributes significantly to WfW through funds generated by tax revenue. Other (minor) funding sources have included international aid funding, the commercial forestry sector, and funds raised by DWAF from water charges (Turpie, *et al.* 2008).

Research suggests that the WfW program has indeed had positive effects on both local livelihoods and ecosystem health through increased provision of water. However, the high costs of on-going follow-up activities necessary to prevent re-growth of cleared areas have led some to question the sustainability of the program from an ecological perspective (Binns, *et al.* 2001). Moreover, the South African forestry industry remains dependent on IAPs, which complicates efforts to achieve 100% eradication (van Wilgen, *et al.* 1998). Finally, it is unclear whether the income benefits generated by short-term (two-year) contracts with WfW are leading to long-term improvements in the livelihoods of workers employed by the program (Binns, *et al.* 2001). Nonetheless, it is a promising example of the role that ecosystem service valuation can play in creating ecologically- and economically-beneficial outcomes.

1.6.2 Water Neutral

Whereas Wunder (2005) described PES as a voluntary transaction between two parties, the WfW is currently funded largely by taxpayers (the "buyers"), while the "sellers" are previously unemployed individuals contracted to remove alien plants on two-year contracts (Turpie, *et al.* 2008). However, DWAF's draft Water Pricing Strategy (2007) promises to increase the proportion of finances drawn from voluntary payments, including water user fees, by allowing for water users to be liable for the full cost of controlling specific invasive alien plants. It also provides for additional water provision and cost recovery for all water user sectors that contribute to invasive alien plant clearing according to their registered water abstraction (DWAF 2007).

The World Wide Fund for Nature (WWF) Water Neutral Scheme is designed to take advantage of this policy shift by working with members of the South African business and industry community to reduce and offset their water consumption. On the one hand, the scheme encourages voluntary monitoring and reduction of operational water consumption. At the same time, it also promotes financial support for alien plant removal through WfW to offset remaining water consumption through the release of equivalent volumes back into natural aquatic ecosystems. Thanks to this combination of actual reductions in operational water use and alien plant removal to offset the rest, participating organizations can claim to be operationally "water neutral" (WWF 2007).

The water neutrality program is modeled after global carbon neutrality programs that allow corporate institutions to "offset" their carbon emissions by investing in projects that directly reduce or prevent carbon emissions equal to the emissions their business will release. In light of the considerable annual water withdrawals attributed to both urban and industrial consumption (3,600 million kiloliters/year) and alien invasive tree species (3,300 million kl/yr) in this "water stressed" country, the WWF-Water Neutral Scheme aims to reduce the impact of both industrial production and alien plant invasion on South Africa's limited freshwater supplies (WWF 2007).

To test this model, in 2007 WWF and South African Breweries Limited (SAB) began piloting the joint implementation of a water neutral project that will offset SAB's operational water use at two of its domestic breweries (WWF 2007). It was necessary to first complete a thorough analysis of current annual operational water use at SAB's breweries in Cape Town and Port Elizabeth, as well as to calculate projected costs of reducing water consumption versus investing in WfW alien plant removal projects. Over the twenty-year life of the SAB water neutral project, it is anticipated that 350 hectares of alien-infested habitat will be cleared and returned to natural habitat; resulting in the equivalent of 360 full-time jobs being created; and roughly one million kilolitres of water per year will be made available for natural ecosystems or other users (WWF 2007).

1.6.3 Working for Wetlands and Working for Woodlands

The success of the WfW program has generated two offspring, Working for Wetlands and Working for Woodlands, which focus on habitat restoration rather than narrowly on service delivery (Turpie, *et al.* 2008). Both are housed within the South African National Institute for Biodiversity (SANBI) and are funded through the Department of Environmental Affairs and Tourism (DEAT). Like their parent WfW, Working for Wetlands (WfWet) and Working for Woodlands (WfWood) form part of the Expanded Public Works Programme (WfWet undated).

WfWet "offers technical expertise to landowners and collaborates with local partners to set rehabilitation objectives with the intention of improving the integrity and functioning of ecosystems," including the restoration of wetland biodiversity and the delivery of hydrological services (Turpie, *et al.* 2008; WfWet undated). WfWood, on the other hand, is primarily concerned with restoring the highly transformed sub-tropical thicket ecosystem once widespread throughout the Eastern Cape with the parallel goal of increasing carbon sequestration (Powell, *et al.* 2006; Turpie, *et al.* 2008).

Although both initiatives are currently funded by taxpayer funds, WfWood, in particular, has the potential to be largely self-funded through revenues earned from the sale of carbon credits on the voluntary market or through the Clean Development Mechanism (CDM) provided for by the Kyoto Protocol (Mills & Cowling 2006; Powell, *et al.* 2006). These mechanisms allow for the creation of a market for carbon that permits energy-intensive companies, typically in the developed world, to off-set their carbon emissions by funding carbon capture (sequestration) elsewhere, often times in the developing world. Although the intricacies of the global carbon market can complicate financial feasibility studies, in general these carbon credits can provide an important source of revenue for funding habitat restoration (Powell, *et al.* 2006).

An off-shoot of the WfWood program, the Subtropical Thicket Rehabilitation Project (STRP) seeks to build off of the "Working for" model by employing contractor teams to harvest and replant *Portulacaria afra* individuals to restore degraded thicket sites while sequestering carbon (Powell, *et al.* 2006). Research suggests that *P. afra* is not only an important species for restoring thicket ecosystems, but also that this plant has a surprising capacity for rapid carbon fixation even in the semi-arid environments where thicket thrives (Mills & Cowling 2006). Although carbon fixation varies considerably according to site- and planting-specific factors, several studies have demonstrated the potential for carbon sequestration through the restoration of degraded thicket sites with *P. afra*: up to 0.42 kg C m^{-1} yr^{-1} (Mills & Cowling 2006; Powell 2008). Plans to certify and sell the carbon sequestered by planting *P. afra* in degraded sites as carbon credits through the CDM or voluntary carbon offset markets could therefore provide a crucial source of self-generated revenue for thicket restoration (Powell 2008).

One noticeable characteristic of the South African PES programs described above is that many focus on the delivery of one specific ecosystem service, such as water flows, biodiversity, or carbon sequestration. Initially the Maloti-Drakensberg Transfrontier Project (MDTP 2008) attempted to quantify multiple ecosystem services provided by mountain grasslands, including enhanced river baseflows in the winter months, sediment reduction, and carbon sequestration. Ultimately, the project determined that only the first two services could be reliably proven to deliver one overarching service, namely improved water flows to downstream users. As such, the first two services could actually be considered intermediate ecosystem services necessary for the production of enhanced water flows.

However, it has been found that "the modification of ecosystems to enhance any one service generally comes at the expense of other services" (IIED 2007). For example, in the case of the MDTP, water flows can be increased through the enhanced basal cover. However, improving the provision of water services from the Maloti-Drakensberg ecosystem necessarily depends on reduced grazing pressure in the communally managed rangelands in the mountains (MDTP 2008). Thus, the enhancement of water provisioning services comes at the expense of incomes derived from herding activities in the ecosystem. To capture these trade-offs among different services, this thesis will quantify changes not only in the magnitude and value of individual ecosystem services, but also in the overall mix of ecosystem services provided by the study area. Based on the results of this valuation exercise, the work attempt to identify an optimal suite of ecosystem service benefits spread equitably among different users.

1.7 Objective and Key Questions

The objective of this thesis is to compare the impacts of alternative levels of land use intensity on the ecological health of the Bathurst commonage and neighboring Waters Meeting Provincial Nature Reserve and examine how the ecological health of different sites affects the supply and value of ecosystem services provided to the residents of the Kowie River catchment.

Key questions include the following:

(i) What is the ecological health of the commonage, managed for natural resource consumption, as compared to adjacent land in the protected Waters Meeting Nature Reserve?

(ii) What is the per hectare value of ecosystem services provided by natural resources subject to alternative levels of land use intensity?

(iii) What are the differences between various levels of land use intensity in terms of the variety and magnitude of ecosystem services provided to the community?

1.8 Structure of the thesis

The next chapter describes important characteristics of the study area, including the socio-economic, land use, and biodiversity context of the Eastern Cape and the geographic, social, and political context of Bathurst, in particular. Chapter Three presents the methods and results of an ecological health evaluation completed in the study area as part of this thesis and briefly discusses its implications for future land management on the study site.

Chapters Four and Five detail the benefit transfer and field-based valuations, respectively, of ecosystem services provided by the study area. Each of these chapters is organized according to ecosystem service, with relevant methods, analysis, results, and discussion addressed for each service in turn along with limitations to the data and methodologies. Finally, Chapter Six compares the values derived from the alternative valuation methods presented in Chapters Four and Five and proposes possible management options for integrating land uses in the study area to facilitate a socially-, economically-, and environmentally-optimal mix of direct and indirect ecosystem goods and services.

2 Chapter Two

The Study Area: Bathurst and Its Environment

2.1 Introduction

This thesis will compare and contrast the impacts of various land uses on direct and indirect ecosystem service values provided by two adjacent pieces of land with alternative land management regimes: the Bathurst commonage and Waters Meeting Nature Reserve (NR). To better understand how each regime impacts and is impacted by regional land use trends, it is important to place the two study sites within the context of the Eastern Cape in general, and the Bathurst area in particular. This chapter will first explore several characteristics of the Eastern Cape that are important for explaining current regional land use trends, including socio-economic factors, primary land uses, and biodiversity. Next, the social and political history of Bathurst will be discussed with special attention given to both historical and contemporary use and governance of the Bathurst commonage and Waters Meeting NR.

2.2 Deliniation of the study area

The Bathurst commonage (33° 30′ S, 26° 46′ E) and Waters Meeting NR (33° 33′ S, 26° 42′ E) (referred to collectively as the "study area/site") are both located within the Ndlambe local municipality (LM) in South Africa's Eastern Cape province. The commonage covers approximately 3,000 hectares (2,989 ha) and is located to the west and south of the village of Bathurst. It is bordered to the north by the Lushington River, to the west by the Kowie River, and to the north and south by Waters Meeting NR (4,247 ha) (Fabricius, *et al.* 2006). As shown in Figure 2.1 below, the study area as a whole is bordered almost exclusively by private farms, with the exception of the village of Bathurst to the east.

Waters Meeting NR is a small provincial nature reserve (shown in green in Figure 2.1) managed by Eastern Cape Parks located along the Kowie River. The western-most portion of the commonage (shown in light pink below) separates the smaller, more trafficked Waters Meeting I (1,445 ha, south) from the larger, less-accessible Waters Meeting II (3,802 ha, north) (Fabricius, *et al.* 2006). While Waters Meeting I is well known by local residents as a picturesque destination for hiking and weekend picnics, there are no signs directing tourists to Waters Meeting II. Since Waters Meeting II would therefore likely contribute little to tourism or direct-use values, this study will be restricted to the more popular Waters Meeting I (hereafter referred to simply as "Waters Meeting NR").

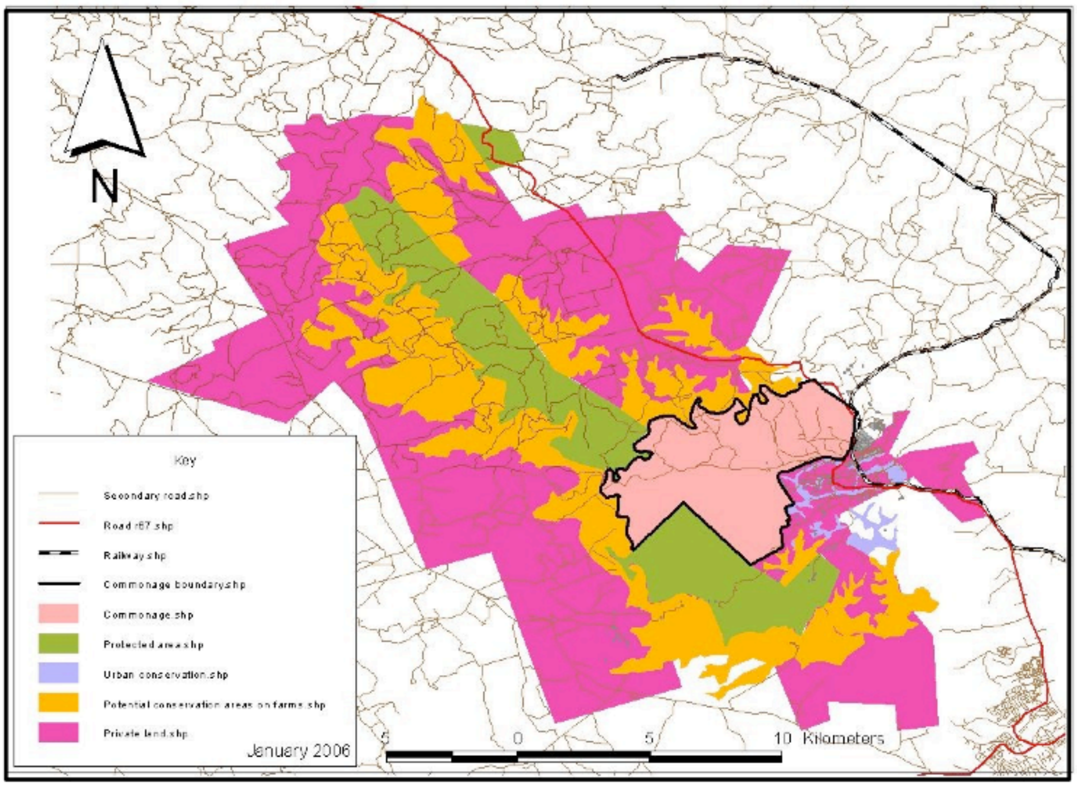

Figure 2.1: Map of the study area

Source: Fabricius, *et al.* 2006

2.3 <u>The Eastern Cape</u>

2.3.1 The socio-economic context

The Eastern Cape is distinguished both by its vast size and by the depth of its poverty. Covering just under 14% of South Africa's total area, the Eastern Cape is the second largest province in the country. It lies along the southeastern coastline between the Western Cape to the west and Kwa-Zulu Natal to the east. The most impoverished of South Africa's nine provinces according to average monthly household expenditures, the Eastern Cape is home to some 6.4 million people, or roughly 14% of all South African citizens (StatsSA 2003).

Encompassing two of the former "homelands" or "Bantustans," the province is populated mainly by people of African descent (88 %), followed by 7 % Coloured, 5 % White, and 0.3 % Indian/Asian people. Although English is the language of commerce, isiXhosa and Afrikaans are also spoken widely here (StatsSA 2003). Another reflection of its political history can be found in the province's land tenure distribution: while the majority of land is privately owned (66.5 %), fully 29.5 % of land in the province is held communally, with the remaining 4 % owned by the state (CSIR 2000). The potential implications of the government's land reform programme on the future distribution of land rights, which were discussed in Chapter One, should not be underestimated (DLA 2005).

2.3.2 The land use context

Despite recent trends toward some eco-tourism related activities, commercial agriculture still dominates the Eastern Cape's landscape and economy (Eastern Cape Business Information Service 2001; StatsSA 2003). Livestock production is the primary agricultural land use, but there is some limited commercial cultivation in areas with higher rainfall or access to irrigation (Nel & Davies 1999). As discussed in Section 1.2.2 in Chapter One, economic pressures, including a decline in the commercial value of animal products, such as wool and mohair, increasing competition from global markets, and the withdrawal of state subsidies for agriculture during the 1990s have all negatively impacted the local agricultural industry (ABSA 2003; Atkinson 2005; Atkinson & Buscher 2006; Simbi & Aliber 2000).

These trends, in turn, have resulted in numerous knock-on effects, including diminished agricultural purchasing power and a concomitant decline in agricultural employment that has led to the eviction of farm workers (Atkinson 2005; Atkinson & Buscher 2006; Nel & Hill 1997). As mentioned above, the migration of former farm workers to nearby towns has placed increasing pressure on local natural resources, especially communal resources such as commonage land, with potentially detrimental impacts for both food security and biodiversity (Atkinson 2005; Atkinson & Benseler 2004; Palmer 2005).

2.3.3 The biodiversity context

The Eastern Cape encompasses a stunning array of biodiversity, particularly as it uniquely encompasses seven of the eight biomes found within South Africa: Forest, Fynbos, Grassland, Nama Karoo, Savanna, Succulent Karoo and Thicket (DEAET 2004). It is home to dozens of species of threatened birds, amphibians and reptiles, as well as the largest number of vegetation types (28) of any province (DEAET 2004; Low & Rebelo 1996). In fact, seven of these vegetation types are endemic (greater than 90 % of their extent occurs only in the Eastern Cape) and the province includes no fewer than five internationally recognized centers of plant endemism, including the Albany, the Cape Floristic region, the Succulent Karoo, the Pondoland, and the Drakensberg (DEAET 2004; van Wyk & Smith 2001).

The semi-arid valleys of the Eastern Cape province, South Africa, support one of these unique vegetation types, known as succulent or semi-arid solid thicket (Lechmere-Oertel, *et al.* 2005a, b; Lloyd, *et al.* 2002). Although the thicket biome was originally thought to be an interface among numerous distinct vegetation types (subtropical forest, Afromontane forest, fynbos, Karoo and grassland), more recent research has led to the understanding that thicket is in fact a precursor to its many surrounding biomes (Cowling 1984; Cowling, *et al.* 2005; Everard 1987; Low & Rebelo 1996; Lubke, *et al.* 1986). Thicket has long been referred to as Valley Bushveld, but it has also been variously depicted as savanna encroached by bushclumps, scrub forest, and a taller, scrubbier variety of fynbos (Acocks 1953; Fabricius, *et al.* 2003).

Ranging from two to three meters in height, thicket vegetation is "dense, perennial, semi-succulent and thorny" (Lechmere-Oertel, *et al.* 2005b quoting Everard 1987; Lloyd, *et al.* 2002). The biome covers approximately 1.7 million hectares of the Eastern Cape and displays a variety of growth forms and a rich diversity of plant species (Cowling 1984; Lloyd, *et al.* 2002). Rather than the clearly distinguished layers typical of forests, thicket vegetation typically features a dense matrix of tall shrubs often characterized by prominent spines of every size and shape that may have evolved in response to prolonged herbivory (Cowling 1984; Lloyd, *et al.* 2002; Vlok, *et al.* 2003). In addition, the succulent shrub *Portulacaria afra* Jacq. (Spekboom) not only plays a critical role in mitigating both droughts and floods, but also serves as a reliable source of fodder for both wild and domestic animals and is a dominant species in many thicket types (Stuart-Hill 1992; Vlok & Euston-Brown 2002).

The Thicket ecosystem also supports a number of critical ecological and evolutionary processes that influence climatic and edaphic conditions (vegetation-soil feedback loops that maintain a microclimate favorable for supporting thicket vegetation), including fire, herbivory, seed dispersal, and soil and water conservation (Vlok & Euston-Brown 2002). However, less than 5 % of this second smallest biome in South Africa is protected within formal conservation areas, leaving much of the remaining thicket areas vulnerable to land transformation for commercial agriculture or intensive resource collection by rural communities (Pote, *et al.* 2006). The implications of land transformation on the ecosystem services provided by Thicket will be discussed further in later chapters.

2.4 <u>The Bathurst context</u>

2.4.1 The geography of Bathurst

Located approximately 250 m above sea level, the town of Bathurst receives an average of 717 mm of rainfall per year (SAWS 2005). It has a mild subtropical climate characterized by mean daytime and nighttime temperatures between 10 and 21 degrees Celsius during the winter months and between 17 and 26 degrees Celsius in the summer.

Vegetation on both the commonage and Waters Meeting NR is dominated by various types of Succulent Thicket that fall within the Albany Centre of Endemism, one of five such centers in the Eastern Cape, which has been described as an interface between six of the seven biomes represented in the province (DEAET 2004; Van Wyk & Smith 2001). The Centre is characterized by its numerous varieties of succulent plants, and it is estimated that fully 15 % of the plant species found here are endemic to the area (Anderson & Van Wyk 1999; DEAET 2004). The future of this unique vegetation system is at risk due to a variety of threats, including both direct influences of human activity, such as overgrazing, agriculture, and urbanization, and indirect human impacts, such as alien plant invasion (Cowling & Hilton-Taylor 1994). In fact, the Eastern Cape Biodiversity Strategy reports that no fewer than five species are already extinct and recommends expanding Waters Meeting NR to prevent further losses (DEAET 2004).

The western half of the commonage and much of Waters Meeting NR are characterized by rugged terrain and steep slopes that support primarily Albany Thicket and Albany Valley Thicket (Fabricius, *et al.* 2006; Lombard, *et al.* 2003). Albany Thicket primarily occurs in moist areas atop Witpoort quartzite formation, whereas Albany Valley Thicket tends to be limited to clayey soils originating from the Weltevrede formation, as well as from Dwyka Tillite. The valleys that support the latter tend to be narrow ravines rather than wide-open river valleys, with additional Albany Valley Thicket supported by other moist micro-sites, including south-facing slopes. In contrast, the hilltops and relatively flat areas on the most eastern section of the commonage (near town) are dominated by Grahamstown Grassland Thicket, a mosaic of Albany thicket and grassland that occurs where soils resulting from Witpoort and Weltevfrede formations have been exposed to intermittent fires (Vlok & Euston-Brown 2002).

A thorough vegetation survey of the Bathurst commonage conducted by Hobson (1993) found various types of Thicket, but also areas of grasslands and wetlands, all interspersed with small areas of riparian forest on the slopes and valley bottoms. Woody plant species common to Albany Thicket that Hobson (1993) identified in the study area include *Schotia latifolia*, *Scutia myrtina*, *Pittosporum viridiflorim, Canthium inerme,* and *Olea europaea* subsp. *africana*, whereas *Euclea undulata*, *Pappea capensis*, *Schotia afria,* and *Sideroxylon inerme* are dominant in the pockets of Albany Valley Thicket. In addition to numerous succulent plants (e.g. *Euphorbiaceae*, *Crassulaceae*, *Lilliaceae*), two threatened cycad species are also found within the commonage (*Encephalartos altensteinii* and *E. latifrons*). On the other hand, the suppression of fire in the grassland areas of the commonage has led to the encroachment of woody shrubs, such as *Acacia karroo, Rhus pallens,* and *Scutia myrtina* (Lloyd, *et al.* 2002; Vlok & Euston-Brown 2002).

The commonage also supports many plants that attract birds, such as Anacardiaceae, Celastraceae, Rutaceae, Ebenaceae, and Oleaceae. In fact, the whole Bathurst area, including the commonage, Waters Meeting NR, and parts of town, is home to over 117 recorded species of birds (Fabricius, *et al.* 2006). Hobson (1993) notes several species of interest to bird watchers, such as the Knysna Woodpecker (endemic to South Africa) (*Campethera notate*), the Cuckoo Hawk (*Aviceda cuculoides*), Crowned Eagle (*Stephanoaetus coronatus*), Fish Eagle (*Haliaeetus vocifer*), Black Sparrow-hawk (*Accipiter melanoleucus*) (also endemic to S.A.) and six species of owl, including both the Cape (*Bubo capensis*) and Giant Eagle (*Bubo lacteus*) Owls.

The study area also supports roughly two-dozen mammal species, including the rare tree hyrax (*Dendrohyrax arboreus*) and the rare and endangered blue duiker (*Philantomba monticola*) (Hobson 1993). Other species common to the both the commonage and Waters Meeting NR include medium-sized mammals such as aardvark (*Orycteropus afer*), bushbuck (*Tragelaphus scriptus*), bushpig (*Potamochoerus porcus*), porcupine (*Hystrix africaeaustralis*), and Chacma baboon (*Papio cynocephalus ursinus*), as well as smaller mammals such as *Cape clawless otter* (*Aonyx capensis*), grey duiker (*Sylvicapra grimmia*), rock hyrax (*Procavia capensis*), scrub hare (*Lepus saxatilis*), Smith's red rock rabbit (*Pronolagus saundersiae*), springhare (*Pedetes capensis*) and vervet monkey (*Cercophecus aethiops*). Thanks to its fences, Waters Meeting NR can also support larger mammals, such as Cape Grysbok (*Raphicerus melanotis*), Greater kudu (*Tragelaphus strepsiceros*), Grey rhebuck (*Pelea capreolus*), and Mountain reedbuck (*Redunca fulvorufula*), which may no longer be present on the commonage (Davenport, pers. comm. 2008; Earle, pers. comm. 2008; Smithers 1983). A bushbuck carcass observed inside the commonage as well as reports of other leopard (*Panthera pardus*) evidence in the surrounding area by respondents to the contingent valuation survey conducted as part of this thesis (section 5.3.2) suggest that there may also be a few remaining leopard on the study site.

In addition to hosting over one hundred bird species, at least twenty mammals, and hundreds of plant species, the Kowie River catchment is home to the endemic and threatened fish *Sandelia bainsii* (Fabricius, *et al.* 2006; Hobson 1993). Known colloquially as the Eastern Cape Rocky, this small fish grows to about 30 cm long and only exists in tributaries of three rivers in the Eastern Cape. Although a systematic sampling in the Kowie River catchment has not recently been conducted, the study area may support some of the few remaining *S. bainsii* individuals in the Lushington River, which runs through the commonage and eventually joins the Kowie River flowing from Waters Meeting NR to Port Alfred (J. Cambray quoted in Fabricius, *et al*. 2006).

According to a local expert, Dr. Jim Cambray of the Albany Museum, the Rocky may go extinct in the next ten years unless management actions are taken to protect it from local threats, such as loss of habitat, invasive alien fish like bass and catfish, and sedimentation in the Kowie. Because the Rocky is so specially adapted to its habitat, it acts as an indicator of the health of the Kowie River as an ecosystem. Therefore, the sharp decline in Rocky numbers observed over the past thirty years in the Kowie River may be an indication of ecosystem decline (Bendix 2001; Cambray 1996; J. Cambray, pers. comm. 2008).

Thus, in addition to playing a critical role in ecosystem services, as mentioned above, the Bathurst commonage and Waters Meeting NR also contain a number of endemic plants, several threatened species, and a variety of bird and mammal life. For all of these reasons, one important recommendation of a recent community land use survey was the expansion of existing conservation protections beyond Waters Meeting NR to include critical parts of the commonage that still host high levels of biodiversity (Fabricius, *et al.* 2006).

Figure 2.2 places the study area within the broader regional conservation context (Cowling, *et al.* 2003; Fabricius, *et al.* 2006; Rouget, *et al.* 2006). In fact, the entire study area falls within the Fish-Kowie Megaconservancy identified by the Sub-Tropical Thicket Ecosystem Planning (STEP) Project as a priority conservation area in the region (Cowling, *et al.* 2003). To ensure long-term conservation of the Thicket biome, this regional conservation programme designed ecological “corridors” to include areas of intact Subtropical Thicket vegetation, elephant habitat (believed to be a keystone species for the biome), protected areas and conservancies, and areas important for ecological processes (Cowling, *et al.* 2003; Rouget, *et al.* 2006).

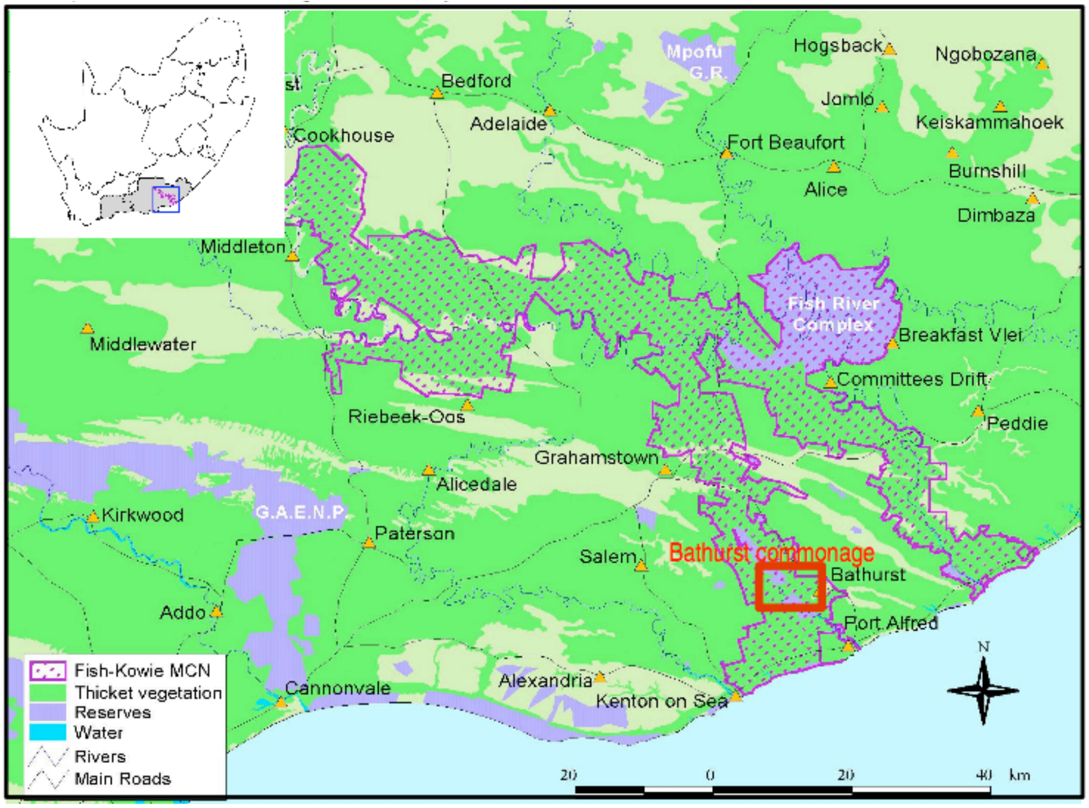

Figure 2.2: The Fish-Kowie Megaconservancy

Source: Fabricius, *et al.* 2006

Crucially, the STEP programme also emphasized the value of "keeping people on the land" and recognized the role that multiple land uses can play in achieving the goals of agricultural production, water management, and nature conservation (Knight, *et al.* 2003; Knight & Cowling 2003). Although the implementation of the Fish-Kowie Megaconservancy has stalled in Bathurst due primarily to the limited capacity of institutions governing natural resources, especially the commonage, the proposed Megaconservancy Network necessary to achieve STEP conservation goals within the corridor remains a useful blue-print for long-term land use planning in the study area (Fabricius, *et al.* 2006).

2.4.2 The social and political implications of a peri-urban settlement

Bathurst (33° 49′ S, 26° 83′ E) falls within Ward 5 of the Ndlambe LM, which is one of the nine local municipalities that together form Cacadu District Municipality. Ward 5 consists of the urban centre of Bathurst, Nolukhanyo township, and Freestone and Wilsons Party settlements (Ndlambe IDP 2007). Located 15 km north of the burgeoning town of Port Alfred, Bathurst is surrounded mostly by private farms, with the exception of the commonage and Waters Meeting NR, making it a unique "crossroads between urban and rural areas" (Fabricius, *et al.* 2006). It is located in the heart of one of South Africa's largest pineapple-growing areas, but chicory and cattle production are also major features of the surrounding agricultural landscape (Ndlambe IDP 2007).

This peri-urban settlement was home to approximately 3,023 people of working age in 1986, but by 2001 the population had doubled to 6,929 people, likely a reflection of the contraction of the local agricultural industry as described in Chapter One (Higginbottom, *et al.* 1995; Ndlambe IDP 2007). Most residents (49 %) have lived in the Bathurst area for between 19 – 64 years, while roughly one third (34 %) have moved there in the past 18 years (Ndlambe IDP 2007). The most recent Ndlambe Integrated Development Plan (Ndlambe IDP 2007) notes that this "influx of farm workers…and people from neighbouring municipalities seeking new economic opportunities…is placing increasing pressure on the housing delivery program and efforts to eradicate informal settlements."

There are approximately 3,621 people capable of working in the Bathurst area, but only 731 of these are actually employed with an additional 39 who find seasonal work, such as harvesting chicory and pineapples, leaving 1,683 unemployed individuals and 1,169 residents classified as 'other' (Ndlambe IDP 2007). The unemployment levels of Ndlambe LM as a whole place it in the middle of Cacadu's nine local municipalities, but its poverty rate is the highest in Cadadu District Municipality, with 64 % of Ndlambe LM residents living in poverty, up from 52 % in 1996 (Cacadu IDP 2007; Global Insight 2006; Ndlambe IDP 2007). The incidence of poverty is calculated by measuring the proportion of people living in households whose total income falls below specific thresholds, ranging from R 893 for a single person household to R 3,314 for a household with eight members (Global Insight 2006). As shown below in Table 2.1, over 16 % of households in the Bathurst area earn no income whatsoever and another 16 % earned between R 1 and R 4,800 (2008 US$ 648) per year (Ndlambe IDP 2007).

Table 2.1: Proportion of households (%) earning annual incomes (Rand)

Census unit	**Annual household income**						
	No income	**1 - 4,800**	**4,801 - 9,600**	**9,601 - 19,200**	**19,201 - 38,400**	**38,401 - 307,200**	**> 307,201**
Ndlambe	18.3	9.0	23.4	19.2	12.2	16.6	1.5
Ward 5	16.1	15.9	13.9				

Sources: Ndlambe IDP 2007; StatsSA 2003

In Ndlambe LM as a whole, just 1.5 % of the population earned more than R 307,200 per year (2008 US$ 38,605) according to the 2001 Census (StatsSA 2003). These high levels of unemployment and poverty in Ndlambe LM are reflected in its residents' heavy dependence on government social grants: nearly 7,000 residents (representing 58 % of households in the local municipality) receive either a child grant (3,927), an old age pension (1,689 beneficiaries), a disability grant (1,544), a foster care grant (181), or 'other' social grant (Ndlambe IDP 2007).

In addition to high levels of poverty and unemployment, there are a number of related factors that complicate local economic development in Ndlambe LM. These include poor economic infrastructure, such as few banking facilities, low rates of property ownership (just 40 % of households own their property), unsettled land claims, and limited functional literacy (54 %). In fact, nearly 13 % of residents in Ward 5 of Ndlambe LM have had no formal schooling whatsoever. Furthermore, age, disability and gender demographics also place Bathurst among the most at-risk settlements in Ndlambe LM. The Bathurst area is home to the second highest proportion of dependents under age 5 in Ndlambe LM, as well as the largest proportion of disabled persons. Over 46 % of households in Ward 5 are headed by females and 15% of its households earn less than R 800 per month, the highest concentrations for each category in Ndlambe LM (Ndlambe IDP 2007). Overall, the Human Development Index (HDI), which measures the overall level of development of an area on a scale of 0 – 1 (1 being high) based on a combination of factors, including life expectancy, literacy, and income, placed Ndlambe LM at 0.52 in 2005, below both the Cacadu District Municipality average (0.57) and the Eastern Cape as a whole (0.53) (Global Insight 2006).

The urban centre of Bathurst is made up of one main street along the R67, the main road to Port Alfred, around which most of Bathurst's few businesses are located. The neighborhoods surrounding this business district are populated primarily by rate-payers, whereas roughly 1,760 households are located in the Nolukhanyo township, which was created early in the 19th century for housing Bathurst's African residents (Higginbottom, *et al.* 1995; Ndlambe IDP 2007). Although official income and other demographic data are not computed below the Ward level, in general the rate-payers who reside in the urban centre tend to be members of relatively affluent households; some are descendents of the 1820s British settlers that used Bathurst as an administrative centre (Fabricius, *et al.* 2006). In contrast, residents of Nolukhanyo township are typically unable to afford municipal rates: according to a recent survey of Nolukhanyo households who access commonage resources, annual cash income was roughly R 18,100, of which over half (52 %) was provided by social grants, roughly 45 % was earned income, and remittances contributed the remainder (Davenport 2008a).

Unfortunately, Bathurst has been plagued by historical divisions along political, economic, and cultural lines. In particular, racial tensions between the relatively small number of rate payers in the town and the typically impoverished majority of non rate-payers were enflamed during the political transition to democracy in the 1990s (Fabricius, *et al.* 2006). In addition to bitterness over the perceived subsidization of common services for the (black) non rate payers by the more affluent (white) rate payers, disagreements over the management of the grazing rights commonage have been a source of pressure for many decades (Higginbottom, *et al.* 1995).

2.4.3 A brief history of the Bathurst commonage

The Royal Commission initially bestowed the Bathurst commonage upon "the people of Bathurst" in 1825. In 1924 the commonage expanded with the addition of 400 ha from the Bathurst Forest Reserve[1]. Since then, its management has fallen to the local authority, which is charged with governing the land "in accordance with local byelaws and the needs of the Bathurst community." Historically, revenues collected from grazing and dipping fees, municipal rates, fines and permits were used to fund the management of the commonage. However, the present governing authority, Ndlambe municipality, neither actively manages the commonage nor collects any revenues from it (Fabricius, *et al.* 2006).

Although livestock grazing has always dominated land use on the commonage, local residents from all sectors have used this common resource to serve a variety of purposes throughout the past two centuries, including timber collection, pineapple cultivation, and human settlement. In fact, the commonage was the initial location of Bathurst's "native reserve" until it was re-zoned to accommodate 12-acre pineapple plots for white farmers in the 1950s. Similarly, the privilege of grazing livestock on the commonage was reserved for white owners only until democracy in 1994 (Fabricius, *et al.* 2006).

From about the 1950s, livestock grazing on the Bathurst commonage was governed by a rotational grazing system created at the behest of local cattle owners. Bank & Hobson (1993)

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[1]This is the south-western portion of the commonage that dissects Waters Meeting NR and is sometimes referred to by its listing in the land register as "Erf 2" (WMNR 2007).

report that in addition to eight dams, watering points served by a system of boreholes and pipes were still equipped for regular use up until the 1990s. In contrast, the last decade has seen the collapse of active management on the commonage and the resulting deterioration of its infrastructure, including the boreholes and fences separating grazing camps (Fabricius, *et al.* 2006). Although commonage guidelines originally stipulated that users were only allowed 20 head of cattle each, it appears that some owners "grossly exceed" these numbers, with one individual grazing perhaps 200 head on the commonage (Fabricius, *et al.* 2006).

A recent report by Fabricius, *et al.* (2006) on the current status of commonage use and management concludes that the land continues to serve a variety of purposes similar to those observed over the past decade, including vegetable cultivation, stone quarrying, municipal waste storage, (illegal) hunting, and natural resource collection. The results of their community consultation process (depicted in Figure 2.3 below) revealed that the commonage is an important source of multiple resources in addition to cattle, including fuel wood, medicinal plants, bush meat, and other living resources for direct users in the Nolukhanyo community. In addition, residents of all socio-economic levels in Bathurst and Nolukhanyo access several abiotic resources, such as soil, water for gardening, and shade; as well as numerous intangible services, including clean air and water, aesthetic beauty, biodiversity, and spiritual connections with nature and one's ancestors (Fabricius, *et al.* 2006).

Nevertheless, the community consultation process confirmed that livestock production, primarily of cattle, remains one of the predominant land uses on the commonage today, as has been the case throughout its history (see Figure 2.4 below). According to the Nolukhanyo Cattle Owners Association and the Ndlambe municipality, 127 cattle owners have registered grazing rights on the commonage. However, expert input from the Bathurst Cattle Owners Association and Mr. Van Deventer, who performs the cattle dipping Bathurst, suggests that there are in addition roughly 70 – 75 unregistered cattle owners, a few of whom may be "absentee owners" who do not reside in Bathurst. There are also perhaps 80 goat owners who graze a total of roughly 500 animals on the same land as the cattle (Fabricius, *et al.* 2006).

While dipping records indicate that cattle numbers on the commonage fluctuate between 500 and 700 head (average of 550), a handful of owners reportedly underrepresented the extent of their herds so grossly that the top four owners may possess nearly 500 head between them. Although these claims could not be verified due to the lack of records and accountability governing cattle management on the commonage, Fabricius, *et al.* (2006) noted that "many respondents and even the cattle owners themselves have agreed that cattle numbers are far too high for the commonage to support them all." Historical stocking regulations on the commonage stipulated a maximum of 20 head per owner, without specifying how many owners were allowed to graze their cattle on the commonage. Based on their sample of 21 cattle owners, Fabricius, *et al*. (2006) estimated that on average owners graze 24 animals each on the commonage, but at least two individuals graze over 100 and 200 head, respectively.

Nonetheless, a household survey of Nolukhanyo commonage 'users' completed by Davenport (2008a) found that fuel wood is the most commonly utilized resource on the commonage. His study revealed not only that roughly 60 % of the households sampled used the Bathurst commonage to access between two and four key livelihood resources annually, but also that 90 % of users sampled utilized the commonage for fuel wood collection, compared to 40 % who grazed livestock (cattle and/or goats) and 33 % who collected wild fruits there.

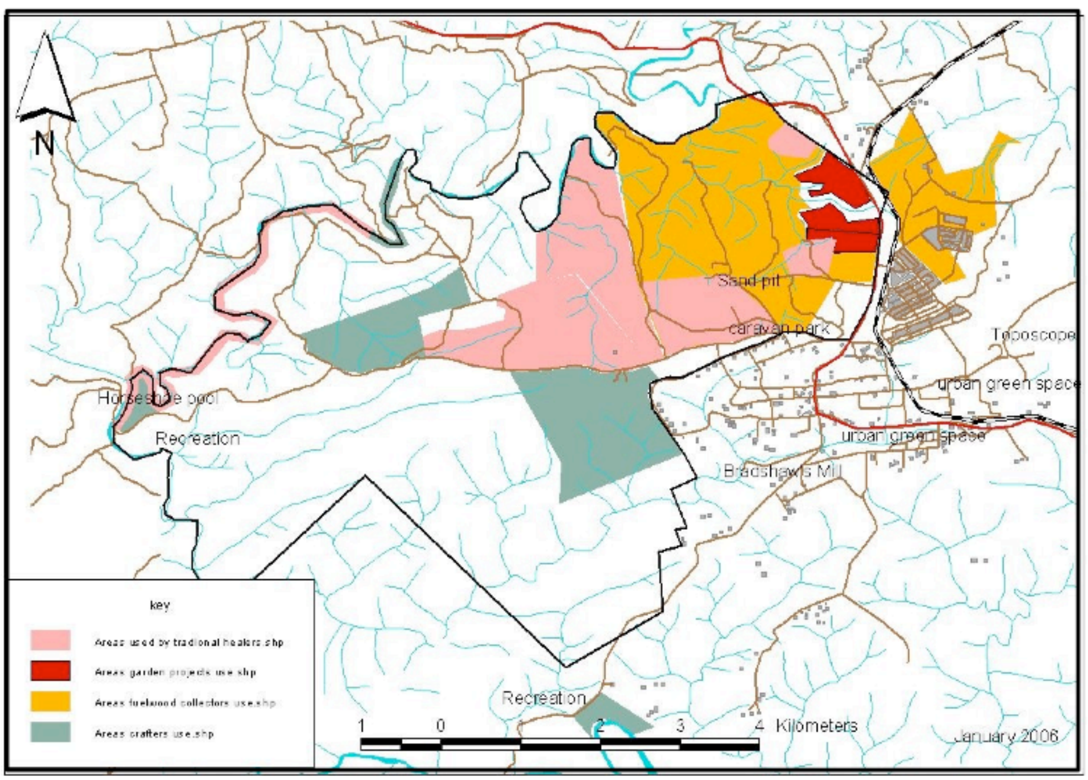

Figure 2.3 Key resource use areas for different users

Source: Fabricius, *et al.* 2006

Although livestock provided the single largest direct use value contribution to user households, contributing on average roughly R 1,836 per year to livestock owners' household incomes, fuel wood contributed the greatest overall value to local livelihoods, on average R

1,641 per year per household across 90 % of sampled households. All other commonage resources sampled contributed less than R 50 per year with the exception of wild fruits, which provided a value of nearly R 196 per year. Across all user households in Nolukhanyo, income (value) derived from commonage resources represented on average about 17 % of their total livelihoods, with the remainder supplied by a combination of employment, social welfare grants, garden yields, and off commonage resources (Davenport 2008a).

Fortunately, Figures 2.3 and 2.4 also verify that the spatial ecology of the commonage lends itself to intensive resource use in the relatively flat and less ecologically diverse areas near the township, while the harsh terrain and steep slopes down to the river beds farther from town virtually prohibit livestock grazing or heavy resource collection in these species-abundant areas (Fabricius, *et al.* 2006). It is also important to emphasize that many of the common resources accessed by Nolukhanyo residents are not readily available locally outside of the commonage, particularly for landless, impoverished households. As such, the Bathurst commonage plays an integral role in the local economy and culture.

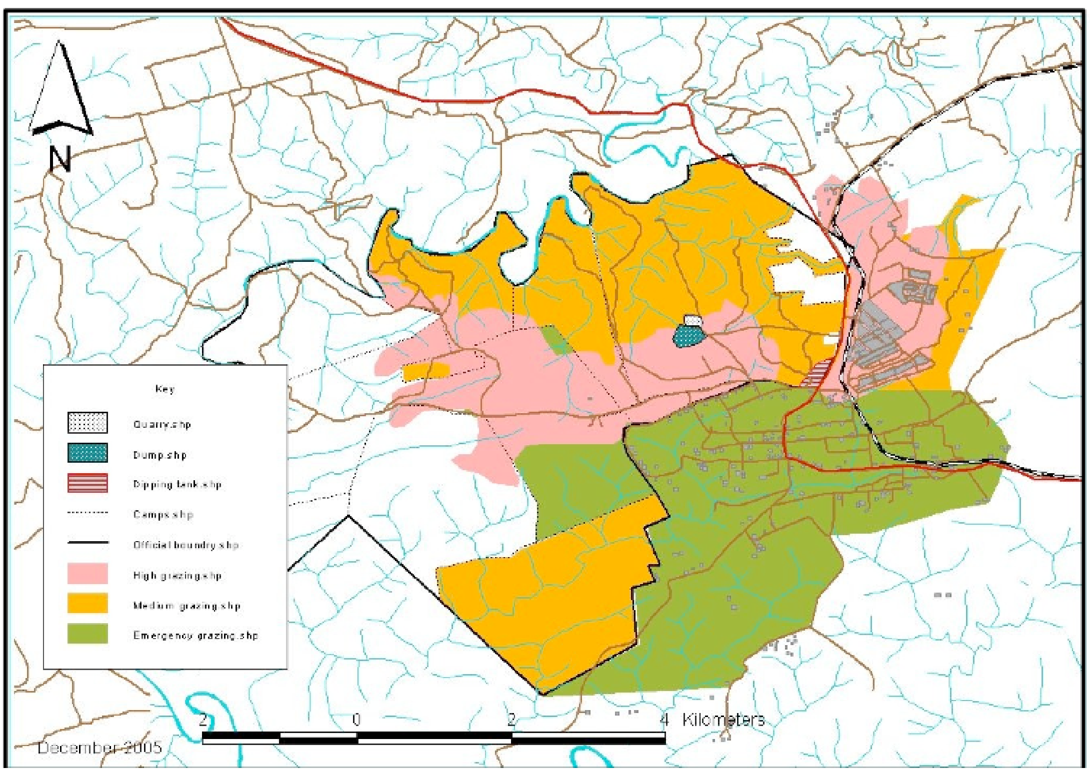

Figure 2.4 Extent of grazing on the commonage

Source: Fabricius, *et al.* 2006

2.4.4 Waters Meeting Nature Reserve

As mentioned above, the Bathurst commonage represents a critical link between the two sections of Waters Meeting Nature Reserve. The 4,247 ha-reserve was originally proclaimed a Forest Reserve in 1897 and later a Nature Reserve in 1952. The land is owned by the state and managed by the Eastern Cape Parks Board for conservation and tourism. With the exception of a particular grass species permitted for collection by traditional craft-makers, there is no direct resource collection allowed within the borders of Waters Meeting NR. Instead, users gain value from a variety of indirect services, including climate regulation, water and air circulation, biodiversity conservation, and recreation (Fabricius, *et al.* 2006). For example, local residents from Grahamstown to Port Alfred frequent the reserve for picnics on the weekend and the occasional paddle up the Kowie River from its mouth at Port Alfred to an overnight hut at the popular Horseshoe Bend in the river.

Moreover, the Sarel-Hayward dam inside Waters Meeting NR supplies Port Alfred with its municipal water (WMNR 2007). Although roughly 14 % of the catchment area for the dam lies within the commonage borders, compared with just under 11 % within Waters Meeting NR, most of the rivers on the commonage are only annual (author's estimation; Fabricius, *et al.* 2006; Powell, pers. comm. 2008). As such, additional sources of water from the Fish River catchment have been diverted to ensure a steady supply of water to the Sarel-Hayward dam (Powell, pers. comm. 2008). Therefore, while it can be argued that the study area is not the primary source of Port Alfred's water supply, it nonetheless plays a vital role in the catchment that delivers this water (Fabricius, *et al.* 2006).

2.5 Towards a shared vision for sustainable land use

The consultative study completed by Fabricius *et al.* (2006) engaged the Bathurst community in a constructive process for identifying an "effective management structure" for the Bathurst commonage and neighboring areas, specifically Waters Meeting NR. A number of projects were proposed by the community to enhance the equity and sustainability of current land uses in the study area, including a craft and tourism centre, a conservancy, a nursery, and specialty (Nguni) or intensive livestock farming. By comparing the impacts of different land use intensities on ecosystem health and quantifying the current and potential values of both direct and indirect services provided by the study site, this thesis will potentially assist decision makers in identifying the most equitable and sustainable land uses so that the area can be managed for an optimal mix of direct and indirect ecosystem goods and services.

3 Chapter Three

Ecological Health Evaluation

3.1 Introduction

Although indigenous herbivores of considerable size have long co-existed with thicket (Kerley, *et al.* 1995; Midgley 1991), this ecosystem "is surprisingly sensitive" to heavy grazing by domestic herbivores, particularly goats (Mills, *et al.* 2005; Stuart-Hill 1991, 1992; Stuart-Hill & Danckwerts 1988). Numerous studies (e.g. Evans, *et al.* 1998; Hoffman & Cowling 1990; Johnson, *et al.* 1999; Kerley, *et al.* 1995, 1999; Moolman & Cowling 1994; Stuart-Hill & Aucamp 1993) have explored the changes caused by domestic livestock in thicket and have determined that long-term effects include "significant changes in vegetation cover and structure accompanied by a significant reduction in diversity of endemic geophytes and succulents" (Lechmere-Oertel, *et al.* 2005b).

The resulting landscape displays a marked loss of species diversity and resembles an "open savanna-like system with a cover of ephemeral grasses and forbs" (Mills, *et al.* 2005). This transformation from a "dense closed-canopy shrubland" into a "pseudo-savanna" takes place within a matter of decades, and sometimes within a single decade depending on the intensity of grazing (Hoffman & Cowling 1990; Kerley, *et al.* 1995; Lechmere-Oertel, *et al.* 2005a, b; Mills, *et al.* 2005). In their review, Lloyd, *et al.* (2002) found that roughly 800,000 ha of semi-arid thicket have already experienced this transformation, while another 600,000 ha are in the process of transformation.

In addition to marked changes in species diversity and plant groups, the transformation of semi-arid thicket also affects a number of ecosystem services, including nutrient cycling, water infiltration, soil moisture retention, and carbon sequestration (Lechmere-Oertel, *et al.* 2005a; Mills, *et al.* 2005). For example, Lechmere-Oertel, *et al.* (2005a) working in the Sundays River thicket found that while nutrients and organic carbon are concentrated beneath patches of perennial shrubs in intact thicket, transformed thicket displayed a distinctly different spatial pattern of soil fertility, with nutrients concentrated beneath canopy trees instead. In fact, the authors found that this pattern becomes homogenized in transformed thicket and overall soil fertility is reduced. Due to a dramatic loss of litter cover, transformation decreases the proportion of the landscape surface available for infiltration from 60 % in intact thicket to just 0.6 % in transformed areas. Transformed thicket also shows lower rates of soil moisture retention (Lechmere-Oertel, *et al.* 2005a).

Furthermore, intact thicket has been found to store a "surprisingly high" amount of carbon for a semi-arid region. Work by Mills, *et al.* (2005) discovered that intact thicket stores an average of 76 tons of carbon per hectare (t C ha^{-1}) in living biomass and surface litter, with an additional 133 t C ha^{-1} stored in soils to a depth of 30 cm. They hypothesize that these high rates of carbon accumulation may be attributed to the dominance of *P. afra*, a succulent shrub that "switches" between C_3 and CAM photosynthesis, produces large quantities of leaf litter (approximately 450 g m^{-2} yr^{-1}) and shades the soil densely.

Compared to intact thicket, Mills, *et al.* (2005) found that transformed thicket stored roughly 35 % less soil carbon to a depth of 10 cm and roughly 75 % less carbon in biomass. This finding has significant implications for thicket restoration, as restored areas could sequester as much as 80 t C ha^{-1}. Based on this research, the Working for Woodlands PES model plans to take advantage of the voluntary and Kyoto-sanctioned carbon markets by selling carbon credits created by planting *P. afra* in transformed ecosystems throughout the Eastern Cape (Powell, *et al.* 2006).

In addition to livestock grazing, the commonage has been subjected to numerous other major disturbances over the past fifty years, including pineapple and vegetable cultivation, human settlement, fuel wood collection, bush meat hunting, and infrastructure development (e.g. rubbish dump, quarry, etc.) (Fabricius, *et al.* 2006). A recent community mapping exercise demonstrated that even within the commonage, however, the intensity of land use varies considerably across time and space. Specifically, Fabricius, *et al.* (2006) identified a "high use" zone and a "low use" zone characterized by different levels of resource collection by the various commonage users. In contrast, the predominant land use on Waters Meeting NR has consistently been conservation for at least the past 100 years, which precludes direct natural resource collection (hence "no use") and limits visitors to indirect uses, such as hiking and wildlife viewing (Fabricius, *et al.* 2006).

To understand how these different land use intensities affect ecosystem health, this chapter will first explore changes in land use over the past fifty years on the study site through an analysis of historical aerial photographs. The discussion will then turn to the results of a field evaluation of several ecosystem health status indicators across the three land use zones (high, low, and no use) for each of three spatial layers: woody plants, herbaceous cover, and soil layer. In addition, the concentration of indigenous and exotic (i.e. cattle) dung pats will be compared across transects to estimate relative differences in grazing pressure, and the steepest slope angle within each transect will be compared to predict relative differences in susceptibility to erosion. The chapter will then discuss possible implications of variable land use intensities on ecological health across the site.

3.2 Data collection and transformation

With the exception of land cover change, which was evaluated through aerial photographs, the methods described below were used to collect ecological health data from 49 transects within three natural resource use zones: 16 transects of 200 m^2 in relatively 'intact' thicket (low use) on the Bathurst municipal commonage; 17 transects of 200 m^2 in relatively 'open' thicket (high use) on the Bathurst municipal commonage; and 16 transects in 'intact' thicket (no direct use) on Waters Meeting Nature Reserve (NR). Due to the relatively fewer number of accessible hiking trails and roads within Waters Meeting I compared with the more intensely-utilized commonage, the length of all but four transects in Waters Meeting NR was reduced to 30 m to facilitate data collection, resulting in 12 transects of 120 m^2 and four transects of 200 m^2. Both the location and direction of each transect were selected randomly.

To select the starting location of each transect, a 10 m x 10 m grid was created in ArcView 3.0 over the most recent available orthorectified aerial photographs of the study site (from 2004). Firstly, all cells within 10 m of the border of the site were excluded from the sampling process to avoid edge effects. Next, twenty-five grid cells were randomly selected within each of the three use zones (high, low, and no use). The upper-right-hand corner of each selected grid cell was taken as the starting point for each transect. A handheld Garmin GPS unit was used to locate the randomly selected starting points in the field; all points located on a slope of more than 30 % were eliminated from the sample due to the danger inherent in data collection. Upon reaching each randomly selected location, a metal stake was spun to determine the direction of the transect.

The first sub-section will detail the methods used to determine land use change using aerial photographs. The methods used for data collection in each of the three spatial layers–woody plants, herbaceous cover, and soil layer, as well as the transect-level characteristics (dung density and slope angle)–will be described in the next four sub-sections.

3.2.1 Land cover transformation

3.2.1.1 Preparation of the aerial photographs and sampling technique

All available aerial photographs covering the entire study site from 1956 to present were obtained from the South African Directorate of Surveying and Mapping. Photographs of the entire study site were available for the following years: 1956, 1965-67[1], 1973, 1990, 1998, and 2004. Only the photographs from 2004 were already orthorectified and georeferenced; as such, all other photographs were cropped in Microsoft Picture Viewer and georeferenced based on at least four points matched to the 2004 image using ArcMap 9.0. In lieu of orthorectifying the older images, at least two of every three photographs of a given location, depending on picture quality, were georeferenced to reduce distortions on the periphery and enhance the accuracy of the resulting overlapped image (Giannecchini, *et al.* 2007).

Following the methodology of Zhang, *et al.* (2005), images were compared to evaluate changes in the extent of vegetation cover over time (Lucas, *et al.* 1993, 2002; Mertens & Lambin 2000). A preliminary review of the photographs was completed to establish the following different land cover types on the study site: woody vegetation, cultivated areas (historical and more recent), grassland, river, roads, and other infrastructure. Although the photographs were scrutinized for areas of heavy erosion, no evidence to support the classification of this cover type was found.

A transparent grid was created in ArcView 3.2 to divide each photograph into squares of equal size (100 m^2) and imported into ArcMap 9.0 for analyzing the georeferenced photographs. All georeferenced photographs were viewed at a constant scale of 1:4,000 to determine land cover. However, picture quality varied considerably across years due to differences in the scale of each annual set of photographs, which varied from 1:20,000 to 1:60,000.

A random sampling of 15.4 % of the grid cells (roughly 424 cells inside the commonage and 200 within Waters Meeting) was created using a script. Randomly selected cells were consulted for each photograph to establish land cover changes over time, and a unique script was run for each viewing of each photograph. Land cover was classified based on the land cover falling on the point beneath the upper-right-hand corner of each randomly selected grid

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[1] The flight paths for 1965, 1966, and 1967 together covered the entire study site. Since no other complete set of aerial photographs were available between 1956 and 1973, the combined 1965-67 set were analyzed.

cell (see below for description of land cover types). Selected grid cells where this corner fell outside the study site were excluded from the analysis. Vegetation types reported by Hobson (1993) and land use zones as defined by a recent community mapping exercise (Fabricius, *et al.* 2006) were consulted as necessary for substantiating the cover type as defined by the selected point in the black and white aerial photographs.

This method was repeated for the commonage and Waters Meeting at least three times for each year to generate an estimation of standard error. Results of this analysis on the commonage and Waters Meeting were recorded separately as a percent of the points sampled and averaged across each of the three analyses per year. These mean annual proportional values were then plotted over time with standard errors. Based on the transition of commonage management to the democratically elected municipal government post-1994, the overall rate of transformation away from woody cover between 1998 and 2004 was considered comparable to the 5-year mean annual rate of deforestation used by Bellassen & Gitz (2008) for determining the value of avoided deforestation. Likewise, the 50-year mean rate of transformation of woody vegetation was used in place of the 20-year mean calculated by Bellassen & Gitz (2008).

3.2.1.2 Classification of land cover types

As previously described, vegetation on the study site is dominated by Albany Thicket, interspersed with Albany Valley Thicket in the narrow ravines, and Grahamstown Grassland Thicket on the hilltops and relatively flat areas of the commonage near Nolukhanyo and the town of Bathurst. Due to the varied but generally limited quality of the historical photographs, it was not possible to distinguish between different types of thicket vegetation. Instead, all points sampled where the land cover was determined to be a tree or shrub based on the presence of dark and/or dense vegetation were classified as 'woody vegetation.'

In contrast, historically cultivated ('old fields'), recently cultivated ('new fields'), and grasslands were characterized by a lack of woody vegetation and distinguished based on the degree to which a given point was covered by non-agricultural[2] vegetation. Points labeled 'new fields' include all agricultural areas that displayed visible signs of plowing and/or did

[2] Since it has been suggested that the proliferation of Grahamstown Grassland Thicket can be attributed at least partly to exposure to intermittent fires, it is possible that this vegetation type is not wholly 'natural' in the sense that it is influenced by human interaction (Vlok & Euston-Brown 2002). Nonetheless, for the purposes of the land cover change exercise, all points where the land cover was dominated by non-woody, non-agricultural vegetation were labeled as grasslands.

not show evidence of the re-growth of non-agricultural vegetation, typically grassland species. In comparison, fields were classified as 'old' if there were no visible signs of plowing and/or the site showed signs of non-agricultural vegetation re-growth (i.e. grassland cover). The 'grasslands' category therefore is distinguished by the dominance of non-woody, non-agricultural vegetation and includes formerly cultivated areas that have thoroughly reverted to non-agricultural vegetation to the point where no visible signs of their former agricultural use remain. Points that fell over open areas not covered by infrastructure or roads (see below) were labeled grassland. Points located in a shadow were assumed to be covered by the land cover class of the adjacent grid cell out of the shadow.

Rivers include both annual and perennial rivers as identified by the presence of water visible from aerial photographs. Points where a river's surface was covered by vegetation were labeled according to the vegetation type, rather than as a river. Roads include two-way dirt roads wide enough for vehicles, one-track drivable roads, as well as hiking and footpaths, and animal (i.e. cattle) trails as visible from above. Points labeled as infrastructure include those falling on non-natural structures (excluding roads), such as boreholes; the quarry and rubbish dump inside the commonage; former homesteads; and the dam inside Waters Meeting.

However, there are several limitations to the aerial sampling method chosen to assess land cover changes over time. Firstly, the accuracy of any aerial sampling method is limited by the resolution and quality of the images analyzed, which can be affected by glare or cloud cover (Zhang, *et al.* 2005). Secondly, using aerial images to determine land cover compromises the classification of areas beneath dense foliage. Finally, the random sampling method chosen does not allow for the characterization of distinct land cover changes at a given point over time. Thus, at best the results track broad land cover changes on the study site.

3.2.2 Woody plants

Indicators of the health of the woody plant layer include the percentage aerial cover, density of woody plants and stems (those < 2 cm and > 2 cm), woody stem diameter (for stems > 2 cm), damage to woody plants, standing stock of above ground woody biomass and carbon content thereof, and density and proportion of sexually mature *Scutia myrtina* stems (as a proxy for nectar production potential). The method used to measure each is discussed below.

3.2.2.1 Percentage aerial cover of woody biomass

The percentage of woody cover was visually estimated within four 1 m^2 plots located every 10 m (6 m for the 120 m^2 transects) along the transect, beginning at 10 m (6 m for the 120 m^2 transects) and ending at 40 m (24 m for the 120 m^2 transects), using a Modified Walker scale (Shackleton, *et al.* 2003). This involved a three-step process whereby the researcher first estimated whether each category covers (a) less than or (b) more than 50 % of the land area. Having narrowed the proportion to between 0 % – 49 % or 50 % – 100 %, the remaining area was again halved to facilitate estimation. Thus, if, in step one, it was determined that more than 50 % of the area was covered by woody cover, then the next step would be to determine whether the proportion of woody cover is between 50 % – 74 % or between 75 % – 100 %. A third estimation was made by halving the new range to, for example, between 75 % – 86 %. Finally, the researcher made a final estimate within this narrowed range. Where the percent of woody cover when viewed from 2 m above the ground was determined to be at least 75 %, a second estimate of aerial cover was made at ground level.

The four estimates of percent woody cover at 2 m above ground and ground-level, where applicable, for each plot per transect were averaged separately across each transect to approximate percent woody cover across the transect as a whole.

3.2.2.2 Density of woody plants and stems

The number of both plants and stems within each transect was counted and divided by the total area of the transect to estimate plant and stem density. Stems were disaggregated into those greater or less than 2 cm to distinguish between established plants (> 2 cm) and woody recruitment (< 2 cm).

3.2.2.3 Basal diameter

Digital calipers (Mitutoyo, series 500, model CD6CX) were used to measure the basal diameter of all stems > 2 cm to two decimal places as close the ground as possible but above any basal swelling/buttress. Where trees were larger than 150 mm in diameter, a dressmaker's tape was used to measure the basal circumference in centimeters above the basal swelling/buttress. Plants with multiple stems > 2 cm were measured for each large stem and the number of stems < 2 cm in diameter were counted.

3.2.2.4 Damage to woody cover

All cut stems per transect were counted, measured for basal diameter (or circumference) as above, and examined to determine whether the plant was still living. Where coppicing was not evident, the plant was assumed dead. Large trees were also examined for bark removal; none was noted.

3.2.2.5 Dead stems

In addition, all dead stems that remained standing were counted and measured for basal diameter (or circumference) as above; decayed stems that had fallen were identified as litter in aerial cover estimates but not counted in the damage estimates or measured.

3.2.2.6 Carbon content of above ground dry woody biomass

To the extent possible given the dense bramble of stems typical of thicket, the life form (tree or shrub) of each stem > 2 cm was determined. Any stem not identified as a tree based on field evaluation was assumed to be a shrub; plants with all stems < 2 cm were assumed to be shrubs. The cumulative basal stem area of shrubs and trees were calculated separately using an mean estimate of 1 cm per stem < 2 cm. Live and dead stems were treated separately to distinguish between the standing stock of living biomass and necromass. Allometric equations developed by Powell (2008) based on sub-tropical thicket sites in the Baviaanskloof Nature Reserve were used to convert the cumulative basal stem area (CBSA) of shrubs and trees into standing stock of carbon in above ground (live and dead) woody biomass (Table 3.1). All carbon and CBSA data were $\log_{10}$ transformed. Following Powell (2008), the carbon content of above ground dry biomass was estimated using conversation ratio of 0.48.

Table 3.1 Allometric equations used to estimate dry biomass/carbon (kg) from CBSA (m^2)

Life form	Reference species	R equation	R^2 value
Shrub	*Putterlickia pyracantha*	Log_{10} y (C (kg) = 1.0622*(Log10 CBSA (m^2)) + 2.7834	0.7784
Tree	*Acacia karoo*	Log_{10} y (C (kg) = 0.9068*(Log_{10} CBSA (m^2)) + 2.5771	0.7326

Source: Powell (2008)

3.2.2.7 Sustainability of current fuel wood collection

Using actual annual household use data collected by Davenport (2008a), the total value of all fuel wood collected on the commonage was estimated by multiplying the value per household (across all user households) by the total number of households collecting fuel wood as estimated by household surveys in Nolukhanyo. This figure was then converted back to actual biomass collected using the mean selling price of R 0.61/kg recorded by Davenport (2008a). This estimation of total biomass collected was repeated for fencing poles and wooden tools, the other two woody resources recorded by Davenport (2008a). The resulting estimation of total annual woody biomass collected from the commonage was then distributed proportionately between the high and low use zones according to their relative area. To increase the probability that current collection activities are sustainable, it was assumed that all wood collected was already dead. These estimates were compared to the annual growth of dead woody biomass on the commonage as estimated by assuming the carbon to biomass ratio of 0.48 used by Powell (2008) and a mean annual growth rate of 1.7 % of standing dead woody biomass (Shackleton 1998). For comparison, the same calculation was made for live woody biomass assuming a mean annual growth rate of 3 % (Shackleton 1993c).

3.2.2.8 Density of sexually mature *Scutia myrtina* stems

Each *Scutia myrtina* stem per transect was counted, measured for basal diameter (or circumference) as above, and examined for flowering or fruiting. However, due to drought at that period no stems were found to be flowering or fruiting during sampling in August – October 2008; instead a sample of 195 *S. myrtina* stems was examined for evidence of flowering or fruiting in May 2009. Roughly 50 plants each in four transects were evaluated at four different sites around Grahamstown (n = 195). For the largest stem of each plant, the basal circumference was measured and the presence of flowers or fruits was noted. These data were then grouped into size classes at 1.5 cm intervals, and the proportion of stems flowering or fruiting in each size class was calculated (Shackleton 1993b). The resulting distribution of flowering/fruiting plants per size class was used to predict the number of sexually mature stems on the study site. This estimate of sexually mature stems per land use zone was divided by the total area of all transects in each zone, respectively, to calculate sexually mature stems per square meter. Finally, the proportion of sexually mature stems within each land use zone was calculated by dividing the number of stems in each size class predicted to be sexually mature based on the Grahamstown data by the total number of stems recorded in each zone.

3.2.3 Herbaceous cover

Within each transect, four 1 m^2 plots were sampled every 10 m (6 m for the 120 m^2 transects) for herbaceous data. In each of the four 1 m^2 plots per transect, the percentage aerial cover of grass, forbs, and litter was estimated along with litter mass, mean grass height, and percent of grass biomass grazed.

3.2.3.1 Percentage aerial cover of grass, forbs, litter

The percentage of grass, forbs/herbs, and litter was visually estimated using the Modified Walker Scale in the 1 m^2 plot as described above.

3.2.3.2 Mean grass height

The height of the grass plant closest to the lower left-hand corner of each 1 m^2 plot was measured in centimeters using a dressmaker's tape. The standing height of the tallest leaf (without extending the blade) was measured from the ground. No flowers were noted so flower height was not measured.

3.2.3.3 Litter mass

All litter (dried organic matter) within each of the four 1 m^2 plots per transect was collected and air-dried for at least 20 days before weighing to determine total litter mass per m^2 in each transect.

3.2.3.4 Percent of grass biomass grazed

The percentage of grass biomass that had been removed by grazing was visually estimated, taking into account both the aerial percentage cover removed and the diameter of chewed stems. It was assumed that thick stems would have been much higher had they not been grazed and thus represented proportionally higher lost biomass compared with thin stems.

3.2.4 Soil layer

Various soil layer characteristics were evaluated, including soil composition by particle type (e.g. stone, clay, silt, sand), cation exchange capacity (CEC), organic carbon content, infiltration, moisture accessibility, ash content, carbon loss on ignition, soil classification, soil compaction, and percent aerial cover of bare ground, rock, and erosion.

3.2.4.1 Soil characteristics

Within each of the four 1 m^2 plots per transect, topsoil samples of the first 0 - 10 cm were collected using a hand trowel. The four soil samples from each transect were bulked in paper bags, thoroughly mixed, air-dried for at least 20 days, and sent to the Döhne Laboratory for analysis. The following indicators were measured by Döhne Analytical Services in Stutterheim (Ras 2008): soil composition by particle type (e.g. stone, clay, silt, sand), cation exchange capacity (CEC), infiltration rate (in the laboratory and not in the field), moisture accessibility, ash content, carbon loss on ignition, organic carbon content, and soil classification.

3.2.4.2 Percentage aerial cover of bare ground, rock, and erosion

The percentage area covered by bare ground, rock, and erosion in each 1 m^2 plot was estimated using the Modified Walker scale as described above.

3.2.4.3 Soil compaction

A pocket penetrometer was used to determine soil compaction (in kPa) in the center of each 1 m^2 plot. The four readings were averaged cross each transect.

3.2.5 Transect-level characteristics

3.2.5.1 Exotic and indigenous dung density

The number dung pats of indigenous and exotic (i.e. cattle) animals per transect were counted separately and divided by the total area of each transect to estimate density per m^2.

3.2.5.2 Slope angle

The percentage steepness of the steepest slope per transect was measured using an Abney level.

3.3 <u>Data analysis</u>

For most normally distributed data (woody plant density, woody stem density < 2 cm and > 2 cm), an ANOVA was used to determine the significance of differences among the three land use zones (high, low, and no use), and post-hoc pairwise comparisons were used to identify zones that were statistically different from one another. Based on a sampling of variables from the field data, it was determined that at least some of the variables did not meet the normality and homogeneity of variances assumptions necessary for using parametric analysis.

The land cover change data were first transformed using an acrsine transformation. Since the data were still not sufficiently normally distributed, the Kruskal-Wallis test was used to determine significant differences between the commonage and Waters Meeting NR. Finally, pairwise comparisons were used to identify significant differences between years for each site and between sites for the 2004 data. Aerial litter cover as viewed at ground level (below 2 m initial estimate) were also arcsine transformed but still not normal. As such, the Mann-Whitney test was used.

The *Scutia myrtina* data were also first transformed using an arcsine transformation and then subjected to two chi-squared tests. First the data were tested as per the 1.5 cm basal circumference intervals; since the result was not significant, all size classes above 12.9 cm basal circumference were merged to ensure that all classes contained at least five stems. This is the size class above which all stems were flowering.

For all other variables that were not normally distributed, the non-parametric Kruskal-Wallis test was also used to determine the significance of differences among land use zones. Where differences were significant at the 5 % level ($p < 0.05$) or higher, pairwise comparisons were then used to determine whether each zone was significantly different from every other zone.

In addition to testing for significant differences across sites (and years), Pearson's correlations were performed between pairs of variables of interest.

3.4 Results

3.4.1 Land cover transformation

As shown in Figure 3.1, differences across the two sites in the proportional land cover area are significant for both woody cover ($t = 19.6$; $p < 0.0001$) and grassland cover ($t = 46.1$; $p < 0.0001$) in 2004. Figure 3.2 and Figure 3.3 below show the proportional area covered by each of the land cover types on the commonage, while Figure 3.4 and Figure 3.5 depict the proportional area of cover classes found on Waters Meeting NR. Figure 3.6 and Figure 3.7 show the annual rates of change (%) in the proportional area of each land cover type between photographs on the commonage and Waters Meeting NR, respectively.

On the commonage, the proportional area covered by new agricultural fields (those with visible signs of cultivation and/or no re-growth of non-agricultural vegetation) ranged from a high of 5.5 $\pm$ 0.019 % of the total area in 1956 to a low of 1.6 $\pm$ 0.010 % in 2004. The area covered by old agricultural fields (those without visible signs of plowing and/or some re-growth of non-agricultural vegetation) varied between 0.5 $\pm$ 0.002 % in 1956 and 5.9 $\pm$ 0.019 % roughly a decade later. In contrast, the area covered by grassland ranged from a minimum of just 21.2 $\pm$ 0.019 % in 1998 to a maximum of 30.9 $\pm$ 0.006 % six years later, whereas the proportional area covered by woody plants fell from a high of 73.2 $\pm$ 0.019 % in 1998 to 63.5 $\pm$ 0.025 % in 2004, barely above the minimum observed woody area of 63.4 $\pm$ 0.027 % in 1965.

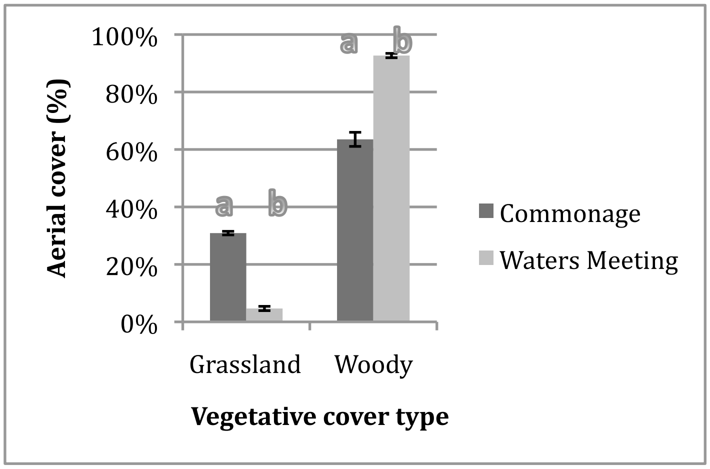

Figure 3.1 Differences in land cover across sites for 2004

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard deviation.

Differences across years in neither woody cover (H = 11.14; $p < 0.05$) nor new fields (H = 9.87; $p > 0.05$) are statistically significant for the commonage. However, both grassland cover (H = 14.34; $p < 0.05$) and old fields (H = 12.77; $p < 0.05$) differ significantly across years. For grassland cover, subsequent year-by-year comparisons revealed that 2004 is significantly different to both 1998 and 1990, but not to any other years (Figure 3.2). However, 1998 and 1990 are not different to one another or to any other years except 2004. For old fields, year-by-year comparisons showed that the only significant difference is between 1956 and 1965; differences between all other combinations of years are not significant (Figure 3.3). Open water, infrastructure, and roads covered a relatively small proportion of the commonage, accounting for less than 2.5 % in any year, and were thus grouped together as the "other" category in the graph.

Woody plants cover the vast majority of the surface area of Waters Meeting, ranging from 86.3 $\pm$ 0.013 % in 1973 to 92.7 $\pm$ 0.008 % in 2004. Differences in woody cover differ significantly (H = 11.17; $p < 0.05$) across years. Year-by-year comparisons revealed that 1973 is different to all other years, and 1965 is different to 1973, 1990, and 2004 (Figure 3.4).

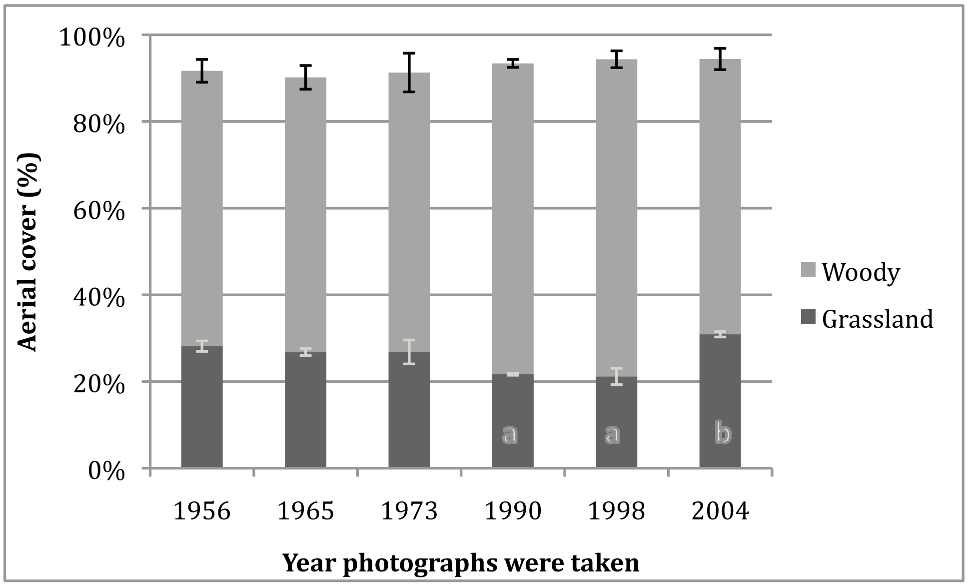

Figure 3.2 Proportional area of vegetative land cover types on the commonage (1956 – 2004)

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between years.

Bars indicate standard deviation.

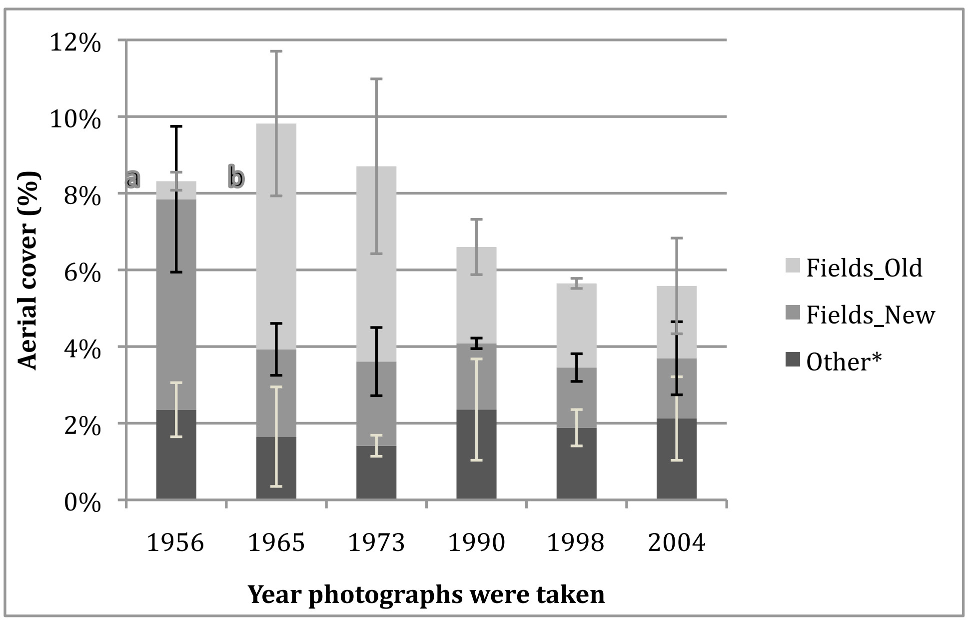

Figure 3.3 Proportional area of other land cover types on the commonage (1956 – 2004)

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between years.

*Other includes roads, rivers, and infrastructure. Bars indicate standard deviation.

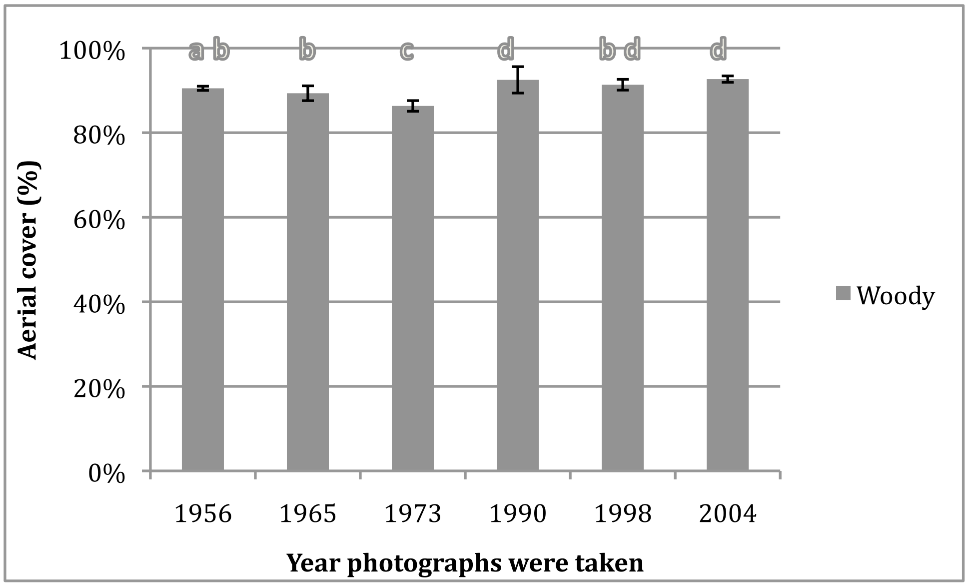

Figure 3.4 Proportional area of woody cover on Waters Meeting (1956 – 2004)

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between years.

Bars indicate standard deviation.

Grassland cover within the reserve also varied significantly (H = 14.01; $p < 0.05$) across years. Subsequent year-by-year comparisons revealed that 1956 is different to 1965, 1973, and 1990; 1965 is different to all other years except 1973; 1973 is different to all other years except 1965; 1990 is different to 1956, 1965, and 1973; 1998 is different to 1965 and 1973; and 2004 is different to 1965 and 1973. As would be expected within the protected area, no new fields were visible from the photographs. However, it appears that at least one old agricultural field may have been incorporated into the reserve. Together, rivers, roads, infrastructure and old fields represented less than 5 % of the total surface area (Figure 3.5).

Further exploration of annual changes in land cover types on the commonage found that between 1956 and 1965, the proportional area covered by new fields fell by 6.5 % annually and followed a continuous decline until 2004, which showed a slight (0.04 %) annual increase since the previous photograph in 1998 (Figure 3.6). Old fields declined at a varying annual rate (-1.6 % to -3.0 %) from 1965 to 2004 following a spike of +128 % between 1956 and 1965 (due to this disproportionately large change, the change between 1956 and 1965 is not shown in Figure 3.6).

The area covered by grassland showed relatively minor fluctuations (between –1.1 % and +0.01 %) until 1998, when this area increased dramatically by a mean annual rate of 7.7 %

until 2004. Woody plant cover also showed small variance (-0.02 % to +0.7 %) until 1998, when the proportional area dropped by on average 2.2 % annually until 2004 (Figure 3.6).

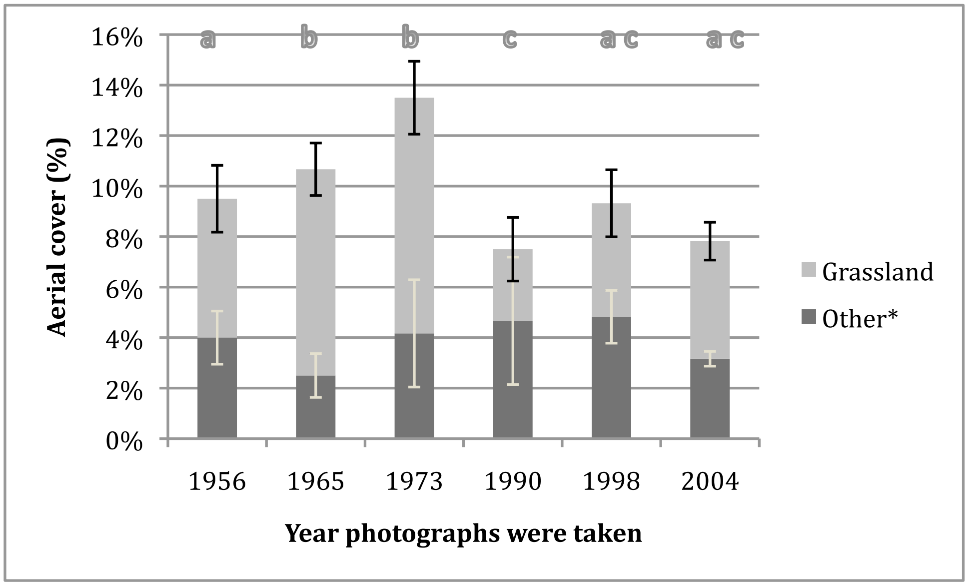

Figure 3.5 Proportional area of other land cover types on Waters Meeting (1956 – 2004)

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between years.

*Other includes old fields, rivers, roads, and infrastructure. Bars indicate standard deviation.

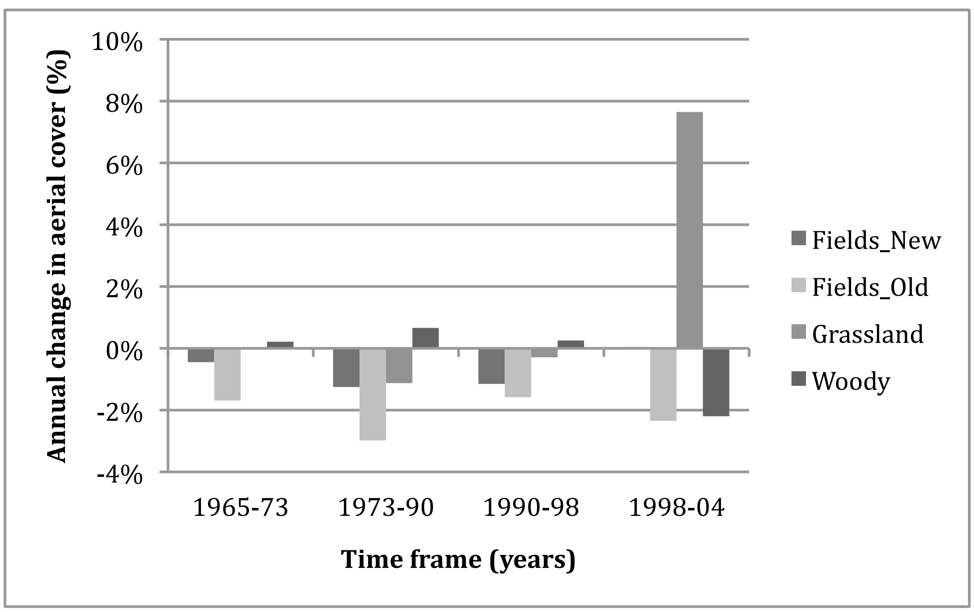

Figure 3.6 Annual rate of cover change since previous photograph of the commonage[1]

[1]Due to the disproportionately large increase in old fields (128 %) between the 1956 and 1965 photographs, this time frame was left out of the figure.

In contrast, the annual rate of change in woody plant cover on Waters Meeting varied by just 1 % over the photographs sampled (-0.42 % to +0.42 %). The annual rate of change in area covered by grassland inside the reserve was more varied, ranging from –4.10 % in the years 1973 – 1990 to +7.32 % between 1990 and 1998 (Figure 3.7).

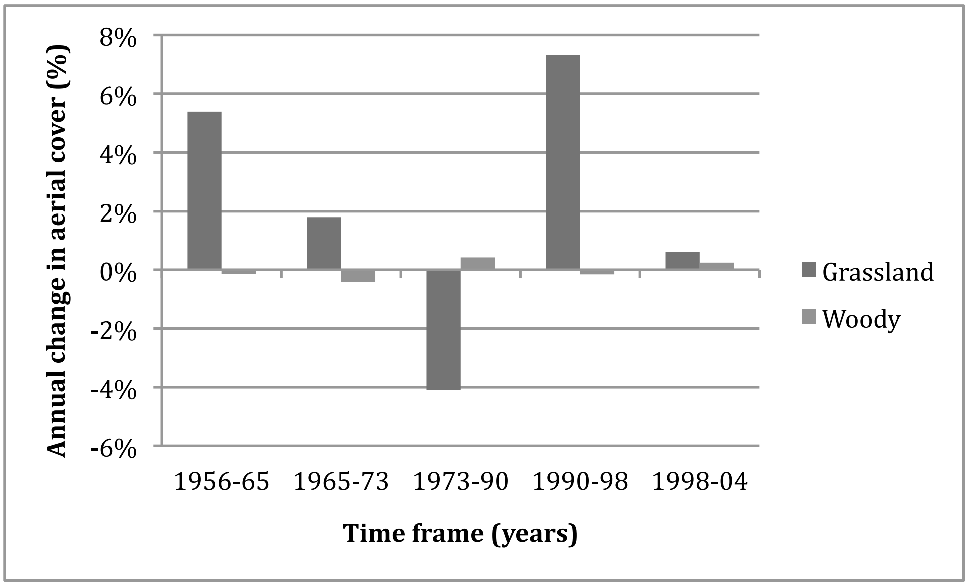

Figure 3.7 Annual rate of cover change since previous photograph of Waters Meeting

3.4.2 Woody plants

Significant ($p < 0.05$ or higher) differences were found between land use zones for all woody plant layer variables except minimum basal diameter of stems > 2 cm. These results (mean $\pm$ standard error) are described below.

3.4.2.1 Percentage aerial cover of woody biomass

The percentage woody aerial cover of woody biomass was significantly (H = 32.41; $p < 0.0001$) lower in the high use zone (18.7 $\pm$ 4.4 %) than in either the low use zone or Waters Meeting NR, neither of which were significantly different to one another (65.4 $\pm$ 7.0 % and 90.2 $\pm$ 3.3 %, respectively).

3.4.2.2 Density of woody plants and stems

The density of woody plants was significantly (F = 13.38; p < 0.0001) lower in the high use zone (0.2 + 0.04 plants/m^2) than in either the low use zone or Waters Meeting NR, which were also significantly different to one another (0.5 + 0.1 plants/m^2 and 0.4 + 0.03 plants/m^2, respectively). The density of stems > 2 cm in diameter was significantly (F = 16.64; p < 0.0001) lower in the high use zone (0.4 + 0.1 stems/m^2) than in either the low use zone or

Waters Meeting NR, which were also significantly different to one another (1.1 + 0.1 stems/m^2 and 0.8 + 0.1 stems/m^2, respectively). Finally, the density of stems < 2 cm in diameter was also significantly ($F = 7.13$; $p < 0.005$) lower in the high use zone (1.2 + 0.2 stems/m^2) than in either the low use zone or Waters Meeting NR, which were not significantly different to one another (2.8 + 0.3 stems/m^2 and 1.9 + 0.3 stems/m^2, respectively). These results are presented in Figure 3.8 below.

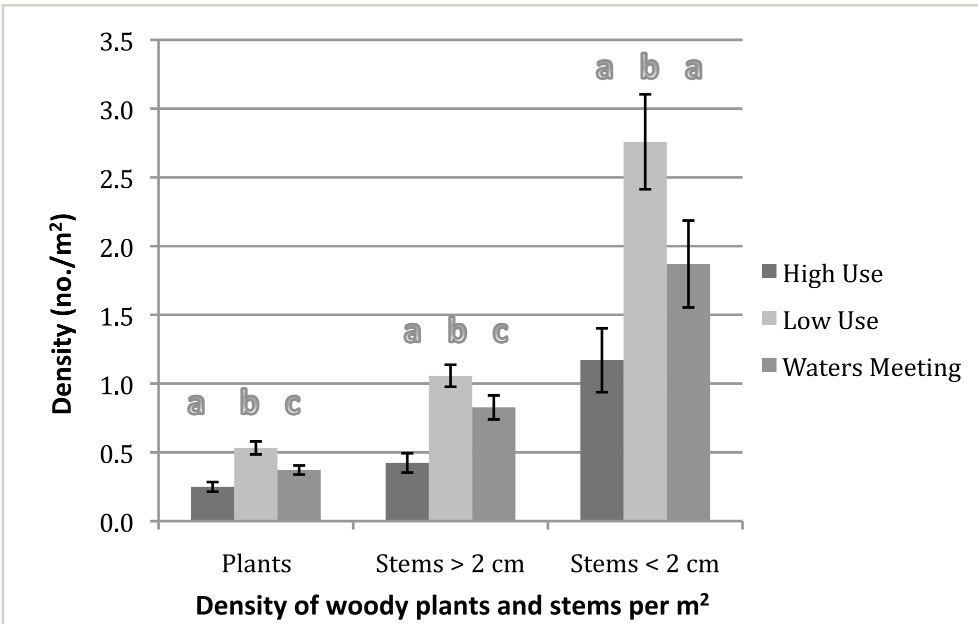

Figure 3.8 Differences[1] in woody plant and stem density among land use zones

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard error.

3.4.2.3 Basal diameter of woody stems > 2 cm

The mean basal diameter of stems > 2 cm was significantly ($H = 17.90$; $p < 0.001$) smaller in the high use zone (42.5 $\pm$ 1.7 mm) than in Waters Meeting NR (65.8 $\pm$ 4.0 mm). However, neither zone was significantly different to the low use zone (51.9 $\pm$ 2.9 mm). The maximum basal diameter of all stems > 2 cm was significantly ($H = 11.63$; $p < 0.05$) lower in the high use zone (199.1 $\pm$ 37.6 mm) than in Waters Meeting NR (407.8 $\pm$ 35.0 mm). Again, neither zone was significantly different to the low use zone (314.8 $\pm$ 39.7 mm). The minimum basal diameter of stems > 2 cm was not statistically different among the three zones (Figure 3.9).

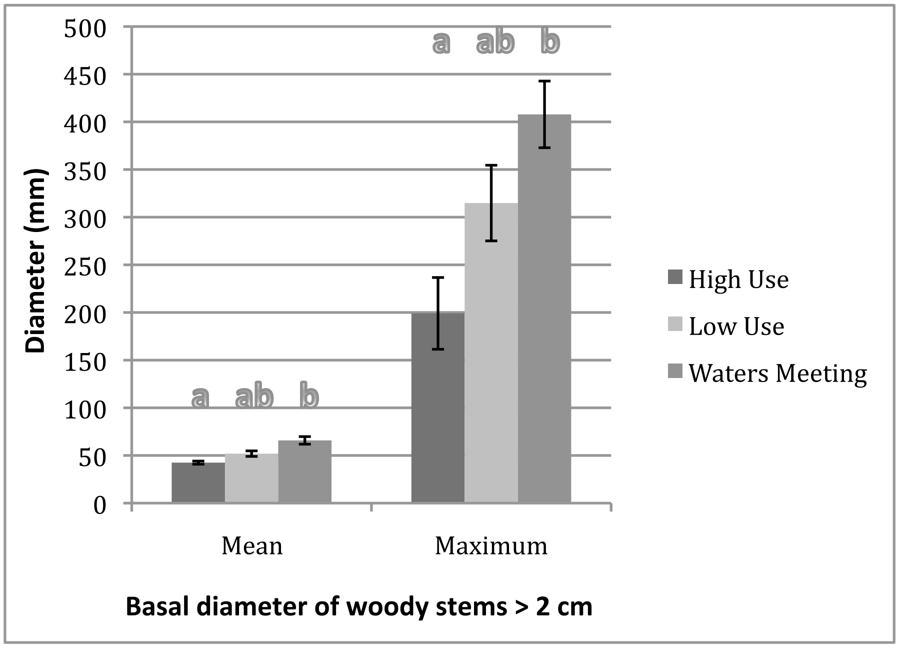

Figure 3.9 Differences[1] in basal diameter among land use zones

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard error.

3.4.2.4 Damage to woody cover

The density of stems > 2 cm damaged by visible cutting was significantly ($H = 20.74$; $p < 0.0001$) lower in Waters Meeting NR (none) than in either the high use zone or low use zone, neither of which were significantly different to one another (0.02 ± 0.005 stems/m^2 and 0.03 ± 0.01 stems/m^2, respectively). In contrast, the density of cut stems that showed coppicing was not statistically significant among the different land use zones (none in the high use zone; 0.003 ± 0.001 stems/m^2 in the low use zone; and none on Waters Meeting NR, where no cut stems were recorded). Large trees were examined for bark removal for medicinal use; none was noted.

3.4.2.5 Density of dead stems > 2 cm

The density of dead stems > 2 cm was significantly ($H = 12.23$; $p < 0.05$) higher in both Waters Meeting NR and the low use zone, which were not significantly different to one another (0.08 ± 0.01 stems/m^2 and 0.07 ± 0.01 stems/m^2, respectively), than in the high use zone (0.03 ± 0.01 stems/m^2). These results along with the density of damaged stems are presented in Figure 3.10.

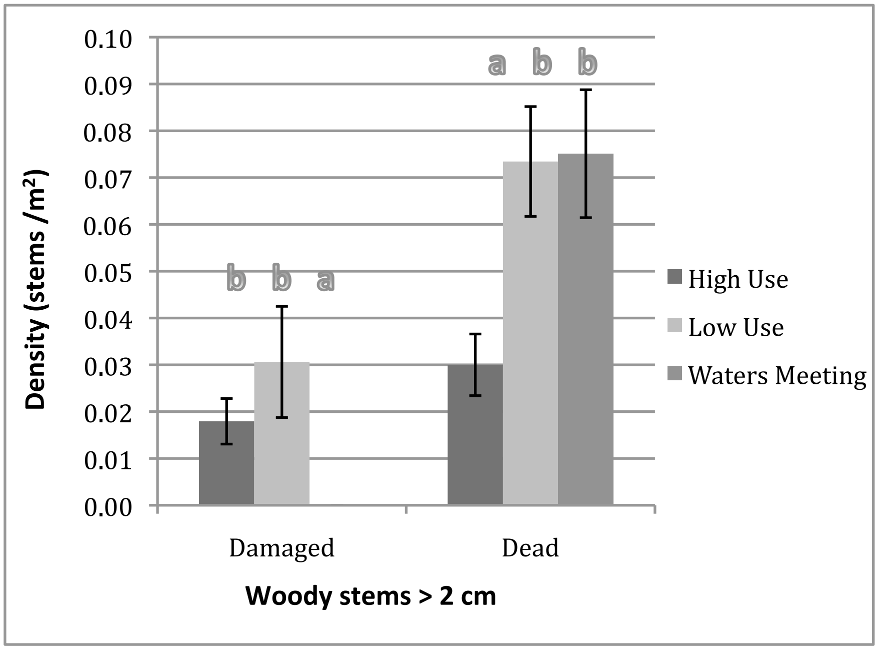

Figure 3.10 Differences[1] in damage to woody stems > 2 cm among land use zones

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard error.

3.4.2.6 Carbon content in above ground woody biomass

The total carbon content of above ground (ABG) woody biomass was significantly ($H = 13.98$; $p < 0.001$) lower in the high use zone ($5.66 \times 10^{-3} \pm 2.28 \times 10^{-3}$ t/m^2 or 56.6 ± 22.8 t/ha) compared with the low use zone ($1.30 \times 10^{-2} \pm 1.91 \times 10^{-3}$ t/m^2 or 129.9 ± 19.1 t/ha). However, the total carbon content of either site was not significantly different from that found on Waters Meeting NR ($6.46 \times 10^{-3} \pm 9.41 \times 10^{-4}$ t/m^2 or 64.6 ± 9.4 t/ha). Of the total ABG woody carbon in the high use zone, just 0.65 ± 0.30 % was contained in dead biomass, compared to the low use zone, where 1.25 ± 1.97 % of total carbon content was in dead biomass, neither of which were significantly different to one another. However, the proportion of total carbon in dead biomass on Waters Meeting NR (4.96 ± 6.13 %) was significantly ($H = 11.58$; $p < 0.01$) higher than that recorded in either of the other two land use zones (Figure 3.11).

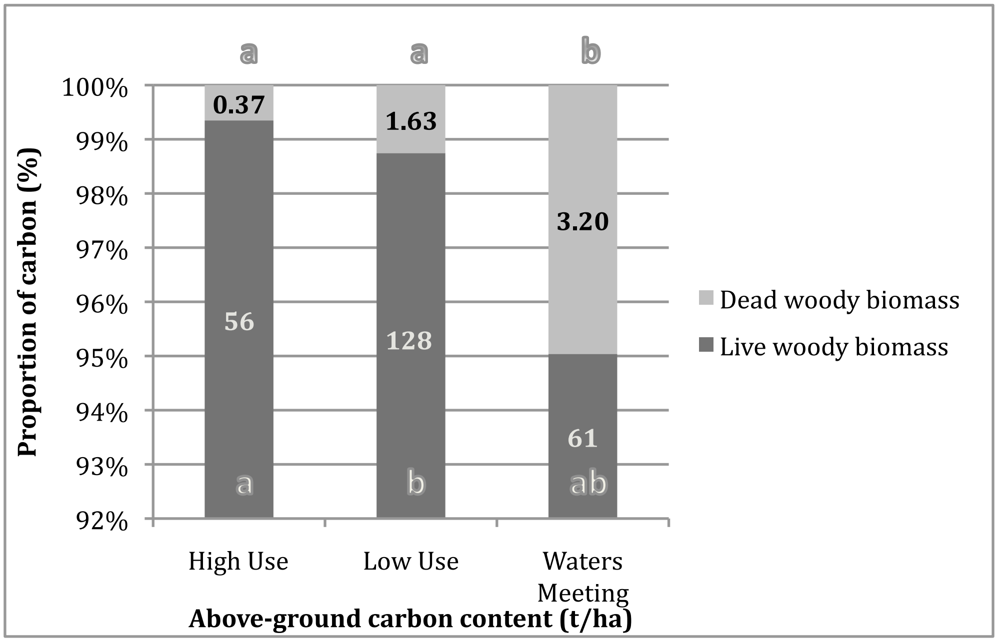

Figure 3.11 Differences[1] in carbon content of above ground woody biomass (t/ha)

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

3.4.2.7 Sustainability of current fuel wood collection

Annual growth of dead woody biomass in the high and low use zones was estimated to be 17.5 MT and 95.3 MT, respectively; in contrast, Waters Meeting was estimated to produce 164 MT of dead wood per year. These figures are small compared to annual use rates of 1,356 MT and 1,674 MT in the high and low use zones, respectively. With use to growth ratios of 7754 $\pm$ 2996 % and 1757 $\pm$ 679 % in the high and low use zones, respectively, it is inevitable that a considerable proportion of the fuel wood currently collected on the commonage comes from live woody stems. As such, use rates were compared to annual growth of live woody biomass.

Annual growth of live woody biomass in the high use zone was estimated to be 4,699 MT, compared with current fuel wood collection totaling 1,356 MT (28.9 $\pm$ 11.1 % of annual growth). In the low use zone, annual live woody biomass growth was estimated to be 13,236 MT compared with 1,674 MT collected (12.6 $\pm$ 4.9 % annual growth). Fuel wood collection alone is thus considerably above the 20 % of annual production typically deemed to be "sustainable" natural resource collection (Fa, *et al.* 2002). Including fencing poles and wooden tools, the collection estimates increase to 1,380 MT (29.4 % growth) in the high use zone and 1,704 MT (12.9 %) in the low use zone. For comparison, annual growth of live woody biomass on Waters Meeting NR was estimated to be 5,544 MT; but wood collection inside the reserve is prohibited.

3.4.2.8 Density of sexually mature *Scutia myrtina* stems

The density of sexually mature *Scutia myrtina* stems was estimated to be just 0.14 stems/m^2 in the low use zone and 0.07 stems/m^2 on Waters Meeting NR[3]. No *S. myrtina* stems were recorded in the high use zone. Alternatively, 72.1 % of all *S. myrtina* stems recorded in the low use zone were estimated to be fruiting or flowering, compared with 72.8 % on Waters Meeting NR. Differences in the size class distribution of stems between the two sites did not differ significantly (chi-square = 0.86 for 1.5 cm size classes; 0.62 for revised size classes).

3.4.3 Herbaceous layer

Significant ($p < 0.05$ or higher) differences were found between land use zones for all of the herbaceous layer variables: percentage aerial cover of grass, forbs, and litter; mean grass height; litter mass per square meter, and percent of grass biomass removed through grazing. These results (mean $\pm$ standard error) are described below.

3.4.3.1 Percentage aerial cover of grass, forbs, litter

There were significant differences in the percentage aerial cover (viewed at 2 meters above the ground) of grass, forbs, and litter among the different land use zones. Grass cover was significantly (H = 24.31; $p < 0.0001$) higher in the high use zone (35.9 $\pm$ 4.7%) than in either the low use zone or Waters Meeting NR, which did not differ significantly from one another (13.5 $\pm$ 4.3 % and 2.0 $\pm$ 0.9 %, respectively). Forb cover was also significantly (H = 12.33; $p < 0.01$) higher in the high use zone (4.5 $\pm$ 1.3 %) than in Waters Meeting NR (0.7 $\pm$ 0.5 %); however, the low use zone (2.9 $\pm$ 1 %) did not differ significantly from either of the other two zones.

The percentage aerial cover of litter (viewed from 2 m) was significantly (H = 7.47; $p < 0.05$) higher in the high use zone compared to Waters Meeting NR (9.2 $\pm$ 2.9 % and 4.9 $\pm$ 2.1 %, respectively), while the low use zone did not significantly differ from either site (6.9 $\pm$ 1.7 %). The percentage aerial cover of litter (viewed from ground level) was measured from the 10 sites in the low use zone and all 16 sites in Waters Meeting NR with > 80 % woody cover in the initial estimate (viewed from 2 m). Litter covered 43.6 $\pm$ 4.2 % in the low use zone and

[3]NB: Due to the very low density of stems in some transects, all stems were summed per site; therefore no standard error estimates were calculated.

52 ± 3.8 % on Waters Meeting NR; differences between the two sites were not significant (U = 47.0).

3.4.3.2 Mean grass height

The mean height of grass was significantly (H = 31.91; $p < 0.0001$) shorter in the high use zone (2.81 ± 0.46 cm) than in either the low use zone or Waters Meeting NR, which were also significantly different to one another (7.47 ± 0.87 cm and 15.81 ± 1.28 cm, respectively).

3.4.3.3 Litter mass

As expected, litter mass per square meter was significantly (H = 26.31; $p < 0.0001$) lower in the high use zone (223.7 ± 71.7 g/m^2) than in either the low use zone or Waters Meeting NR, neither of which were significantly different to each other (925.6 ± 207.2 g/m^2 and 1,306 ±140.5 g/m^2, respectively).

3.4.3.4 Percent of grass biomass grazed

The percent of grass biomass grazed was significantly (H = 33.55; $p < 0.0001$) higher in the high use zone (49.2 ± 4.7 %) than in either the low use zone or Waters Meeting NR, both of which were also significantly different to each other (9.2 ± 3.0 % and 0.1 ± 0.1 %, respectively).

3.4.4 Soil layer

Significant ($p < 0.05$ or higher) differences were found between land use zones for the following soil layer variables: percent sand, clay, and silt; infiltration; accessible moisture; cation exchange capacity (CEC); organic carbon; total carbon (loss on ignition); density of exotic and indigenous dung per square meter; percent aerial cover of bare ground and litter; mean height of grass; percent of grass removed by grazing; soil compaction, soil classification based on texture; and litter mass per square meter. Details of these results are described below (mean ± standard error).

3.4.4.1 Soil characteristics

The high use zone was significantly different from both the low use zone and Waters Meeting NR, which were not significantly different to one another, for the following soil characteristics: percent sand, percent clay, infiltration, accessible moisture, and organic carbon (Walkley Black method). As shown below in Figure 3.12, the high use zone (82.9 ± 1.9 %) had a significantly (H = 22; $p < 0.0001$) higher proportion of sand than both the low use zone and Waters Meeting NR, which were not significantly different to one another (69.7 ± 2.1 % and 67.3 ± 1.4 %, respectively). In contrast, mean percent clay was significantly (H = 24.51, $p < 0.0001$) lower in the high use zone (11.6 ± 1.22 %) than in either the low use zone or Waters Meeting NR, which were not significantly different to one another (16.6 ± 0.9 % and 14.2 ± 0.7 %, respectively).

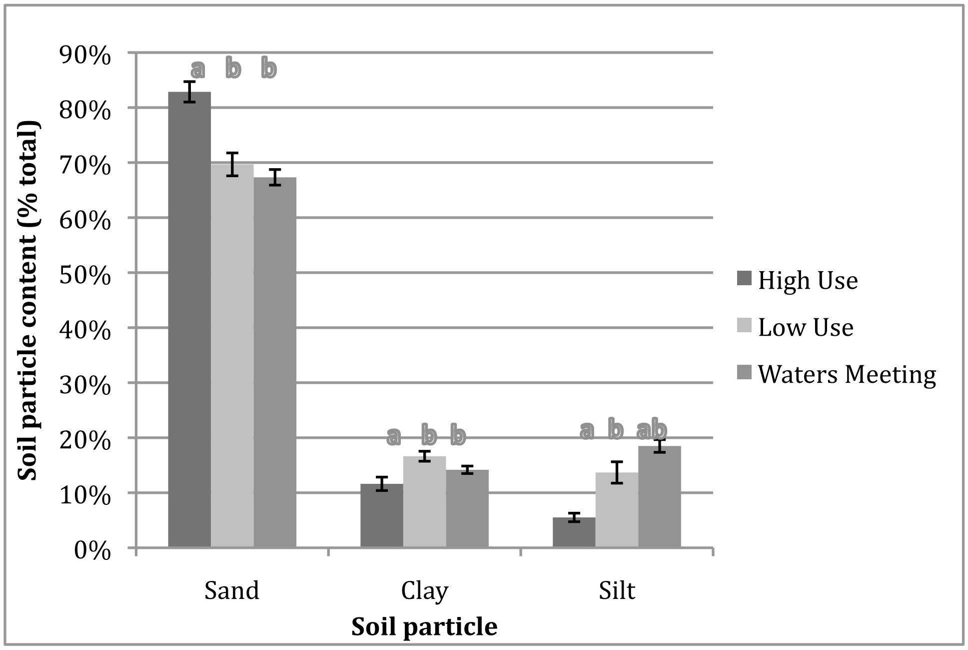

Figure 3.12 Differences[1] in soil composition by particle across land use zones

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard error.

The rate of infiltration as measured in the laboratory was significantly (H = 20.56; $p < 0.0001$) higher in the high use zone (10.12 ± 0.5 mm/hr) than in either the low use zone or Waters Meeting NR, which were not significantly different to one another (7.06 ± 0.4 mm/hr and 6.88 ± 0.3 mm/hr, respectively). Accessible moisture was significantly (H = 22.77; $p < 0.0001$) lower in the high use zone 78.12 ± 7.1 mm/m) than in either the low use zone or Waters Meeting NR, which were not significantly different to one another (118.06 ± 4.8 mm/m and 124.44 ± 2.5 mm/m, respectively). Finally, the organic carbon content as measured by the Walkley Black method was significantly (H = 27.16; $p < 0.0001$) lower in the high use zone (1.2 ± 0.2 %) than in either the low use zone or Waters Meeting NR, which were not significantly different to one another (4.7 ± 0.9 % and 5.7 ± 0.4 %, respectively). Differences in the percent stone composition among sites were not significant at $p < 0.05$;

stone content in the high use zone was zero, with 8.9 ± 2.5 % in the low use zone and 5.8 ± 1.8 % in Waters Meeting NR.

Significant differences were found between two or more sites for the following soil characteristics: percent silt, cation exchange capacity (CEC), and total carbon (loss on ignition). Silt composition was significantly (H = 22, $p < 0.05$) lower in the high use zone than in the low use zone (5.5 ± 0.8 % and 13.7 ± 1.9 %, respectively), while the percent silt in Waters Meeting NR soil samples (18.5 ± 1.2 %) was not significantly different from either of the two commonage land use zones. CEC was significantly (H = 19.94; $p < 0.0001$) lower in the high use zone (2.14 ± 0.53) than in Waters Meeting NR (8.38 ± 1.18), while the low use zone (4.63 ± 0.68) was not significantly different to either of the other two zones. Figure 3.13 shows that total carbon content (% organic matter) as measured by loss on ignition was significantly (H = 33.34; $p < 0.0001$) lower in the high use zone (3.6 ± 0.4 %) than in either the low use zone or Waters Meeting NR, which were also significantly different to one another (9.8 ± 1.6 % and 18.5 ± 1.3 %, respectively).

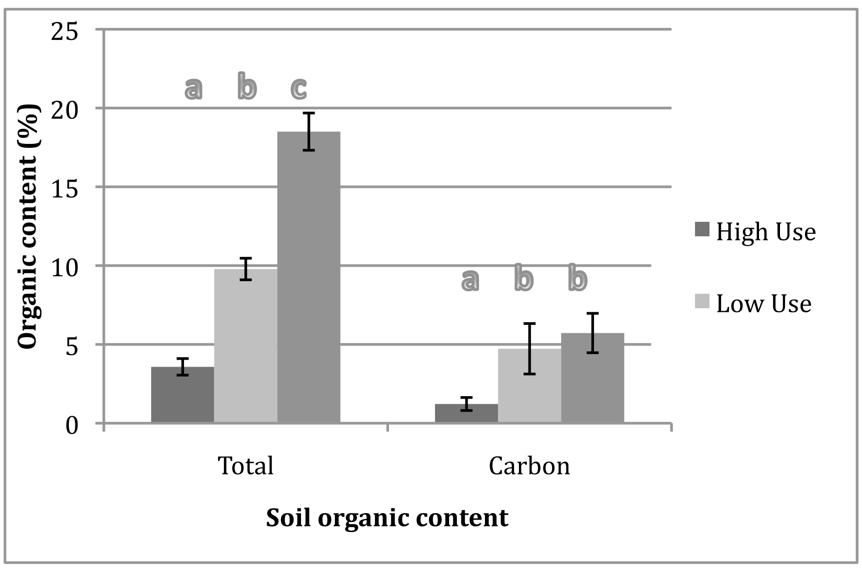

Figure 3.13 Differences[1] in soil organic content across land use zones

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard error.

The soil in the high use zone was classified as 2.18 ± 0.3, meaning that it is a loamy sand. The soil in both the low use zone and Waters Meeting can be classified as sandy loam, with mean figures of 3.13 ± 0.1 and 3.0 ± 0.0, respectively.

3.4.4.2 Percentage aerial cover of bare ground, rock, and erosion

The percentage aerial cover of bare ground was significantly (H = 26.23; $p < 0.0001$) higher in the high use zone (29.3 ± 3.7 %) than in either the low use zone or Waters Meeting NR, which were not significantly different to each other (10.8 ± 3.2 % and 1.7 ± 0.8 %, respectively). The percentage area covered by neither rock cover (1.2 ± 1.1 %; 0.5 ± 0.3 %; 0.5 ± 0.3 % in the high use, low use, and Waters Meeting NR, respectively) nor erosion (0.9 ± 0.9 %; 1.6 ± 0.9 %; none in the high use, low use, and Waters Meeting NR, respectively) differed significantly among sites (Figure 3.14).

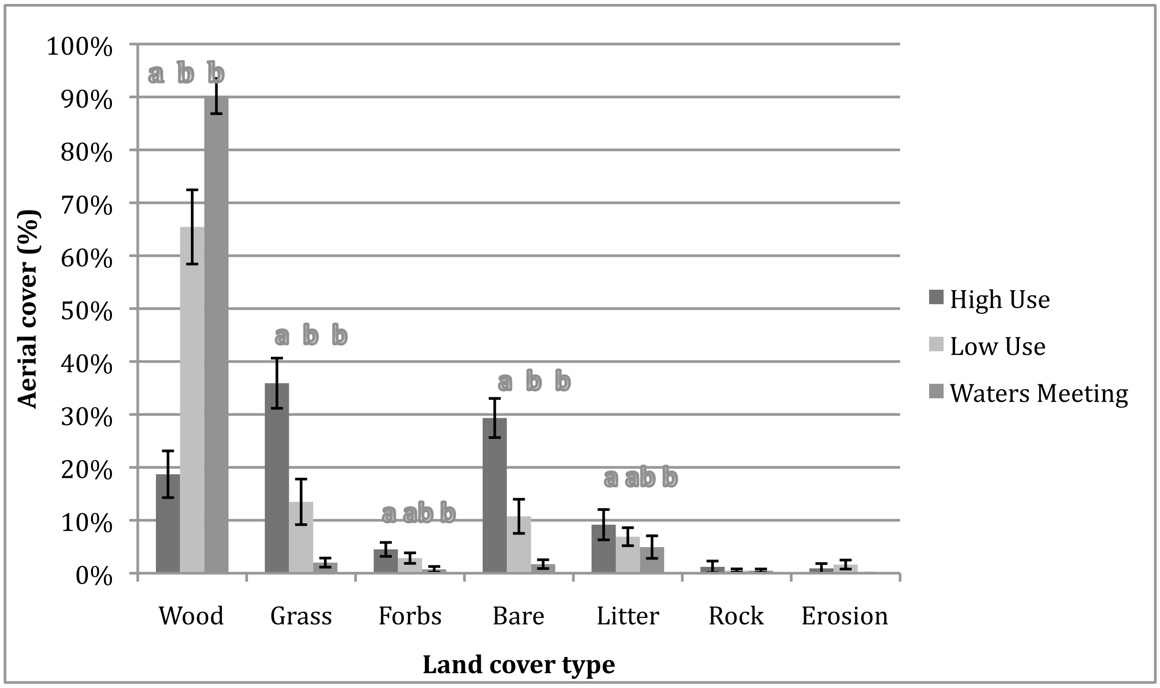

Figure 3.14 Differences[1] in proportional aerial cover by type across land use zones

[1]Unlike superscripts indicate significant differences (at $p < 0.05$ or higher) between two sites.

Bars indicate standard error.

3.4.4.3 Soil compaction

The level of soil compaction differed significantly (H =19.01; $p < 0.001$) among the three land use zones, with the high use zone having a higher mean level (3,657 ± 201 kPa) than either the low use zone or Waters Meeting NR, which did not differ significantly from one another (2,786 ± 539 kPa and 1,683 ± 224 kPa, respectively).

3.4.5 Transect-level characteristics

Differences among land use zones for both the density of exotic and indigenous dung and the slope angle were significant at $p < 0.001$ or higher between at least two zones. These results are reported below.

3.4.5.1 Density of exotic and indigenous dung

The density of dung pats per m^2 from exotic species (i.e. cattle) was significantly (H = 17.71; $p < 0.001$) higher in the high use zone than on Waters Meeting NR (0.1 $\pm$ 0.03 pats/m^2 and none, respectively), but neither differed significantly from the low use zone (0.03 $\pm$ 0.01 pats/m^2). In contrast, the density of dung pats per m^2 from indigenous species (mostly ungulates) was significantly (H = 26.1; $p < 0.0001$) lower in the high use zone (0.004 $\pm$ 0.003 pats/m^2) compared with either the low use zone or Waters Meeting NR, which were not significantly different to one another (0.031 $\pm$ 0.009 pats/m^2 and 0.1 $\pm$ 0.01 pats/m^2, respectively).

3.4.5.2 Slope angle

The angle of the steepest slope per transect was significantly (H = 18.46 and 17.25, respectively; $p < 0.001$ for both) lower in the high use zone (4.5 $\pm$ 1.1 %) than either the low use zone (16.7 $\pm$ 3.8 %) or Waters Meeting NR (26.5 $\pm$ 4.0 %), which did not differ significantly from one another.

3.5 <u>Correlations</u>

With respect to soil composition and characteristics, the percent total sand composition was significantly positively correlated with the infiltration rate ($r^2 = 0.9344$, $p < 0.0001$), which is reasonable since infiltration was measured in the lab and not the field. In contrast, sand composition was significantly negatively correlated with accessible moisture ($r^2 = 0.9652$, $p < 0.0001$).

As would be expected, percent litter cover at ground level (under the 2 m canopy) was significantly positively correlated with litter mass ($r^2 = 0.7756$, $p < 0.0001$); however, percent litter cover at the 2 m level, stem > 2 cm density, plant density were not correlated with litter mass ($r^2 = 0.0962$, $p > 0.5$; $r^2 = 0.2459$, $p > 0.08$; and $r^2 = 0.2536$, $p > .07$, respectively). Litter mass itself was significantly positively correlated with both % soil organic matter and cation exchange capacity ($r^2 = 0.6583$, $p < 0.0001$ and $r^2 = 0.5914$, $p < 0.0001$, respectively).

Both percent grass chewed and bare soil cover at 2 meter viewing were significantly positively correlated with soil compaction ($r^2 = 0.6164$, $p < 0.0001$ and $r^2 = 0.4884$, $p < 0.0001$). However, exotic dung density was not correlated with soil compaction ($r^2 = 0.2723$, $p > 0.05$). Percent woody cover was significantly negatively correlated with grassy cover ($r^2 = 0.8568$, $p < 0.0001$).

Interestingly, none of the following woody plant variables were significantly correlated with carbon content in live woody biomass: density of stems > 2 cm ($r^2 = 0.4429$, $p > 0.1$), density of stems < 2 cm ($r^2 = 0.2179$, $p > 0.1$); mean diameter of stems > 2 cm ($r^2 = 0.0264$, $p > 0.8$); maximum diameter > 2 cm ($r^2 = 0.0211$, $p > 0.8$); % woody cover ($r^2 = 0.0889$, $p > 0.5$). Live carbon content was not significantly correlated with either the organic matter content of the soil as measured by the Walkley Black method ($r^2 = 0.1165$, $p > 0.7$) or with total soil carbon content as measured by loss on ignition ($r^2 = 0.0101$, $p > 0.9$).

3.6 <u>Implications of land use intensity on the ecological health of the study site</u>

This section will discuss the implications of the results for each of the land use zones with a view toward relating land use patterns to ecological health outcomes. Although this research does not attempt to quantify the impact of some external factors, such as fire regime and climate, the small area of the study site (less than 5,000 ha) suggests that contextual factors, such as climate and geology, should vary more or less consistently across the entire site. Without precise historical data on land use, vegetation, climate, and fire regimes, however, it is not possible to make defensible conclusions about causal relationships between various land uses and ecological indicators. It is also not within the scope of this study to separate the effects of historical land uses, especially cultivation, from contemporary land uses, including grazing and fuel wood collection. For the purposes of this study, "contemporary" is defined as the period post-1994, during which time land use on the commonage has consisted mainly of cattle grazing, fuel wood collection, and other direct natural resource collection activities, such as for medicinal herbs, reeds for weaving, and wild foods as described in Chapter Two (Davenport 2008a; Fabricius, *et al.* 2006; Higginbottom, *et al.* 1995). Nonetheless, the results presented here suggest that the overall ecological health of the high use zone is significantly different from either the low use zone or Waters Meeting NR, which do not differ significantly on most indicators. Possible implications of the different land use intensities documented on the study site are discussed.

3.6.1 Implications of land use intensity on land cover change over time

Although the demonstration of causal relationships between historical land uses and ecological health on the commonage is also beyond the scope of this study, several trends emerge from the land cover analysis that may be linked to changes in land use over time. Firstly, transition from new to old agricultural fields is evident throughout the series of photographs, with the very slight increase in new fields between 1998 and 2004 likely attributable to a recent vegetable production project initiated near Nolukhanyo (Fabricius, *et al.* 2006). Historically, Fabricius, *et al.* (2006) also report that 12-acre (4.86 ha) farms were allocated to white residents of Bathurst for pineapple production in the 1950s, which may account for the relatively larger proportion of new agricultural fields in the 1956 photograph compared with more recent images. Between 1956 and 1965, however, the area covered by old fields increased on mean 128 % annually, suggesting that many fields fell into disuse during this decade. Nonetheless, it is worth noting that Allsopp (1999) observed that the transformation of disused (old) fields in the semi-arid Karoo back to natural vegetation can require several decades, indicating that cultivation in particular can have long-lasting impacts on ecosystem functioning.

Interestingly, following a very small decline (0.02 %) in the area covered by woody plants from 1956-65, this land cover type grew at a small but positive annual rate from 1965-1998. Between 1998 and 2004, however, the area covered by woody plants on the commonage fell by 2.2% annually. At the same time, the proportional area covered by grasslands declined at a modest but reliable annual rate for most of the years between 1956 and 1998 (from -0.5 % in 1956 – 1965 to -1.1 % in the years 1973 – 1990), with the rather insignificant exception of a negligible 0.01 % increase from 1965 – 1973. In contrast, the grassland area grew by 7.6 % annually between 1998 and 2004. Part of the increase in grassland coverage on the commonage could be attributed to the transformation of old fields into naturally vegetated areas. While the proportional annual decrease in old fields from 1998 – 2004 (2.4 %) is comparable to annual rates of change in other decades, it is far outweighed by the significant increase in grasslands during this period (7.7 %).

Furthermore, the change in grassland coverage from 1998 – 2004 is in sharp contrast to the modest decreases that characterized annual rates of change in both old field and grassland cover over the preceding decades. As such, it is more likely that the drop in woody plant cover from 1998 – 2004 is, in fact, correlated to the significant increase in grassland cover during that time. Without data on historical land use and the underlying factors influencing natural resource consumption on the commonage, it is difficult to test whether these trends were influenced by natural resource collection as compared to other causes, such as changes in climate or socio-economic conditions (Giannecchini, *et al.* 2007). Still, these trends should be noted carefully, particularly in light of the noted deterioration of commonage management since the mid-1990s (Fabricius, *et al.* 2006).

Although climate, geography, and other abiotic factors certainly influence land use change, and therefore impact land cover, research suggests that interactions between human society and natural systems play a significant and perhaps central role in landscape transformation, especially in rural Africa (Geist & Lambin 2002; Giannecchini, *et al.* 2007; Moreira, *et al.* 2001). Studies that analyze the drivers of land use change frequently cite numerous interacting causes, including demographic, economic, technical, institutional, and cultural factors (Geist & Lambin 2002). A recent study of these trends in three villages located in the Bushbuckridge area of South Africa noted that while increases in population density were "undoubtedly fundamental" in the process of land cover transformation, other factors were also important, including the deterioration of institutional control over natural resource use in the village commons, the decline of formal employment, decreasing available land, and an increase in goat ownership (Giannecchini, *et al.* 2007). Moreover, their work supports other studies that have demonstrated that while poverty often plays a causal role in land use, and therefore land cover, change, the relationships between poverty and natural resource use are by no means straightforward (Cavendish 2000; Twine, *et al.* 2003a, b). It is thus important to consider site-specific socio-economic, institutional, and cultural characteristics over time to accurately determine the main factors influencing land cover change.

Without precise historical data on natural resource use, climatic conditions, and socio-economic factors potentially impacting land use on the commonage, it is not possible to make defensible conclusions about the causal factors leading to these two trends. Nonetheless, it is worth noting that livestock on the commonage were reportedly actively managed from the 1950s to the mid-1990s through a system of rotational grazing with well-maintained fences and watering points (Fabricius, *et al.* 2006). Since then, however, resource collection on the commonage has been characterized by a lack of management that has resulted in the deterioration of this infrastructure and institutional controls. Moreover, the opening of the commonage to formerly excluded Nolukhanyo residents may have resulted in increased demand for primary natural resources, especially fuel wood. Future research on the impact of natural resource collection on the commonage should aim to quantify historical use patterns and their drivers, such as socio-economic and institutional changes, to compare and contrast with existing data on current trends in commonage use and users (Geist & Lambin 2002).

3.6.2 Implications of land use intensity on the woody plant layer

Pote, *et al.* (2006) classify woody plant species as a "keystone element" of an ecosystem through, for example, "the provision of fodder for wild and domestic animals, nesting sites and habitats for avifauna and reptiles, micro-habitats for the germination of other species, and nutrient pools within a harsh landscape" (Belsky, *et al.* 1989; Jeltsh, *et al.* 1996). Although woody plant species represent an important component of the semi-arid subtropical thicket ecosystem, Kerley, *et al.* (1995, 1999), hypothesized that these woody strata have become denser in modern times due to the historical elimination of large indigenous browsers. At the same time, early research on succulent thicket suggested that this ecosystem is characterized by low resilience (Fabricius 1997; Kerley, *et al.* 1995) implying that it is unlikely to recover from major disturbances. Since research indicates that woody biomass represents a key resource not only for ecosystem structure and functioning, but also for rural communities as a source of fuel wood and building material (Pandey 2002; Pote, *et al.* 2006; Shackleton, *et al.* 2004), it is important to carefully consider the impacts of human demand for woody plants on the subtropical thicket ecosystem.

Fuel wood collection has been shown to have significant impacts on woody biomass, including a reduction in woody density, biomass, and species richness (Motinyane 2002; Pote, *et al.* 2006; Shackleton 1993b). In addition, rural communities demonstrate preferences for both deadwood (Shackleton 1993b) and easily harvested medium-sized branches (Pote, *et al.* 2006), thereby modifying woody community structure towards seedlings and large trees and reducing the amount of litter in the ecosystem. Of note, these impacts are often found in increasing intensity with decreasing distance to a human settlement (Motinyane 2002; Pote, *et al.* 2006).

Nonetheless, it is also important to remember that environmental variability also plays a role in woody biomass supply (Reid & Ellis 1995; Sullivan 1999). For example, Sullivan (1999) showed that even in arid areas of Namibia (rainfall < 95 mm per annum), woody community structure was more influenced by environmental variability than by rural human communities. In fact, Shackleton (1993b) found an increase in the absolute density and overall proportion of seedlings compared with other size classes in Transvaal lowveld vegetation within a communal fuelwood harvesting area compared with a neighboring protected area, suggesting that the regenerative capacity of the harvested area was not significantly compromised by this activity. The resilience of this arid veld ecosystem is further evidenced by the high proportion (77.3 %) of coppicing among stems of at least 16 cm diameter (Shackleton 1993b).

Based on a snapshot of ecological health data, the results of this study clearly suggest that differences in land use intensity between the high use zone and Waters Meeting NR have had a significant impact on woody plants. The aerial cover of woody plants in the high use zone represents less than one quarter of that measured on Waters Meeting NR. Moreover, plant and stem density (of both stems < 2 cm and > 2 cm) were all significantly smaller in the high use zone compared with both Waters Meeting NR and the low use zone. This corroborates evidence from Pote, *et al.* (2006) and Motinyane (2002), who both found a strong gradient of increasing woody plant density with increasing distance away from the settlement. The low use zone actually had a slightly (but significantly) higher density of stems larger than 2 cm in diameter (1.1 $\pm$ 0.1 stems/m^2) compared with Waters Meeting NR (0.8 $\pm$ 0.1 stems/m^2), perhaps due to increased coppicing due to harvesting.

Nonetheless, estimated fuel wood collection rates of nearly 29 % and 13 % of annual live woody biomass growth in the high and low use zones, respectively, suggest that current extraction rates of woody biomass may be jeopardizing the sustainable replacement of this natural resource on the commonage. Moreover, the higher grazing and trampling pressure on the commonage as measured by grassy biomass removal (49.2 $\pm$ 4.7 % in the high use zone compared with 9.2 $\pm$ 3.0 % and 0.1 $\pm$ 0.1 % in the low use zone and Waters Meeting NR, respectively) and exotic (cattle) dung density (0.1 $\pm$ 0.03 pats/m^2 in the high use zone compared with 0.03 $\pm$ 0.01 pats/m^2 and zero in the low use zone and Waters Meeting NR, respectively) may negatively impact recruitment of new woody stems over time.

Mean basal diameter and maximum basal diameter in the high use zone were both significantly lower than in Waters Meeting NR; however, the low use zone did not differ significantly from either the high use zone or Waters Meeting NR for these measures. This may indicate that a higher proportion of larger stems have been removed from the high use zone, thus modifying the woody community structure as demonstrated by Pote, *et al.* (2006) and Shackleton (1993b). In contrast to Waters Meeting NR, where no damaged stems were recorded, the high use zone had a significantly higher density of damaged stems. On the other hand, more than twice as many dead stems > 2 cm were found on Waters Meeting NR. As would be expected, both of these measures indicate that both live and naturally dead woody plants inside the reserve are less likely to be subject to direct natural resource collection. Notably, however, no bark removal was noted in any land use zone, indicating that demand for bark as a traditional medicine may be quite low.

This evidence is corroborated by the significantly higher carbon stock found in dead woody biomass in Waters Meeting NR compared with both the low and high use zones on the commonage. On the other hand, the low use zone has a significantly higher total carbon stock than the high use zone, but the total carbon stock found on Waters Meeting NR was not significantly different to either of the two commonage zones. Furthermore, the carbon stock results described here for the commonage are noticeably higher than those recorded from other degraded thicket ecosystems in the Eastern Cape. The total carbon stock in above ground woody biomass of the intact subtropical thicket on Waters Meeting NR was estimated to be 64.6 $\pm$ 9.4 t C ha^{-1} compared with 56.6 $\pm$ 22.8 t C ha^{-1} and 129.9 $\pm$ 19.1 t C ha^{-1} in the high and low use zones of the commonage, respectively.

In contrast, Mills, *et al.* (2005) reported 40 $\pm$ 3 t C ha^{-1} in above ground biomass in intact thicket and 7 $\pm$ 1 t C ha^{-1} in thicket transformed by goat pastoralism near Kirkwood, located just outside Addo Elephant National Park and approximately 100 km to the northwest of the study site for this thesis. Similarly, Powell (2008) recorded 26.5 $\pm$ 3.9 t C ha^{-1} in the woody plant layer of intact subtropical thicket within the core region of the Bavianskloof NR (roughly 220 km to the southwest of Bathurst) compared with just 3.5 $\pm$ 0.8 t C ha^{-1} and 4.0 $\pm$ 0.7 t C ha^{-1} in old agricultural lands and degraded subtropical thicket, respectively. The large variance (range: 0.02 – 388 t C ha^{-1}) in live carbon content measured among sites in the high use zone may partly explain this unexpected result. Regardless, it is undeniably an indication of the need for further research to characterize the structure of geographically distinct subtropical thicket ecosystems.

Interestingly, no significant difference was found between the proportion of sexually mature *Scutia myrtina* stems in the low use zone and Waters Meeting NR. However, no *S. myrtina* stems were recorded in the high use zone, perhaps indicating that these species are selected preferentially for collection. Alternatively, the absence of *S. myrtina* in the patchwork of woody plant species interspersed between grassland in the high use zone may suggest that this species prefers the relatively closed 'intact' thicket ecosystem found in the more distant (relative to the township) reaches of the commonage and Waters Meeting NR to the transformed 'open' thicket in the high use zone nearest to Nolukhanyo. In light of the observed importance of this species to honey production in the Grahamstown area (Galpin 2007), this may preclude successful bee keeping initiatives in the high use zone. Moreover, this again seems to substantiate the findings of Pote, *et al.* (2006) and Motinyane (2002) and suggests that land use on the commonage is generally characterized by decreasing intensity with increasing distance from human settlement.

3.6.3 Implications of land use intensity on the herbaceous layer

Evidence of the impact of grazing on mean grass height and the proportion of grassy biomass removed was significantly higher in the high use zone than either the low use zone or Waters Meeting NR, which were also significantly different to one another. This corroborates the results of the community mapping exercise on the commonage (Fabricius, *et al.* 2006) and suggests that differences in ecological health indicators among zones on this relatively small (< 5,000 ha) land area may be reliable indicators of the differential impacts of different land use intensities. For example, aerial cover of grassy biomass was significantly higher in the high use zone compared with either the low use zone or Waters Meeting NR, which were not significantly different to each other. Moreover, aerial cover of grassy biomass was significantly negatively correlated with woody cover.

Since natural sub-tropical thicket ecosystems are dominated by succulent and woody plants (Lechmere-Oertel, *et al.* 2005a, b), the higher incidence of grass in the high use zone may be an indication that the intensity of land uses in this zone, including cultivation, cattle grazing, and intensive resource collection, may have a significant impact on ecosystem function. As increased land use intensity tends to remove vegetation, whether through cultivation, livestock grazing, or fuel wood collection, these activities would be expected to decrease the system's capacity to produce and accumulate organic matter in the soil by reducing the driving force for producing the litter that is a prerequisite for soil organic carbon (Allsopp, *et al.* 2007b). For instance, Mills, *et al.* (2005) found that carbon contained in litter in intact thicket accounted for roughly 5 % of total ecosystem carbon storage. Whereas, thicket transformed by goat grazing had roughly 63 % of the litter biomass of the intact system, and litter accounted for less than 1 % of ecosystem carbon in the transformed system (Mills, *et al.* 2005). In fact, litter mass per m^2 in the high use zone is an order of magnitude smaller than in Waters Meeting NR and less than one quarter of that collected in the low use zone. Moreover, litter mass on the commonage and Waters Meeting was found to be significantly positively correlated with both percent soil organic matter and CEC, indicating that differences in litter cover may, in turn, have considerable effects on the soil properties observed in each land use zone.

In addition to considering the possible role of livestock grazing on these indicators, it is also important to keep in mind the impact of other land uses, including cultivation and fuel wood collection. For example, research in old fields in the semi-arid Karoo ecosystem found "dramatically lower levels of organic matter and nitrogen in cultivated soils, even after several years of fallow" (Allsopp 1999). Moreover, fuel wood collection in the high use zone is more than twice the 20 % of annual production deemed sustainable; even in the low use zone woody biomass may be subject to unsustainable removal rates for fuel wood. Therefore, although it is clear that differences in land use intensity are correlated with ecological health, further research on the impacts of specific land uses on ecological health would be required to quantify the extent to which the various land uses on the commonage impact the herbaceous layer.

3.6.4 Implications of land use intensity on the soil layer

It is important to remember that the different zones on the study site are characterized by varied patterns of land use that have included cultivation, fuel wood collection, human settlement, and livestock grazing. While each of these land uses may have quantifiable impacts on soil attributes, numerous studies suggest that livestock grazing has considerable affects on soil attributes, which in turn impacts ecosystem functioning (e.g. Allsopp 1999; Derner & Schuman 2007; Han, *et al.* 2008; Liebig, *et al.* 2006). As such, this section will focus on the demonstrated impacts of grazing on soil properties, including decreased infiltration (e.g. Mwendera & Saleem 1997; Rietkerk, *et al.* 2000; Savadogo, *et al.* 2007; Zhao, *et al.* 2007), increased bulk density (e.g. Renzhong & Ripley 1997; Steffens, *et al.* 2008; Zhao, *et al.* 2007), and decreased nitrogen (e.g. Han, *et al.* 2008; Pineiro, *et al.* 2006; Snyman & Du Preez 2005; Steffens, *et al.* 2008). In addition, the decrease in above ground vegetative biomass and litter caused by grazing (e.g. Han, *et al.* 2008; Savadogo, *et al.* 2007) and fuel wood harvesting (e.g. Motinyane 2002; Pote, *et al.* 2006; Shackleton 1993b) has been shown to result in decreased soil organic matter (e.g. Renzhong & Ripley 1997) and decreased soil organic carbon (e.g. Han, *et al.* 2008; Snyman & Du Preez 2005; Steffens, *et al.* 2008; Zhao, *et al.* 2007).

However, Han, *et al.* (2008) note that there are conflicting reports on the impact of grazing on soil organic carbon. For example, Reeder & Schuman (2002) found higher soil carbon in grazed pastures than protected enclosures. Their research suggested that the lower carbon content in ungrazed areas could be attributed to the restriction of carbon in excess litter and an increase in the proportion of annual forbs and grasses that lack the fibrous rooting systems necessary to facilitate the formation of soil organic matter. On the other hand, research elsewhere has reported no effect of grazing on soil carbon (Dormaar, *et al.* 1977; Milchunas & Lauenroth 1993; Renzhong & Ripley 1997) or a decrease in soil carbon levels (Derner, *et al.* 1997; Frank, *et al.* 1995; Snyman & Du Preez 2005). As such, it is important to document site-specific differences in soil properties according to different land use regimes.

Significant differences in soil chemistry and physical properties across land use zones on the study site suggest that the different levels of land use intensity (due to livestock grazing as well as various other land uses) have differential impacts on soil compaction, accessible moisture, soil fertility as measured by cation exchange capacity (CEC), soil organic matter and soil organic carbon. Based on research in arid and semi-arid rangelands, it is hypothesized that animal trampling may play a key role in soil compaction, which in turn affects the soil's ability to absorb rainfall and nutrients (Renzhong & Ripley 1997; Steffens, *et al.* 2008; Zhao, *et al.* 2007). The significantly larger amount of force necessary to penetrate the soil crust in the high use zone compared with the low use zone and Waters Meeting NR, which were not statistically different to one another, indicates that soil compaction is significantly higher in the high use zone.

Moreover, the high use zone demonstrated a significantly higher proportion of bare soil compared to both the low use zone and Waters Meeting NR (one-third and an order of magnitude smaller than the high use zone, respectively) and also showed more evidence of grazing as measured by grassy cover, grass height, and percent of grassy biomass removed by grazing, than either the low use zone or Waters Meeting. The finding that both percent grass chewed and bare soil cover were significantly positively correlated with soil compaction would therefore seem to support the hypothesis that increased intensity of livestock grazing in the high use zone has resulted in increased soil compaction compared with the low use zone and Waters Meeting NR. This would also suggest that rainfall would be significantly less likely to be intercepted by vegetation and therefore more likely to leave the system as runoff rather than being absorbed into the soil (Zhao, *et al.* 2007). Increased runoff, in turn, could result in higher rates of erosion and concomitant nutrient loss in the high use zone compared to the other two zones (Allsopp 1999).

Furthermore, soil composition in the high use zone is characterized by a significantly higher proportion of sand particles (mean = 82.48 $\pm$ 1.88 % in high use zone; 71.30 $\pm$ 2.35 % in low use; and 66.83 $\pm$ 1.42 % in Waters Meeting NR) and significantly lower clay composition compared with both the low use zone and Waters Meeting NR, which were not significantly different to one another. Moreover, percent total sand composition was significantly positively correlated with the infiltration rate as measured in the lab. Of note, no visible evidence of major erosion, such as gullies, was detected in any of the zones. In light of the fact that (a) transects in the high use zone are characterized by significantly less steep slopes than either the low use zone or Waters Meeting NR and (b) percent clay composition was not found to be correlated with the angle of the steepest slope, it is therefore reasonable to assume that this trend may be at least partly attributed to differences in land use among the three zones.

Since sand composition was significantly negatively correlated with accessible moisture, it is therefore unsurprising that soil moisture accessibility is significantly lower in the high use zone compared with either the low use zone or Waters Meeting NR, which were not significantly different to one another. Likewise, both cation exchange capacity (CEC) and percent organic carbon were significantly higher on Waters Meeting NR than the high use zone, although the low use zone did not differ significantly from either other site for these indicators. Stroosnijder (1996) demonstrated that decreased organic matter content in the soil led to reduced macroporosity, which in turn would limit available sites for cations to bind to the soil. As discussed above, the reduction in soil fertility as measured by CEC may be attributed at least in part to the reduced organic matter detected in the high use and low use zones compared to Waters Meeting NR (all significantly different to one another), which may in turn be affected by the reduced litter mass collected in the high use zone compared with the low use zone and Waters Meeting NR.

3.6.5 Implications of land use intensity for transect-level characteristics

Densities of exotic (i.e. cattle) and indigenous dung were recorded in each land use zone to ground-truth the differing use intensities as recorded by the community mapping exercise for the commonage (Fabricius, *et al.* 2006). As would be expected, the density of exotic dung in the high use zone is significantly higher than on Waters Meeting, where none was found; exotic dung density in the low use zone is between these two sites but not significantly different to either of them. As mentioned, these two measurements provide an index of relative grazing pressure, which may lead to negative ecological impacts through, for example, increased trampling and browsing pressure, thereby increasing soil compaction and reducing plant recruitment. In contrast, a negligible amount of indigenous dung was found in the high use zone (0.004 ± 0.003_pats/m^2), compared with significantly higher densities in both the low use zone and Waters Meeting NR (0.031 ± 0.009 pats/m^2 and 0.068 ± 0.014 pats/m^2, respectively), which were not significantly different to one another. This observation would therefore appear to validate the results of the community land use mapping exercise undertaken by Fabricius, *et al.* (2006) in Nolukhanyo. The generally comparable ecosystem health indicators observed in both the low use zone and Waters Meeting NR may therefore be an indication that current land use management in the "emergency grazing" (low use) zone can sustain ecosystem functions comparable to those supported by the natural resources protected within Waters Meeting NR.

At the same time, the significantly steeper slopes recorded in Waters Meeting NR and the low use zone, which were not statistically different to one another, compared to the high use zone suggest that increased land use intensity in these steeper zones could potentially compromise their ecosystem health. While it is beyond the mandate of this work to define the threshold of sustainability, research in other communal rangelands has demonstrated that slopes tend to be degraded more easily than bottomlands (Allsopp, *et al.* 2007b; Fynn & O'Connor 2000). As such, it could be particularly important to manage the relatively flatter high use zone of the commonage such that sufficient fodder (and water) is available, even during the dry winter, to preclude grazing along the steeper slopes of the low use zone.

3.7 <u>Implications for future land use management on the study site</u>

Without more precise temporal data on changes in ecological health indicators within the study site, it is difficult to assess the sustainability of current land uses on the commonage. However, based on household use data for fuel wood collection (Davenport 2008a) and biomass data collected from the study site, it appears that this land use may already be compromising the sustainability of the woody strata on the commonage. As such, it will be important to quantify the extent of other land uses on the commonage, in particular livestock grazing and bush meat harvesting, especially in light of the significant returns to livestock rearing on the commonage (Davenport 2008a). Accurate data on the annual growth of various resources will also increase the accuracy of this initial assessment of the sustainability of fuel wood extraction and support better-informed management of the commonage resources.

In the meantime, assuming that the three land use zones began with similar ecosystems and that other drivers of ecological change, such as climate and fire regime, are roughly comparable across the roughly 4,500 ha study site, it is possible to make some general distinctions among the three land use zones. In particular, nearly all of the ecological health indicators measured suggest that the high use zone is currently of a lower ecological health than the low use zone and Waters Meeting NR. In contrast, most indicators measured by this study do not differ significantly between the low use zone and Waters Meeting NR, perhaps indicating that historical and current land use intensity in this part of the commonage has not significantly impacted its ecological health compared with the 'intact' ecosystem in the protected area.

Given that the mean grass height in the high use zone was less than one-third of that measured in the low use zone and the mean percent of grassy biomass grazed more than five times higher than the low use zone, and the density of cattle dung was 70 % higher in the high use zone, it is reasonable to assume that differences in the ecological health of these two areas of the commonage may be attributed at least in part to differences in the intensity of natural resource collection. In light of these observations, the significant differences in woody cover, woody plant density, density of dead stems, aerial cover of grass and bare ground, litter mass, soil composition and characteristics, and density of indigenous dung in the high use zone compared with the low use zone may indicate that increased land use intensity in the high use zone has had significant implications for ecological health even within the 2,989 ha of the Bathurst municipal commonage.

On the other hand, although the significantly shorter mean grass height and higher proportion of grassy biomass grazed in the low use zone compared with Waters Meeting NR suggest that this part of the commonage is, in fact, subjected to higher land use intensity than the protected nature reserve, differences in the ecological health indicators measured within these two zones were often *not* significant. In the woody plant layer, the aerial cover of woody plants, mean and maximum basal diameter of woody stems, density of dead stems, and proportion of sexually mature *Scutia myrtina* stems did not differ significantly between the two sites. Similarly, aerial cover of grass, forb, bare ground, and litter did not differ significantly between the low use zone and Waters Meeting NR; nor did litter mass. In the soil layer, only one indicator (total carbon content as measured by loss on ignition) differed significantly between the two sites; however, organic carbon content according to the Walkley Black method did *not* differ significantly. In fact, the densities of neither exotic nor indigenous dung were significantly different between the two sites. In addition to corroborating the outcome of the community land use mapping exercise (Fabricius, *et al.* 2006) these results may indicate that land use intensity in the low use zone of the commonage has not significantly impacted the ecological health of this zone compared with Waters Meeting NR.

The implications of these results on future management of land uses on the study site are potentially substantial. The indicators measured suggest that the impact of historical and current land use intensities have potentially jeopardized the ecological health of the high use zone, which is nearest to Nolukhanyo township and has been subjected to perhaps the most significant disturbances over its history, including intensive cultivation, livestock grazing, and fuel wood collection. In contrast, based on the status indicators measured, it appears that the ecological health of the low use zone is roughly comparable to that of Waters Meeting NR, even though resource collection on the commonage is typified by unlimited access. Assuming demand for natural resources in the low use zone remains roughly constant due to its distance from the township and relatively more rugged terrain, governance agencies should therefore concentrate on implementing land use management and resource monitoring in the high use zone in the short term.

At the same time, future increases in the intensity of land use in the low use zone could be particularly damaging to its ecological health in light of the steeper slopes that characterize sites in the low use zone compared to those in the high use zone. Moreover, while it is beyond the scope of this study to estimate the sustainability of all major land uses, the results strongly indicate that fuel wood collection on the commonage may currently be unsustainable. As such, it will be crucial to quantify the annual production rates of other natural resources in order to assess the sustainability of collection activities on the commonage.

Finally, it is expected that the varied ecological health status across the study site is both a reflection of current resource extraction and an indication of differences in the magnitude of direct and indirect ecosystem services flowing from each land use zone. Building from this ecological health assessment, the next two chapters aim to value these differences with a view toward understanding the trade-offs between the two land use management strategies on the site.

4 Chapter Four

Ecosystem Service Value Transfer

4.1 Introduction

Following the evaluation of ecosystem health described above, an ecosystem service value transfer exercise was conducted to estimate the value of various services provided by the study site based on research conducted in as similar social and ecological settings to the extent possible. In situations where "primary data collection is not feasible due to budget and time constraints, or when expected payoffs to original research are small," ecosystem service values (ESVs) calculated for other policy contexts can be adjusted to accommodate the context at hand (EPA 2000). This process, known as "value transfer", involves identifying existing literature that values ecosystem services provided by ecological resources similar to those present at the policy site and transferring those values from the original site to the policy site (Desvousges, *et al.* 1998; Loomis 1992; Troy & Wilson 2006). Although primary data collection will usually present more accurate estimations, value transfer allows for a rapid estimation of economic values provided by a specific ecosystem service at lower expense and has become an important policy tool (Iovanna & Griffiths 2006; Troy & Wilson 2006). It also helps to identify ESVs that will potentially contribute most to the total value and thereby guide where primary research and empirical data collection should be directed.

Ecosystem services for valuation were chosen to encompass a variety of beneficiaries, types of services, and scales. In particular, an effort was made to address each of the two main constituencies in Bathurst: the poor, non-rate paying citizens of Nolukhanyo and the relatively more affluent rate payers of Bathurst. For this reason, both *direct* and *indirect* ecosystem service values (ESV's) were estimated based on value transfer from existing literature for a variety of services provided directly to the Bathurst community, as well as indirect services that benefit users throughout the Kowie River catchment area.

Following trends in international ESV literature, provisioning services, such as the production of fuel wood, honey, and fodder for cattle, were valued at the municipal[1] level because the primary users of these services are Nolukhanyo residents (Hassan 2003; Turpie 2003). In contrast, many indirect services, such as regulating services including erosion control and

[1] As mentioned, the Ndlambe LM includes the towns of Bathurst and Port Alfred; for the purposes of determining the value of provisioning services, only Bathurst residents will be considered "users."

climate regulation (via carbon sequestration), as well as cultural services like aesthetic and spiritual values, are available to a wider community of resource users (Hein, *et al.* 2006). It has been argued that erosion control on the commonage and Waters Meeting NR plays a direct, if not primary, role in the provision of clean water to Port Alfred residents (Fabricius, *et al.* 2006). Similarly, avoided deforestation within the study area could prevent the release of carbon into the atmosphere that potentially impacts a much wider scale of constituents.

Typically, more affluent residents of Bathurst would be most likely to take advantage of the scenic vistas provided by the study area purely for recreational or aesthetic value (Fabricius, *et al.* 2006). However, as is evidenced by a phone survey conducted as part of the field valuation process, residents of both Grahamstown and Port Alfred also access these benefits from the Bathurst commonage and Waters Meeting NR. Meanwhile, Nolukhanyo residents find spiritual value in the sacred pool created by the Lushington River in the commonage, but so do believers from several neighboring towns (Bernard, pers. comm. 2008; Fabricius, *et al.* 2006; Mali, pers. comm. 2008; Mbumba, pers. comm. 2008). Therefore, these indirect services will be valued at the scale of the Kowie River catchment to recognize the services both up- and downstream users gain from good land management in the study area.

Although it is important to recognize that some ecosystem services, such as climate regulation and nature conservation, have a national or even international constituency (Brown 1997, 1998; Hein, *et al.* 2006;), this thesis will focus on quantifying services provided to people living in the Kowie River catchment area. The reasons for this choice of scale are two-fold. Firstly, most valuations of regulating and supporting services have focused on broad benefits at the regional or national scale (Hassan 2003; Turpie 2003; Turpie, *et al.* 2008), while provisioning services have typically been valued at the household level (e.g. Davenport 2008a; Shackleton, *et al.* 2000, 2002). As a result, there is a paucity of information about ecosystem service values at an intermediate scale between individual households and national or regional landscapes. Secondly, there is a considerable degree of uncertainty surrounding the valuation of cultural and other indirect ecosystem services (Edwards & Abivardi 1997). To reduce the size of the error inherent in these estimations, the valuation of these services was limited to a catchment-level scale.

4.2 Selection of services for benefit transfer

The following direct services were chosen based on their relative contributions to the livelihoods of Nolukhanyo residents as determined by recent household surveys (Davenport

2008a): bush meat, cattle, fuel wood, honey, and medicinal plants. Only one indirect service was chosen based on its expected value to Bathurst residents and on the availability of international literature for value transfer to the study area: willingness to pay (WTP) to protect endangered species (Eastern Cape Rocky and leopard).

It is anticipated that residents of the Kowie catchment derive value from several other indirect goods and services, including avoided sedimentation, avoided deforestation, tourism, and the sacred pool on the commonage. In fact, in some contexts, non-use values, such as those derived from spiritual resources or aesthetic beauty, may constitute a considerable component of the total economic value of a given site (Georgiou, *et al.* 1997). However, due to the limited availability of data necessary for comparing the study site with the original context from which published values were calculated, it was decided not to include these ESVs in the benefit transfer exercise.

Moreover, there are important methodological, cultural, and ethical constraints that affect the value transfer of sacred resources across different sites (Adamowicz, *et al.* 1998; Bernard, pers. comm. 2008; Fox 2002). The values of sacred resources are endogenous to a "specific social environment" and often vary depending on the taboos or roles governing their use by specific individuals according to age, gender, or social status. This seriously complicates the estimation of aggregate values for a single resource or ecosystem based on community members' values, let alone the transfer of aggregate values across sites with different socio-environmental contexts (Adamowicz, *et al.* 1998).

It has also been argued that sacred resources are "sacrosanct and non-negotiable" such that one good, in this case a sacred pool, cannot be directly substituted for another, such as a nearby pool also associated with spiritual value, because of the nature of belief systems governing these resources (Adamowicz, *et al.* 1998; Bernard, pers. comm. 2008). Even if this were not the case, the small area of the sacred pool on the commonage compared with the site's overall area and its limited use as indicated by local *amasangoma* (traditional healers) would likely result in a relatively small per hectare value averaged across the site (Mbatha, pers. comm. 2008; Mbumba, pers. comm. 2008). In light of these considerable methodological and cultural constraints and the modest per hectare value expected, it was decided not to attempt a valuation of the sacred pool on the commonage.

4.3 Benefit transfer of ecosystem service values

The following sections detail the methods, results, and discussion for each ecosystem service valued through benefit transfer, respectively. Where possible, the value of the estimated standing stock on the study site was differentiated from the value of annual production of benefits; representing the capital stock and annual returns respectively. In general, for each service the current value of the estimated standing stock on the commonage and Waters Meeting NR was calculated based on approximate carrying capacities transferred from the available literature on the specific resource being valued or a comparable substitute (e.g. same genus). Annual production estimates were likewise based on published data for each resource or similar taxa.

Where appropriate, the carrying capacity of the commonage (or Waters Meeting NR) was discounted (or augmented, in the case of Waters Meeting NR) by the demonstrated impact of different land use changes according to published comparisons of intact and transformed semi-arid ecosystems. Where direct comparisons for a given resource across intact and transformed sites were not available, carrying capacity was adjusted based on the assumed impacts of documented changes in aboveground biomass on the resource in question (Jaramillo, *et al.* 2003; Mills, *et al.* 2005; Penzhorn, *et al.* 1974). For example, the standing stock of various indigenous mammal species was estimated based on published data for natural (intact) habitats. Therefore, the standing stock of each mammal studied was discounted in the commonage valuation to reflect the impact of relatively more open vegetation on the availability of food and shelter in contrast to the protected habitats located within the densely vegetated Waters Meeting NR. In contrast, the standing stock of natural resources collected on the commonage that were calculated based on current use estimates as reported by household user surveys (Davenport 2008a), such as fuel wood and medicinal plants, were augmented to reflect the increased availability of aboveground biomass likely supported by the protected area within Waters Meeting NR.

It is important to note that differences in land uses across sites could significantly impact the productivity, as well as the standing stock (carrying capacity), of ecosystem goods and services. Unfortunately, sufficient data on the impacts of different land uses on the annual production of ecosystem services studied were unavailable. As such, it was not possible to estimate differences in annual production as a result of land use impacts on the commonage as compared to Waters Meeting NR. This necessarily limits the usefulness of the annual production data to the benefit transfer exercise, as the difference in annual production between the two sites is therefore equal to the proportional difference in area and the discounted standing stock.

Despite this constraint, it was decided to include annual production in the benefit transfer exercise in order to provide some measure, albeit crude, of the sustainability of current or future use of ecosystem services generated by the study site, such as fuel wood provision. Since field data collection for this thesis was concentrated within a four-month period and primarily focused on indirect services not covered by previous studies on the site (e.g. Davenport 2008a), the sustainability of current direct resource collection patterns will be predicated on annual production and consumption rates as estimated by the benefit transfer exercise.

Furthermore, the sustainability of any PES enterprise will depend more on the rate of annual production of the service than on the standing stock of 'natural capital' available. Of course, the cost of exploiting a given ecosystem good or service will also affect the profitability of any PES project. Nonetheless, PES depends fundamentally on the availability of a sustainable (i.e. renewable) supply of the ecosystem good or service in question. To enable some prediction of the sustainability of current and future benefit capture, it was decided to estimate the value of both the annual production and the standing stock of ecosystem goods and services, despite the limitations discussed here.

Finally, prevailing per unit prices from the available literature were used to translate estimates of standing stock and annual production into per hectare monetary values. The primary aim of this thesis is to demonstrate the total value of ecosystem goods and services derived from the study site, rather than to evaluate potential business opportunities for local residents. Thus, all values will be reported gross, rather than net.

4.3.1 Bush meat

Recent analyses of community use of the Bathurst commonage indicate that hunting is among the livelihood activities practiced there (Davenport 2008a; Fabricius, *et al.* 2006). Globally, bush meat hunting has been shown to provide a crucial source of protein for impoverished communities, albeit not always amongst the poorest households (de Merode, *et al.* 2004; Scoones, *et al.* 1992). Earlier research indicated that the poorest households were disproportionately reliant on wild foods to supplement their diets (Dei 1989; Scoones, *et al.* 1992). In fact, de Merode, *et al.* (2004) note that the potential importance of wild foods, including bush meat, to vulnerable households is recognized by many humanitarian agencies, which often use reliance on wild foods as a proxy for impending famine (de Waal 1988; Young 1992). However, more recent studies have challenged this accepted wisdom (de Merode, *et al.* 2004; East, *et al.* 2005; Godoy, *et al.* 1995; Wilkie *et al.* 2005).

For example, research in the Democratic Republic of Congo (DRC) showed that the poorest households were unable to take advantage of bush meat either as a source of food or additional income due primarily to their lack of funds to purchase either the equipment necessary for hunting or traded bush meat (de Merode, *et al.* 2004). Similarly, Godoy, *et al.* (1995) found that middle-income households in Nicaragua were most dependent on wild foods, while work in West and Central Africa (East, *et al.* 2005; Wilkie *et al.* 2005) has demonstrated that increasing wealth leads to higher demand for fresh bush meat compared with, for example, less expensive frozen meats. Still, several other studies (e.g. Chenevix-Trench 1997; Wickramasignhe, *et al.* 1996) have reported no income effects on the extent to which households depend on wild foods, an indication of the "complex array of social and economic factors that determine differential access to wild resources both within and between communities" (de Merode, *et al.* 2004).

The actual contribution of bush meat to rural household diets may be relatively modest. In fact, de Merode, *et al.* (2004) reported that bush meat consumption of just 0.04 kg per day represented only 3.1 % of the total value of food consumed by all households, compared with 6.2 % for fish or 9.6 % for wild plants. This is somewhat low compared with previous studies of agricultural households in the Congo basin, which found a rate of 0.13 kg per day (Wilkie & Carpenter 1999). However, there are likely a number of context-specific factors that contribute to bush meat consumption, including household wealth, the availability of alternative meats, and access to political and social networks (de Merode, *et al.* 2004; East, *et al.* 2005). As such, it is difficult to make generalizations about the contribution of bush meat to rural households outside the original study sites.

Regardless of the extent to which households depend on bush meat and other wild foods throughout months of plenty, it is clear that their dependence increases during the "lean" or "hungry" season before the harvest, when agricultural products are scarce and vulnerability to hunger is highest (Chambers 1997; Dei 1989; de Merode, *et al.* 2004). De Merode, *et al.* (2004) found that consumption of agricultural products fell by nearly half during the lean season, but consumption of bush meat increased by 75 %. This confirms earlier work by Toulmin (1986) and Dei (1989), which indicated that rural households in Central Mali and

Ghana, respectively, increased their consumption of wild foods during the hungry season, sometimes by more than double, to maintain sufficient nutrition until the next harvest.

Nonetheless, wild foods, especially bush meat, appear to be even more important as a crucial source of income for isolated households wealthy enough to afford the necessary equipment (de Merode, *et al.* 2004; Eves & Ruggiero 2000; Noss 2000). In fact, de Merode, *et al.* (2004) found that over 90 % of bush meat production is sold at market in their study site, an agricultural village in northeastern DRC. In contrast, less than 25 % of either wild plants or agricultural produce are sold at market. As a proportion of total sales, bush meat represented 25 % and fish accounted for 39 %, compared with just 2 % for wild plants (de Merode, *et al.* 2004). Especially in remote rural areas with few alternative opportunities for income generation, bush meat thus provides a crucial source of cash to purchase important commodities and assets, such as medical supplies and fishing nets.

The contribution of bush meat to rural incomes is further accentuated during the "hungry season" before the harvest when many households face serious resource constraints (Chambers 1997; Dei 1989; de Merode *et al.* 2004). Dei (1989) found that the economic contribution of wild foods to rural households in Ghana more than doubled during this time. De Merode, *et al.* (2004) also showed that bush meat incomes increased substantially (155 %), though the difference was not statistically significant. Thus, bush meat provides a crucial source of both food and income to remote households with few alternative livelihood options, especially during periods of heightened vulnerability.

On the other hand, there is growing concern from both the conservation and development fields that bush meat extraction for subsistence use and trade may be outstripping supply in certain areas, particularly in West and Central Africa (Fa, *et al.* 2000, 2002, 2004, 2006). Bush meat hunting has been deemed the most widespread form of resource extraction from tropical forests, since hunting penetrates even the largest and least accessible habitats (Peres & Terborgh 1995). Concern regarding the impacts of this activity on ecosystems has led to considerable interest from both the conservation and development communities (Cavendish 2000; Fa, *et al.* 2000; 2003; Milner-Gulland & Bennett 2003; Pattanayak & Sills 2001).

Although various conservation organizations have blamed subsistence and commercial hunting for resource degradation, regional evidence suggests that this is not always the case (Fa, *et al.* 2002, 2006). For example, while current extraction rates for 60 % of the 53 mammal species surveyed by Fa, *et al.* (2002) in the Congo basin appear to be significantly above the 20 % of annual production deemed to be "sustainable," hunting in the Amazon was found to be sustainable for all 24 mammal species considered. Given the importance of site-specific socio-economic and ecological factors to the determination of sustainability, it will be imperative to document these trends in the study site. This section will focus on the potential production of bush meat on the commonage and Waters Meeting NR to estimate the potential value of bush meat as a source of income to the Nolukhanyo community.

4.3.1.1 Methods

Because detailed species surveys have not been completed in the study area for at least the past ten years, the quantity of bush meat available was estimated using data on average stocking and population growth rates for the animals likely to be found in the study area and considered edible by Nolukhanyo residents. Recent studies on bush meat trade show that mammals account for the vast majority of all carcasses and total biomass (de Merode, *et al.* 2004; Fa, *et al.* 2006). Mammals represented 95 % of the carcasses documented in local bush meat markets and 96.2 % of the total biomass documented in Nigeria and Cameroon (Fa, *et al.* 2006) and over 90 % of total bush meat biomass documented in a village of northeastern Democratic Republic of Congo (de Merode, *et al.* 2004). Since reptiles, birds, and amphibians therefore represented such a small proportion of total bush meat trade, it was decided to exclude these taxonomic groups from the calculations while recognizing that they are consumed by members of poor households (e.g. McGarry & Shackleton 2009). Considering both the distribution of available habitats in the study site and local preferences for bush meat, the following species were selected for valuation (see Table 4.1 for Latin names): aardvark (antbear), duiker (blue and grey/common), bushbuck, bushpig, Cape grysbok, Chacma baboon, greater kudu, grey rhebuck, mountain reedbuck, porcupine, hyrax (rock and tree), scrub hare, Smith's red rock rabbit, springhare, and vervet monkey (Bothma 1989, 2002; Davenport, pers. comm. 2008; Earle, pers. comm. 2008; Smithers 1983).

To estimate the current number of animals on the study site, a variety of resources were consulted to find average density or stocking rates of each mammal. Where density or recommended stocking rates were not available, the inverse of territory size was used to approximate potential stocking rates. With the exception of some ungulate species, information on the extent to which individual territories overlap was mostly unavailable. As such, it was assumed that all intraspecific territories are exclusive, i.e. do not overlap. The estimated number of animals per hectare was then multiplied by the average mass of an individual of each species (both sexes) to predict average mass (kg) per hectare (Bothma

1989, 2002; Smithers 1983). This number was first multiplied by the area (hectares) of the commonage and Waters Meeting NR, respectively, to predict the total mass of mammals on each site, and then multiplied by the average dressing weight of each animal to predict the total available bush meat in the study area.

Table 4.1 Edible mammal species in the study area

Common name	Latin name
Aardvark (Antbear)	*Orycteropus afer*
Blue duiker	*Philantomba monticola*
Bushbuck	*Tragelaphus scriptus*
Bushpig	*Potamochoerus porcus*
Cape clawless otter	*Aonyx capensis*
Cape grysbok	*Raphicerus melanotis*
Chacma baboon	*Papio cynocephalus ursinus*
Greater kudu	*Tragelaphus strepsiceros*
Grey (Common) duiker	*Sylvicapra grimmia*
Grey rhebuck	*Pelea capreolus*
Mountain reedbuck	*Redunca fulvorufula*
Porcupine	*Hystrix africaeaustralis*
Rock hyrax (Dassie)	*Procavia capensis*
Scrub hare	*Lepus saxatilis*
Smith's red rock rabbit	*Pronolagus saundersiae*
Springhare	*Pedetes capensis*
Tree hyrax	*Dendrohyrax arboreus*
Vervet monkey	*Cercophecus aethiops*

Sources: Davenport, pers. comm. 2008; Earle, pers. comm. 2008; Smithers 1983

Next, data on annual population growth were collected from numerous sources to estimate the number of additional animals potentially added to the study site each year net of all births and deaths, in-migration, and emigration out of the area. For species without readily available data, including many of the smaller mammal species, such as rabbits, and even some large mammals, including the aardvark, annual population growth rates for species of similar mass and common taxonomy, or at least similar mass, were transferred to the species in question. The new individuals were then weighed as adults to account for both births and in-migration to the study site from neighboring farms. Using the same dressing weight percentages identified for the previous estimate, the average dressed weight (kg) of each additional animal was multiplied by R8/kg (Shackleton, *et al.* 2002) to estimate the value of annual production.

The resulting estimations of average standing stock and annual production were assumed to be the values for intact thicket and were discounted according to changes in vegetation cover between sites. Three studies, two on subtropical thicket in South Africa (Mills, *et al.* 2005; Penzhorn, *et al.* 1974) and one on a dry forest in Mexico (Jaramillo, *et al.* 2003), directly compare aboveground vegetative biomass production across 'intact' and 'degraded' sites in the same ecosystem. On average, the degraded sites were found to produce just 29.3 % of the aboveground vegetative biomass produced by intact sites.

To ensure a conservative estimate of the impact of vegetation loss on animal production capacity, it was assumed that animal production decreases less dramatically than vegetation cover between the unharvested ('intact') Waters Meeting NR and harvested ('degraded') commonage. Thus animal production was assumed to decline by 80 % of the vegetation cover loss, or 56.6 %, on the commonage compared to the protected Waters Meeting NR. Moreover, even in areas of the commonage characterized by 'intact' vegetation, it was assumed that edge effects, fragmentation, and general disturbances caused by human resource use would result in a further 10 % decrease in the value of bush meat produced on the commonage. It is worth noting that the relationship between land use intensity and species density is not always linear (e.g. Ogutu, *et al.* 2009; Söderström, *et al.* 2003). However, data on species-specific relationships in the study area were not available, and these data would not, in any case, make a significant difference on the overall conclusions across all species. As such, it was decided not to address this as part of the valuation exercise. These final mass estimates (kg) were then multiplied by the average price (R8) per kilogram of bush meat as identified by Shackleton, *et al.* (2002) to estimate the value (in 2001 Rand) of the bush meat currently on the study site and average annual production. All figures were adjusted to constant 2008 Rand using the South African Consumer Price Index (StatsSA 2009) and converted to constant 2008 US dollars using average annual exchange rates (SARS 2008).

4.3.1.2 Results

The total standing stock value of all bush meat was estimated to be R 4,252,744 $\pm$ 540,638 (US$ 513,852 $\pm$ 65,478) on Waters Meeting NR and R 4,468,367 $\pm$ 569,389 (US$ 541,178 $\pm$ 68,960) on the commonage (Table 4.2). This gives a per hectare standing stock value of R 2,936 $\pm$ 374 (US$ 356 $\pm$ 45.31) for Waters Meeting NR, while that on the commonage was valued at just R 1,495 $\pm$ 190 (US$ 181 $\pm$ 23.07) after adjusting for the assumed impact of natural resource harvesting on habitat availability (Table 4.3). The total value of annual production of bush meat on Waters Meeting NR was estimated to be R 858,086 $\pm$ 110,320 (US$ 103,925 $\pm$ 13,361) while annual production on the commonage was only slightly higher at R 900,804 $\pm$ 116,234 (US$ 109,099 $\pm$ 14,077). The per hectare value of annual production was roughly R 594 $\pm$ 76.35 (US$ 71.92 $\pm$ 9.25) on Waters Meeting NR and R 301 $\pm$ 38.89 (US$ 36.50 $\pm$ 4.71) on the commonage.

Table 4.2 Total value of bush meat on Waters Meeting NR and the commonage (Constant 2008 Rand, US$)

Animal	Standing stock value				Annual production value			
	Waters Meeting NR		Municipal Commonage		Waters Meeting NR		Municipal Commonage	
	R	US$	R	US$	R	US$	R	US$
Bushpig	2,072,920	251,058	2,183,155	264,409	466,407	56,488	491,210	59,492
Bushbuck	1,224,206	148,267	1,289,307	156,152	140,784	17,051	148,270	17,957
Greater kudu	352,506	42,693	371,251	44,963	52,876	6,404	55,688	6,745
Cape grysbok	131,799	15,963	138,807	16,811	15,157	1,836	15,963	1,933
Blue duiker	126,793	15,356	133,535	16,173	19,019	2,303	17,117	2,073
Scrub hare	69,460	8,412	73,153	8,860	65,987	7,992	69,496	8,417
Mountain reedbuck	57,713	6,990	60,782	7,361	16,737	2,027	17,627	2,135
Grey duiker	50,535	6,120	53,222	6,446	10,107	1,224	10,644	1,289
Rock hyrax	44,460	5,385	46,824	5,671	111	13.46	117	14.18
Smith's RR rabbit	40,101	4,857	42,234	5,115	38,096	4,614	40,122	4,859
Springhare	31,071	3,763	32,724	3,963	29,518	3,575	31,088	3,765
Porcupine	10,603	1284	11,167	1,352	3.61	0.44	3.80	0.5
Chacma baboon	9,617	1,165	10,128	1,227	385	46.59	405	49.07
Tree hyrax	7,940	962	8,362	1,013	20	2.40	21	2.53
Grey rhebuck	6,995	847	7,367	892	1,749	212	1,842	223
Cape clawless otter	3,233	392	3,405	412	808	97.90	851	103
Aardvark	2,733	331	2,878	349	314	38.06	331	40.09
Vervet monkey	59.65	7.22	62.82	7.61	8.59	1.04	9.05	1.10
Total	**4,242,744**	**513,852**	**4,468,367**	**541,178**	**858,086**	**103,925**	**900,804**	**109,099**
Standard Deviation	**540,638**	**65,478**	**569,389**	**68,960**	**110,320**	**13,361**	**116,234**	**14,077**

Sources: Prices adjusted to 2008 from Shackleton, *et al.* (2002); species data from various sources: see references

Table 4.3 Per hectare value of bush meat on Waters Meeting NR and the commonage (Constant 2008 Rand, US$)

Animal	Standing stock value				Annual production value			
	Waters Meeting NR		Municipal Commonage		Waters Meeting NR		Municipal Commonage	
	R	US$	R	US$	R	US$	R	US$
Bushpig	1,435	174	730	88.46	323	39.09	164.34	19.90
Bushbuck	847	103	431	52.24	97.43	11.80	49.61	6.01
Greater kudu	244	29.55	124	15.04	36.59	4.43	18.63	2.26
Cape grysbok	91.21	11.05	46.44	5.62	10.49	1.27	5.34	0.65
Blue duiker	87.75	10.63	44.68	5.41	13.16	1.59	5.73	0.69
Scrub hare	48.07	5.82	24.47	2.96	45.67	5.53	23.25	2.82
Mountain reedbuck	39.94	4.84	20.34	2.46	11.58	1.40	5.90	0.71
Grey duiker	34.97	4.24	17.81	2.16	6.99	0.85	3.56	0.43
Rock hyrax	30.77	3.73	15.67	1.90	0.08	0.01	0.04	0.00
Smith's RR rabbit	27.75	3.36	14.13	1.71	26.36	3.19	13.42	1.63
Springhare	21.50	2.60	10.95	1.33	20.43	2.47	10.40	1.26
Porcupine	7.34	0.89	3.74	0.45	0.00	0.00	0.00	0.00
Chacma baboon	6.66	0.81	3.39	0.41	0.27	0.03	0.14	0.02
Tree hyrax	5.49	0.67	2.80	0.34	0.01	0.00	0.01	0.00
Grey rhebuck	4.84	0.59	2.46	0.30	1.21	0.15	0.62	0.07
Cape clawless otter	2.24	0.27	1.14	0.14	0.56	0.07	0.28	0.03
Aardvark	1.89	0.23	0.96	0.12	0.22	0.03	0.11	0.01
Vervet monkey	0.04	0.00	0.02	0.00	0.01	0.00	0.00	0.00
Total	**2,936**	**356**	**1,495**	**181**	**594**	**71.92**	**301**	**36.50**
Std Deviation	**374**	**45.31**	**190**	**23.07**	**76.35**	**9.25**	**38.89**	**4.71**

Sources: Prices adjusted to 2008 from Shackleton, *et al.* (2002); species data from various sources: see references

4.3.1.3 Discussion

The top ten contributors to standing stock value on the study site account for 98 % of the total value. These animals are, in descending order, bushpig, bushbuck, greater kudu, Cape grysbok, blue duiker, scrub hare, mountain reedbuck, grey (common) duiker, rock hyrax (dassie), and Smith's red rock rabbit. This is largely a function of either high average body weight per individual or high density per hectare: the four largest ungulate species (bushbuck, kudu, grysbok, and reedbuck) contribute 42 % of the total value of standing stock.

These findings are similar to results documented by Fa, *et al.* (2006) in Nigeria, where 80 % of the estimated number of carcasses documented in local bush meat markets were, in descending order, brush-tailed porcupine, blue duiker, bay duiker, guenons, and grasscutter rat. By weight, however, the top five species (four ungulates and one rodent) contributed over 68 % of the biomass documented in Nigerian markets. For comparison, in Cameroon the top five species (in order: brush-tailed porcupine, blue duiker, giant pouched rat, tree pangolin, and grasscutter rat) documented in local bush meat markets represented just over 51 % of total biomass (Fa, *et al.* 2006). The authors noted that the relative abundance of both brush-tailed porcupine and blue duiker likely contribute to their prominence in local markets.

Similarly, Fa, *et al.* (2005) surveyed published data from 36 sites in seven African countries and found that ungulates contributed 73.2 % of all hunted animals, while rodents and primates accounted for 12.2 % and 12.0 %, respectively. The largest species by mass (15.0 kg – 99.9 kg) constituted over half (54.4 %) of the total hunted biomass (Fa, *et al.* 2005). However, it is worth noting that smaller prey may be under-represented in markets since usually larger prey is sent to market (Fa & Garcia Yuste 2001) and small prey is consumed directly.

Due primarily to the limited availability of density and annual production data for non-ungulate species, but also to the topography and undocumented hunting on the study site, there are a number of possible sources of error in these calculations. As already discussed, the very nature of transferring published values from their original context to the study site reduces the accuracy of any estimation of ecosystem service value. While recommended stocking rate and reproduction data are generally available for ungulate species (Bothma 1989, 2002), both density and annual production estimates were sometimes based on fairly limited information about the average territory size and reproduction rates of non-ungulates.

For several small mammals, the available literature listed only a wide range of known territory sizes, and so the average of these published values was used to approximate actual territory size on the study site. As such, it is possible that the per hectare value estimates are either over- or under-estimates, depending on where the median or mode lies compared to the average. No annual production data were available for some small, rapidly reproducing species, such as hares. In these cases, annual production rates were transferred from animals of similar mass and/or taxa, which could introduce additional error into the estimations depending on how actual reproduction rates for each species compare to the reference species.

Moreover, available data rarely documented the extent to which individual species' territory ranges overlap, if at all. In all cases, it was assumed that home ranges of individuals of the same species were exclusive (i.e. non-overlapping), so to provide a conservative estimate of value. However, it may be the case that some less-territorial individuals actually have ranges that overlap with other individuals of the same species, depending on resource availability. If it is the case that individual territories of the same species do overlap, then slightly more individuals would be expected to live within a given piece of land and therefore contribute higher value than that estimated by assuming exclusive territories.

At the same time, the literature does not specify whether or not the territory size for an individual species was measured in areas of mixed species or only other individuals of the same species. If it were measured in a mixed species setting, then the identity and distribution of the other species present would also likely impact the actual territory available for the individual in question. This lack of data therefore precludes a reliable prediction of whether the estimated values are above or below actual observable values on the study site.

Furthermore, the topography of parts of the commonage may render these areas less accessible to hunting than even the protected areas within Waters Meeting NR. It is also almost certain that available habitats on the study site differ from those that occur in the original sites where density and reproduction data were collected. In light of these uncertainties associated with transferring published values to the study site, it is difficult to establish whether the current stock of bush meat is greater or less than that predicted by the literature.

Although difficult to quantify without detailed population surveys, hunting on the study site may affect the distribution and availability of game, particularly for large (> 15 kg) species (Davenport 2008a; Fa, *et al.* 2005, 2006). While recent research among Bathurst communities confirms that hunting, though technically illegal, occurs on the commonage (Fabricius, *et al.* 2006), whilst hunting within Waters Meeting NR Nature Reserve is officially prohibited and undocumented. However, it is possible that animals caught on the commonage may have spent time inside Waters Meeting NR and escaped (or been pursued) through broken fences along the contiguous borders shared by the two parcels of land. Since it is impossible to quantify hunting pressure on the study site based on the transfer of published values, this exercise likely overestimates the value of bush meat, in particular for larger species.

Large animals, in particular large ungulates, tend to have slower reproduction rates and lower densities than smaller species, leaving them more vulnerable to hunting and over-exploitation (Fa, *et al.* 2005). According to this exercise, the value of the eight largest species by mass (kudu, bush pig, aardvark, Cape grysbok, bushbuck, Chacma baboon, mountain reedbuck, and porcupine) account for roughly 47.5 % of the total standing stock value per hectare. This is in line with research on bush meat hunting throughout afrotropical forests, which has demonstrated that large-bodied species (15.0 – 99.9 kg) account for over half (54.4 %) of the total hunted biomass (Fa, *et al.* 2005). However, in areas faced with high hunting pressure and a concomitant reduction in large animal (ungulate) populations, rodents and small antelope typically account for a larger proportion of bush meat carcasses, though not necessarily total biomass (Cowlishaw, *et al.* 2005; Fa 2000; Fa, *et al.* 2000). Due to sustained hunting pressure on the study site, it is unlikely that many large species still exist in the densities recommended or predicted by the literature. As such, the values estimated here should be taken as upper bounds of the actual values of standing stock and annual production for bush meat on the study site.

4.3.2 Livestock

The value of small-scale livestock production to households living on or near communal areas has been well recognized throughout southern Africa (Dovie, *et al.* 2006). In contrast to commercial livestock production, which aims primarily to maximize slaughter value, communal livestock production provides a number of important direct and indirect values that contribute substantially to overall livelihoods (Cousins 1996; Dovie, *et al.* 2006; James, *et al.* 2005; Shackleton, *et al.* 2001, 2005). Apart from limited cash sales, the direct use value of livestock accrues primarily from draught power (whether used by the owner or hired out), transport, milk (typically for home consumption), dung (used as a sealant in construction and also for burning and fertilizer), meat, and hides. Cattle, in particular, also provide indirect financial value in the form of savings stored in the herd in the absence of rural commercial banking options. Although typically less important than these financial services, livestock also contribute indirect value through cultural services, such as ceremonial slaughter and bride-wealth payments[2] (Andrew, *et al.* 2003; Barrett 1992; Shackleton, *et al.* 2000, 2001).

In light of these multiple and on-going services, it is perhaps unsurprising that cattle sales are often limited to periods of extreme vulnerability, such as drought (Barrett 1992; Riethmuller 2003). In contrast, sales of goats are more commonly used to fund semi-regular purchases, such as school fees, household items, capital for trading and housing projects, and less frequently for ceremonies and celebrations (Barrett 1992; Dovie, *et al.* 2006). Thus, while livestock sales potentially offer owners a crucial source of cash, especially in isolated rural areas, communal livestock owners typically manage their herds to maximize overall returns from a number of services, rather than focusing strictly on commercial slaughter value (Andrew, *et al.* 2003; James, *et al.* 2005; Shackleton, *et al.* 2001).

Moreover, research on communal livestock production systems has demonstrated the importance of multiple direct use values, including milk, draught power, transport, and manure, and indirect values, such as bride-wealth payments, to *both* livestock owners *and* non-owners in the community through gifts and local trade (Dovie, *et al.* 2006; Shackleton, *et al.* 2005). In fact, Shackleton, *et al*. (2005) found that in the villages of the Bushbuckridge region, South Africa, between 40 % and 60 % of non-owning households received one or more of these services free or at a reduced rate from owning households. Even urban livestock owners can provide nutritional and income-generation services to the community through the production of milk and eggs (FAO 2001; Riethmuller 2003). Therefore, livestock production, especially of cattle, contributes wider social and economic benefits to the community beyond the minority of owner households that can be valued based on the cost of replacing these services with, for example, commercially procured fertilizer or milk (Cousins 1996; Shackleton, *et al.* 2001, 2005).

4.3.2.1 Methods

As with fuel wood production, the gross value of livestock (cattle and goats) per household in Nolukhanyo was estimated based on household user surveys; the values of sheep, horses, donkeys, chickens and pigs were not included in these livestock estimations due to low

[2] Known in the local language isiXhosa as *lobola*, the practice of the bride's family donating livestock to the groom's family is common throughout southern Africa (Andrew, *et al*. 2003; Shackleton, *et al*. 2001).

numbers reported by households (Davenport 2008a). The value of standing stock was estimated by multiplying the number of each animal by the mean price offered to small-scale herders in nearby Grahamstown (Davenport & Gambiza 2009). The following values were used in this exercise: for cattle, bulls were valued at R 5,400; cows were R 2,800; oxen R 3,450; and calves R 900. For goats, billies were valued at R 800; does at R 520; wethers R 700; and kids R 100 (Davenport 2008a). In addition to current (2007) stock numbers, households surveyed also reported their stock losses since 2006 due to death or theft. Several households owning either cattle or goats reported net losses over the 2006 – 2007 seasons; these data points were dropped from the analysis as per Davenport (2008a).

Annual production value was calculated by adding the value of incremental herd growth between 2006 and 2007 to the values of various benefits obtained from livestock on the commonage, including milk, meat sales, dung, and live animal sales. Annual herd growth was calculated as the net difference in value between the standing stock of cattle and goats in 2007 as compared to 2006, excluding inflation and negative values reported (Davenport 2008a). The value of livestock benefits generated annually, whether consumed at home or sold for cash, were calculated by Davenport (2008a) using the following prices: milk was valued at R 4/litre (mean selling price reported by surveyed households); dung at R 0.23/kg (Dovie, *et al.* 2006); the values of both meat and cash sales varied according to the prices reported by each household surveyed.

Mass estimates were approximated using the following conversions: one truck load of dung was assumed to weigh 300 kg; a wheel barrow 40 kg; and a 5 litre bucket 2.2 kg (Dovie, *et al.* 2006). Although transactions in both skins and transport services provided by livestock were solicited, no respondent reported receiving benefits from either service in 2007 (Davenport 2008a). See Davenport (2008a) for further details on livestock valuation, bearing in mind that the methodology reported here differs from the original study.

Next, these values calculated per livestock-owning household were aggregated across all households in the Nolukhanyo community. The average[3] value of each benefit (cattle or goat standing stock or annual production) per livestock-owning household was first multiplied by the number of households receiving that particular benefit (i.e. the number of households

[3] To ensure a reasonable value per user household and reduce standard variation, only non-zero values were used to calculate the mean and standard deviation of each reported benefit. The average value of each benefit was then multiplied across only those households reporting that particular benefit. This differs from the methodology employed by Davenport (2008a) and complicates data comparison with the original study.

owning either cattle or goat for standing stock; or the number of households reporting each individual benefit, e.g. milk, or dung, etc. for annual production). The resulting total value reported across all sampled households receiving each particular benefit was then divided by the total number of households in the sample (30) to estimate average value per commonage-user household.

Based on the finding that fully 70 % of households in Nolukhanyo collect one or more resources at least once annually from the Bathurst commonage (Davenport 2008a), the per commonage-user household value of each benefit was then multiplied by the total estimated number of commonage-user households in Nolukhanyo (1,232) to calculate gross total value of standing stock and annual production for the commonage. To facilitate aggregation with other ESVs, the resulting estimates were then converted to livestock value per hectare by dividing these figures by the total area of the Bathurst commonage in hectares (2,989). All figures were adjusted to constant 2008 Rand using the South African Consumer Price Index (StatsSA 2009) and converted to constant 2008 US dollars using average annual exchange rates (SARS 2008).

Waters Meeting NR does not currently support livestock production and this is unlikely to change in the near future due to restrictions placed on resource collection inside the reserve. Although small-scale collection of some resources, such as deadwood for fuel or medicinal plants, could potentially be compatible with the reserve's conservation mandate, the demonstrated impacts of livestock on subtropical thicket could seriously compromise the sustainability of this livelihood strategy within the reserve (e.g. Lechmere-Oertel, *et al.* 2005b; Mills, *et al.* 2005). Even if livestock grazing were allowed within the reserve, potential production would be severely limited by the availability of adequate fodder, which is largely restricted to small grassland areas along the Kowie River. As such, livestock production values were only estimated for the commonage.

4.3.2.2 Results

The gross per hectare value of the standing stock of livestock currently utilized on the commonage was estimated to be R 3,818 ± 4,889 (US$ 462 ± 592), while the gross annual production per hectare was R 873 ± 0.79 (US$ 106 ± 0.10). This yields gross total values of R 11,411,223 ± 14,612,605 (US$ 1,382,048 ± 1,769,777) for standing stock of cattle and goats and R 2,610,373 ± 2,354 (US$ 316,150 ± 285) for their annual production (Table 4.4).

Table 4.4 Value of livestock on the commonage (Constant 2008 Rand, US$)

Livestock service	Per hectare value		Total value	
	R	US$	R	US$
Standing stock of cattle	3,635	440	10,866,414	1,316,065
Standard deviation	4,768	577	14,251,044	1,725,988
Standing stock of goats	182	22.08	544,809	65,983
Standard deviation	121	14,65	361,561	43,790
Total standing stock	**3,818**	**462**	**11,411,223**	**1,382,048**
Standard deviation	4,889	592	14,612,605	1,769,777
Total annual production	**873**	**106**	**2,610,373**	**316,150**
Standard deviation	0.79	0.10	2,354	285

Source: Author's own calculation based on data collected by Davenport (2008a)

4.3.2.3 Discussion

With a total standing stock value more than an order of magnitude greater than any other direct use value estimated by the benefit transfer method for the study site, livestock undeniably represent a crucial component of the total economic value derived from natural resources on the commonage. The multiple values generated by livestock, including meat, milk, dung, and live sales, and the communal nature of these goods and services, whether redistributed through cash sales or as gifts, likely enhance the overall contribution of cattle and goats to total economic value (Andrew, *et al.* 2003; Barrett 1992; Shackleton, *et al.* 2000, 2001).

Moreover, the values reported here likely under-represent the total economic value of livestock to the Nolukhanyo community due to the lack of available data[4] on cultural and spiritual values, such as those derived from traditions like bride-wealth (*lobola*) and ceremonial slaughter, applicable to the study site. As previously discussed, spiritual and cultural service values are particularly difficult to either quantify or transfer due to their contextual specificity (Adamowicz, *et al.* 1998; Edwards & Abivardi 1997). Were these less certain but markedly valuable services included in this exercise, the total economic value derived from livestock produced on the commonage would be higher. Furthermore, thanks to the availability of livestock production data collected from the study site, this benefit transfer exercise is, for the most part[5], not subject to the same caveats that accompany the transfer of values from one context to another.

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[4] Davenport (2008a) only collected household data on direct use values.

[5] Davenport (2008a) transferred some benefit prices (e.g. dung, live animal values) from outside Bathurst.

Nonetheless, the limited number of data points reported by surveyed households (Davenport 2008a) does introduce some uncertainty into the reported values, especially for the standing stock estimates. Only five households out of the 30 sampled (16.7 %) reported owning goats; although, fully one-third of households (10) surveyed reported owning at least one type of cattle (Davenport 2008a). This small number of data points results in large standard deviations; it may also be an indication of the concentration of wealth among a few wealthy livestock owners, therefore potentially compromising the validity of community-wide value estimates. At the same time, the fact that 75 % of livestock owners reported net negative growth in their herds between 2006 and 2007 suggest that high risks are associated with livestock production on the commonage. As such, while livestock are a clearly an important source of value on the commonage, their contribution to overall community income may be hampered by inequitable distribution and/or high levels of inter-annual fluctuation.

4.3.3 Fuel wood

The dependence of rural communities in Africa on wood as their primary source of energy has been well documented (Campbell, *et al.* 2003; Dovie, *et al.* 2002; Madubansi & Shackleton 2007). Fuel wood can be used to meet a number of domestic energy needs, including for heating, cooking, and lighting, in both urban and rural households (Dovie, *et al.* 2002; Madubansi & Shackleton 2007). Even in South Africa, where the government has implemented an ambitious electrification program, evidence suggests that many poor households, even in urban areas, continue to rely heavily on fuel wood up to a decade or more after being connected to the country's electricity supply (Campbell, *et al.* 2003; Shackleton, *et al.* 2001). This is likely due to a combination of financial constraints, including the high cost of electrical appliances and monthly electricity bills (Howells, *et al.* 2005), and social considerations, such as the traditional importance of fuel wood collection in South Africa (Shackleton, *et al*. 2007b). Due to the limited availability of alternative energy sources within the spatial and/or financial accessibility many households, especially the poor and rural households, fuel wood is likely to remain a crucial energy source for the immediate future (Madubansi & Shackleton 2007).

Therefore, it is unsurprising that fuel wood collection represents a major provisioning service provided by the Bathurst commonage to residents of Nolokhanyo township (Davenport 2008a). In fact, Davenport (2008a) found that over 90 % of households who use at least one resource from the commonage annually ("user households") collect fuel wood from the commonage. This suggests that fully 64 % of the 1,760 households in Nolukhanyo, or 1,126 households, depend on fuel wood harvested locally to meet at least part of their annual energy demands. As such, it was decided to quantify the contribution of fuel wood to the per hectare value represented by natural resources on the study site.

4.3.3.1 Methods

Using household data on the direct use value of fuel wood collected from the Bathurst commonage (Davenport 2008a), the value of fuel wood collected per hectare was estimated by multiplying the average value per user household (R 1,641) by the number of households in Bathurst that use fuel wood collected on the commonage (1,126) to give an estimate of the total value of fuel wood currently utilized. This total value was then divided by the total area of the commonage (2,989 ha) to estimate current value per hectare. Since the household interviews did not specify the source of fuel wood, it was assumed that all fuel wood collected was deadwood to facilitate comparison with available literature (e.g. Shackleton 1993b, 1998).

Research on adjacent sites, one harvested and one a protected area, in a semi-arid South African savanna found that both species richness and the amount of deadwood were significantly reduced in the harvested site. Shackleton (1993b) estimated that the impact of these two resource collection regimes resulted in significantly more deadwood collected from the unharvested (3.9 % total biomass) site compared with the harvested (0.7 % total biomass) site, a relative difference of over 457 %. However, the landscape of parts of the commonage and Waters Meeting NR includes steep slopes and river valleys that would pose a serious challenge for resource collection. As such, not all deadwood produced in the study area is equally available for resource collection.

Research in savanna ecosystems suggests that more than 77 % of deadwood produced is not harvestable by hand because it is either located too high, is too big to carry, or is too small to be useful (Mudekwe 1997). Shackleton (1998) accounts for the accessibility of deadwood by hiring local residents to harvest deadwood from 28 plots in three protected areas over three years. His calculations include only the deadwood that is "utilizable," defined as that which is: detached, attached at ground-level, or is attached at less than 2.5 m above the ground; has a stem circumference of greater than 5 cm; and can be broken off by hand and carried.

Nonetheless, it is much more difficult to quantify the accessibility of fuel wood, or, for that matter, any other natural resource, on the study site based on the transfer of data from existing literature due to its unique topography and species. In light of this limitation, the per hectare value of fuel wood collected on the commonage as measured by household use surveys was augmented by only 40 % of the difference between the unharvested and harvested sites measured by Shackleton (1993b). Since over 457 % more deadwood was collected at unharvested sites compared with harvested sites, the annual production of deadwood measured on the commonage was augmented by 40 % of this difference, or roughly 183 %, to estimate deadwood production on Waters Meeting NR (Shackleton 1993b). This augmented value per hectare was then multiplied by the total area of the reserve (1,445 ha) to estimate the total value of fuel wood on Waters Meeting NR. Finally, based on the assumption that annual "utilizable" deadwood represents 1.7 % of total biomass (Shackleton 1998), the value of annual fuel wood production (available dead wood) on each site was estimated by multiplying the total value of fuel wood on each site by 1.7 %.

4.3.3.2 Results

Based on Davenport's (2008) commonage user surveys, the per hectare value of fuel wood collected on the municipal commonage (MC) was estimated to be R 618 (US$ 74.90), giving a total value of R 1,848,422 (US$ 223,868) for the standing stock of fuel wood collected on the commonage. Assuming that deadwood available for collection annually represents 1.7 % of biomass (Shackleton 1998), the recurring value of fuel wood collected on the commonage is expected to be R 31,423 (US$ 3,806). In contrast, assuming that the unharvested Waters Meeting (WM) NR protects a higher proportion of deadwood than the harvested commonage, but with limited accessibility, it is expected that the value for fuel wood in the reserve would be roughly R 1,131 (US$ 137) per hectare. This results in an estimated total standing stock of fuel wood worth R 1,634,011 (US$ 197,900), with annual production valued at R 27,778 (US$ 3,364). Total standing stock on the study site was thus estimated to be R 3,482,434 (US$ 421,768), and total annual production was valued at R 59,201 (US$ 7,170) (Table 4.5).

Table 4.5 Total and per hectare value of fuel wood (Constant 2008 Rand, US$)

Land use zone	Per hectare value		Total value	
	R	US$	R	US$
MC Standing stock	618	74.90	1,848,422	223,868
MC Annual production	10.51	1.27	31,423	3,806
WM Standing stock	1,131	137	1,634,011	197,900
WM Annual production	19.22	2.33	27,778	3,364
Total Standing stock	**1,749**	**212**	**3,482,434**	**421,768**
Total Annual production	**29.74**	**3.60**	**59,201**	**7,170**

Sources: Davenport 2008a; Shackleton 1993b

4.3.3.3 Discussion

As with each of the ESV transfers, there are a number of considerations to keep in mind when interpreting these numbers. The initial per hectare value of fuel wood used here is based on the fuel wood actually collected by residents of Nolukhanyo township. This could result in unobservable, potentially contrasting, errors in the calculations. Since the household surveys of fuel wood consumption did not specify the wood's source, using the values derived from Davenport (2008a) could potentially *over*estimate the amount of deadwood actually available for harvesting if residents are currently collecting live wood for use as fuel. This assumption will be tested in the next chapter based on field observations of the extent of cutting on the site. At the same time, were it the case that local residents are currently *under*-exploiting the available deadwood, the estimation cited here could, in fact, be an underestimate of the value of fuel wood in the study area. Without directly measuring annual biomass production, it is difficult to test this assumption. Thus, the site-specific nature of "sustainable" harvesting, where annual biomass harvesting remains below annual biomass production, precludes an accurate determination of the sustainability of current resource use patterns (Shackleton 1998).

Furthermore, it has been shown that fuel wood harvesters exhibit significant preferences for particular species and size classes (Pote, *et al.* 2006; Shackleton 1993b; Shackleton, *et al.* 2007a). Pote *et al.* (2006) determined the following five key species harvested for fuel wood in thicket: *Acacia karoo*, *Coddia rudis, Diospyros dichrophyia, Olea europaea* subsp. *africana,* and *Ptaeroxylon ubliquum*. In addition, Shackleton, *et al.* (2007a) identified *Scutia myrtina* as another important fuel wood source in the thicket biome. Without site-specific biomass estimations and household use data that identify the species and state (live or dead) of fuel wood, it is impossible to know which of these scenarios is correct.

Finally, Shackleton (1993b) showed that stems with a circumference above 10 cm were actively selected over smaller ones, and stems between 11 and 25 cm were the most frequently collected. However, since Shackleton (1998) defines "utilizable" deadwood as all stems greater than 5 cm in circumference, it is possible that stems between 5 and 10 cm may be "utilizable" but would not be preferentially selected by fuel wood harvesters. The extent to which this discrepancy affects the actual value of fuel wood collected would depend on the local demand for fuel wood, which itself is a function of not just dead wood availability, but also the cost of alternative fuels and climatic conditions (Eberhard 1986). Again, it is difficult to make a reasonable determination about the extent to which local resource users collect deadwood with a circumference of 5 – 10 cm without detailed user surveys.

Thus, it was necessary to make several assumptions about the availability, source, and size of fuel wood harvested on the study site to estimate its value using existing literature. While this estimation is therefore contingent on the validity of these assumptions, it can be considered a first best approximation of the value based on available figures. In light of the overwhelming dependence of user households on fuel wood collected from the commonage (91 % of all users, or 64 % of all households in Nolukhanyo township), it is unsurprising that this resource represents a significant ecosystem service in terms of value provided to the local community. The next chapter will attempt to refine these estimations based on site-specific transects used to determine the carbon sequestration potential from reducing deforestation on the commonage.

4.3.4 Honey

Wild honey represents one of several non-timber forest products (NTFPs) utilized by rural households for consumption and sales (Andrew, *et al.* 2003; Campbell, *et al.* 2002; Shackleton & Shackleton 2004). Although the total contribution of NTFPs to livelihoods in terms of home consumption and sales varies, evidence from Tunisia indicates that NTFPs, including fodder for livestock, charcoal production, honey, and tobacco cultivation, contributed up to 73 % of household incomes in 1999 (Daly-Hassen & Ben Mansoura 2005). Whereas, Ambrose-Oji (2003) reported that NTFPs contributed less than 20 % to total livelihoods in Cameroon. In southern Africa, studies indicate they represent 15 – 30 % of incomes (Shackleton, *et al*. 2007b).

In fact, research from around the world indicates that values of honey, like those of other NTFPs, vary widely according to a number of factors, including prevailing market prices, local abundance of flowering plants and honeybee populations, and opportunity costs for collection (Croitoru 2007; Gubbi & MacMillan 2008; Shackleton & Shackleton 2004). In a survey of NTFP values in the Mediterranean region, Croitoru (2007) found that although the average annual per hectare value of honey in the region as a whole was just € 1, its relative importance to northern Mediterranean countries, such as Slovenia, Cyprus, and Greece, was evidenced by annual values of € 5-10/ha. However, these values paled in comparison to Lebanon (€ 98) and Egypt (€ 97), where high local prices (Liberia) and smaller forested areas result in significant value per hectare earned from local honey collection (Croitoru 2007).

Furthermore, Gubbi & MacMillan (2008) recorded mean daily revenue of US$ 3.15 $\pm$ 4.19 per day[6] per honey collector near the Periyar Tiger Reserve in Kerala state, India, where average wages for agricultural labor range from US$ 2.09/day in Kerala to US$ 1.38/day in nearby Tamilnadu state (DES 2003, 2005).

In their survey of NTFP collection recorded by 14 studies in South African savannas, Shackleton & Shackleton (2004) found that over half (50.5 %) of all households surveyed collected wild honey, with a range from zero to 96.7 % of households utilizing this natural resource. Among households surveyed, only 5.6 % of households bought or sold honey in Limpopo, while in KwaZulu-Natal, up to 15 % and 6.4 % of households purchased or sold honey, respectively (Shackleton & Shackleton 2004). In light of this wide variation, it is important to evaluate the contribution of each NTFP within the local socio-economic and ecological context.

In addition to honey production, natural ecosystems also provide pollination services to both wild plants and commercial crops. Based on the assumption that the indigenous fynbos vegetation supports approximately 50 % of honey production in South Africa's Western Cape, the value added from pollination services provided by 15,000 hives to the Western Cape deciduous fruit industry was estimated to be roughly 1998 R 593,400, or over R 1 million in 2008 terms (US$ 127,404) assuming no growth in the industry over the past ten years (Hassan 2003; Turpie, *et al.* 2001). However, some argue that the managed pollination services in the Western Cape, where beekeepers actively relocate their hives to orchards in exchange for a fee, is in fact a commercial input to agricultural crop production (Allsopp, *et al.* 2008; Cook, *et al.* 2007; DFPT 2005).

Furthermore, in contrast to the Western Cape, where annual production of deciduous fruits, including apples, grapes, peaches, pears, and plums, totals over four million metric tones (4,290,337 MT), farmers in the Eastern Cape produce just 97,241 MT per annum (Shabalala & Mosima 2002). In fact, the few remaining crop farms in the immediate vicinity of the study site typically cultivate pineapple, a crop that is propagated asexually (Chan 2008; Fabricius, *et al.* 2006). As such, there is likely to be less scope for achieving anywhere near the value added by managed pollination services in the Western Cape. Finally, there is no evidence that smallholders in South Africa or elsewhere have achieved the physical capital

[6] Although data were collected on the number of days per trip, no data on the number of trips per year was recorded (Gubbi & MacMillan 2008). Thus, it was not possible to quantify annual per hectare value.

(e.g. trucks), scale (beekeepers often transfer hundreds of hives to a single fruit farm), and management skills necessary to successfully negotiate with farmers and engage in this enterprise (Timmermans 2005). This is not to suggest that it would be impossible for them to do so; merely that existing experience does not provide sufficient evidence to accurately quantify the potential income from this activity. Instead, this section will focus on the potential honey revenue from the study site based on the experience of several small-scale honey production projects in South Africa.

4.3.4.1 Methods

Potential annual honey production on the study site was estimated from actual annual production rates obtained by small-scale honey producers in two provinces of South Africa: the Eastern Cape (specifically the area known formerly as the Transkei homeland) and KwaZulu-Natal (Timmermans 2005). Timmermans (2005) collected production data from four small-scale honey projects covering between one and three years of production per project. Data from one project at Bushbuckridge were incomplete and so were excluded from this analysis. Data on the three remaining projects, namely Lutubeni Project, Lethimpumelelo Trading Cooperative, and the SAPPI Honey Project[7], included the number of hives, total production (kg), and the price (Rand) at which processed honey was sold. Using these data, honey production per hive (kg/hive) and the revenue per hive (R/hive) were calculated for each project. All prices were converted to 2008 US dollars ($) using inflation data from the South African Consumer Price Index (StatsSA 2009) and average annual exchange rates from the South African Revenue Service (SARS 2008).

The industry norm of one hive per two hectares of forest was used to estimate the number of hives that could be supported by vegetation on Waters Meeting NR and the commonage. To account for the lower density of vegetation on the commonage, the number of hives on the commonage was discounted according to the proportional loss of vegetation estimated during the fuel wood exercise (i.e. 56.6 %) with an additional 10 % decrease calculated to account for habitat fragmentation and edge effects on honey production on the commonage. The average revenue per hive (2008 currency) estimated from the three small-scale honey projects

[7] Although Group 1 trained in 2001 under the SAPPI Honey Project collectively produced more than twice as much honey in 2002 as Group 2, who harvested their first honey in 2002, all honey from the two groups was sold together. To account for the substantial difference in production between groups, the total value of combined sales was adjusted according to the relative honey contribution (total kg) of each group to the total mass of honey collected, or roughly 70 % by Group 1 and 30 % by Group 2 (Timmermans 2005).

as reported by Timmermans (2005) was then multiplied by the projected number of hives each land parcel could support to estimate the total potential annual revenues from honey production on the study site. Since the purpose of this exercise is to estimate benefit flows on the study site, rather than design a business plan for exploiting honey production, production costs, such as capital equipment, training, and marketing equipment, were excluded from the calculations.

The total value of the 'standing stock' of honey production on the commonage was assumed to be the value of hives that could be supported by the study site using the one hive per two hectares norm multiplied by the approximate value per hive of R 393 (converted INR 2003 estimate to 2008 Rand; US$ 47.62) (StatsSA 2009; Timmermans 2005). The value of hive depreciation was excluded due to insufficient data availability and in keeping with the gross benefit calculations.

4.3.4.2 Results

Table 4.6 reports the range of production data reported by Timmermans (2005). Average revenue per kg honey produced was R 25.49 $\pm$ 8.86 (US$ 3.09 $\pm$ 1.07), in line with the prevailing 2008 price (R 25 – 27/kg) for bulk honey reported to the National Agricultural Marketing Council (NAMC 2008). Average revenue per hive was estimated to be R 208 $\pm$ 111 (US$ 25.19 $\pm$ 13.45), with a range from R 55 (US$ 6.72) achieved by the Lutubeni Project in 2002 to R 432 (US$ 52.31 earned by SAPPI Group #1 that same year (Timmermans 2005). Average honey production per hive was estimated to be roughly 11.3 kg $\pm$ 4.7, with a range from 3.8 to 15.0 kg.

The commonage was estimated to support 1,522 hives on its relatively more open 2,989 ha of vegetation, while Waters Meeting NR could potentially support roughly 723 hives in its densely forested 1,445 ha (Table 4.7). Excluding depreciation, this gives a standing stock value of R 598,366 (US$ 72,470) for the commonage and R 284,076 (US$ 34.405) for Waters Meeting NR, or roughly R 882,442 (US$ 106,875) for the entire study site. Using an average revenue of R 208 $\pm$ 111 (US$ 25.19 $\pm$ 13.45) per hive, the estimated total annual revenue from honey production on the commonage was R 316,493 $\pm$ 168,954 (US$ 38,331 $\pm$ 20,463), compared with roughly R 150,256 $\pm$ 80,211 (US$ 18,198 $\pm$ 9,715) from Waters Meeting NR. In total, the study site could potentially support annual honey production revenues of R 466,749 $\pm$ 249,165 (US$ 56,529 $\pm$ 30,177).

Table 4.6 Actual revenues obtained from small-scale honey production projects in South Africa (Constant 2008 Rand, US$)

Project	Location	Year	Honey	Hives	Honey	Revenue per kg		Revenue per hive	
Name	Province, locale		kg/hive	#	Total kg	R	US$	R	US$
Lutubeni Project	Transkei, Mthatha	2002	15.0	60	150	22.19	2.69	55	6.72
Lutubeni Project	Transkei, Mthatha	2003	9.3	42	375	7.93	0.96	71	8.58
Lutubeni Project Private*	Transkei, Mthatha	2004	11.1	30	113	34.55	4.18	164	19.82
Lutubeni Project Retail*	Transkei, Mthatha	2004	11.1	30	176	36.07	4.37	211	25.56
Lethimpumelelo Trading	KZN, Nyalazi	2004	3.8	60	225	31.89	3.86	151	18.26
SAPPI Group 1	KZN, Ixopo District	2001	15.0	240	3,750	20.17	2.44	315	38.17
SAPPI Group 1	KZN, Ixopo District	2002	21.3	330	7,014	20.32	2.46	432	52.31
SAPPI Group 2	KZN, Ixopo District	2002	12.2	240	2,938	20.32	2.46	249	30.13
SAPPI Groups 1,2,3**	KZN, Ixopo District	2003	8.1	810	6,569	27.84	3.37	226	27.34
SAPPI Groups 1,2,3**	KZN, Ixopo District	2004	6.1	810	4,976	33.59	4.07	206	24.99
Smallholder average	**Eastern Cape/KZN**	**--**	**11.3±4.7**	**291±299**	**2,926±2,721**	**25.49±8.86**	**3.09±1.07**	**208+111**	**25.19±13.45**

Source: Timmermans 2005

*Of the 338 kg harvested by the Lutubeni Project members in 2004, a private buyer purchased 113 kg in bulk, while 176 kg was sold in individual 500 g bottles.

The remaining 49 kg were not sold as of the time of reporting in 2005 (Timmermans 2005).

**Combined total production for all three groups trained in 2001, 2002, and 2003, respectively (Timmermans 2005).

Table 4.7 Expected annual revenues and standing stock value of honey production on study site (Constant 2008 Rand, US$)

Land use zone	Area	Hives	Revenue per hive		Annual production (R)		Annual production (US$)		Standing stock	
	ha	#	R	US$	Value	Std Dev*	Value	Std Dev*	R	US$
Commonage	2,989	1,522	208	25.19	316,493	168,954	38,331	20,463	598,366	72,470
Waters Meeting NR	1,445	723	208	25.19	150,256	80,211	18,198	9,715	284,076	34,405
Total Study Site	**4,434**	**2,244**			**466,749**	**249,165**	**56,529**	**30,177**	**882,442**	**106,875**

Sources: Based on values from Timmermans 2005, NAMC 2008

*Standard deviation reflects the standard deviation in revenues transferred from Timmermans (2005) and does not include standard deviation in number of hives.

4.3.4.3 Discussion

As with all ESV transfers, differences between the context of the original site(s) and the study site affect the applicability of the transferred value (Troy & Wilson 2006). To avoid exaggerating the potential productivity of local honey production on the study site, only data from small-scale projects, rather than commercial honey enterprises, were used to estimate potential profits. However, the three small-scale projects for which data are reported are all located within or adjacent to commercially- or state-managed forest plantations of eucalyptus, gum, and wattle trees (GBM 2007; SAPPI 2005; Timmermans 2005). Since data on the honey production capacity of indigenous subtropical thicket were not available, the values reported here should be treated with caution due to likely differences in the density of flowering plants for bee forage.

Moreover, the project investment design may affect overall profitability, particularly with respect to the funding of initial costs, including training and the purchase of equipment for handling the bees and processing the honey. In many cases the project organizer, such as SAPPI Forest Products, the Department of Water Affairs and Forestry (DWAF) or the Agricultural Research Council (ARC), funded some or all of the initial start up costs through grants (e.g. ARC; DWAF) or loans (e.g. SAPPI). In the case of both the Lutubeni Project and the Lethimpumelelo Trading Cooperative, all initial costs, including hive boxes, harvesting and processing equipment, and participant training, were covered by grants from the DWAF and ARC, for Lutubeni, and the Institute of Natural Resources (INR) and the Department of Trade and Industry (DTI), for Lethimpumelelo. Although payments made to members of the Lethimpumelelo Trading Cooperative were not reported, members of the Lutubeni Project earned an average of R 300 over the first three years of the project, although this was skewed heavily by a payment of R 700 in 2004 (Timmermans 2005).

In contrast, beekeepers in the SAPPI project were offered small (R10,000) loans to cover the purchase of equipment and protective clothing. Each year, the beekeepers contribute 25 % of their individual incomes to loan repayment, and these fees are used to fund the donation of two extra hives for each hive purchased by the bee keepers, as well as various running costs associated with the honey processing facility. In addition to funding the purchase and maintenance of equipment for the project through small loans, SAPPI also sponsored an intensive training program for all participants free of charge. Average earnings per beekeeper during the 2001 – 2004 seasons were over R 4,553 per year (nominal), an order of magnitude larger than the payouts to members of the Lutubeni Project. Still, as of the fifth year of operating the beekeeping project in Ixopo, SAPPI had invested over R 1,370,000 for a return of just R 465,316 from honey sales. Therefore, it is clear that in all cases the profitability of these small-scale projects depends heavily on outside funding of the initial start up costs and training (Timmermans 2005). As such, this finding should be considered in any future promotion of beekeeping on the study site.

Regardless of the model of initial investment (grant vs. loan), all projects reported problems with high transport costs and equipment theft. In the case of Lutubeni, beekeepers from remote areas distant from the location of the actual apiaries were selected for participation in the project. The project did not monitor or reimburse transportation costs; as such, net profits to members would be even lower than those reported. Similarly, transport costs reportedly represent a significant proportion of project management budgets for the Lethimpumelelo Trading Cooperative and the SAPPI Honey Production project. In fact, faced with high transport costs and the ecological and social carrying capacity of SAPPI's project, the program was limited to 36 participants, less than half of the original target group of 100 beekeepers (Timmermans 2005).

In addition, all projects have faced other major setbacks, including theft and vandalism, high turnover rates among beekeepers, hive predation and disease, and unfavorable ecological conditions, such as a major drought in 2003 in KwaZulu-Natal that negatively affected the availability of nectar and pollen for the bees. Incredibly, the combination of theft by people and baboons, pesticide spraying, and forest fires resulted in the loss of roughly 75 % of the active hives in the Lutubeni Project during the 2002/03 season, and caused a 38 % drop in honey production per hive from 15.0 kg/hive in 2002 to 9.3 kg/hive in 2003. The widely reported theft and vandalism of hives may actually be an indication that there is additional demand for honey production projects beyond the current supply (Timmermans 2005). In light of these numerous and noteworthy challenges, the inter-annual variation of profits from beekeeping on the study site are likely to be high. However, the inclusion of several consecutive years of data from the same project should reduce the magnitude of error in the calculations presented here.

Moreover, although the overall average revenue per kg of honey produced (R 25.49 $\pm$ 8.86) was in line with national bulk honey prices (R 25 – 27) as reported by NAMC (2008), there was significant variation in the prices offered to the novice beekeepers in the three projects surveyed. At worst, the members of Lutubeni Project received R 7.93/kg (constant 2008) for their honey in 2003, compared with R 36.07/kg the very next year. Despite the feeling among members of the Lethimpumelelo Trading Cooperative that the prices offered by the commercial beekeeper that mentors the group were below those accessible via direct retail sales, the group acknowledged that this offer was a reliable sales option (Timmermans 2005). Thus, smallholder beekeeping profits may not reach their potential without marketing training and adequate access to markets.

Furthermore, as demonstrated by the SAPPI Honey Production project, designing appropriate incentive structures to encourage individual entrepreneurship and responsibility can potentially lead to considerable incomes from beekeeping. Nonetheless, expected profits from beekeeping on the study site would likely be highly variable and modest for at least the first few seasons due to the high initial costs and various unpredictable challenges, such as theft and climate variation (Timmermans 2005). As such, beekeeping should ideally be promoted as one of several potentially sustainable livelihood strategies rather than as a stand-alone income-generating activity.

4.3.5 Medicinal plants

In a country where residents outnumbered medical doctors by 17,400:1 just before its first democratic elections in 1993, it is unsurprising that a significant proportion of the population meets at least some of their health care needs through traditional medicine (Pretorius, *et al.* 1993). Cocks & Dold (2002) note that the combination of "high population growth, rapid urbanization, unemployment, and the high cultural value of traditional medicines" continues to fuel demand for medicinal plant species. Although medicinal plant collection is typically concentrated in rural areas where various plant species thrive, Cocks & Dold (2006) have also demonstrated the importance of wild plants to the cultural lives of *urban* Xhosa people..

In addition to supporting traditional belief systems, the harvesting of medicinal plants also represents a noteworthy livelihood strategy (Cocks & Dold 2002, 2006; Shackleton, *et al.* 2001). The national trade in medicinal plants in 1998 was estimated at approximately 20,000 t/yr, worth a total annual value of roughly R 479 million today (Mander 1998). Cross-border trade of indigenous plants used for traditional healing is also reportedly thriving throughout Africa (Cunningham 1997). The astonishing value generated by this 'hidden economy' is supported by the equally vast scale of medicinal plant users in South Africa (Cocks & Dold 2002). Mander (1998) estimated that 27 million South Africans regularly used wild plants for medicinal purposes, roughly 67 % of the population at the time (StatsSA 1998). Although the traded values of individual species vary widely (Cocks, *et al.* 2004; Cocks & Bangay 2006), Shackleton, *et al.* (2001) reported that overall, medicinal plant use was valued at between R 66 and R 500 per household per year across seven study areas in South Africa.

Nonetheless, there are concerns that medicinal plant collection could prove unsustainable in light of the high and increasing demand for indigenous plants, informal and unregulated nature of harvesting activities, and high degree of commercialization of the trade in medicinal plants (Botha *et al.* 2004; Cocks & Dold 2006; Cunningham 1997; Dold & Cocks 2002; Sims-Castley 2002). Harvesters surveyed by Cocks, *et al.* (2004) in four major towns in the Eastern Cape (Port Elizabeth, East London, Umtata, and Queenstown) reported collecting 220 species, of which 166 were particularly important for trading. Medicinal plants are most often harvested from the wild with minimal or no management, especially in communal areas and public lands, such as municipal commonage, which potentially jeopardizes the sustainability of this livelihood strategy (Sims-Castley 2002). Furthermore, based on their survey of six Eastern Cape medicinal plant markets, Cocks & Dold (2002) conclude that 93 % of the species traded were harvested 'unsustainably' because all or at least crucial parts (e.g. bark) of the plants were removed entirely, resulting in the death of the plant. An assessment of the sustainability of current harvesting on the commonage is beyond the scope of this benefit transfer exercise. Still, it is worth noting that without alternative plant sources, such as cultivation, or better regulation of wild harvesting activities, the future availability of certain indigenous plant species and the livelihoods dependent on those species could be at risk (Davenport 2008a; Sims-Castley 2002; Wiersum, *et al.* 2006).

4.3.5.1 Methods

Similar to fuel wood production, the total value of standing stock of medicinal plants on the commonage was estimated by multiplying the average value reported by household user surveys by the number of households reporting medicinal plant collection on the commonage (Davenport 2008a). The per hectare value of medicinal plants collected on the commonage as measured by household use surveys was augmented by 40 % of the difference in above-ground biomass between the 'intact' and 'degraded' sites as reported by international literature to account for the impact of resource collection on the commonage vegetation (Jaramillo, *et al.* 2003; Mills, *et al.* 2005; Penzhorn, *et al.* 1974). The total standing stock value of each site was then divided by its total area to calculate value per hectare.

Finally, the value of annual production was estimated based on the growth characteristics of the fourteen plant species identified by user households. Eight of the species reported by users were woody plants (shrubs or trees) while the remaining six were forbs, climbers, and succulent plants (Davenport 2008a). Since species-specific growth rate data were not available, the per hectare values of standing stock on the commonage and Waters Meeting NR were first multiplied by the proportional representation of each life form as shown in Table 4.8 (57 % woody plants; 46 % non-woody plants) and then multiplied by an average growth rate of 3 % per annum for woody plants (Shackleton 1993c) and 15 % for other species. The per hectare values were then multiplied by the area of each land parcel to estimate total annual production. All figures were converted to constant 2008 Rand using the South African CPI (StatsSA 2009) and to 2008 US dollars using average annual exchange rates (SARS 2008).

Table 4.8 Medicinal plants used by Nolukhanyo households surveyed

Vernacular name[1]	Suggested botanical name[1]	Life form[2]	Harvested[3]	Vegetation[3]
Impendulo	*Rubia petiolaris*		Root	Valley Thicket
Imphepho	*Helichrysum odoratissimum*	Forb	Leaf & stem	Grassland
Inceba	*Polygala serpentaria*	Shrub	Root	Valley Thicket
Iperepes	*Clausena anisata*	Shrub/small tree	Leaves	
Irooiwater	*Bulbine latifolia*	Succulent	Rhizome	Valley Thicket
Isidumo	*Ilex mitis*	Tree	Bark	Forest
Mayisake	*Cissampelos capensis*	Shrub	Bark	Valley Thicket
Ubulawu	*Scabiosa columbaria*	Forb	Root	Grassland
Uchithibunga	*Rhoicissus digitata*	Climber	Tuber	Forest & valley thicket
Uchithibunga	*Rhoicissus tridentata*	Climber	Tuber	Forest & valley thicket
Umhlonyane	*Artemisia/Marrubium* spp.	Shrub		
Umnonono	*Strychnos henningsii*	Tree	Bark	Forest
Uphuncuka	*Crassula/Talinum* spp.	Shrub		

Sources: [3]Cocks & Dold 2002; [1]Davenport 2008a; [2]Various sources: see references

4.3.5.2 Results

The total standing stock of medicinal plants on the commonage was estimated to be R 49,667 (US$ 6,706), with a per hectare value of R 17 (US$ 2.24). The standing stock of medicinal plants on Waters Meeting NR was estimated to contribute R 27,068 (US$ 3,655) overall and R 19 (US$ 2.53) per hectare. Annual production on the commonage was R 4,310 (US$ 582) and R 2,349 (US$ 317) on Waters Meeting NR. Annual production per hectare was R 1.44

(US$ 0.19) and R 1.63 (US$ 0.22) on the commonage and Waters Meeting NR, respectively (Table 4.9).

Table 4.9 Total and per hectare value of medicinal plants (Constant 2008 Rand, US$)

Land use zone	Standing stock value				Annual production value			
	Per hectare		Total		Per hectare		Total	
	R	US$	R	US$	R	US$	R	US$
Commonage	17	2.24	49,667	6,706	1.44	0.19	4,310	582
Waters Meeting NR	19	2.53	27,068	3,655	1.63	0.22	2,349	317
Total	**35**	**4.77**	**76,735**	**10,361**	**3.07**	**0.41**	**6,660**	**899**

Source: Davenport 2008a

4.3.5.3 Discussion

Due to the small sample size of the user survey and limited species data available, conclusions about the total value of medicinal plants on the study site should be made with caution. Firstly, although surveyed households identified fourteen different plant species collected on the commonage, each species was cited only once, making it difficult to gauge the relative importance of one or more species. For this reason, Davenport (2008a) used an average value of R23/kg across all species reported (Cocks, *et al.* 2004; Cocks & Bangay 2006). However, mean values reported at six different Eastern Cape markets for the species used by Nolukhanyo households range from R41/kg for *Ilex mitis* bark to R105/kg for *Cissampelos capensis* bark. As such, better species use data based on a larger sample size could allow for a more accurate valuation of medicinal plants on the commonage.

Additionally, due to the lack of available data on annual growth rates, or most other botanical details beyond life form, for the medicinal plants reported by Davenport (2008a), the relative proportion of woody and non-woody plants by number of species cited were used here to calculate annual production. Species-specific growth rates would therefore likely affect the annual production calculation, which would also allow for some assessment of the sustainability of current harvesting rate. Nonetheless, the small sample size of medicinal plants reported by surveyed households would complicate accurate estimation of both sustainability and the contribution of medicinal plants to total economic value on the study site.

Furthermore, of the thirty households interviewed by Davenport (2008a), only five (roughly 17 %) reported collecting medicinal plant on the Bathurst commonage, a use rate higher than only clay and sweepers (13 % each) and well below Mander's (1998) estimate that 67% of

South Africa's population used medicinal plants. Arguably, not all households who use medicinal plants necessarily collect them on the commonage. Nonetheless, it would appear that users represent a somewhat smaller proportion of the community than has been found elsewhere (Davenport 2008a). Moreover, at just R 28 $\pm$ 93 per household across the community, medicinal plants fall in the bottom third of the resources studied in terms of contribution to household incomes in Nolukhanyo (Davenport 2008a) and well below inflation-adjusted figures of R 66 – R 500 reported from seven different locations by Shackleton, *et al.* (2001). As such, even if the conservative value or growth rates used here were doubled, the overall contribution of medicinal plants to the total economic value of the study site would likely remain modest.

Nonetheless, these data represent only the traded value of medicinal plants and do not account for either the replacement cost of the services they provide or their sacred value. An extensive evaluation of the medicinal properties of the reported plant species and estimation of the costs necessary to replace these functions with traditional medicines were beyond the scope of this valuation exercise. Nor does this valuation exercise account for the sacred properties of medicinal plants due to the methodological and ethical constraints discussed already (Adamowicz, *et al*. 1998; Edwards & Abivardi 1997). However, Cocks & Dold (2006) maintain that recognizing the cultural and spiritual values of medicinal plants would provide an important incentive for biodiversity conservation, itself another inherent source of the overall value of medicinal plants. Therefore, it should be noted that the modest values reported here might not adequately capture the total economic value of medicinal plants on the study site.

4.3.6 Willingness to pay to protect endangered species

Each of the benefit transfer exercises thus far has focused exclusively on direct use or consumption values of natural resources. However, non-use and non-consumptive values of wildlife are a potentially significant source of indirect benefits to human livelihoods that, if valued, can provide additional motivation for careful land-use planning and conservation (Allen & Loomis 2006). Perhaps the most well recognized method for capturing the non-use values of biodiversity is contingent valuation (CV), which elicits respondents' willingness-to-pay (WTP) for a particular ecosystem good or service. The accuracy of CV estimates is dependent on a number of factors, including the respondents' familiarity with the resource being valued, their inherent biases, and the substitutability of various goods and services (Allen & Loomis 2006; Loureiro & Ojea 2008; Martin-Lopez, *et al.* 2007). Nonetheless, CV has been an important source of information on the benefits of endangered species conservation and has played a key role in conservation policy at the U.S. Environmental Protection Agency (EPA) and elsewhere.

This section will focus on the valuation of two endangered species found in the study area: leopard (*Panthera pardus*) and Eastern Cape Rocky (*Sandelia bainsii*). In contrast to the leopard, a well-known, so-called "charismatic" species with an extensive, although threatened, range throughout Africa and parts of Asia, the Eastern Cape Rocky is a relatively unknown (to the public) fish that is only found in tributaries of three rivers in the Eastern Cape, one of which is the Kowie River (Cambray 1996). According to a local expert, the Rocky may go extinct in the next ten years unless management actions are taken to protect it from local threats, such as loss of habitat, invasive alien fish like bass and catfish, and sedimentation in the Kowie (J. Cambray, pers. comm. 2008). Although free-roaming leopards have been nearly eliminated from the local landscape over the past 150 years of permanent settlement in the area, recent sightings of leopard kills on farms near the study site suggest that there may be a few remaining leopards living (or at least moving) outside the protection of Waters Meeting NR (Cole, pers. comm. 2008). In order to better gauge potential support for conservation initiatives to protect these two locally endangered species, this benefit transfer exercise will estimate local WTP based on a survey of international CV studies of similar endangered species.

4.3.6.1 Methods

This benefit transfer exercise assumes that residents living in towns within the Kowie River catchment area—Grahamstown, Bathurst, and Port Alfred—one of the last known habitats of the Eastern Cape Rocky and local leopard, are willing to pay some amount to ensure the continued existence of these two animals in the catchment. A survey of internationally[8] published values for *individual* species as established by contingent valuation (CV) methods was conducted to estimate an average willingness to pay for the conservation of the Eastern Cape Rocky and leopard. Surveys that asked respondents to value multiple species together were excluded. Based on a recent meta-analysis of CV applications to species conservation that found a significant influence of survey year on reported WTP, only surveys conducted during or after 1994 were considered (Richardson & Loomis 2009). Only surveys of fish species were used to calculate estimated willingness to pay for the Rocky, whereas surveys of

[8] It was necessary to do an international search because there are no previous CV studies on *individual* species in South Africa.

"charismatic" species, including marine mammals, iconic birds, and large terrestrial carnivores, were used to estimate the potential value of leopard to residents living in the Kowie catchment.

All monetary values reported in the international literature were first converted from foreign currency to US dollars (if applicable) using exchanges rates quoted either by the source or from the U.S. Central Intelligence Agency (CIA) World Factbook (2009). Historical US prices were then standardized to 2008 US dollars using the U.S. Bureau of Labor Statistics Consumer Price Inflation (CPI) index (2009).

It has long been hypothesized that external conditions, such as personal income, can have a significant impact on environmental behavior (Bostedt, *et al.* 2008; Guagnano, *et al.* 1995; Kotchen & Reiling 2000; Mohai 1985). To account for the potential impact of income on respondents' willingness to pay (WTP), the reported mean WTP in 2008 US dollars of each survey was expressed as a percentage of the average annual household income[9] of respondents (in 2008 US dollars) as reported by the study or based on public income data (e.g. Statistics Sweden 2007; US Census Bureau 2008b) available for the study area where sample data were unavailable.

The resulting WTP expressed as a percentage of average income was then multiplied by the average annual household income in Makana and Ndlambe local municipalities (LM) to estimate WTP per household. As already mentioned, Bathurst and Port Alfred form part of the Ndlambe LM, while Grahamstown falls within the Makana LM. However, both local municipalities encompass several smaller towns in the rural areas surrounding these three settlements. Unfortunately, disaggregated *household* income data were not available at the sub-municipal level for each of these towns; instead, data for the local municipalities were used (StatsSA 2005).

Moreover, census data on household incomes are reported according to the number of households reporting incomes within a given range (StatsSA 2005). To transform municipal data into a single estimate of average annual household income for the study area, the midpoint of each income range was multiplied by the number of households reporting for that range. For each municipality, the sum of these aggregate income data for all income ranges

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[9] It is important to note that much of the literature does not specify whether the reported annual income was gross, net after taxes and other payments, or disposable (with the notable except of all the Swedish studies).

was then divided by the total number of households in the municipality to estimate a local municipality average annual household income. The estimated WTP values per household for Makana LM and Ndlambe LM were then multiplied by the total number of households in each local municipality, respectively, and the resulting estimates of WTP by local municipality were summed to estimate total WTP for the conservation of the Rocky or leopard in the Kowie River catchment in 2008 US dollars.

Adult education levels and the proportions of the sampled population living in rural versus urban areas were also compared between Makana LM and Ndlambe LM and national census data for the two best-represented countries in the international literature, namely Sweden and the United States. To best capture the distribution of the adult population (defined as persons aged 20 years or older for South Africa and as persons between 25 – 64 years of age in Sweden and the US) that had attained *at most* a given level of education (measured in years or "grades"), national data that report the highest level of education attained were used rather than reported sample means (Statistics Sweden 2007; StatsSA 2005; US Census Bureau 2007).

In addition to the different definition of 'adult' used by Statistics South Africa, the data available from Statistics Sweden (2007) reports primary and secondary education statistics as aggregated data, thereby complicating comparisons with data from South Africa and the United States. To overcome the various requirements necessary to achieve tertiary (post-secondary) degrees in different countries, data from all three countries were aggregated into one category (Tertiary Education) for this analysis. Still, direct comparisons between the adult education level data sets should be avoided; rather, the relative magnitude of differences in the reported data will be examined.

Rough estimations of the proportion of the sampled populations living in rural and urban areas were calculated based on official population statistics. Not every study reviewed investigated the effects of rural versus urban residency on WTP. Moreover, studies that did explore this variable typically distinguished between a "rural" population and an "urban" population and therefore were designed to sample each of these populations separately. To overcome this sampling bias, national rural-urban population data were used for comparison with the study area.

For South Africa, disaggregated population data for Ndlambe LM were used to sum the total number of inhabitants in the municipality's two "urban" areas: Port Alfred and Alexandria, identified as the primary economic centres according to the Ndlambe Integrated Development Plan (2007). As already mentioned, disaggregated population data for Makana LM were not available, so rural-urban population data for Makana LM were not estimated. For Sweden, the total population of residents living in the six counties collectively referred to as the "carnivore area" in the work of Ericsson, *et al.* (2008) was considered the "rural population;" following the sampling method for this study, the rest of Sweden was considered to be "urban."

Rural-urban population data for the United States were last collected in the 2000 Census (US Census Bureau 2009). For this census, the United States Census Bureau (2008a) defined "rural" as all areas outside of urban areas; urban areas are defined to include core census block groups or blocks that have a population density of at least 1,000 people/sq. mi. and surrounding census blocks that have an overall density of at least 500 people/sq. mi. As with the education data, direct comparisons among data sets should be avoided; however, broad trends can still be useful for contrasting between the study area and populations sampled by the international literature.

Finally, the distribution of population according to both age and gender was compared among Makana LM and Ndlambe LM, Sweden, and the United States (Statistics Sweden 2007; StatsSA 2005; US Census Bureau 2007). Data expressing population by age were grouped according to the ranges reported by the Swedish data, which included the fewest age categories. Unlike the education and rural-urban occupancy data, data on the gender and age distribution of each location is directly comparable, bearing in mind that the most recent census in South Africa was completed in 2001.

To account for the effect of different land uses on available habitat within the catchment, this price was adjusted in a similar way to the method already described for discounting the annual production of bush meat in intact vs. transformed thicket. Since degraded semi-arid vegetation is estimated to contribute on average 29.3 % of total aboveground biomass, the contribution of resources provided by the commonage was assumed to represent 29.3 % of the total expressed WTP values (Jaramillo, *et al.* 2003; Mills, *et al.* 2005; Penzhorn, *et al.* 1974). It was assumed that the remaining value (70.7 % of the total WTP) is provided by resources protected within the intact vegetation on Waters Meeting NR.

The estimation of the value of standing stock for this exercise was complicated by the availability of only two studies that calculated respondents' WTP a lump sum for species conservation, rather than their annual WTP. Due to this data limitation, and the fact that the lump sum WTP estimates for wolf conservation (Chambers & Whitehead 2003) were less than half those elicited by annual response surveys (Boman & Bostedt 1999; Ericsson, *et al.* 2008), it was decided not to attempt the valuation of standing stock for species conservation. As such, only the estimated value of annual contributions for conservation is included as a contribution to the total economic value of the study site.

4.3.6.2 Results

The available literature on individual species valuation shows a wide variation in respondents' WTP that represents at most roughly one half of one percent (0.53 %) of their annual household income (see Table 4.10 – Table 4.13). Estimates of WTP per household range from just R 101 (US$ 12.28); 0.02 % of average annual household income) reported by New Mexico households for squawfish conservation (Cummings, *et al.* 1994) to over R 1,718 (US$ 208; 0.46 % of average annual household income) for wolf conservation pledged by residents living outside of the six-county "carnivore area" in Sweden (Ericsson, *et al.* 2008). On average, respondents reported a household WTP of roughly R 681 ± 479 (US$ 82.49 ± 57.99; 0.25 % ± 0.16 of average annual household income) to protect individual species (Table 4.13).

Disaggregating these WTP estimates according to the species being valued shows that respondents were significantly less willing to pay to conserve fish species (R 595 ± 398 or US$ 72 ± 48.15/household/year; 0.26 % ± 0.16 of mean annual income) than for large terrestrial carnivore conservation: households sampled in Sweden reportedly would contribute on average R 1,479 ± 249 or US$ 179 ± 30.13 annually for local wolf conservation, representing 0.44 % ± 0.02 of their annual disposable household income (Table 4.6). Based on five surveys soliciting WTP for birds and marine mammals, it appears respondents were least generous to these typically "charismatic" animals: mean WTP per household was just R 46 ± 333 or US$ 55.81 ± 40.28 a year, roughly 0.13 % ± 0.04 of average annual household income (Table 4.5).

As shown in the tables below, transferring these reported values for individual species conservation to the context of the Kowie River catchment using mean WTP per household as a percentage of average annual household income results in notable potential sums of money for the conservation of leopard and Eastern Cape Rocky in the area. Average annual household incomes (2008 US$) in the international literature ranges from R 106,545 (US$

12,904) in Taiwan to over R 803,405 (US$ 97,300) in Orange County, California. This affluence sharply contrasts with the average annual household income in Makana (R 3,744 or US$ 453) and Ndlambe (R 3,514 or US$ 426) local municipalities (LM). As a result, it would be expected that households in the catchment would contribute roughly R 9 (US$ 1) per year towards individual species conservation. Nonetheless, aggregated across all households in the catchment, this would amount to roughly R 317,090 (US$ 38,404) *per year* for local species conservation, a considerable sum despite relatively low household incomes compared to, for example, the United States, Sweden, and Taiwan (see Table 4.15).

Considering only the four studies that estimated 15 different values for various fish species, Kowie households would be expected contribute only R 9.38 (US$ 1.14) on average, or roughly 0.26 % of their annual household incomes, towards the conservation of the Eastern Cape Rocky (Table 4.10). However, in aggregate this seemingly insignificant household WTP would total about R 227,289 (US$ 27,528) annually to ensure the continued existence of the Rocky in local rivers. Based on five estimates of WTP for bird and marine mammal conservation reported by three studies, it appears that households in the Kowie River catchment would only be WTP an estimated R 4.74 (US$ 0.57) annually (0.13 % income) to protect these charismatic animals, or roughly R 162,616 (US$ 19,695) annually for the catchment as a whole (Table 4.5). In contrast, local households would be expected to annually pay as much as R 16.13 (US$ 1.95), on average, for leopard conservation (0.44 % of annual household income) based on three surveys of Swedish citizens' WTP for wolf conservation (Boman & Bostedt 1999; Ericsson, *et al.* 2008). In total, this would provide R 552,915 (US$ 66,965) for leopard conservation annually (Table 4.6).

Only two recent studies measuring three different values of once-off (lump sum) WTP for a single species are reported in international literature (Table 4.14.). Estimates of once-off WTP range from R 281 (US$ 34) pledged by Maine residents to protect the Peregrine falcon to roughly R476 (US$ 57) promised by Minnesota residents for gray wolf conservation. On average, respondents were WTP R 411 $\pm$ 113 (US$ 49.75 $\pm$ 13.64) as a one-time contribution for individual species conservation, or roughly 0.10 % $\pm$ 0.03 of their annual income. Based on these three reported values of once-off WTP, it would be expected that the average household in Makana would contribute R 3.78 (US$ 0.46) as a one-time payment for species conservation, while Ndlambe households would be expected to pay just R 3.55 (US$ 0.43). Still, again these modest contributions sum to an estimated value of R 125,642 (US$ 15,217) across all households.

Table 4.10 Household annual willingness to pay for fish species (Constant 2008 Rand, US$)

Authors	Survey yr	Species	WTP (annual)		Survey region	Avg annual income		WTP as % income	
			Rand	US$		Rand	US$		
Bell, *et al.* 2000[a]	2000	Salmon	1,206	146.08	Grays Harbor, WA H	256,711	31,091	0.47	
			797	96.47	Grays Harbor, WA L	256,711	31,091	0.31	
			1,229	148.86	Willapa Bay, WA H	232,271	28,131	0.53	
			789	95.51	Willapa Bay, WA L	232,271	28,131	0.34	
			504	61.10	Coos Bay, OR H	247,149	29,933	0.20	
			415	50.26	Coos Bay, OR L	247,149	29,933	0.17	
			800	96.93	Tillamook Bay, OR H	253,589	30,713	0.32	
			247	29.91	Tillamook Bay, OR L	253,589	30,713	0.10	
			1,166	141.20	Yaquina Bay, OR H	250,130	30,294	0.47	
			764	92.56	Yaquina Bay, OR L	250,130	30,294	0.31	
Cummings, *et al.* 1994	1994	Squawfish	101	12.28	New Mexico	307,333	37,222	0.02	
Stanley 2005	2001	Riverside fairy shrimp	247	29.90	Orange Co., CA	803,405	97,303	0.03	
Tseng & Chen 2008	2006	Taiwan trout	141	17.09	Taiwan	106,545	12,904	0.13	
			224	27.10	Taiwan	106,545	12,904	0.21	
			292	35.40	Taiwan	106,545	12,904	0.27	
Average HH WTP	**Unit**	**Species**	**Avg**	**Std Dev**		**Avg**	**Std Dev**	**Avg**	**Std Dev**
	US$	Fish	72.04	48.15		31,571	19,698	0.26	0.16
	Rand	Fish	595	398		260,672	162,642		
Overall WTP		**Species**	**Rand**	**US$**		**Rand**	**US$**		
Makana LM	Estimated	EC Rocky	9.68	1.17	Makana households	3,744	453		
Ndlambe LM	Estimated	EC Rocky	9.08	1.10	Ndlambe households	3,514	426		
Kowie Catchment	**Total Est**	**EC Rocky**	**321,415**	**38,928**	**All households**	-			

[a]Bell, *et al.* (2000) characterize "high income" as > US$ 30,000 and "low income" as < US$ 30,000.

Table 4.5 Household annual willingness to pay for bird and marine mammal species (Constant 2008 Rand, US$)

Authors	Survey yr	Species	WTP (annual)		Survey region	Average annual income		WTP as % income	
			Rand	US$		Rand	US$		
Loomis & Ekstrand 1997	-	Mexican spotted owl	453	54.82	US households (hh)	293,048	35,492	0.15	
Loureiro & Ojea 2008[b]	2005	Guillemot	203	24.58	Spanish households, uninformed	146,659	17,762	0.14	
			216	26.16	Spanish households, informed	139,688	16,918	0.15	
Giraud, *et al.* 2002	2000	Stellar sea lion	412	49.85	AK statewide hh	770,959	93,373	0.05	
		Stellar sea lion	1,021	123.63	US households	667,577	80,852	0.15	
Average HH WTP	**Unit**	**Species**	**Avg**	**Std Dev**		**Avg**	**Std Dev**	**Avg**	**Std Dev**
	US$	Birds & marine mammals	55.81	40.28		48,880	35,955	0.13	0.04
	Rand	Birds & marine mammals	461	333		403,586	296,870		
Overall WTP		**Species**	**Rand**	**US$**		**Rand**	**US$**		
Makana LM	Estimated	Birds & marine mammals	4.90	0.59	Makana households	3,744	453.45		
Ndlambe LM	Estimated	Birds & marine mammals	4.59	0.56	Ndlambe households	3,514	425.55		
Kowie Catchment	**Total Est**	**Birds & marine mammals**	**162,616**	**19,695**	**All households**	**-**			

[b]Loureiro & Ojea (2008) compared WTP between respondents living in the same coastal areas near the site of a recent oil spill that damaged the local guillemot population according to whether or not they were informed about other guillemot populations in Northern Europe.

Table 4.6 Household annual willingness to pay for large land carnivore species (Constant 2008 Rand, US$)

Authors	Survey yr	Species	WTP (annual)		Survey region	Average annual income		WTP as % income	
			Rand	US$		Rand	US$		
Boman & Bostedt 1999	1993-1994	Wolf	1,222	147.97	Swedish households	287,017	34,761	0.43	
Ericsson, *et al.* 2008[c]	2004	Wolf	1,498	181.39	Sweden: carnivore reg.	331,344	40,130	0.45	
		Wolf	1,718	208.11	Rest of Sweden	376,783	45,633	0.46	
Average HH WTP	**Unit**	**Species**	**Average**	**Std Dev**		**Avg**	**Std Dev**	**Avg**	**Std Dev**
	US$	Large land carnivores	179.16	30.13		40,175	5,436	0.44	0.02
	Rand	Large land carnivores	1,479	249		331,714	44,884		
Overall WTP		**Species**	**Rand**	**US$**		**Rand**	**US$**		
Makana LM	Estimated	Leopard	16.65	2.02	Makana households	3,744	453.45		
Ndlambe LM	Estimated	Leopard	15.62	1.89	Ndlambe households	3,514	425.55		
Kowie Catchment	**Total Est**	**Leopard**	**552,915**	**66,965**	**All households**	**-**			

[c]Ericsson, *et al.* (2008) surveyed two different populations: households living in the six provinces with recorded populations of all four large carnivores in Sweden (wolf, bear, lynx, and wolverine), known as the "carnivore area", and households living throughout the rest of Sweden.

Table 4.13 Household annual willingness to pay for any species (Constant 2008 Rand, US$)

	Unit	Species	WTP (annual)		Survey region	Avg annual income		WTP as % income	
Average HH WTP			**Avg**	**Std Dev**		**Avg**	**Std Dev**	**Avg**	**Std Dev**
	US$	Any species	82.49	57.99		36,456	23,192	0.25	0.16
	Rand	Any species	681	479		301,006	191,494		
Overall WTP		**Species**	**Rand**	**US$**		**Rand**	**US$**		
Makana LM	Estimated	Any species	9.55	1.16	Makana households	3,744	453.45		
Ndlambe LM	Estimated	Any species	8.96	1.08	Ndlambe households	3,514	425.55		
Kowie Catchment	**Total Est**	**Any species**	**317,090**	**38,404**	**All households**	**-**			

Finally, Table 4.8 disaggregates these values according to the relative biomass estimated to have been produced by the (degraded) commonage and (intact) Waters Meeting NR, respectively. Thus, commonage resources annually contribute roughly R 94,127 (US$ 11,400) to the overall catchment value of R 321,415 (US$ 38,928) for fish conservation; R 47,622 (US$ 5,768) of the R 162,616 (US$ 19,695) estimated for bird and marine mammal conservation; R 161,921 (US$ 19,611) of the large land carnivore total of R 552,915 (US$ 66,965); and R 92,860 (US$ 11,247) towards the R 317,090 (US$ 38,404) expected for conservation of a non-specified species in the catchment.

In contrast, the natural resources protected within Waters Meeting NR would be expected to annually contribute R 227,289 (US$ 27,528) towards fish conservation, R 114,994 (US$ 13,927) for bird and marine mammal conservation, R 390,994 (US$ 47,355) for large land carnivore conservation, and R 224,230 (US$ 27,157) for the conservation of a non-specified species. Using lump sum WTP estimates, the commonage and Waters Meeting NR would contribute R 36,794 (US$ 4,456) and R 88,848 (US$ 10,761), respectively, of the R 125,642 (US$ 15,217) estimated for catchment-wide species conservation.

Adult education levels[10] for regions sampled internationally are also markedly higher than those prevalent throughout the study area (Table 4.9). Whereas nearly 12 % and 19 % of adults in Makana LM and Ndlambe LM, respectively, have completed no formal education, this figure is less than 0.5 % for the United States (StatsSA 2005; US Census Bureau 2007). Statistics Sweden (2007) doesn't even list this category in their English-language tables available online. The highest proportion of adults in both Makana LM (57 %) and Ndlambe LM (49 %) have completed between grades 7 and 11 (i.e. Standards 5 – 9), whereas this level accounts for roughly 28 % of Swedish adults and just 9 % of American adults.

[10] NB: There are a number of differences in the categorization of education levels across the countries cited in Table 3 as noted in the footnotes; as such, direct comparisons among data should be made with caution.

Table 4.7 Household lump sum willingness to pay for individual species (Constant 2008 Rand, US$)

Authors	Survey date	Species	WTP (lump)		Survey region	Average annual income		WTP as % income	
			Rand	US$		Rand	US$		
Chambers & Whitehead 2003	2001	Gray wolf	477	57.81	Ely, MN residents	396,147	47,979		0.12
		Gray wolf	474	57.44	St. Cloud, MN res.	393,869	47,703		0.12
Kotchen & Reiling 2000	1997	Peregrine falcon	281	34.00	ME residents	451,577	54,692		0.06
Average HH WTP	**Unit**	**Species**	**Avg**	**Std Dev**		**Avg**	**Std Dev**	**Avg**	**Std Dev**
	US$	Any species	49.75	13.64		50,124	3,958	0.10	0.03
	Rand	Any species	411	113		413,864	32,680		
Overall WTP		**Species**	**Rand**	**US$**		**Rand**	**US$**		
Makana LM	Estimated	Any species	3.78	0.46	Makana households	3,744	453.45		
Ndlambe LM	Estimated	Any species	3.55	0.43	Ndlambe households	3,514	425.55		
Kowie catchment	**Total Est**	**Any species**	**125,642**	**15,217**	**All households**		**-**		

Table 4.8 Contribution of the commonage and Waters Meeting NR to total value (Constant 2008 Rand, US$)

Species	Catchment		Commonage		Waters Meeting NR	
Annual WTP	**Rand**	**US$**	**Rand**	**US$**	**Rand**	**US$**
Fish	321,415	38,928	94,127	11,400	227,289	27,528
Birds & marine mammals	162,616	19,695	47,622	5,768	114,994	13,927
Large land carnivores	552,915	66,965	161,921	19,611	390,994	47,355
Any species	317,090	38,404	92,860	11,247	224,230	27,157
Lump sum WTP	**Rand**	**US$**	**Rand**	**US$**	**Rand**	**US$**
Any species	125,642	15,217	36,794	4,456	88,848	10,761

Sources: Author's own calculations based on sources cited in Table 4.12 – Table 4.14.

Although there are some inconsistencies in the way Sweden reports its primary and secondary education data (see footnotes to Table 4.9), tertiary education figures for all three countries provide perhaps the most robust comparison between adults in the study area and those in the US and Sweden. The best-represented group in both the US (58 %) and Sweden (36 %) are adults with some tertiary (i.e. post-secondary) education, including two- or four-year associate or bachelors degrees, respectively, as well as masters and doctoral degrees. In contrast, only 3 % of adults in Ndlambe LM have completed any tertiary education, with a slightly higher proportion (5.4 %) of adults in Makana LM, the home of Rhodes University.

Table 4.17 shows that the proportion of people living in rural settlements in the study area (nearly 48 %) is significantly higher than in the US, where only 21 % of citizens were classified as "rural" in the 2000 Census (Ndlambe IDP 2007; US Census Bureau 2009). Similarly, Swedish population figures for the six counties classified as "rural" carnivore areas (Broberg & Brannland 2008; Ericsson, *et al.* 2007) show that they represent roughly 16 % of the national population (Statistics Sweden 2007). Again, due to different definitions of "rural" and "urban," direct comparisons should be avoided; still, the magnitude of differences between the study area and those areas sampled in international WTP literature is noteworthy.

In general, the South African residents in the study area tend to be more rural, less educated, and significantly less wealthy than the respondents sampled by the international literature. As will be discussed further in the next section, it appears that all of these demographic characteristics may play a role in determining respondents' WTP (Bostedt, et al. 2008; Broberg & Brannland 2008; Jorgensen, et al. 2001).

Table 4.9 Adult education levels (% population with education level) across study regions

Location	Census year	Age category	No school	Up to grade 6[a]	Grade 7-11[b]	Secondary graduate[c]	Tertiary education	Population
Makana	2001	20 yrs+	11.9	21.2	56.6	5.4	4.9	46,123
Ndlambe	2001	20 yrs+	18.5	24.9	48.7	4.9	3.0	34,498
US	2007	25-64 yr	0.3	2.4	9.4	30.3	57.6	158,284,000
Sweden	2007	25-64 yr	-	15.8	28.3	18.4	36.0	4,838,227

Sources: Statistics Sweden 2007; StatsSA 2005; US Census Bureau 2007

[a]Sweden groups all persons with no more than 9 years of education together; direct comparisons between the data for this education level in Sweden and other places should not be made.

[b]For the US and SA, this category includes persons who began 12th grade but did not receive a diploma/certificate; for Sweden it includes persons who have completed 3+ years of secondary school.

[c]Sweden does not separate persons who have completed 3+ years of secondary school from graduates.

Table 4.17 Population by rural or urban residence

	Rural		Urban		Total
	Population	**% total**	**Population**	**% total**	**Population**
Ndlambe LM	27,238	47.6%	30,003	52.4%	57,241
United States	59,061,367	21.0%	222,360,539	79.0%	222,360,539
Sweden	1,429,755	15.6%	7,753,172	84.4%	9,182,927

Sources: Ndlambe IDP 2007; Statistics Sweden 2007; US Census Bureau 2009

NB: See the Methods section (4.3.6.1) for a discussion on various definitions of "rural" versus "urban".

4.3.6.3 Discussion

As discussed already, the transfer of ecosystem service values from one context to another is an inherently imperfect process because of the many contextual details specific to the original study site (Loomis 1992; Troy & Wilson 2006). Due to the chosen method of transferring willingness to pay as a percentage of annual household income, the most obvious difference between the original study sites and the Kowie River catchment is the huge disparity in average household wealth. At US$ 453.45 and US$ 425.55 for Makana LM and Ndlambe LM, respectively, annual average household incomes in the study area are just over 1 % of those reported in international literature (overall average of US$ 41,880.60). However, applying the mean % WTP (as a proportion of income) across all households, results in estimates of catchment-wide support for conservation of US$ 29,389 for the Eastern Cape Rocky to US$ 66,965 for the leopard.

It is worth noting that the method used to estimate average income from South African census data, which report the number of households per income group, may be an underestimate of average household wealth because there is no upper-bound for the highest income category that starts at R 2,457,601, or roughly US$ 309,130 (StatsSA 2005). However, the top four income brackets (households earning over R 307,201 or roughly US$ 38,605 annually) represent just 1.5 % and 1.4 % of the total sampled population in Makana and Ndlambe, respectively. In contrast, between 18 – 19 % of all households in Makana and Ndlambe reported earning no (0) annual income. Since the midpoint for this range is also zero, the estimated average annual income for the study area used in this benefit transfer exercise is more likely to be significantly overestimated.

Furthermore, as discussed in previous sections, the disaggregation of values contributed by the commonage and Waters Meeting NR according to estimated differences in biomass production is a significant oversimplification of the complexity inherent to these ecosystems.

Firstly, the relationship between biomass production and animal density, especially for fish, is unlikely to be straightforward due to the complex interactions between various ecosystem services (e.g. Costanza, *et al.* 1997). Secondly, this method does not account for the extent of suitable habitat for various species within each site; leopards, birds, and fish each have different habitat requirements.

Finally, community user surveys indicate that resource collection is concentrated near Nolukhanyo township such that vegetation further from the settlement is more 'intact' than that found within easy access of resource users (Fabricius, *et al.* 2006). Since this level of site specificity was difficult, if not impossible, to capture with a desktop study, it was decided to instead use an estimation of gross biomass production as a basis for comparing the relative contribution of the commonage and Waters Meeting NR. However, the resulting estimations should be taken as a first approximation due to these ecological and site-specific complexities.

Beyond the significant differences in income between the study area and international samples and oversimplification of site differences, the limitations on benefit transfer derived from contingent valuation (CV) are potentially even more significant because of the uncertainty inherent to this methodology (Mitchell & Carson 1989). CV attempts to measure the respondent's most likely action when presented with a set of hypothetical scenarios and valuation questions that typically require some understanding of "issues that are complex and unfamiliar to the respondent" (Boman 2009). Contingent valuation is therefore highly sensitive to the biases of individual respondents (Richardson & Loomis 2009). As such, when transferring benefits derived from CV surveys, it is as important to consider the way in which the data were collected as it is to consider contextual differences between sites.

Even a cursory review of international contingent valuation literature reveals a stunning array of survey methods: the questions can be communicated by mail, on the phone, or in person; the willingness to pay question may follow a number of formats, such as open-ended, multiple bounded, or dichotomous choice; and the prompt may ask respondents to estimate their value as a yearly or a once-off (lump sum) payment (Garrod & Willis 1999). Nonetheless, the mean WTP as a percentage of income was remarkably similar across the different studies, context and species, lending credence to the approach used to estimate WTP for the East Cape Rocky and leopard in this desktop study. The rationale behind each of these methods will be discussed in further detail in the next chapter with respect to the design chosen for the contingent valuation survey of residents of the Kowie catchment.

What is important to note here, however, is the complexity that these many variations introduce into transferring benefits valued using diverse methods. Therefore, the values estimated by benefit transfer for leopard and Eastern Cape Rocky in the Kowie River catchment provide, at best, a rough idea of how actual residents in the area might assess the worth of these local natural resources. Keeping all this in mind, there are, nevertheless, a number of interesting international trends in households WTP for conservation of individual species that are worth mentioning here, including the impact of a species' 'charisma' on respondents' willingness to pay for its continued survival; the extent and nature of respondents' personal interactions with the species in question; and the socio-economic profile of respondents (Bostedt, *et al.* 2008; Richardson & Loomis 2009; Wattage & Mardle 2008).

'Charismatic megafauna,' a term often applied to large, relatively well-known animals from which humans perceive to derive some utility, have consistently been shown to provoke higher WTP than their less popular neighbors based on regression analyses of CV surveys (Metrick & Weitzman 1996, 1998; Richardson & Loomis 2009). In fact, a recent meta-analysis of 31 CV studies with 67 WTP estimations completed by Richardson & Loomis (2009) concluded that charismatic species are valued 115 - 180 % higher than non-charismatic species. Since there is no comprehensive definition of what precisely makes an animal 'charismatic,' authors typically use proxy variables, such as taxonomic class for vertebrates, to determine the degree to which this distinction influences respondents' willingness to pay to preserve a charismatic species.

Regression analyses of factors that influence willingness to pay for species conservation in developed nations have demonstrated that, all other things equal, charismatic mammals, birds, and fish (although not classified as 'charismatic') tend to elicit a higher willingness to pay than uncharismatic amphibians and reptiles (Metrick & Weitzman 1996, 1998; Richardson & Loomis 2009). In fact, Metrick & Weitzman (1998) point out that even the United States Fish and Wildlife Service implicitly discriminates against uncharismatic species, reserving just 5 % of its recovery budget for these creatures. The trivial perceived benefits typically attributed to uncharismatic species compared with charismatic megafauna are arguably inappropriate, however, given the interdependency between members of both groups in delivering ecosystem goods and services (Brown & Shogren 1998). Thus, Crocker & Tshirhart (1992) contend that the former should be treated as "intermediate goods" in the production of better-loved species.

Based on the results of international meta-analyses, it would appear that respondents still value charismatic animals more highly than uncharismatic ones. Richardson & Loomis (2009) conducted a twenty-year review of both annual and lump sum WTP estimates for animals in the United States, and found reported WTP figures between 83 % and 221 % higher for so-called charismatic species than for the single reptile species reviewed, the sea turtle. Including the results from Sweden and Taiwan not reviewed by Richardson & Loomis (2009), the average annual household WTP for charismatic species was only 27 % higher than that for fish (not shown). Still, estimated WTP for leopard was 128 % higher than estimated WTP for the Rocky.

Of note, two of the studies representing between them thirteen of the fifteen reported WTP values for fish asked respondents to assess the worth of Pacific salmon (Bell, *et al.* 2000) and the Taiwan trout (Tseng & Chen 2008), both iconic fish that represent a sense of local pride. Because the Eastern Cape Rocky does not enjoy these distinctions, the estimated annual catchment-wide WTP of R 321,415 (US$ 38,928) could be an overestimate.

Furthermore, it is also important to consider the relationship between the respondents and the animal being valued (Boman & Bostedt 1999; Bostedt, *et al.* 2008; Richardson & Loomis 2009). Boman & Bostedt (1999) note that some animals, particularly large carnivores, could be considered environmental "goods" as well as "bads." For example, "when wolves kill livestock, their existence value comes into direct conflict with human interests" (Boman & Bostedt 1999). In fact, Petty, *et al*. (1992) demonstrated that "direct personal experience" with wolves and other large carnivores results in stronger but also more central attitudes toward their conservation.

Nonetheless, several recent studies on large carnivores in Sweden, including wolves, wolverines, bear, and lynx, have demonstrated the negative impact of proximity to these animals on respondents' WTP by comparing values reported by respondents in areas with known populations of these animals to those from the rest of Sweden (Bostedt, *et al.* 2008; Broberg & Brannlund 2008; Ericsson, *et al.* 2007, 2008). Both Bostedt, *et al.* (2008) and Ericsson, *et al.* (2007) concluded that, after controlling for other influences on respondents' willingness to pay, such as education and income, urban populations had a higher WTP than the "carnivore areas", where respondents are more likely to be engaged in ranching or farming and could potentially be negatively affected by the presence of wolves.

While a similar study of respondents' WTP to increase wolf populations in northern Minnesota showed noticeable differences in environmental attitudes between residents living in wolf territory and urban dwellers, these biases did not result in a significantly different WTP between the two groups (Chambers & Whitehead 2003). Overall, Richardson & Loomis (2009) found in their survey of CV research that "visitors" (i.e. people living outside the primary habitat of the animal being assessed) were on average willing to pay 250 % of the value reported by local households, confirming earlier work by the authors (Loomis & Larson 1994; Loomis & White 1996). In light of this, it is expected that the limited direct exposure of relatively more urban residents in Grahamstown and Port Alfred to wild leopard, for instance, may result in higher value assessments than those offered by more rural Bathurst residents whose livelihoods are more likely to come into conflict with leopard conservation.

In addition to the effect of potential threats to rural livelihoods on WTP, it has been shown that hunters and other "consumptive" users of wildlife typically report less support for conservation activities than those who are not consumers (Kellert 1985; Kellert, *et al.* 1996). In general, Ericsson & Heberlein (2003) demonstrated that older males living in rural areas were more likely to participate in consumptive activities, such as hunting. In fact, the results of numerous CV surveys have found that, all other things equal, male respondents have lower WTP than females and older respondents tend to be less WTP than younger ones (Bostedt, *et al.* 2008; Broberg & Brannland 2008; Jorgensen, *et al.* 2001).

Other socio-economic characteristics, such as education and income level, also tend to influence respondents' WTP. As would be expected, several contingent valuation surveys have concluded that WTP increases with increasing income levels (Bostedt, *et al.* 2008; Broberg & Brannland 2008; Ericsson, *et al.* 2007; Ericsson & Heberlein 2003; Jorgensen, *et al.* 2001; Stanley 2005; Williams, *et al.* 2002). Interestingly, however, in their survey of Swedish residents, Bostedt, *et al.* (2008) found that education level had the strongest impact on respondents' WTP to protect large carnivores, followed by rural or urban residency.

Both income and education levels in the study area are well below the average for populations sampled by international WTP literature (i.e. US and Sweden), while the proportion of households residing in rural areas is considerably higher. In the international literature, urban, well-educated, high-earners have reported the highest willingness to pay for carnivore conservation in Sweden (Bostedt, *et al.* 2008; Broberg & Brannland 2008). It is also worth noting that several studies have found a high inter-correlation among these variables (Broberg

& Brannland 2008; Ericsson, *et al.* 2007). Thus, it is possible that as respondents' education levels increase, their potential to earn a higher income rises, and they are therefore more likely to move out of rural areas and into urban centers with a higher concentration of well-paid jobs (Broberg & Brannland 2008).

Nonetheless, a closer comparison of the rural-urban residency and income figures reported by the international literature challenges these hypotheses. In the two studies that set out to directly compare rural to urban annual WTP (Chambers & Whitehead 2003; Ericsson, *et al.* 2008), the difference in WTP as percentage income between the two population groups is less than 0.01 %. Furthermore, surveys by Bell, *et al.* (2000) also complicate the hypothesis that rural, less wealthy, less-educated populations have lower WTP than urban, wealthier, and better-educated populations. They gauged WTP for salmon conservation among residents of five counties in Oregon and Washington (USA) that are predominantly rural, lower income, and less well-educated than the rest of the counties in each state. Contrary to expectations, residents of the five counties surveyed by Bell, *et al.* (2000) were WTP 25.7 % more than the average respondent surveyed by all studies sampled in the international literature review; this despite the fact that annual incomes in the counties were 17.6 % lower than the overall average of respondents in the sampled studies.

Thus, the effect of income on respondents' WTP is not always straightforward (Hanemann 1984; Wattage & Mardle 2008). In another example, a regression analysis of a recent survey of residents' WTP for wetland conservation in Sri Lanka found that income did *not* significantly influence WTP (Wattage & Mardle 2008). The authors hypothesize that the non-significance of the income variable could be the result of wide variation in reported incomes (Wattage & Mardle 2008). In light of this inconclusive evidence, it was decided not to discount WTP based on the high proportion of rural residents and low incomes that characterize the study area.

Lastly, the WTP estimations for leopard and the Eastern Cape Rocky are based on the results of a very limited number of studies. Only two CV surveys completed in the past fifteen years report respondents' annual WTP to protect an individual large carnivore, the wolf (Boman & Bostedt 1999; Ericsson, *et al.* 2008). Although a total of 15 reported values of annual WTP to protect were used to estimate WTP for the Eastern Cape Rocky, these data come from only four studies on individual fish species. This is mainly the result of the very limited number of CV surveys conducted on individual species in the past fifteen years. Nonetheless, estimations drawn from such a small sample are likely to have high standards of error due to the significant variation among the studies.

In conclusion, there are a number of limitations to the transfer of individual species values estimated by international CV literature from their original socio-economic contexts to the study area. While the method of comparing respondents' willingness to pay to their annual household income accounts for one of the single most defining characteristics of different study sites, estimating WTP using the very low average annual household income of potential respondents in the Kowie River catchment results in an almost insignificant annual household WTP estimate (R 9.25 or US$ 1.12 per household per year). However, when aggregated across all households in the catchment, even this modest contribution per household could well represent a potentially viable funding stream.

4.4 Conclusions

A comparison of the contributions of annual production from each individual ecosystem service value (ESV) to total economic value on the study site reveals the importance of both livestock and wild animals, valuable for both their consumption as bush meat (direct use) and conservation (indirect/non-use) values (Table 4.18 – Table 4.20). Livestock account for fully 63 % of total value on the commonage and contribute more than twice as much value as the next most valuable ESV, bush meat. Livestock also represent the highest individual value on the study site as a whole (45 % of total value), even without any value from Waters Meeting NR. Since the cultural and spiritual values of livestock were excluded from this valuation exercise, the actual contribution of livestock to overall value may, in fact, be even higher.

Tables 4.18 – 4.20 suggest that, besides livestock, there may be opportunities for additional income generation from the study site through increased exploitation of several other direct ESVs that are currently only marginal income sources for Nolukhanyo households (Davenport 2008a). Bush meat represent the second highest value on the study site (30 % of total value) and is the largest single source of value on Waters Meeting NR, contributing over half (52 %) of the estimated total value of the reserve. While honey production is relatively more important (#3 rank, only 8 % of total value) on the commonage, leopard and Eastern Cape Rocky conservation are ranked the 2nd and 3rd highest values on Waters Meeting NR, representing 37 % of total value.

Table 4.18 Contribution of ESVs to economic value on the commonage (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Livestock	2,610,373	316,150	873	106	11,411,223	1,382,048	3,818	462
2	Bush meat	900,804	109,099	301	37	4,468,367	541,178	1,495	181
3	Honey	316,493	38,331	106	13	598,366	72,470	200	24
4	Leopard[1]	161,921	19,611	54	7	-	-	-	-
5	EC Rocky[1]	94,127	11,400	31	4	-	-	-	-
6	Fuel wood	31,423	3,806	11	1	1,848,422	223,868	618	75
7	Medicinal plants	4,310	582	1	0	49,667	6,706	17	2
	TOTAL	**4,115,141**	**498,979**	**1,378**	**167**	**18,326,378**	**2,226,270**	**6,148**	**745**

[1]As mentioned above, the value of standing stock for species conservation was not attempted due to the limited number of case studies that elicited respondents' WTP as a lump sum, rather than an annual, value.

Table 4.10 Contribution of ESVs to economic value on Waters Meeting NR (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Bush meat	858,086	103,925	594	72	4,242,744	513,852	2,936	356
2	Leopard[1]	390,994	47,355	271	33	-	-	-	-
3	EC Rocky[1]	227,289	27,528	157	19	-	-	-	-
4	Honey	150,256	18,198	104	13	284,076	34,405	197	24
5	Fuel wood	27,778	3,364	19	2	1,634,011	197,900	1,131	137
6	Medicinal plants	2,349	317	2	0	27,068	3,655	19	3
7	Livestock[2]	0	0	0	0	0	0	0	0
	TOTAL	**1,654,403**	**200,687**	**1,147**	**139**	**6,160,831**	**749,812**	**4,282**	**519**

[2]The value of livestock production on Waters Meeting NR was not estimated due to land use restrictions.

Table 4.20 Contribution of ESVs to economic value on the study site (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Livestock	2,610,373	316,150	589	71	11,411,223	1,382,048	2,574	312
2	Bush meat	1,758,890	213,025	397	48	8,711,111	1,055,029	1,965	238
3	Leopard[1]	552,915	66,965	125	15	-	-	-	-
4	Honey	466,749	56,529	105	13	882,442	106,875	199	24
5	EC Rocky[1]	321,415	38,928	72	9	-	-	-	-
6	Fuel wood	59,201	7,170	13	2	3,482,434	421,768	785	95
7	Medicinal plants	6,659	899	2	0	76,735	10,361	17	2
	TOTAL	**5,776,203**	**699,666**	**1,303**	**158**	**24,563,944**	**2,976,082**	**5,540**	**671**

On the other hand, fuel wood represents less than one percent (0.8 %) of value generated by the measured ESVs on the commonage and only 1.7 % of total value on Waters Meeting NR. This contrasts sharply with Davenport's (2008) household use surveys, which found that fuel wood (R 1,706 per commonage-user household) was the second largest source of on-commonage income per household after livestock (R 1,836). However, his work reflects realized value, whereas the transfer approach adopted in this chapter reflects potential value.

As such, there may be considerable scope for expansion of income from underexploited ESVs on the commonage, such as bush meat and honey, subject to the production costs incurred in accessing these services and actual natural resource stocks on the site. The significance of production costs should not be underestimated; commercial bee keeping in South Africa typically requires production costs of R 11 – 15/kg (constant 2008 Rand), which would erode even the modest revenues earned by small-scale honey producers as documented by Timmermans (2005). Moreover, the lack of systematic population data for any wildlife species on the study site means that any project promoting direct resource collection should proceed with caution and ideally only after such data are available to reduce the risk of overexploitation and unsustainable resource collection that outstrips annual production.

In addition to direct ESVs, leopard and Eastern Cape Rocky conservation represent a significant potential source of value on the study site. Although relatively less important on the commonage compared with direct ESVs, conservation accounts for fully 37 % of the total value of ESVs estimated for Waters Meeting NR, or 15 % of total value for the study site as a whole. While it is beyond the focus of this thesis to suggest specific conservation projects for the study site, this value transfer exercise suggests that, if mobilized appropriately, local support for wildlife conservation could contribute significant funding for a PES project on the study site to conserve the endangered Eastern Cape Rocky and/or leopard.

It should nevertheless be kept in mind that these values were transferred from sites with different social, economic, and ecological contexts than the study site. This precludes any evaluation of sustainable use rates and gives, at best, an indication of the relative contribution of different ESVs to the total economic value of the site. To improve the accuracy of key non-extractive ESVs (WTP and honey production) and estimate the contribution of indirect services to total economic value, the next chapter will detail the methodology and results of ESV estimations based on data collected on and around the study site. This exercise aims to capture a more focused picture of the indirect ESVs generated by the study site with a view toward identifying opportunities for future PES projects that provide fiscal incentives for promoting non-extractive and potentially economically and environmentally sustainable land uses without compromising local livelihoods.

5 Chapter Five

Ecosystem Service Field Valuation

5.1 Introduction

In light of the demonstrated importance of commonage resources in providing direct income generation opportunities to support local livelihoods (Anderson & Pienaar 2003; Andrew, *et al.* 2003; Cartwright, *et al.* 2002; Davenport 2008a), it is unsurprising that conservation accounted for just 6 % of the total economic value generated by the ecosystem services valued through the benefit transfer exercise. Despite the recognition that commonage land also generates indirect services, such as through recreation and sacred resources (Anderson & Pienaar 2003; Cartwright, *et al.* 2002; Ingle 2006), the valuation of these cultural and other indirect benefits typically depends on the application of alternative costing methods, such as contingent valuation, that are considered more susceptible to error than the market valuation of direct goods and services (Edwards & Abivardi 1998). Although commonage management, to the extent that it exists, has historically focused on the provision of these direct ecosystem service values (ESVs), especially livestock production, wood for fuel and building, and, in the case of the Bathurst commonage, cultivation (Andrew, *et al.* 2003; Fabricius, *et al.* 2006; Millennium Assessment 2005), it is expected that indirect and non-extractive (e.g. honey production) services, which do not deplete natural capital stocks, could potentially generate significant economic values that may at least partially offset those derived from the direct collection of commonage resources.

As previously discussed, the payments for ecosystem services (PES) approach to land use management rewards land use practices that result in quantifiable improvements in ecosystem health and thus increase the production of ecosystem services, be they direct, such as increased woody biomass production for fuel wood, or indirect, for example, improvements in watershed health that enhance aquifer replenishment (Kosoy, et al. 2008; Pagiola et al. 2002; Pagiola & Platais 2007). In fact, a number of countries have experimented with pro-poor payments for watershed services (PWS), including Costa Rica, Ecuador, Bolivia, India, South Africa, Mexico, and the United States (Forest Trends, *et al.* 2008). Pro-poor PES projects must be carefully designed and implemented to avoid potential risks including, in particular, misunderstandings of the terms of the contract and its implications for long-term land use options among local landholders, asymmetric market information that may unfairly advantage well-informed 'buyers' of environmental services over local community 'sellers,' and the "loss of control and flexibility over local development options and directions." Nevertheless, evidence from existing pro-poor PWS projects suggests that they can minimize trade-offs between poverty reduction and watershed services goals while transferring wealth, "often from wealthier urban areas to poorer rural areas," and empowering local land users as "valued service deliverers" (Agarwal & Ferraro 2007; Asquith, *et al.* 2007; Bruijnzeel & van Noordwijk 2007). Moreover, a recent primer on implementing PES projects co-authored by Forest Trends, The Katoomba Group, and the United Nations Environment Program (2008) notes that pro-poor PES projects can generate short-term benefits for local land users, including augmented cash income, expanded experience with external markets, and enhanced knowledge of sustainable resource use practices; as well as wider long-term advantages, such as improved ecosystem resilience and higher land productivity.

While the commonage will likely remain a critical source of direct income generation opportunities for Nolukhanyo residents for at least the near future, it is hypothesized that a well-designed PES project on the study site could generate additional revenues for local development by compensating local users for reducing natural resource extraction rates to ensure the long-term sustainability of natural capital stocks (Turpie, *et al.* 2008). For instance, fuel wood users who collectively agree to reduce their harvesting rate could be rewarded with money earned from the sale of carbon credits accruing from avoided deforestation (e.g. Bellassen & Gitz 2008). At the same time, reducing wood off-take would also increase the likelihood that future generations will be able to rely on commonage resources to support their basic fuel needs. Alternatively, there may be potential for encouraging lower-impact land uses or at least more carefully regulated natural resource collection on the commonage through voluntary conservation payments from Kowie River catchment residents. It is also expected that local revenues from ecotourism on Waters Meeting NR could be significantly expanded and shared with local residents who forego direct natural resource collection in the protected area.

To assess the potential revenues from PES projects on the study site, it was necessary to value selected indirect and non-extractive ecosystem services that would be expected to generate noteworthy revenues for local land users without depleting the natural capital stock. The following sections detail (i) the selection of ecosystem services for field valuation, (ii) the methodology used to value each service using a variety of field-based data sources and valuation techniques, and (iii) the total economic value derived from these services on the study site with a view toward identifying likely income generation opportunities through PES projects.

5.2 Selection of services for field valuation

The following ecosystem services valued in Chapter Four were selected for field valuation on the basis of (i) expected magnitude of the potential revenue stream based on the benefit transfer exercise and (ii) the availability of sufficient site-specific data on current and/or potential utilization to estimate annual and standing stock values: honey production and willingness to pay to protect endangered species. Although the value of medicinal plants was explored through fieldwork, insufficient data in terms of both plant density and species present were available for comparing values across different land use zones. In addition, two indirect services were valued through the field exercise: avoided deforestation and ecotourism. Thus, unlike the benefit transfer exercise, which focused primarily on direct service values, the field valuation exercise was designed to capture indirect benefits derived from the natural resources on the study site and refine the values estimated in Chapter Four. In so doing, this chapter aims to identify sustainable revenue streams that generate annual benefits without depleting the underlying capital stock.

As discussed in Chapter Four, it is recognized that, on the one hand, sacred resources contribute significant value to human societies. At the same time, they are endogenous to a "specific social environment" and are typically not directly substitutable, which precludes a meaningful valuation through the travel cost method (Adamowicz, *et al.* 1998; Bernard, pers. comm. 2008). Moreover, interviews with traditional healers (*amasangoma*) in Nolukhanyo revealed that only a handful of families access the sacred pool at Penny's Hoek on the commonage in any given year (Mbatha, pers. comm. 2008; Mbumba, pers. comm. 2008). As such, and in light of the sacred pool's small area proportional to the commonage area, this resource was excluded from the field valuation exercise.

5.3 Field valuation of ecosystem services

The following sections detail the data collection, transformation, and analysis; results; and discussion for each of the ecosystem services valued through field research. Where relevant, values for both the standing stock and annual production of the resource were estimated. As with the previous chapter, this exercise will estimate gross revenues (vs. net of costs). However, unlike the benefit transfer exercise, which substituted differences in the standing stock and annual production of ESVs based on available literature, the field valuation exercise makes use of ecological data collected on the study site and therefore should improve the accuracy of values estimated in the previous chapter. Nonetheless, the methods used in the field valuation exercise also rely much more heavily than the previous chapter on indirect valuation methods, including the estimated value of hypothetical changes in wood collection behavior and hypothetical contributions for the conservation of endangered species native to the study area. The estimates produced through these methods are therefore necessarily limited by the theoretical nature of the scenarios proposed (Allen & Loomis 2006; Loureiro & Ojea 2008; Martin-Lopez, *et al.* 2007). Further research will thus be required to refine these estimates based on actual changes in ESVs resulting from alternative natural resource management schemes on the study site.

5.3.1 Honey production

Although no commercial honey production currently takes place on the study site, the experience of commercial beekeepers in the Kowie River catchment suggests that honey collection could be an economically and ecologically sustainable livelihood on the study site. Building on their collective experience, this section will reevaluate the potential value of honey production on the study site based on actual production and revenue figures reported by three prominent commercial beekeepers working in the local thicket biome. Like other NTFPs, honey collection potentially represents an alternative livelihood strategy that can augment rural incomes while also serving as an incentive for woodland conservation (Dovie, *et al.* 2002; Kaushal & Melkani 2005; Shackleton & Shackleton 2004, 2005; Shahabuddin & Prassad 2004). However, there is some indication that improper harvesting methods, such as fires caused by excessive use of smoke to deter bees from attacking, and over-harvesting of wild honey can lead to ecosystem degradation and resource depletion (Bhattacharaya, *et al.* 2002; Ganesan 2003; Rai & Uhl 2004; Timmermans 2005). As such, it will be important for beekeeping promotion projects to include thorough training on proper harvesting methods, ideally combined with some ecological education that emphasizes the role of beekeeping in providing honey and other services, such as pollination.

5.3.1.1 Methods

In contrast to the honey valuation described in Chapter Four, which relied primarily on secondary production and revenue data reported from outside the thicket biome in the Eastern Cape and Kwa-Zulu Natal (Timmermans 2005), this valuation of honey production will depend predominantly on figures reported by local beekeepers working in the thicket biome. Seven prominent beekeepers in the Eastern Cape were initially contacted to establish the following details of their activities: (i) average number of hives per hectare; (ii) average honey yield per hive, as well as the range over time; (iii) average revenue earned from honey production; (iv) the top five most important nectar species in the thicket biome; and (v) the identity of one or more species whose nectar is available during bottlenecks when other sources are not available. Based on initial phone interviews, it was determined that only three beekeepers (G. Cambray, Friderichs, and Moxham) kept bees in the thicket biome; each of these gentlemen was interviewed in person to characterize their activities and experience of honey production in thicket.

Based on the number of hives maintained by each beekeeper and total annual income derived from honey, two of these beekeepers could be considered commercial producers, while the third produces honey primarily as a hobby. Since the aim of this exercise is to quantify maximum revenue potential from the study site, this analysis relies primarily on data from the two commercial beekeepers while keeping in mind that new beekeepers who are less well capitalized will likely have lower earning potential (at least initially) than the experienced beekeepers interviewed through this survey.

Although this valuation was originally envisaged to include revenue received from pollination services, none of the three beekeepers surveyed were actually receiving payments for these services. Mr. Friderichs (pers. comm. 2008) currently keeps half of his hives on citrus farms in the Amatola region; however, he actually pays the farmers in kind according to the number of cages located on each farm. Moreover, production of the major bee-pollinated crops (apples, grapes, citrus, peaches, pears, and plums) totalled just 3,442 tons in the Eastern Cape in 2002, compared with 4.25 million tons in the Western Cape (Shabalala & Mosima 2002). Thus, the market for pollination services in the Eastern Cape is unlikely to be as vibrant as that of the Western Cape, where an estimated R 11.5 million (US$ 1.8 million) was paid for hive rentals in 2005 (Allsopp, *et al.* 2008). As such, and in keeping with the benefit transfer exercise, this field valuation exercise focuses on potential revenue derived from honey production alone.

To ensure a conservative estimate of the honey production value derived from thicket, a number of assumptions were included in the benefit calculations. Firstly, the carrying capacity of hives per hectare was averaged across both good and bad production years to account for the substantial fluctuations in annual honey production, which is highly dependent on the appropriate timing of temperature and rainfall. Secondly, since beekeepers sometimes move their hives according to the forage available in a given area by season (and bees migrate within their territory to nectar sources as they become available), honey yield per hive was discounted according to the proportional time per year the bees were estimated to spend foraging in thicket vegetation.

Finally, the relative density of *Scutia myrtina* was used to discount the carrying capacity of Waters Meeting NR and the high use zone against the low use zone, which contains the highest density of *S. myrtina*. As reported in Chapter Three, no *S. myrtina* stems were recorded in the high use zone of the commonage, compared with 0.143 stems/m^2 in the low use zone and 0.074 stems/m^2 on Waters Meeting NR[1]. Thus, the hive carrying capacity of Waters Meeting NR was discounted by 48 % compared with the low use zone, which was assumed to support the average number of hives per hectare reported by the surveyed thicket beekeepers. The revenue potential of specialty (e.g. organic), retail (e.g. supermarket), and small-scale (e.g. market stalls, local restaurants) honey sales were calculated separately using the assumptions outlined below in Table 5.1. For the purposes of estimating the total economic value of the study site, the value of small-scale sales will be used to approximate the revenue potential of the study site. Nonetheless, the revenue potential of retail and or specialty (e.g. organic) honey sales should not be ignored in development planning in the area.

Table 5.1 Assumptions used to estimate revenues from honey production

Season	Carrying capacity (ha/hive)	Yield (kg/hive)	Revenue (R/kg)
Good	1	30	50 (specialty)
Standard	1.5	20	24 (retail)
Poor	2	6	20 (small-scale)

Sources: Based on personal communication with G. Cambray, Friderichs, & Moxham (2008)

In keeping with the methodology used in the benefit transfer exercise, the total value of the 'standing stock' of honey production on the commonage was assumed to be the value of hives that could be supported by the study site. However, this new estimation of standing stock makes two revisions to the previous methodology based on the experience of local beekeepers: (i) standing stock value includes the costs of cages and netting around apiaries that house 30 hives each, and (ii) all hive and apiary construction is done by the beekeepers themselves, rather than purchased commercially, to minimize costs and take advantage of

[1] NB: The difference between the low use zone and Waters Meeting NR was not significant at 5 % level.

local labor. As in the benefit transfer exercise, the value of hive (apiary) depreciation was excluded from the estimation of standing stock value to harmonize results from the other gross benefit calculations.

5.3.1.2 Results

The carrying capacity of the high use zone (1,338 ha) was assumed to be zero due to the lack of *S. myrtina* recorded there. In contrast, the low use zone was estimated to support 1,101 hives within its 1,652 ha, and roughly 462 hives were estimated for Waters Meeting NR (1,445 ha). The potential value of standing stock of apiaries (30 hives plus cage and netting = R 3,500) that could be supported was thus estimated to be R 128,489 (US$ 15,562) in the low use zone and R 53,947 (US$ 6,534) on Waters Meeting NR, for a total of R 182,436 (US$ 22,095) on the study site.

For specialty sales (R 50/kg), annual production in the low use zone of the commonage was estimated to be R 573,511 ± 265,023 (US$ 69,472 ± 32,098) and on Waters Meeting NR this figure would be R 240,833 ± 111,271 (US$ 29,168 ± 13,476), for a total of R 814,444 ± 376,294 (US$ 98,640 ± 45,574) derived from the study site as a whole (Table 5.2). For retail sales (R 24/kg), annual production values were estimated to be R 275,333 ± 107,700 (US$ 33,346 ± 13,044) and R 115,600 ± 45,218 (US$ 14,001 ± 5,477) for the low use zone and Waters Meeting NR, respectively, for a total of R 390,933 ± 152,919 (US$ 47,347 ± 18,520) derived from the study site as a whole (Table 5.3). For small-scale sales (R 20/kg), annual production was estimated to be worth R 229,444 ± 85,756 (US$ 27,789 ± 10,386) in the low use zone and R 96,333 ± 36,005 (US$ 11,667 ± 4,361) on Waters Meeting NR, for a total of R 325,778 ± 121,761 (US$ 39,456 ± 14,747) on the entire study site (Table 5.4).

Table 5.2 Estimated revenues from *specialty* honey sales (Constant 2008 Rand, US$)

Land use zone	Hives	Annual production (R)		Annual production (US$)		Standing stock	
	#	Value	Std Dev*	Value	Std Dev*	Rand	US$
Low Use	1,101	573,611	265,023	69,472	32,098	128,489	15,562
Waters Meeting	462	240,833	111,271	29,168	13,476	53,947	6,534
Total	**1,564**	**814,444**	**376,294**	**98,640**	**45,574**	**182,436**	**22,095**

Sources: Based on personal communication with G. Cambray, Friderichs, & Moxham (2008)

*Standard deviation of annual production reflects the variation in annual honey yield due to climate.

Table 5.3 Estimated revenues from *retail* honey sales (Constant 2008 Rand, US$)

Land use zone	Hives	Annual production (R)		Annual production (US$)		Standing stock	
	#	Value	Std Dev*	Value	Std Dev*	Rand	US$
Low Use	1,101	275,333	107,700	33,346	13,044	128,489	15,562
Waters Meeting	462	115,600	45,218	14,001	5,477	53,947	6,534
Total	**1,564**	**244,007**	**130,259**	**77,994**	**15,776**	**182,436**	**22,095**

Sources: Based on personal communication with G. Cambray, Friderichs, & Moxham (2008)

*Standard deviation of annual production reflects the variation in annual honey yield due to climate.

Table 5.4 Estimated revenues from *small-scale* honey sales (Constant 2008 Rand, US$)

Land use zone	Hives	Annual production (R)		Annual production (US$)		Standing stock	
	#	Value	Std Dev*	Value	Std Dev*	Rand	US$
Low Use	1,101	229,444	85,756	27,789	10,386	128,489	15,562
Waters Meeting	462	96,333	36,005	11,667	4,361	53,947	6,534
Total	**1,564**	**325,778**	**121,761**	**39,456**	**14,747**	**182,436**	**22,095**

Sources: Based on personal communication with G. Cambray, Friderichs, & Moxham (2008)

*Standard deviation of annual production reflects the variation in annual honey yield due to climate.

5.3.1.3 Discussion

Given that no *Scutia myrtina* stems were recorded in the high use zone of the commonage, which represents nearly 45 % of the total commonage area, it is interesting that the new estimated value of honey production on the commonage (assuming small-scale sales) is only 27.5 % lower than the value reported in Chapter Four. Due to the lower (but not significantly so) proportion of *S. myrtina* stems in Waters Meeting NR compared with the low use zone on the commonage, the new estimate of honey value on Waters Meeting NR also dropped from the Chapter Four value, by nearly 36 %. Overall, the field-based estimate of honey production for small-scale sales on the study site decreased by roughly 30 % from the value estimated through benefit transfer.

Of course, these estimates reflect the numerous assumptions used in their calculation. In particular, based on the careful honey collection records maintained by Mr. Moxham (pers. comm. 2008), it is clear that honey yields vary dramatically by site and year. While the vegetation on the study site is primarily thicket, a number of other vegetation types can be found both within the study site and on nearby land parcels, including riverine forests, grasslands, and even pineapples, which are a common crop in the Bathurst area. While it would be premature to make sweeping conclusions from such a small data set (Mr. Moxham has recorded honey collection data for nine years at three different sites), it is worth noting that the site where bees have access to pineapples in addition to inland bush and riverine vegetation produced roughly 66 % more honey over the 9-year period than a site characterized by gum trees and indigenous bush and 52 % more than the site covered by semi-coastal scrub (Moxham, pers. comm. 2008). Mr. Friderichs (pers. comm. 2008) also pointed out the gap in honey production between his hives kept on inland citrus farms compared with those located along the coast, which produce 50 % less honey due to high winds.

In general, honey production is dependent on the availability of sufficient nectar sources for the bees at all times of year. Corroborating the observations of Galpin (2007), two of the beekeepers surveyed cited *Scutia myrtina*, also known locally as cat-thorn (English), droogie (Afrikaans), and isiPingo (isiXhosa), as the single most important nectar source in the local thicket biome (G. Cambray, pers. comm. 2008; Friderichs, pers. comm. 2008). In fact, Mr. Friderichs (pers. comm. 2008) maintains that *S. myrtina* produces the best honey in South Africa. For this reason, *S. myrtina* density was measured in the field within each of the three land use zones. Unfortunately, no significant differences were found between the low use zone and Waters Meeting NR; though no *S. myrtina* stems were recorded in the high use zone, suggesting that there may be limited honey production potential in this zone.

As shown in Table 5.5, other important sources of nectar in the thicket biome include *Pappea capensis*, *Schotia afra, Euphorbia* species*, Portucularia afra,* and *Acacia karroo*; minor nectar sources include *Ptaeroxylon obliquum*, *Ziziphus mucronata*, *Rhigozum trichotomum*, and *Euclea undulata* (G. Cambray, pers. comm. 2008; Galpin 2007). It should be noted, however, that honey produced from *Euphorbia* nectar leaves a burning taste when consumed; for this reason honey collection trays can be removed when these trees are in bloom (July – August) (G. Cambray, pers. comm. 2008; Galpin 2007). Non-thicket species include bottlebrush, blue gums, peppers, honeysuckle, and jacaranda (Moxham, pers. comm. 2008).

In addition to differences in vegetation, all beekeepers surveyed noted the significant influence of climate, and in particular well-timed and sufficient rainfall, on nectar production, which in turn affects honey yields (G. Cambray, pers. comm. 2008; Friderichs, pers. comm. 2008; Moxham, pers. comm. 2008). Thicket also sometimes produces poor honey flows when there is too much rain or misty weather (G. Cambray, pers. comm. 2008). According to the collection records kept by Mr. Moxham (pers. comm. 2008), the standard deviation in honey yields between 2001 and 2007 exceeded average production by over 28 % at one site and by nearly 14 % at a second site, while the standard deviation at his third apiary site was roughly equal to average honey yields over the period for which complete records were available (2001 – 2007). Similarly, Mr. Friderichs (pers. comm. 2008) reported that his honey yields have a range of 20 tonnes between the best and worst years, with yields in good years approximately 44 % higher than in average years, and yields in bad seasons about 67 % lower than average seasons.

Table 5.5 Important sources of nectar for honey production in thicket

Latin name	Common names (English/Afrikaans/Xhosa)	Importance; notes
Scutia myrtina	Cat-thorn (E), Droogie (A), isiPingo (X)	Critical
Schotia afra	Karoo Farmers Bean (E)	Medium
Portucularia afra	Spekboom (E)	Medium
Pappea capensis	Jacket plum (E)	Medium
Acacia karroo	Thorn tree (E)	Medium
Euphorbia triangularis	River Euphorbia (E), Riviernaboom (A), umHlonthlo (X)	Medium; honey leaves burning sensation
Ptaeroxylon obliquum	Sneezewood (E)	Minor
Ziziphus mucronata	Ziziphys (E)	Minor
Rhigozum trichotomum	Driedoring (A)	Minor
Euclea undulata	Guarri (E), Gewone Gwarrie (A), umGwar (X)	Minor
Rhus pallens	iNhlokolotshane (X)	Not clear nectar or pollen
Olea europea subspecies *africana*	Wild Olive (E), Olienhout (A), umNquma (X)	Not clear nectar or pollen
Diospyros dichrophylla & D. whyteana	Star Apple (E) family	
Sideroxylon inerme	White Milkwood (E), Witmelkhout (A), umQwashu (X)	Honey reported to have unsavoury taste

Sources: Based on pers. communication with G. Cambray, Friderichs, & Moxham (2008); Galpin (2007)

In light of the significant vegetation- and climate-dependent variations in honey yields, honey production should be promoted as one of multiple income-generating activities, rather than as a primary livelihood option, for local entrepreneurs. Nonetheless, assuming each entrepreneur maintains an average of seven hives (G. Cambray, pers. comm. 2008), the commonage could support an annual revenue of R 1,458 $\pm$ 545 (US$ 177 $\pm$ 66) for each of roughly 157 beekeepers, with an additional 66 supported by Waters Meeting NR. Moreover, with appropriate management and transportation, the Bathurst beekeepers could potentially earn additional revenue of about R100/hive/week for providing pollination services to farmers (G. Cambray, pers. comm. 2008). Given the significant cost and logistics involved with implementing this long-distance pollination service, it was decided to exclude pollination services from the revenue calculation. Nonetheless, it is worth noting that Western Cape fruit farmers paid an estimated total of US$ 1.8 million (R 11,465,367) for pollination services in 2005 (Allsopp, *et al.* 2008).

5.3.2 Willingness to pay to protect endangered species

The four most commonly applied contingent valuation (CV) methods are open-ended, payment card, dichotomous choice, and bidding game. Open-ended questions ask the respondent to indicate their maximum WTP for a specified good without any initial bid prompt; e.g. “How much would you be willing to pay for this good?”. The open-ended format avoids the error introduced by suggested bids, the so-called ‘framing effect’. However, the literature diverges on whether this format produces more or less conservative results (Heinzen & Bridges 2007; Whynes, et al. 2004). In contrast, the payment card format provides the respondent with a context for identifying their maximum WTP by selecting the value from a list of presented prices. Still, research suggests that the range selected can have a significant effect on the results (Heinzen & Bridges 2007). The payment card format is thus subject to ‘range bias’ introduced by the fact that the survey must present a maximum and minimum price set exogenously (Boyle, *et al.* 1996).

In its most basic form, the dichotomous choice format asks respondents to indicate their acceptance/rejection of a single proposed bid; e.g. “Would you pay $100 for this good?” Dichotomous choice questions have been widely used and accepted in the CV literature since the United States National Oceanic and Atmospheric Administration (NOAA) Panel recommendations (Arrow, *et al.* 1993). However, respondents tend to exhibit ‘yea saying’ in this valuation method, meaning that they tend to respond positively to a hypothetical scenario regardless of the information presented. Moreover, since respondents are not able to choose the bid value, respondents’ answers are subject to a ‘starting point bias’; in other words, respondents’ WTP is influenced by the magnitude of the (initial) bid presented (Heinzen & Bridges 2007). Increasing the number of bid choices offered to the respondent as additional binary (yes/no) questions has been shown to result in substantial improvements in the consistency of WTP estimates (Hanemann, *et al.* 1991; Langford, *et al.* 1996). However, research suggests that efficiency gains tend to decrease as the number of bids increases (Cooper & Hanemann 1995; Scarpa & Bateman 2000). In general, this method has been deemed more appropriate for estimating demand for a good, rather than maximum WTP (Heinzen & Bridges 2007).

As with dichotomous choice, the bidding game presents respondents with an initial price (“bid”) that they can either accept or reject. If the respondent accepts this initial bid, then they are presented with increasing prices until the respondent says “no,” indicating that they have reached their maximum WTP (Whynes, *et al.* 2004). Although this format can introduce some bias based on the value of the starting bid, respondents typically have more time to contemplate their final answer thanks to the repeated nature of the elicitation method, which increases the accuracy of the result. The bidding game is also subject to 'yea-saying' bias, but evidence suggests that this format is less prone to this bias than the dichotomous choice method (Heinzen & Bridges 2007). Given these several advantages, the bidding game format was selected to elicit respondents' WTP based on their responses to increasing bids for leopard and Eastern Cape Rocky conservation, respectively.

5.3.2.1 Data collection

A contingent valuation (CV) questionnaire designed to gauge respondents' willingness to pay to protect local leopard and Eastern Cape Rocky ('Rocky') and their habitat was administered by phone to approximately 130 residents of the three primary settlements located along the Kowie River catchment, namely Grahamstown (n = 50), Bathurst (n = 41), and Port Alfred (n = 41). Although it is recognized that flagship species are not necessarily good indicators of overall ecosystem health (Simberloff 1998), Smith & Sutton (2008) recently noted, "[environmental conservation] agencies increasingly use flagship species (those popular, relatively large, charismatic animals) as tools to trigger concern for the species and motivate community members to conserve the flagship species and its habitat". As such, despite its scarcity in the study area, the well-known and charismatic leopard was chosen for the WTP survey as an example of a potential flagship species for the catchment. The Rocky, a less well-known and less charismatic but nevertheless highly endangered endemic species, was selected to test whether WTP for ecosystem conservation is influenced by the level of 'charisma', popularity, or vulnerability of the flagship species selected (Home, *et al.* 2009).

Based on the widespread correlation between WTP and income, it was assumed that those households without a landline telephone would not have sufficient income to donate annually to a conservation fund; the survey was thus limited to residents with landline phones. Telephone numbers were selected using three random numbers generated by Microsoft Excel to identify each household according to the page number, column, and row in which the telephone number was located in the phone book. Telephonic surveys were carried out during all times of day (morning, afternoon, evening) during both week and weekend days to ensure an unbiased sample.

The questionnaire included four distinct sections: (i) demographics, (ii) interactions with local wildlife and landscapes, (iii) conservation opinions, and (iv) the CV questions for leopard and

Rocky, respectively, which were each presented separately and in the same order for all surveys. Wherever possible, questions were presented in a closed (i.e. fixed response) manner, and all conservation opinion questions used a Lickert scale to indicate the respondent's level of agreement with the statement. Data collected from each section are described below.

Demographic attributes collected included age, gender, primary language, highest education level, occupation, and years lived in the catchment area. The next set of questions solicited respondents' interactions with local wildlife and landscapes, including the frequency and nature of their visits to the Kowie River (if any), their most and least favorite aspects of the Kowie, and their personal experience of leopard and the Eastern Cape Rocky, including their knowledge of livestock being killed by leopard in the area. The third section prompted responses on a Lickert scale (1 = totally disagree; 2 = slightly disagree; 3 = neutral; 4 = slightly agree; 5 = totally agree) to several statements on the importance of local conservation efforts, the impact of changes in river quality on the respondent's use or enjoyment of the river, and their opinion of whether residents should contribute financially to the protection of the Kowie River overall and endangered wildlife in the area, in particular. The final section used a bidding game format to elicit respondents' WTP for each animal (leopard and Rocky) in turn following short narratives explaining the importance of these species to the local ecosystem and various threats to their survival in the catchment area (see Appendix 1 for the full survey instrument).

The narratives that follow were read aloud to each respondent preceding the CV questions:

Leopard:

> The paragraph below presents a **completely hypothetical** scenario for research purposes only. Please consider the questions that follow from the perspective of your **entire household**, keeping in mind the money you have to spend on all your household expenses.
>
> The leopard is the only major predator in the Kowie River valley, and it plays an important role in regulating natural food chains. Although leopards used to roam freely along the Kowie River, they have been eliminated from many areas due to a number of factors, including illegal hunting and incompatible land uses that have reduced its territory size. Suppose a fund was created to ensure the future survival of leopard in the Kowie River valley. Possible interventions might include monitoring the movement of leopard in the valley to alert farmers of their presence and/or increasing available habitat.

Rocky:

> Again, the paragraphs below present a **completely hypothetical** scenario for research purposes only. Please consider the questions that follow from the perspective of your **entire household**, keeping in mind the money you have to spend on all your household expenses, including food, electricity, transportation, and donations to other conservation-related causes.
>
> The Eastern Cape Rocky is a small fish that grows to about 30 cm long and only exists in tributaries of three rivers in the Eastern Cape, one of which is the Kowie River. According to a local expert, the Rocky may go extinct in the next ten years unless management actions are taken to protect it from local threats, such as loss of habitat, invasive alien fish like bass and catfish, and sedimentation in the Kowie. Overgrazing on the Bathurst commonage and poor crop management on farms adjacent to the river can lead to soil erosion and silt up the Kowie. Because the Rocky is so specially adapted to its habitat, it acts as an indicator of the health of the Kowie River as an ecosystem. Therefore, the sharp decline in Rocky numbers observed over the past thirty years in the Kowie River may be an indication of ecosystem decline.
>
> Suppose that an organization was formed to clean up the Kowie River and protect the Eastern Cape Rocky. Possible interventions might include encouraging farmers along the river to adopt erosion control practices and reduce their use of harmful chemicals and/or more careful regulation of cattle on the commonage.

To estimate their WTP, respondents were then asked whether or not they would be willing to contribute anything to the conservation of the (leopard, Rocky) on an annual basis, whether time, money, or in-kind donations. Those who indicated they would not be WTP were asked to explain why. Those who responded positively to the first CV question were next asked to elaborate upon each of their pledges by specifying hours and/or money that they would donate to the cause of reducing threats to these species' continued existence and enhancing the conservation of the river as a whole (for the Rocky). Monetary donations were proposed beginning with R 20 per year; respondents who indicated they would be willing to pay R 20 were then asked to respond to bids increasing in 20 Rand increments until the respondent indicated that he/she would not donate the proposed bid. Similarly, respondents were prompted to specify the time they would donate to either (a) administrative or other non-labor intensive conservation initiatives or (b) labor-intensive work, such as clearing the river of invasive fish, by responding yes/no to the following incremental bids: 2 hrs/yr; 2 hrs/mo.; 2 hrs/wk; > 2 hrs/wk. In-kind donations, such as work vehicles or fishing equipment, proved too abstract a concept for respondents and did not elicit any positive responses.

5.3.2.2 Data transformation and analysis

Estimated willingness to pay (WTP) for each of leopard and Rocky was averaged across all respondents to predict mean WTP for at least moderately wealthy households in the catchment (i.e. those that can afford a landline telephone). These WTP estimates were then multiplied by the total number of households with landline phones as estimated from the phone book entries to give the total value for leopard and Rocky conservation in the catchment, respectively. Finally, these total catchment values were discounted according to the proportional contribution of biomass measured within each land use zone on the study site. All values were converted to 2008 US$ using the annual average of the daily US$/Rand exchange rate (SARS 2008).

In order to compare WTP to respondent attributes, a number of variables were transformed. All occupation classes were converted to a scale of 1 – 5 as shown in Table 5.6 below based on Ward & Parker (2008), who used publically available average income figures by occupation class. Self-employed individuals, including farmers with at least 12 years of schooling, were categorized as medium income; those with less than 12 years were placed in the low income class. Education levels were translated into years of schooling according to the education system in South Africa or Zimbabwe where appropriate. For consistency, it was assumed that students who have matriculated completed 12 years of school; technical qualifications (e.g. diploma) 14 years; a bachelor's (BA) degree 12 + 3 = 15 years; Honours 16 years, Master's degree 18 years; doctoral degree 19 years. Finally, an index of "eco-friendliness" was calculated based on responses to 17 questions dealing with respondents' interactions with wildlife and their conservation opinions; unanswered questions (including "don't know") were assigned a value of zero. The index has a theoretical maximum of 75 and a minimum of 0.

Table 5.6 Income classification based on occupation

Class	Annual salary (R)		Occupation examples
	Minimum	Maximum	
None	0	0	No job/formal income
Other	0	0	Pensioner, student, housewife
Low	1	76,800	Domestic worker, self employed < 12 yrs education
Medium	76,801	307,200	Teacher, sales clerk, secretary, bookkeeper, technician, municipal employee, self employed ≥ 12 yrs education
High	307,201	n/a	Engineer, accountant, lawyer, professor, other specialized jobs

Source: Based on Ward & Parker (2008)

Since the objective of this exercise was to calculate the contribution of the commonage to catchment-wide value of leopard and Rocky conservation, responses from all towns were aggregated together. Significant relationships between WTP for leopard and/or Rocky and respondent attributes (town, occupancy, age, gender, education, frequency of visits to the Kowie River, encounters with Rocky/leopard) were first explored through a principal components analysis (PCA) to identify variables of interest for further testing. Correlations between these variables and leopard and/or Rocky WTP, as appropriate, were then tested in Stata 9.0 using either a one-way analysis of variance (Anova) for categorical variables or a multivariate linear regression for continuous variables. Where Anova results indicated significant correlation between WTP and one or more categorical variables, the influence of each category (of each variable) on WTP was tested using interaction expansion for a linear regression. Income and eco-friendliness were tested subsequent to PCA analysis using Anova and linear regression. Interactions between WTP for leopard and Rocky were also tested using linear regressions. Correlations between respondent attributes, including WTP for the other animal being valued, and WTP for the subset of respondents WTP > 0 for leopard or Rocky were explored separately from the entire sample (WTP > 0).

5.3.2.3 Results

As shown in Table 5.7, average household annual WTP for leopard conservation ranged from R 101 (US$ 12.28) in Grahamstown to R 164 (US$ 19.88) in Port Alfred, with a catchment average of R 128 (US$ 15.50). Household WTP for Rocky conservation ranged from R 105 (US$ 12.67) in Bathurst to R 197 (US$ 23.91) in Port Alfred, with a catchment average of R 144 (US$ 17.41).

Table 5.7 Annual household willingness to pay to protect endangered species $\geq$ 0 (Constant 2008 Rand, US$)

Location	Landline phones	Households surveyed	Phones surveyed	Household WTP for leopard		Household WTP for Eastern Cape Rocky	
	#	#	%	R	US$	R	US$
Grahamstown	4,480	50	1	101	12.28	197	23.91
Bathurst	420	41	10	124	15.04	105	12.67
Port Alfred	2,520	41	2	164	19.88	117	14.21
Catchment	7,420	132	2	128	15.50	144	17.41

As shown in Table 5.8, across all households WTP > 0 for leopard conservation (n = 56 or 42.4 % of all sampled), the median WTP = R 110 (US$ 13.32), mode = R 100 (US$ 12.11), and mean = R 302 (US$ 36.58); the mean was likely influenced by two respondents whose

WTP = R 2,600 (US$ 314.89) and R 2,000 (US 242.23), respectively. Excluding these outliers results in mean WTP = R 228 (US$ 27.61) and median WTP = mode WTP = R 100 (US$ 12.11). Across all households WTP > 0 for Rocky conservation (n = 64 or 48.5 % of all sampled), median WTP = mode WTP = R 100 (US$ 12.11) and mean WTP = R 296 (US$ 35.85); mean WTP was likely influenced by two outliers with WTP = R 5,000 (US$ 605.57) and R 2,000 (US$ 242.23), respectively. Excluding these outliers gives mean WTP = 193 (US$ 23.37) and median WTP = mode WTP = R 100 (US$ 12.11). In other words, on average respondents were WTP roughly 18 % more for leopard conservation than for Eastern Cape Rocky conservation, but approximately 14 % more respondents were WTP for Rocky conservation than for leopard conservation.

Table 5.8 Annual household willingness to pay to protect endangered species > 0 (Constant 2008 Rand, US$)

	Household WTP for leopard		**Household WTP for EC Rocky**	
WTP > 0	**Rand**	**US$**	**Rand**	**US$**
Mean	302	36.58	296	35.85
Median	110	13.32	100	12.11
Mode	100	12.11	100	12.11
WTP > 0 Excl. outliers	**Rand**	**US$**	**Rand**	**US$**
Mean	228	27.61	193	23.37
Median	100	12.11	100	12.11
Mode	100	12.11	100	12.11

Table 5.9 displays the estimated contribution of each land use zone to leopard and Rocky WTP, respectively, based on their proportional contribution to total biomass production on the study site. Estimated contributions to leopard and Rocky conservation, respectively, were R 207,380 (US$ 25,116) and R 275,872 (US$ 33,412) from the high use zone; R 475,954 (US$ 57,644) and R 633,150 (US$ 76,683) from the low use zone; and R 236,729 (US$ 28,671) and R 314,915 (US$ 38,140) from Waters Meeting NR.

Table 5.9 Annual willingness to pay to protect endangered species (Constant 2008 Rand, US$)

Land use zone	**Total WTP for leopard**		**Total WTP for EC Rocky**	
	Rand	**US$**	**Rand**	**US$**
High Use	207,380	25,116	275,872	33,412
Low Use	475,954	57,644	633,150	76,683
Waters Meeting	236,729	28,671	314,915	38,140
Catchment	920,062	111,432	1,223,937	148,235

The following figures display the results of principle component analysis on the relationship between WTP and various respondent attributes for leopard and Rocky, respectively. As shown in Figure 5.1, over 50 % of the variance in leopard WTP (including zero values) can be accounted for by the first two axes, which cover respondents' demographic attributes (age, gender, education, and language). Although leopard WTP is not positively related to any respondent attribute, there is a suggestion of a negative correlation to age.

To test these relationships, regressions were performed between leopard WTP and age alone, and age in combination with gender, occupancy, and whether or not they visit the Kowie River (as a proxy for conservation attitude). When tested against leopard WTP (including zero values) alone, age accounts for a very small proportion of the variance ($R^2 = 0.04$, df = 124) but has a significant negative effect on leopard WTP (coefficient = -4.27, $p = 0.017$). When tested against WTP in combination with gender, Kowie visits, and years lived in the catchment (occupancy), the model accounts for a still insignificant amount of the variance ($R^2 = 0.0465$) but age still has the largest (and only significant) effect of any of the variables tested (coefficient = -4.12, $p = 0.025$).

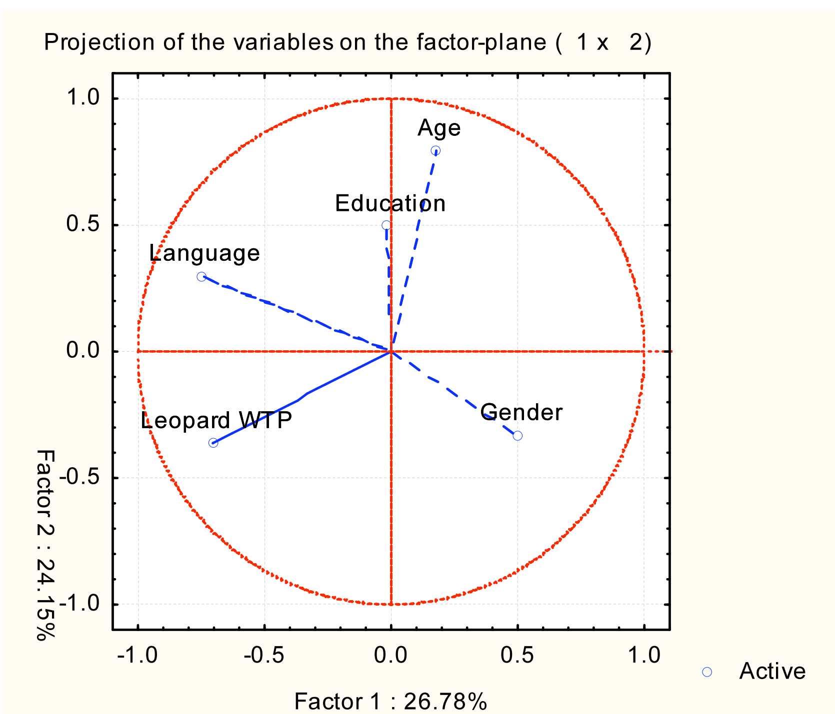

Figure 5.1 Leopard WTP (*including* zero values) and demographic attributes

Figure 5.2 shows the relationship between leopard WTP (excluding zero values, n = 56) and all these demographic attributes plus several proxies for conservation attitudes, including frequency of visits to the Kowie River and encounters with leopard in the catchment (seen in the wild, heard of them in the area, read about their local presence, etc.). While the degree of variance account for by the analysis is fairly low (roughly 35 %), WTP appears to be strongly positively related to language and knowledge/experience of leopard. A two-way Anova with language and encounters with local leopard found that language does have a significant correlation to leopard WTP (F = 4.31, Prob > F = 0.0090).

Contrary to expectations, experience of leopards in the catchment did not appear to have a significant correlation with WTP to protect them (F = 0.11, Prob > F = 0.98). However, these two variables explain for a rather small proportion of overall variance in leopard WTP (R^2 = 0.22). Using an expanded regression with each different language category, the "other" language category, i.e. those external to the study area, such as Zimbabwean English and Belgian, was found to have a strongly and significantly positive effect on leopard WTP (coefficient = 927.5, p = 0.001); though it accounts for a small proportion of variance (R^2 = 0.21). Subsequent Anova testing found that both English and Afrikaans are more significantly correlated with income (F = 8.55, Prob > F = 0.004 and F = 4.87, Prob > F = 0.03, respectively) compared with other languages (F = 3.15, Prob > F = 0.08).

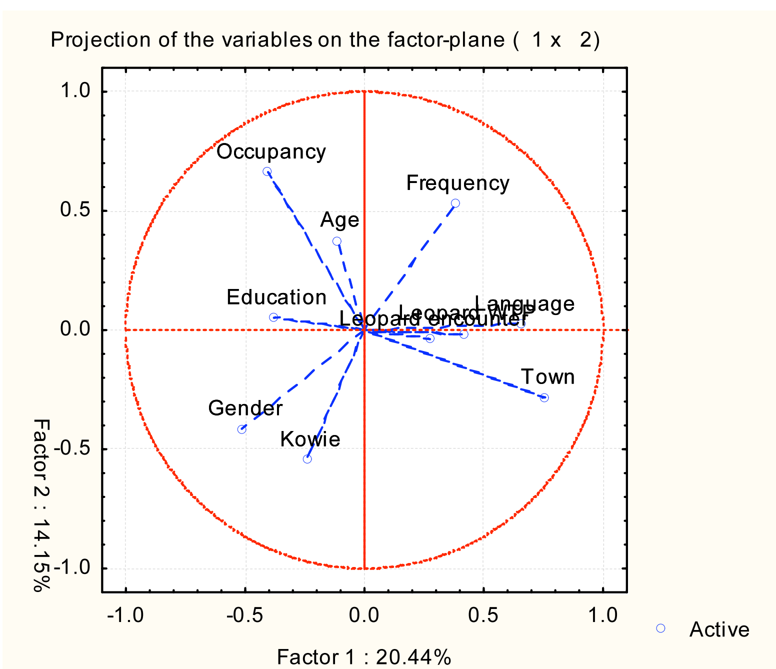

Figure 5.2 Leopard WTP (*excluding* zero values) and all respondent attributes

Figure 5.3 depicts the relationship between WTP for Eastern Cape Rocky conservation (including zero values) and all respondent attributes tested, including demographic variables and conservation orientation variables (presence and frequency of visits to the Kowie River, encounters with Rocky). Although the proportion of variance explained is low (roughly 32 %), WTP appears to be negatively correlated to age and probably experience of the Kowie River. A one-way Anova between WTP and Kowie River visits found no significant correlation (F = 0.53, Prob > F = 0.59). However, a linear regression with age found a significantly negative effect on Rocky WTP (coefficient = -8.03, p = 0.002); albeit age accounts for an insignificant amount of WTP variance alone (R^2 = 0.07).

Similarly, Figure 5.4 depicts relationships between Rocky WTP (excluding zero values, n = 52) and several accounts for a low amount of overall variance (roughly 26 %) but suggests a strongly negative association of WTP with age, and a weakly negative association with Rocky experience. A linear regression with both of these variables accounts for a small amount of variance (R2 = 0.16) but confirms a strongly negative effect of age on WTP (coefficient = -20.41, p = 0.006). No category of Rocky experience (e.g. seen in the wild, etc.) had a significant effect on Rocky WTP.

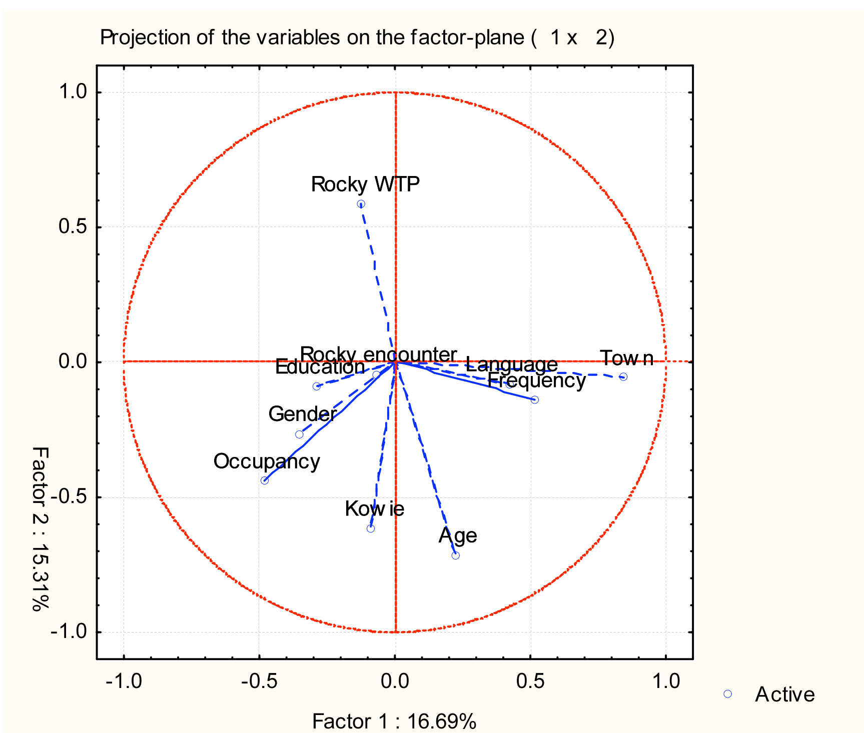

Figure 5.3 Eastern Cape Rocky WTP (*including* zero values) and all respondent attributes

An expanded linear regression was used to test for effects of Rocky WTP, education, income, gender, town, language, age, and eco-friendliness on leopard WTP (incl. zero values). While the proportion of variance explained is low ($R^2 = 0.28$), both Rocky WTP and eco-friendliness have significantly positive effects on leopard WTP (coefficient = 0.21, p = 0.000 and coefficient = 6.71, p = 0.029, respectively). Contrary to expectations, subsequent linear regressions on Rocky WTP (incl. zero values) using these same respondent attributes with leopard WTP did not yield significant results for either income or eco-friendliness. However, both leopard WTP and age show significant effects on Rocky WTP (coefficient = 0.83, p = 0.000 and coefficient = -11.73, p = 0.001, respectively). The proportion of variance explained is low ($R^2 = 0.33$).

An expanded linear regression on leopard WTP (including zero values) using demographic attributes and conservation profiles as measured by education, income, gender, language, town, occupancy, age, existence and frequency of Kowie River visits, encounters with leopard, and Rocky WTP (incl. zero values), revealed that, in addition to Rocky WTP, which had a slightly positive (and significant) effect on leopard WTP (coefficient = 0.22, p = 0.000), visits to the Kowie River at least two weekends per month has a significantly positive effect on WTP (coefficient = 291.52, p = 0.02 and coefficient = 175.65, p = 0.12, for visits 2 – 3 weekends per month and every weekend, respectively). These variables only account for a small proportion of WTP variance ($R^2 = 0.32$).

Likewise, an expanded linear regression of these same respondent attributes (substituting encounters with Rocky rather than leopard) on Rocky WTP (excluding zero values) produced a somewhat robust model ($R^2 = 0.63$) and suggested some surprising results. Education had a significant and negative effect on Rocky WTP (coefficient = -216.26, p = 0.044), as did more frequent visits to the Kowie River (coefficient = -2103, p = 0.01; coefficient = -1647, p = 0.032; coefficient = -2090, p = 0.042; and coefficient = -2339, p = 0.014 for less than four visits per year, roughly one visit every three months, 2 – 3 weekends per month, and every weekend, respectively). Subsequent category-expanded regression of WTP for Rocky conservation (excluding zeros) with frequency of visits, overnight visits, and reasons for visiting the Kowie River found that visits every weekend had a significant and positive effect on WTP (coefficient = 891, p = 0.003), while visiting the river for fishing had a significant and negative effect (coefficient = -887, p = 0.020). The overall proportion of variance was fairly robust ($R^2 = 0.665$), although the number of observations was low (n = 22) due to the omission of category 1 for each variable.

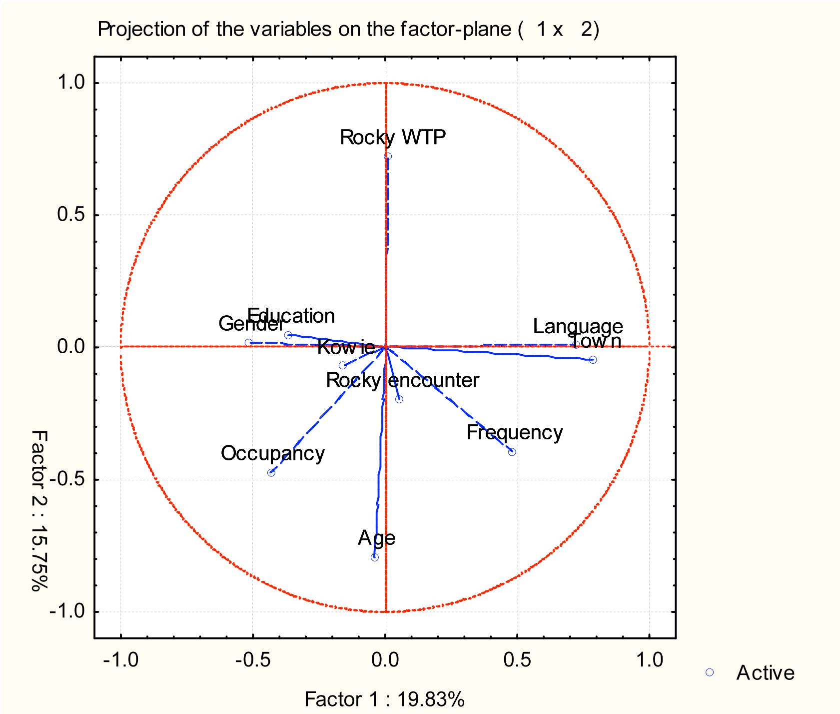

Figure 5.4 Eastern Cape Rocky WTP (*excluding* zero values) and all respondent attributes

Surprisingly, no significant correlations were found between income and either leopard or Rocky WTP (including or excluding zero values). These results are presented in Table 5.10.

Table 5.10 Results of Anova for income effect on WTP for leopard or Rocky conservation

Dependent variable	No. observations	R^2	F value	Probability > F
Leopard (incl. zero)	132	0.03	0.85	0.50
Rocky (incl. zero)	132	0.01	0.27	0.90
Leopard (excl. zero)	56	0.05	0.74	0.57
Rocky (excl. zero)	53	0.06	0.71	0.60

As would be expected, the best indicator of WTP for either Rocky or leopard seems to be WTP to protect the other species in question. A category-expanded linear regression of Rocky WTP as a function of leopard WTP (excluding zero values), education, income, gender, frequency of Kowie River visits, town, and language explained nearly all variance ($R^2 = 0.99$). Leopard WTP (excluding zero values) has a coefficient of 1.01 ($p \leq 0.0001$); no other variables are significant at $p = 0.05$. Reversing the WTP variables gives a coefficient of 0.98 ($p \leq 0.0001$, $R^2 = 0.99$) for Rocky WTP.

5.3.2.4 Discussion

Based on the international literature cited in Chapter Four, it was expected that several respondent attributes would influence WTP for the conservation of leopard and/or Eastern Cape Rocky, including income (Bostedt, *et al.* 2008; Broberg & Brannland 2008; Ericsson, *et al.* 2007; Ericsson & Heberlein 2003; Jorgensen, *et al.* 2001; Stanley 2005; Williams, *et al.* 2002), education (Bostedt, *et al.* 2008), conservation attitudes (Kotchen & Reiling 2000), the relationship between the respondent and the animal being valued (Bostedt, *et al.* 2008; Boman & Bostedt 1999; Richardson & Loomis 2009), and the 'charisma' of the animal being valued (Metrick & Weitzman 1996, 1998; Richardson & Loomis 2009). Although no significant income effects were detected for any of the WTP measures in this study (including and excluding zero values), several other variables did have a significant influence on either leopard or Rocky WTP, including conservation attitudes, education, and, interestingly, the age of the respondent. Variables found to influence each of the measures of WTP (including and excluding zero values) for leopard and Eastern Cape Rocky are explored below.

Variables with a significant and positive effect on overall (including zero values) leopard WTP include the frequency of visits to the Kowie River, eco-friendliness, and overall Rocky WTP, while age has a significant and negative effect. The significant effect of conservation attitudes, as measured by both frequency of visits and responses to numerous conservation opinion questions, on WTP for leopard conservation confirms the work of Kotchen & Reiling (2000), who found that pro-environmental attitudes result in higher estimates of WTP to protect endangered species. Contrary to the literature (Bostedt, *et al.* 2008; Broberg & Brannland 2008; Ericsson, *et al.* 2007; Ericsson & Heberlein 2003; Jorgensen, *et al.* 2001; Stanley 2005; Williams, *et al.* 2002), however, neither education nor income had a significant effect on overall WTP for leopard. As discussed in more detail below, the sampling bias introduced by the exclusion of households without landline telephones may have reduced the influence of socio-economic variables on WTP. Moreover, the method of estimating respondent income based on the occupation of the respondent alone while soliciting *household* WTP may have also obfuscated income effects.

Also contrary to expectations (e.g. Richardson & Loomis 2009), urban-rural residence as measured by town did not significantly affect overall WTP for leopard conservation. Based on research elsewhere that has demonstrated the negative impact of direct exposure to large carnivores on WTP to protect these potentially dangerous species (Boman & Bostedt 1999; Bostedt, *et al.* 2008), it was hypothesized that 'urban' respondents in Grahamstown and Port

Alfred would have higher WTP than ‘rural’ residents in Bathurst in correlation with relatively less direct potentially negative exposure to wild leopard in urban areas. However, the limited direct exposure of respondents across all towns may have confounded this bias. Only two respondents out of 132 reported having personally seen evidence of wild leopard, both of whom reside in ‘urban’ Port Alfred. Moreover, over 80 % of respondents in each town reported *not* having heard of leopard killing livestock in the area within the past five years. In fact, the only respondent who indicated that their livestock had been killed by leopard reported a WTP value more than 56 % higher than the mean to protect this predator. In light of the apparently minimal negative impact of leopard on livestock in the area, the urban-rural distinction appears less relevant to overall leopard WTP.

Narrowing the analysis to only those respondents with leopard WTP > 0 reveals that foreign nationality (e.g. Zimbabwean or European) has a significant and positive effect on WTP. This is an interesting result, since length of occupancy in the Kowie River catchment did not have a significant effect on WTP. Given the significant correlation between income and language, it is possible that foreigners wealthy enough to resettle in South Africa have more disposable income to support local wildlife conservation. However, this needs to be explored further in light of the stronger correlation between English and Afrikaans and income compared with ‘other’ languages.

In contrast, only overall leopard WTP has a significantly positive effect on overall Rocky WTP, while age has a significantly negative effect. Among those respondents who were WTP > 0 to protect the Eastern Cape Rocky, and contrary to expectations, both education level and the frequency of visits to the Kowie River actually had a significantly *negative* effect on WTP. This result is surprising in light of the international literature (Bostedt, *et al.* 2008; Kotchen & Reiling 2000). It may be the case that less educated residents were less able to understand the hypothetical nature of the CV question, especially in light of language differences. Exploring the relationship between visits to the Kowie River and WTP for Rocky conservation further, it appears that neither overnight visits nor number of reasons for visiting cited have a significant effect on WTP for Rocky conservation (among those respondents with WTP > 0 for Rocky conservation). However, fishing showed a significant and strongly negative effect on WTP for Rocky conservation. Given the notable proportion of variance ($R^2 = 0.665$) in Rocky WTP > 0 explained by the respondents’ frequency of visits, overnight visits, and different reasons for visiting, it is possible that the negative influence of fishermen's attitudes on Rocky WTP can explain the apparent inconsistency between frequency of visits and WTP for Rocky conservation.

None of the analyses showed a significant income effect on either leopard or Rocky WTP, including or excluding respondents whose WTP ≤ 0. By limiting the sample population to only those households with a landline telephone, the survey purposefully introduced income bias and therefore increased the likelihood that respondents would have WTP > 0. Nonetheless, it is rather surprising that relative income level had no effect on WTP, given the widely reported income bias (Bostedt, *et al.* 2008; Broberg & Brannland 2008; Ericsson, *et al.* 2007; Ericsson & Heberlein 2003; Jorgensen, *et al.* 2001; Stanley 2005; Williams, *et al.* 2002). However, recent research on WTP for conservation in another developing country (Sri Lanka) suggests that the relationship between respondents' income and WTP is not always simple, perhaps due to large variance in sampled incomes (Wattage & Mardle 2008).

Moreover, some detail was lost in the method of estimating income based on reported occupation of the respondent while soliciting *household* WTP. In particular, students, pensioners, and housewives were all classified as having no income, even though they are presumably supported by some source of income (e.g. parents, private or state pensions, or husbands, respectively). In lieu of an income effect, the strongly negative effect of age on WTP for both leopard and Rocky suggests that pensioners do, in fact, have lower WTP than younger, working-age respondents. This result is consistent with other CV surveys on WTP for conservation (Bostedt, *et al.* 2008; Broberg & Brannland 2008; Jorgensen, *et al.* 2001). Subsequent research should thus differentiate students, pensioners, and housewives by estimated "household" income, including funds from parents/husbands and/or pensions.

Finally, it is worth exploring the influence of the 'charisma' of the animal being valued on respondents' WTP for its conservation. A recent meta-analysis 31 CV surveys reporting the results of 67 separate WTP estimations found that respondents value charismatic mammals, birds, and fish (though not classified as 'charismatic') 115 - 180 % more highly than 'uncharismatic' animals, such as amphibians and reptiles (Richardson & Loomis 2009). However, as reported in the previous chapter, the inclusion of results for charismatic megafauna in Sweden (Bostedt, *et al.* 2008; Broberg & Brannland 2008) and Taiwan trout (Tseng & Chen 2008) not reported by Richardson & Loomis (2009), narrows the gap between charismatic animals and fish to 27 %.

Excluding two outliers from each group, respondents who were WTP to protect either Rocky or leopard (but not necessarily both) in the Kowie River catchment reported a WTP for leopard conservation of 18.1 % higher than for the Rocky, which is in line with the results reported by the international literature. However, including the outliers narrows the gap between leopard and Rocky to just 2 %. In fact, across all households, including those with WTP = 0 for one or both species, WTP for Rocky conservation was actually 12.3 % *higher* than for leopard conservation.

These results may seem counterintuitive at first glance, especially given the fact that less than 10 % of respondents reported having seen the Rocky either in a museum or the wild and only a further 17 % had ever heard of the rather inconspicuous fish at all. In fact, four respondents alluded specifically to the “personality” or “sex-[iness]” of leopard as compared to Rocky. However, it should be noted that the hypothetical scenarios presented for each animal differed in three important ways. Firstly, the leopard was presented as a potential threat to livestock, whereas the Rocky was described as particularly vulnerable to both human disturbances and invasive, predatory fish. Secondly, while the leopard is endangered locally, the highly concentrated range of the Eastern Cape Rocky, which is only known to exist in three rivers and their tributaries, increases the threat of the species’ extinction due to very localized impacts on the Kowie River. Thirdly, while both animals were presented as being integral components of the Kowie River catchment, the Rocky was described as being a potential indicator of overall river health; moreover, its long-term conservation was explicitly linked to the amelioration of the Kowie River ecosystem as a whole.

In light of these differences, it is therefore less surprising that 14.2 % more respondents were WTP for Rocky conservation than for leopard conservation. Of the 10 respondents who were WTP more for Eastern Cape Rocky conservation than for leopard conservation, respondents cited the heightened vulnerability due to the extremely limited range of the Rocky (n=2) or the dire consequences of Rocky extinction (n=3) as justification for increased conservation support for the Rocky. Overall (i.e. including respondents who pledged equal amounts for leopard and Rocky conservation or who pledged more for leopard conservation), seven respondents cited the distinction of the Rocky as an indicator of overall river health and/or the importance of supporting the conservation of the river as a whole in justifying their WTP for Rocky conservation. For comparison, among the 15 respondents who were WTP more for leopard conservation than for the Rocky, respondents tended to cite lack of knowledge of the fish (n=4) or the inevitability of extinction (n=2) as justification. When asked whether they would approve of using their leopard donation for projects that aim to improve overall river health, however, 4 of the 10 respondents who initially reported zero WTP for Rocky conservation responded that this would be an acceptable use of their donation.

Above all, the best predictor of WTP for either leopard or Rocky conservation was WTP for the other animal being valued. Fully 81 % of respondents reported WTP equal amounts for both leopard and Rocky conservation, reflected in the strong influence of WTP for leopard on WTP for Rocky conservation (and vice versa) as revealed through linear regressions. Future research should thus explore the possible 'yea-saying' bias of respondents who reported WTP > 0 for leopard by soliciting WTP for the two animals through either administering separate surveys for each animal, so that each respondent values only leopard or Rocky but not both, or by reversing the order in which the animals are presented in the survey. This could refine the results obtained for each individual species and potentially differentiate the two values more clearly.

Finally, by assuming that all respondents who indicated they were not WTP for either leopard or Rocky conservation have WTP = 0, the results reported here could overestimate average WTP across all residents of the Kowie River catchment, as some respondents may have negative WTP (Boman & Bostedt 1999). For example, farmers who lose livestock to leopard or who agree to use fewer chemicals on their crops may prefer to be paid for the trouble of coexisting with leopard or changing their production methods to reduce runoff into the catchment, respectively. Therefore, future surveys could improve the accuracy of this catchment-wide conservation valuation exercise by targeting different respondent groups according to their anticipated behavioral changes required to achieve enhanced conservation outcomes and solicit respondents' willingness to accept payment or donate funds for leopard or Rocky conservation as appropriate.

5.3.3 Avoided deforestation

Since the proclamation of the Kyoto Protocol in 1997, the significant contribution of deforestation to global greenhouse gas (GHG) emissions has gained increasing attention. It is estimated that tropical deforestation accounts for as much as 20 % of global carbon dioxide emissions, making it second only to fossil fuels in terms of global GHG emissions and the single largest source of GHG emissions in the developing world (Houghton 2005). Despite the inclusion of financial incentives to reduce global carbon dioxide emissions through tree planting (afforestation and reforestation, 'AR') as part of the Clean Development Mechansim

(CDM), the Kyoto Protocol excluded incentives to *prevent* deforestation. As noted by the Nobel Prize winning economist Joseph Stiglitz (2005), this significant omission effectively created perverse incentives for countries with large forested areas to reap a double benefit by harvesting the timber and then selling carbon credits for trees replanted in the formerly forested areas. To encourage countries to instead maintain their forests, a subsequent meeting of the Conference of Parties (COP13) held in Bali in 2007 considered options for creating a system of payments for Reduced Emissions from Deforestation and Forest Degradation (REDD) (UNFCCC 2007a).

The basic premise of REDD is that it rewards countries with financial incentives for preventing forest loss that would have otherwise occurred in the absence of such incentives, known as the 'business as usual' or BAU scenario (Estrada 2007; Federative Republic of Brazil 2007; Santilli, *et al.* 2005). Alternative financing proposals have been forwarded for discussion by the UN Framework Convention on Climate Change (UNFCCC), including the compensation fund proposed by Brazil (Federative Republic of Brazil 2007) and a compensated reduction (CR) scheme (Santilli, *et al.* 2005). However, the success of any compensation fund would presumably be limited by the size of the funding pool (Bellassen & Gitz 2008).

In contrast, the CR mechanism for implementing REDD relies on global carbon markets to provide the motivation for reducing deforestation by authorizing developing countries who have voluntarily reduced their emissions from deforestation below their historical baseline to sell carbon certificates to public and private investors after these reductions have been measured and certified (Santilli, *et al.* 2005). Although payments at a national scale have been proposed to avoid the problem of 'leakage' at the national level, whereby reduced deforestation in one area of the country shifts pressure to deforest other areas (Bouyer & Merckx 2007), the inclusion of small-scale AR projects in the CDM may leave room for incorporating small-scale REDD projects implemented by local communities into the post-Kyoto regime to be negotiated by the COP in Copenhagen in December 2009 (Coomes, *et al.* 2008).

In addition to rewarding reduced carbon emissions, REDD projects have the potential to create positive externalities through increased financing for biodiversity conservation and human development (Ebeling & Yasué 2008). In fact, small-scale AR projects were included in the CDM as a means to promote sustainable development through carbon trading (Aune, *et* *al.* 2005; Lipper & Cavatassi 2004; Smith & Scherr 2003). Despite the demonstrated ability of small-scale AR projects to reduce carbon emissions, however, evidence suggests that their ability to produce benefits beyond carbon sequestration may have been overestimated (Gundimeda 2004; Locatelli & Pedroni 2006; Minang, *et al.* 2007; Pfaff, *et al.* 2007). In fact, the feasibility assessment for a proposed AR scheme in a low-income indigenous community in eastern Panama suggests that the significant economic costs and risks inherent in the upfront investments necessary to accomplish tree planting and maintenance over the life of the AR contract (in this case, 25 years) can be prohibitive for low-income landholders (Coomes, *et al.* 2008).

Nonetheless, recent research on the costs and benefits of adopting REDD projects, even on a small scale, suggests that rewarding communities for avoiding deforestation can potentially offer sufficient financial incentives to make this land use option more attractive for poor rural communities than alternative land uses, such as cattle production or shifting cultivation, while simultaneously circumventing some of the challenges associated with implementing CDM-AR projects (Bellassen & Gitz 2008; Coomes, *et al.* 2008). In fact, the results of opportunity cost analyses in Cameroon (Bellassen & Gitz 2008) and Panama (Coomes, *et al.* 2008) suggest that the net present value (NPV) of avoided deforestation far outstrips annual incomes from shifting cultivation and cattle production, respectively.

Assuming a 5 % discount rate over the five-year period of a CR scheme in Cameroon, Bellassen & Gitz (2008) found that local farmers could earn nearly ten times as much income annually by reducing deforestation by just 5 % of the historical rate. They estimated a NPV per hectare income of 2008 US$ 23,437 ± 890 (R 169,273 ± 6,425) and a total annual NPV of 2008 US$ 143 million ± 5.5 million (R 1.2 billion ± 46 million) over the 6,100 ha (5 %) that would remain forested out of the 122,000 ha potentially lost under the BAU scenario. For comparison, the per hectare NPV generated by the methods of shifting cultivation practiced in Cameroon was estimated to be just US$ 2,448 ± 363 (R 20,216 ± 2,995), or roughly US$ 15 million ± 2.2 million (R 123 million ± 18 million) annually across the entire CR area of 6,100 ha.

Using the same 5 % discount rate, Coomes, *et al.* (2008) find that the average smallholder in the project area of Ipeti Emberá region of eastern Panama could earn a net per hectare benefit (NPV) of roughly US$ 13,985 (R 115,471) from the sale of carbon credits and teak harvested from a plantation at the end of a 25-year contract period for CDM-AR, while cattle production actually produces a net negative NPV of US$ -420 (R 3,468) per hectare. Compared with the revenue generated under the CDM-AR scheme, the avoided deforestation scheme produces relatively modest per hectare NPV net benefits of roughly US$ 776 (R 6,407). However, most of the benefits produced under the CDM-AR scheme derive from the sale of high value teak timber at the end of the project, rather than from carbon credits.

Moreover, CDM-AR projects face a number of challenges, including high labor demand for tree planting and maintenance, sunk costs and illiquid assets that cannot be easily disposed of in times of financial stress (i.e. used as insurance), and risks generated by production and future prices for both the carbon and timber (Coomes, *et al.* 2008). In contrast, by compensating local landholders on an annual basis for avoided deforestation in the short-term (5 – 10 years), rather than for long term (15 – 25 years) afforestation or reforestation, REDD projects preclude the additional labor or input costs necessary for tree planting while maintaining the option value and insurance function of the land and trees; i.e. if either party decides to break the contract, the landholder can still benefit from the existing stock of trees (Coomes, *et al.* 2008). In addition, community leaders surveyed by Coomes, *et al.* (2008) preferred avoided deforestation over CDM-AR for its ability to not only preserve the ecological integrity of extant forests, but also promote distributional equity, since all landholders who elect to conserve their trees would be eligible to receive payments under a REDD project, rather than subsidizing only those who benefitted from deforestation in the first place to reforest their property under a CDM-AR scheme. As such, it is expected that the implementation of a CR scheme on the Bathurst commonage could potentially provide fiscal incentives to reduce deforestation and ensure the sustainability of future fuel wood supplies in a way that maximizes both ecological integrity and distributional equity.

5.3.3.1 Methods

The value of avoiding transformation (deforestation) of the remaining vegetation on the commonage was estimated using data on the annual rate of thicket transformation over time as calculated from aerial photograph analysis and the difference in carbon content in above-ground (ABG) woody biomass (including dead standing stems), litter, and soil between transformed thicket in the high use zone and 'natural' thicket in the low use zone of the commonage (see Chapter Three). Since the low use zone was found to have a higher carbon content than Waters Meeting NR, the carbon content of the low use zone was taken as the maximum or 'natural' level.

For the purposes of this exercise, the historical (and business as usual) rate of deforestation was calculated based on the average annual change in woody plant cover since commonage management was shifted to emphasize access for Bathurst's poor, landless black residents (i.e. the average annual difference in the proportional area covered by woody plants between the 1998 and 2004 aerial photographs available). Because Waters Meeting NR demonstrated a slight (0.24 %) increase in woody cover over this period, it was excluded from the valuation exercise. Thus, only the high use zone is estimated to have potential for revenue generation through a compensated reduction (CR) scheme. Since CR revenue accrues annually based on demonstrated changes in carbon content over time, no estimation of standing stock was included in this valuation exercise.

According to the terms of CR as described in Bellassen & Gitz (2008), Equation 5.1 was used to estimate revenue from avoided deforestation over the course of a 5-year commitment period.

Equation 5.1 Annual net present value of revenue per hectare for avoided deforestation

$$R' = \frac{\text{Total revenue for “avoided emissions” compared to historical threshold}}{\text{Total land under “conservation”}}$$

$$= \frac{(D_0 - D'_5) * 5 * \text{Carbon price} * \Delta\text{carbon} * ß * (1-\mu)^5}{(D_5 - D'_5) * 5}$$

$$= \frac{(D_0 - D'_5)}{(D_5 - D'_5)} * C_p * \Delta c * ß * (1-\mu)^5$$

Where

D_0	is the historical area of annually deforested land
D_5	is the area of annually deforested land during the 5 years under BAU
D'_5	is the area of annually deforested land during the 5 years under CR
C_p	is the price of one metric tonne (T) of carbon
Δc	is the difference of time-averaged carbon content between intact thicket and thicket transformed by livestock grazing
μ	is the discount rate
ß	is the conversion factor from C to $C0_2$ equivalent

The discount rate is included in the equation because this model assumes that compensation will only follow after five years of commitment to the conservation scheme, based on the length of the first commitment period of the Kyoto Protocol and the second period of the

European Trading Scheme (Bellassen & Gitz 2008). Also, according to Bellassen & Gitz (2008),

> "(D_0 –D'_5) in Equation 5.1 represents a 'dilution' of the compensated (D_5 –D'_5) revenue: under a CR scheme, compensation is awarded at the national level for every hectare preserved under the historical baseline (D_0 –D_5). However, if deforestation pressure is expected to increase ($D_5 > D_0$), a policy that attempts to bring it under control would have to spread this revenue over more land (D_5 –D'_5) than the hypothetical surface corresponding to the revenue (D_0 –D'_5)". (Bellassen & Gitz 2008)

The difference between the standing stock of carbon in the high and low use zones of the commonage was calculated using field data on above-ground woody biomass, litter biomass, and soil carbon collected on site. Assumptions and inputs into the model are provided in Table 5.11.

Table 5.11 Summary of inputs used in the estimation of avoided deforestation value

Parameter	Estimate used in model	Source
$D_0 = D_5$ = area of annually deforested land under historical and business as usual scenarios	18.7 ha = 2.2 % of the 850 ha covered by woody plants	1998, 2004 photographs
$D`_5$ = area of annually deforested land under CR assuming $D_5' = 0.95 \times D_5$ i.e. 5 % reduction in deforestation rate	18 ha = 0.95 * 18.7 ha	Bellassen & Gitz 2008
C_p = Price of one metric tonne (T) of carbon	245 Rand	ECX June 2008 Month End Settlement Future Price for December 2012 contracts[2]
Δc = difference of time-average carbon content between high and low use zones of commonage	Total Δc (t) = Δc (wood) + Δc (litter) + Δc (soil)	Data collected from the site
μ = discount rate	5 %	Sankhayan & Hofstad 2001
ß = conversion factor from C to CO_2 equivalent	3.67	Bellassen & Gitz 2008

The total carbon content of woody plant (live and standing dead) biomass and litter (dead biomass), respectively, was assumed to be 48 % of the dry mass (IPCC 1996; Mackdicken 1997). Equation 5.2 was used to estimate soil organic carbon (SOC) in mass per hectare based on the soil organic carbon percent (SOCP) and moist bulk density of the soil (Grossman, *et al.* 2001).

To estimate SOC based on SOCP, it was thus first necessary to calculate the moist bulk density of each of the three land use zones. The volume of each sample was measured in situ to the nearest millilitre (ml) from a standard glass measuring cylinder. The soil samples were then oven-dried for 10 days and weighed to the nearest hundredth of a gram. Moist bulk

[2] Futures prices for 2013 (i.e. when CR payments would be paid) were not available for 2008 contracts. Converted from euros using exchange rate of 12.7 Rand to the euro (Reuters 2009).

density was then calculated by dividing the oven-dry weight by the field volume (Grossman, *et al.* 2001).

Equation 5.2 Estimation of soil organic carbon (SOC) content in kg/m^2

$$SOC = \frac{\underline{L \times SOCP \times p(1 - (V_{>2}/100))}}{10}$$

Where

SOC	is soil organic carbon in kg/m^2
SOCP	is soil organic carbon percent
L	is the thickness of the soil layer in cm
p	is the moist bulk density of the < 2 mm (excluding stones) fabric in g/cm^3
$V_{>2}$	is the percent volume of soil particles > 2mm (i.e. stones)

It should be noted that since the volume was measured at slightly drier than field capacity, the bulk density estimate may be slightly higher than reported here (Grossman, *et al.* 2001). Moreover, it was not possible to incorporate standard deviation into the estimation of soil organic carbon due to the different sites from which bulk density and the other parameters (SOCP, stone volume, soil depth) were measured. Nevertheless, due to the small contribution of soil organic carbon to the overall difference in carbon content between the different land use zones, these factors are not expected to make a significant difference to the overall valuation results.

5.3.3.2 Results

As shown in Table 5.12, the total difference in time-average carbon content between the high (transformed) and low ('natural') use zones of the commonage was estimated to be 77.0 $\pm$ 43.3 t, of which 73.3 $\pm$ 41.9 (95.2 %) is attributed to live and standing dead woody biomass, 3.4 $\pm$ 1.3 t (4.4 %) is from litter (fallen dead biomass), and 0.36 t (0.5 %) is from soil organic carbon. Over the 1.7 ha that would be remain forested under the CR scheme (5 % of the business as usual scenario of 29.4 ha annually deforested), this difference in carbon content could be expected to earn roughly R 53,470 $\pm$ 30,045 (US$ 6,476 $\pm$ 3,639) per hectare or roughly R 49,961 $\pm$ 28,073 (US$ 6,051 $\pm$ 3,400) in total annual revenue for the high use zone (Table 5.13). This translates into R 58.8 $\pm$ 0.04 (US$ 7.12 $\pm$ 4.0) per hectare spread across the entire high use zone (1,338 ha).

Table 5.12 Difference in time-averaged carbon content (t) between high and low use zones

Difference in carbon content (mean ± SE)			
Wood	**Litter**	**Soil**	**Total**
73.3 ± 41.9	3.4 ± 1.3	0.36	77.0 ± 43.3

Table 5.13 Annual revenue (mean ± SE) from CR scheme for the high use zone

R' per ha over CR area		Total R' over high use zone		R' per ha over high use zone	
Rand	**US$**	**Rand**	**US$**	**Rand**	**US$**
53,470 ± 30,045	6,476 ± 3,639	49,961 ± 28,073	6,051 ± 3,400	58.8 ± 0.04	7.1 ± 4.0

5.3.3.3 Discussion

It is worth noting that the rate of deforestation as measured by changes in woody cover in aerial photographs suggest that the commonage is losing woody biomass at a rate 1.6 times higher than the average rate recorded for members of the Southern African Development Community (SADC) for which data were available (FAO 2005; Table 5.14). This rate of change is all the more alarming given the demonstrated importance of the commonage for local fuel wood provision (Davenport 2008a). In light of the crucial role that woody biomass, in particular, plays in both sequestering carbon and supporting local livelihoods, it is recommended that an alternative fuel source, such as a woodlot, be developed for local users .

Table 5.14 Annual deforestation rates (% total) in forested area of SADC member states

Country	Annually deforested area	
	1990 – 2000	**2000 – 2005**
Angola	0.2	0.2
Botswana	0.9	1.0
Lesotho	3.2	2.7
Malawi	0.9	0.9
Mauritius	0.3	0.5
Mozambique	0.3	0.3
Namibia	0.9	0.9
Seychelles	0	0
South Africa	0	0
Swaziland	0.9	0.9
Tanzania	1.0	1.1
Zambia	0.9	1.0
Zimbabwe	1.5	1.7
Average	0.8	0.9
High Use (HU)	2.2	
HU over Avg	1.6	1.6

Source: FAO 2005

Given the small area deforested annually under the business as usual scenario (18.7 ha), the total value of avoided deforestation for the high use zone is rather modest (R 78,651 ± 3,639) compared to the R 1.2 billion ± 46 million estimated for Cameroon, where 122,000 ha of forest are lost annually (Bellassen & Gitz 2008). In fact, even if all 19 ha remained forested each year, the annual income from CR would remain unchanged. Nevertheless, and in light of the fact that the current rate of fuel wood extraction on the commonage may outstrip average annual woody plant growth, CR represents an interesting revenue source for the high use zone that could potentially be used to fund the development of a woodlot to ensure a sustainable fuel source for the future (e.g. Ham 2000). In addition to providing an alternative fuel source to reduce pressure on commonage trees, a woodlot also offers opportunities for local job creation and skills development through the hiring and training of woodlot managers. Thus, a well-implemented CR project could potentially offer double ecological benefits (reduced deforestation from CR commitments and decreased demand for natural wood in favor of woodlot poles) while contributing to local community development.

These advantages notwithstanding, implementing REDD projects, whether through CR or another financing mechanism, still faces a number of challenges that must be overcome in order to ensure that the benefits of avoiding deforestation or degradation accrue to the local landholders whose land use choices impact national carbon emissions. For example, the choice of a historical baseline from which to predict the business as usual (BAU) scenario is particularly important, since overestimating future deforestation trends based on historical rates could result in “hot air” by rewarding emission reductions that would have happened without the project. On the other hand, some countries with historically low deforestation rates could experience rapid forest loss under a realistic BAU due to increased pressure from economic and demographic growth (Ebeling & Yasué 2008). Moreover, although rewarding REDD at a national scale should resolve the leakage problem, there is still a need to address the permanence of reductions through, for instance, the creation of an insurance fund that can be accessed if a country defaults on its REDD contract (Bellassen & Gitz 2008; Bellassen & Leguet 2007).

There is also a need to deal with the upfront opportunity costs to conservation that could prevent landholders from shifting away from alternative land uses, such as cultivation, cattle production or, in this case, fuel wood collection. The World Bank Forest Carbon Partnership was explicitly created to address this problem by providing advance financing to help countries successfully transition to more forest-friendly land uses (Bellassen & Gitz 2008).

Nonetheless, the success of any REDD project in terms of generating either carbon emissions or positive development outcomes requires significant government capacity to, for instance, ‘enforce land use regulations, improve road planning, [and] implement payments for ecosystem services (PES) schemes’ that distribute national revenues from emissions reductions to local landholders (Ebeling & Yasué 2008). The numerous challenges facing modern commonage management in South Africa (Atkinson 2005; Atkinson & Benseler 2004; Buso 2003) therefore imply that municipalities will first have to prove their ability to control commonage resources effectively before any REDD project could be successfully implemented. REDD implementation will also require “appropriate liability, responsibility, enforcement, and verification mechanisms” (Bellassen & Gitz 2008).

While there are clearly a number of issues that need to be addressed, experience with implementing pro-poor PES programs to reduce deforestation in Costa Rica (Pagiola 2008) and Colombia (Scherr, *et al.* 2004) suggests that these obstacles can be overcome, even in resource-poor contexts. Furthermore, the growing support for REDD ahead of the upcoming COP in Copenhagen implies that a wide variety of stakeholders agree that this new avenue for reducing global GHG emissions could potentially generate enough revenue to at least partially offset the opportunity costs of foregoing some level of consumption of the natural resources upon which millions of the world’s poor depend for their livelihoods (Bellassen & Gitz 2008; Coomes, *et al.* 2008; White & Martin 2002).

5.3.4 Ecotourism

Over the past decade, ecotourism has become an increasingly important revenue source in the arid and semi-arid regions of South Africa. These climates, which are generally unsuited for cultivated agriculture, have traditionally supported domesticated livestock production. However, the livestock industry in South Africa has faced numerous obstacles during this period, including the deregulation of the agricultural sector, which resulted in lower (but more internationally competitive) prices; uncertainty due to land reform; and increased vulnerability to losses caused by stock theft, climate change, and bush encroachment, whereby woody species overtake grassy ones, often due to overgrazing (ABSA 2003). Faced with these structural changes, many livestock farmers have begun farming wild game as an alternative to domestic stock, with revenue opportunities from the sales of live animals, hunting, and game viewing. By 2003, ABSA estimated that South Africa was home to roughly 5,000 game ranches and 4,000 mixed game and livestock farms. Furthermore, the total area covered by these private game reserves (PGRs) represented fully 13 % of South

Africa's total land area of 1.2 million km^2, more than double the area covered by all official conservation areas, which cover about 6 % of the total area (ABSA 2003; StatsSa 1996).

Given the significant income and employment creation produced by South Africa's ecotourism industry, this land use potentially represents a significant and sustainable source of revenue for the study site. Nationally, the ecotourism industry produces roughly R 1 billion in revenue annually, with an additional R 1 billion estimated in indirect revenue created by related industries, such as airlines, taxidermy, 4x4 trails, outdoor equipment, and hotels (ABSA 2003). Furthermore, the relatively small proportion of the R 45 billion generated by the South African tourism industry in 1998 (including business travel) attributed to ecotourism gives some indication of the potential for growth in this specialized industry, which grew at an annual rate of 5 % during the period 1993 - 2003 (ABSA 2003). Between 1998 and 2002, the tourism industry as a whole represented 6 % of gross domestic product (GDP), compared with just 3.4 % attributed to agriculture (Carruthers 2008). Nationally, it is estimated that the tourism industry as a whole creates one permanent job for every eight tourists who visit (NAMC 2006).

Recent research on the revenue and employment generated by industries located in the subtropical thicket biome of the Eastern Cape found that luxury private game reserves (PGRs) produced a gross income per hectare[3] of R 2,941 and employed roughly 0.01 people per hectare, compared with R 734/ha and 0.005 jobs/ha generated by public conservation areas (e.g. national parks), and just R 234/ha produced by sheep production for mohair[4] (Sims-Castley 2002). Gross per hectare revenue earned by a representative farm that converted from livestock to game farming quadrupled from R 151 for livestock production to R 605 for game ranching. The report also notes that the professional hunting industry in the Eastern Cape had a turnover of more than R 67 million in the 2000/2001 hunting season, with related industries contributing to a gross total income of R 179 million that year (Sims-Castley 2002). By 2008, the average gross revenue per hectare earned by nine luxury PGRs sampled in the Eastern Cape had declined slightly to R 2,629/ha (Snowball & Antrobus 2008). In contrast to these extraordinary revenues, no game viewing or hunting activities are currently offered at Waters Meeting NR. It is also worth noting that the average per hectare revenue reported by the nine reserves surveyed over the period between 1999 and 2008 varied by nearly 24 % of the mean.

[3] All prices converted to constant 2008 Rand (South African CPI 2009; South African Reserve Bank 2009).

[4] Employment figures were not available for mohair production.

In addition to its revenue potential, ecotourism could also potentially generate significant employment on the study site. The same survey of nine PGRs in the Eastern Cape cited above found that, on average, the conversion from livestock farming to game ranching results in 3.5 times the number of jobs at 2.3 times the average employee salary compared with livestock operations before conversion (Snowball & Antrobus 2008). These are particularly relevant statistics in the Bathurst area, where more than 46 % of residents capable of working are unemployed and nearly 46 % of residents earn R 9,600 or less per year (Ndlambe IDP 2007).

The relative revenue generated by different wildlife consumption activities, including live game auctions, hunting, and game viewing, varies widely across South Africa and depends primarily on distance to a major airport and the presence of a formidable group of large and notoriously difficult to hunt animals known in the industry as the 'Big Five': elephant (*Loxodonta africana*), white (*Ceratotherium dimum*) and black (*Diceros bicornis*) rhinoceros, Cape buffalo (*Syncerus caffer*), lion (*Panthera leo*), and leopard (*Panthera pardus*) (ABSA 2003). In general, game areas located in close proximity to major cities that primarily feature opportunities for viewing the Big Five attract mostly international 'leisure' tourists, while more distant areas with abundant buck species tend to target domestic hunters (ABSA 2003). However, the dichotomy between game viewing and hunting is by no means absolute, as many game areas in South Africa successfully combine hunting and game viewing activities, including several private game reserves (PGRs) in the Eastern Cape (Snowball & Antrobus 2008).

In light of the fact that intensive ecotourism, which typically precludes direct access to natural resources, such as fuel wood or fodder for livestock production, would likely be incompatible with existing management objectives on the commonage, the valuation of ecotourism on the study site was limited to Waters Meeting NR. At the same time, despite the Strategic Management Plan's (WMNR 2007) overt reference to the tourism value of the Kowie River and explicit goal of effectively marketing the reserve's attractions to visitors, the existing ecotourism value of Waters Meeting NR is likely underutilized. Waters Meeting NR alone represents fully 70 % of a specific variety of subtropical thicket (Xeric Kaffrarian) that is formally conserved, a vegetation type unique to South Africa of which only 3 % is protected (WMNR 2007). The reserve also features a scenic overlook above the turn in the Kowie River known as Horseshoe Bend (Figure 5.5), as well as abundant wildlife, including numerous game (Table 5.15) and bird (Table 5.16) species (Davenport 2008b; WMNR 2007).

Table 5.15: Game species *likely* to be found within Waters Meeting Nature Reserve

Common name	Latin name
Blue duiker	*Philantomba monticola*
Bushbuck	*Tragelaphus scriptus*
Bushpig	*Potamochoerus porcus*
Cape grysbok	*Raphicerus melanotis*
Greater kudu	*Tragelaphus strepsiceros*
Grey (Common) duiker	*Sylvicapra grimmia*
Grey rhebuck	*Pelea capreolus*

Sources: Davenport, pers. comm. 2008; Earle, pers. comm. 2008; Smithers 1983; WMNR 2007

Table 5.16 Local bird species of interest to 'Birders' in the Bathurst area

Common name	Latin name
African Black Duck	*Anas sparsa*
African Finfoot	*Podica senegalensis*
Black-bellied Starling	*Lamprotornis corruscus*
Booted Eagle	*Aquila pennatus*
Cape Rock-Thrush	*Monticola rupestris*
Chorister Robin-Chat	*Cossypha dichroa*
Crowned Eagle	*Stephanoaetus coronatus*
Dark-backed Weaver	*Ploceus bicolour*
Denham's Bustard	*Neotis denhami*
Green-Pigeon	*Treron calvus*
Grey Cuckooshrike	*Coracina caesia*
Mocking Cliff-Chat	*Thamnolaea cinnamomeiventris*
Mountain Wagtail	*Motacilla clara*
Narina Trogon	*Apaloderma narina*
Rock Kestrel	*Falco rupicolis*
Trumpeter Hornbill	*Bycanistes bucinator*

Source: Davenport 2008b

Nevertheless, the reserve was historically established to protect native milkwood (*Mimusops sp.*), sneezewood (*Ptaeroxylon obliquum*), and wild olive tree (*Olea euopaea* subsp. *africana*) from intense demand for these resources for fuel, boat building, and agriculture in the late 19th century (WMNR 2007). Thus, its mandate excludes the explicit conservation of the 'Big Five' animals that play a key role in attracting wealthy international tourists (Snowball & Antrobus 2008). At present, only a few wild leopards still roam through the reserve, but no visitors or staff interviewed reported having actually seen these elusive cats.

Despite these natural attractions, existing tourism infrastructure on the reserve is limited to picnic facilities and a 16-bed overnight hut on the banks of the Kowie River (WMNR 2007). A private company based in Port Alfred actually operates the canoe trip that takes visitors to the overnight hut via the Kowie River; the company reportedly devotes half its revenues to the reserve's conservation budget (Powell, pers. comm. 2009). Apart from the canoe trip, no fee-based activities, such as game viewing or guided hikes, are currently offered to visitors.

Revenue on the reserve is thus limited to returns from a nominal entrance fee, However, the entrance gate is not consistently staffed due to limited staff capacity on the reserve (five field rangers are responsible for monitoring the entire area, including the remote northern part of the reserve (3,261 ha) and the smaller Kowie Nature Reserve (174 ha) located to the southeast of Waters Meeting I NR on the banks of the Kowie River), which likely reduces the potential revenue from gate fees. Thus, it is expected that the reserve's existing natural resources and tourism infrastructure could support significantly higher revenues than those currently earned.

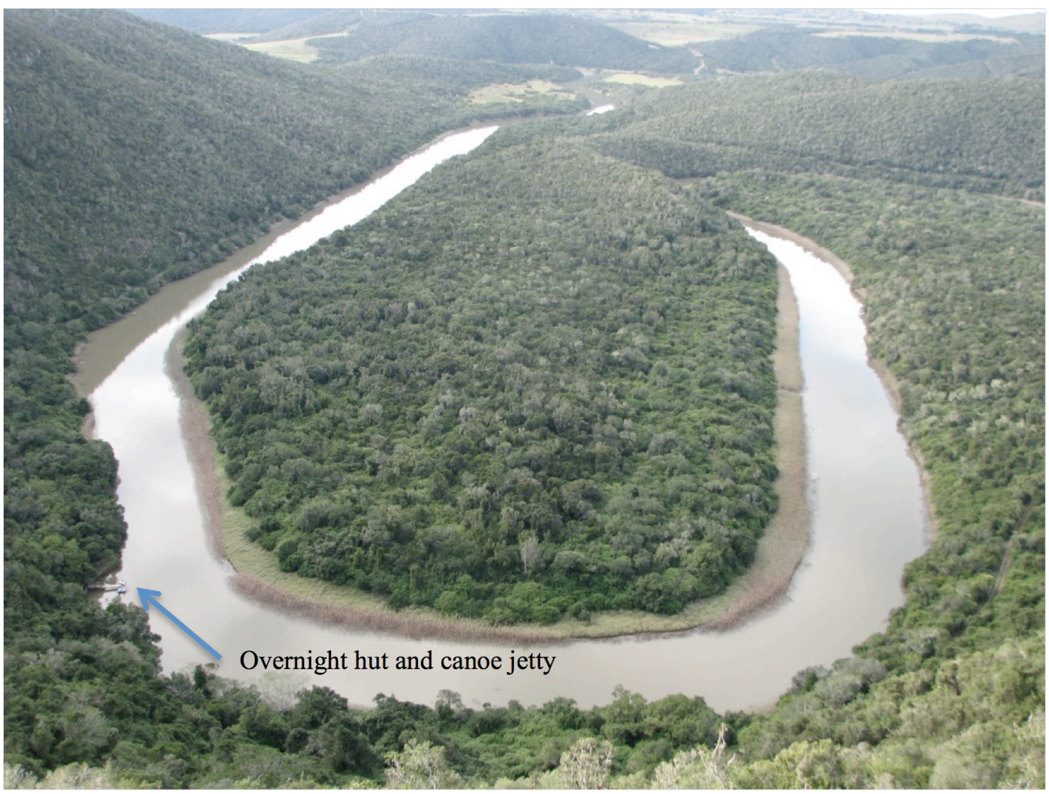

Figure 5.5 View from the overlook above Horseshoe Bend in Waters Meeting NR

The introduction of more of the Big Five and other high-value game animals could potentially significantly increase the theoretical value of annual production (and land) on Waters Meeting NR (ABSA 2003; Snowball & Antrobus 2008). However, this scenario is unlikely given (i) the reserve's small area (less than 1,500 ha where tourism infrastructure is accessible); (ii) its limited game viewing capacity due to dense vegetation and steep terrain that is largely inaccessible to vehicles; (iii) the potential conflict with neighboring land uses (aggressive wild animals could endanger both humans and livestock on the commonage); and (iv) incompatibility with the reserve's Strategic Management Plan (WMNR 2007), which emphasizes the development of additional hiking trails and canoe facilities that would be jeopardized by the presence of aggressive wild game in a small reserve. Moreover, although hunting represents perhaps 8 % of total annual revenues earned by luxury private game reserves in the Eastern Cape (Snowball & Antrobus 2008), this activity is incompatible with the reserve's conservation mandate and so was excluded from the valuation exercise.

As such, the ecotourism valuation was undertaken with the following assumptions:

(i) The management of the commonage for direct resource collection at present and for the foreseeable future precludes the development of intensive ecotourism on the site.

(ii) Existing exploitation of ecotourism on Waters Meeting NR undervalues its potential.

(iii) Given its small area, lack of high value game, and neighboring land uses, the ecotourism potential of Waters Meeting NR is more comparable to other small, low capital reserves that target low-budget domestic tourists than to large luxury reserves that promote Big Five ecotourism primarily to wealthy international visitors, especially given the abundant supply of the latter in the study area.

(iv) Future ecotourism development on Waters Meeting NR should focus on low impact activities, such as additional facilities for low budget overnight accommodation, hiking, and canoeing, as described in the reserve's Strategic Management Plan (2007), and will exclude hunting.

The discussion that follows details the methods employed to estimate the *potential* ecotourism value of Waters Meeting NR based on data collected from a variety of game reserves in the area supplemented with published data on the tourism industry in South Africa.

5.3.4.1 Methods

To estimate the potential value of ecotourism on Waters Meeting NR, an electronic survey was administered by email to 37 private and public nature reserves in the Eastern Cape that are covered at least in part by the subtropical thicket that dominates the natural vegetation on the study site. The same survey was completed by the Reserve Manager at Waters Meeting NR. The reserves solicited represented a variety of business models, from small, family-owned operations focusing primarily on the domestic market to large, luxury private game reserves that target wealthy international tourists, as well as various sizes of national and provincial parks and nature reserves in the area. A special effort was made to solicit survey responses from members of the Indalo association of PGRs, for which aggregate data on gross costs and revenues were available (Langholz & Kerley 2006; Snowball & Antrobus 2008).

The questionnaire consisted of eight sections covering (i) general information on the reserve, such as size, ownership, previous land use, and Tourism Council of South Africa grading; (ii) natural capital stocks, such as the primary vegetation types, number of the 'Big Five' (i.e. elephant, rhinoceros, buffalo, cheetah/leopard, and lion) represented, and special natural features (e.g. riverine ecosystem, dense natural vegetation, etc.); (iii) visitor attractions offered, including any tariffs charged and approximate number of annual visitors for each activity; (iv) capital investments, such as land value, area of all buildings, fees charged for the use of non-accommodation buildings, and visitors per year to each building; (v) accommodation, including types offered (e.g. lodge, guest house, camping, etc.), total number of beds per night, rate per person per night, and annual occupancy; (vi) infrastructure, such as number and extent of roads/dams/bridges/etc., number and age of heavy machinery and vehicles, value of game animals and ecosystem rehabilitation; (vii) operations details, including number of salaried staff employed, the percentage of marketing costs directed at different regional and international markets, and approximate annual gross revenue using a matrix of suggested income ranges; and (viii) reactions to statements about the most influential factors on visitor satisfaction and the state of ecotourism development in the Eastern Cape (see Appendix 2 for the full survey instrument).

In total five public and private game reserves (excluding Waters Meeting NR) responded to the electronic survey with at least partial answers (Table 5.17). Their data were critical inputs into the estimation of ecotourism value on Waters Meeting NR, and their cooperation is generously acknowledged. Nonetheless, due to the extremely low response rate (13.5 %), it was necessary to consult additional sources with detailed data on the profitability of game reserves in Eastern Cape and the southern African region more generally (e.g. ABSA 2003; Barnes & Jones 2008; Bothma, *et al.* 2009; Snowball & Antrobus 2008).

Data on the reserve size, accommodation offered (number of beds and rate per person per night), Tourism Council grading, and activities offered by all thirteen members of the Indalo group of luxury private game reserves sampled by Snowball & Antrobus (2008), were collected through their websites (Table 5.18). These data were also collected for reserves that responded to the electronic survey as necessary to supplement information provided directly. The methods used to estimate standing stock and annual production values for Waters Meeting NR are detailed below.

Table 5.17 Game reserves sampled through electronic survey for this study

Reserve name	Ownership	Big Five	Market
Addo Elephant National Park[1]	Public	Yes	Mixed
Assegaai Trails	Private	No	Domestic
Belton Hiking Trails	Private	No	Domestic
Kariega Game Reserve[2]	Private	Yes	International
Kragga Kamma Game Park	Private	Yes	Domestic

[1]The data reported for Addo Elephant National Park exclude all private concessions within the park.

[2]Kariega Game Reserve was the only Indalo member reserve to respond to the electronic survey.

Table 5.18 Indalo luxury private game reserves sampled through their websites

Private game reserve (PGR)	Website
Amakhala	www.amakhala.co.za
Blaauwbosch	www.blaauwbosch.co.za
Bushman Sands	www.bushmansands.co.za
Hopewell Lodge	www.hopewell-lodge.com
Kariega	www.kariega.co.za
Kuzuko Wilderness Lodge	www.kuzuko.co.za
Kwandwe	www.kwandwereserve.co.za
Lalibella	www.lalibella.co.za
Pumba	www.pumbagamereserve.com
Riverbend Lodge	www.riverbendlodge.co.za
Samara	www.samara.co.za
Shamwari	www.shamwari.co.za
Sibuya	www.sibuya.co.za

Standing stock:

The value of standing stock on Waters Meeting NR was estimated under two different scenarios: low (current) and high (potential) capital investment. The *current* per hectare value of standing stock based on a low capital scenario was estimated using actual data on the extent of existing natural and built resources on the reserve. The *potential* per hectare value under the high investment scenario was assumed to be roughly comparable to the average per hectare standing stock value reported by nine of the twelve Indalo PGRs focused on the luxury international tourist market (Snowball & Antrobus 2008), except that the value of game was held constant across the two scenarios to reflect the limited scope for introducing high tourism value but potentially aggressive Big Five game animals to Waters Meeting NR as described above.

The *current* value of standing stock on Waters Meeting NR was assumed to be equal to the value of all existing capital investments, including land and fencing, buildings and infrastructure, vehicles, and game (ABSA 2003). Up-front capital costs for an average ecotourism enterprise in South Africa (ABSA 2003) were adjusted to 2008 prices (StatsSA 2009) and used to calculate the value of standing stock on Waters Meeting NR based on its existing infrastructure and wildlife assets. It is expected that the actual value of the fencing, buildings and vehicles, in particular, would be significantly lower due to their regular use. However, in keeping with the other valuation exercises, the cost of depreciation was excluded from the estimation of standing stock. It should nevertheless be kept in mind that the value reported here is thus likely an overestimate of the actual value of the standing stock of resources on Waters Meeting NR.

The land value of Waters Meeting NR was assumed to be roughly equal to the minimum value (R 3,000/ha) of commercial farmland in the Bathurst area (Powell, pers. comm. 2009), while the value of existing fences on the reserve was calculated for the perimeter only (21,897 m) using an estimated cost of R15/m (ABSA 2003) inflated to 2008 prices. To calculate the current (low capital) value of all buildings on the reserve, the approximate area (Powell, pers. comm. 2009) of all service (staff housing) and tourism (overnight hut) buildings was multiplied by the inflation-adjusted cost of constructing staff housing (R 2,000/m^2) and tourist accommodation (R 2,300/m^2 assuming modest construction) based on ABSA (2003). No data on the extent of roads or other infrastructure[5] were available. The value of all service vehicles currently owned by the reserve was calculated by adjusting the R 200,000 cost for a single cab 4x4 vehicle (ABSA 2003) for inflation. No tourism (i.e. game-viewing) vehicles were reported. Light equipment (lawnmower and brush cutter) currently owned by the reserve was valued at roughly R 5,000 based on prevailing prices published on online websites hosted by South African retailers.

Since no systematic wildlife sampling has been carried out recently in the reserve, the game stocking capacity of Waters Meeting NR was estimated using a formula developed for the Eastern Cape Department of Economic Development and Environmental Affairs based on an estimated carrying capacity of six hectares per large animal unit (LAU) (Hahndiek, pers. comm. 2009). The relative proportions of graze (i.e. grass) and browse (i.e. woody plants)

[5] Waters Meeting NR contains the Sarel-Hayward dam, which is the primary water source for Port Alfred. Since this infrastructure can thus not be exploited for ecotourism, it was excluded from the valuation.

available were estimated based on the proportional areas covered by grass (4.8 %) and woody plants (95.2 %), respectively, in the most recent aerial photographs of the region (2004). The value of the game that could be supported by the vegetation within Waters Meeting NR at 60 % of the maximum stocking rate was calculated using game auction prices reported by ABSA (2003) inflated to 2008 prices using the South African CPI (StatsSA 2009). Only those large mammals (mostly buck) considered likely to exist in the study area were included in the game valuation (Table 5.15).

In contrast, the *potential* (high capital) estimation of standing stock assumed that the value of game remained constant from the current (low capital) scenario, while the value of land, buildings, infrastructure, and vehicles was calculated based on the average per hectare values reported by nine well-capitalized PGRs targeting international tourists (Snowball & Antrobus 2008). Although the land value of a luxury PGR with all Big Five game animals would likely exceed that of Waters Meeting NR without these animals, the cost of purchasing commercial farmland in the region of the nine reserves sampled by Snowball & Antrobus (2008) was assumed to be roughly comparable to that of the study site, which lies within an approximately 120 km radius of the Indalo reserves.

Annual production:

Since insufficient information was available to estimate the gross revenue potential of most of the five reserves sampled based on either the survey instrument or published literature, data were collected from the public websites of the thirteen Indalo PGRs in the Eastern Cape and the four non-Indalo reserves that responded to the electronic survey to estimate relationships between (i) the total area of the reserve in hectares, (ii) the total number of accommodation beds offered nightly, (iii) the 'attractiveness' of the reserve as measured by the number visitor attractions (e.g. game viewing, spa facilities, hiking or other outdoor recreation activities) offered, and (iv) the relative market value of the reserve as estimated by the number of stars awarded by the Tourism Council of South Africa, and the (all-inclusive) accommodation rate per person and rate per bed per night. Where prices differed between high (November-April) and low (May-October) seasons, the high season prices were used to account for the maximum potential effect on gross revenue. Where reserves offered multiple types of accommodation, the highest number of stars awarded to any given unit was used to estimate the Tourism Council grading, while the accommodation (bed) rate was calculated as a weighted average based on the relative proportion of beds offered at each rate, subject to available information. Operating costs were excluded in keeping with the methodology of other valuation exercises for this thesis.

Since most of the Indalo reserves offer all-inclusive accommodation rates that cover game drives and meals, it was necessary to discount the *accommodation rate per person* to estimate the *rate per bed* for comparison with non-Indalo reserves. To do so, the exclusive day visitor cost charged for game drives and meals on each of the three reserves advertising day visitor costs was divided by the all-inclusive accommodation rate on said reserve, and the resulting proportion was averaged across all three reserves for which data were available. This proportion (roughly 30 %) was then used to discount any all-inclusive accommodation rate charged by an Indalo member reserve to estimate the rate per bed per night.

These data were subjected to multiple linear regressions with three alternative dependent variables: total number of beds, density of beds per hectare, and accommodation rate per person per night. Based on its ability to account for the highest proportion of variance (R^2), the equation estimating total number of beds on the reserve based on reserve area, maximum number of stars, number of Big Five animals present, total number of tourism activities offered, total (all-inclusive) accommodation fee, and rate per person per night, was selected to predict the ecotourism value of Waters Meeting NR based on three different scenarios: current and potential under both low and high capital scenarios. Regression equations considered for predicting these relationships are presented in Table 5.19.

The following independent variables remained constant for all three estimations of annual production according to actual data on Waters Meeting NR: area of the reserve (1,445 ha), maximum number of Tourism Council stars (0), number of Big Five animals present (1), number of activities (game viewing, hiking, canoeing), and approximate number of day visitors per year (5,000). The remaining variables (accommodation fee, rate per bed per night, and occupancy rate) were estimated for each scenario as follows. See Table 5.20 for all model inputs used.

The total annual revenue generated under the *high capital* investment scenario was estimated based on the nightly all-inclusive accommodation rates and average occupancy rates as reported by the Indalo reserves. However, the maximum annual revenue under the high capital scenario as estimated by the model was discounted using two different rates (40 % and 80 %) to account for the lower game density, reduced capacity for game viewing due to dense thicket vegetation, and lack of all Big Five species on Waters Meeting NR compared to the

Indalo reserves, which would likely reduce the revenue potential of Waters Meeting NR. Given these multiple and significant differences in revenue-earning capacity between Waters Meeting NR and the Indalo reserves, only results from the low capital scenario will be used to calculate the total annual production value of natural resources in Waters Meeting NR. Thus, the discount rate used for the high capital valuation will not affect the overall conclusions.

Annual revenues generated under the low capital and current scenarios were estimated using several assumptions. Based on detailed revenue data reported by nine Indalo reserves (Snowball & Antrobus 2008), it was assumed that at least 80 % of reserve revenues were generated by accommodation and game-viewing activities, with a further 9 % (roughly) generated by other activity tariffs, conferences/weddings, and live game sales. It also was assumed that the remaining revenue reported by the international, well-capitalized Indalo reserves that was attributed largely to hunting fees was equal to zero for both the current and low capital scenarios and based on (i) the acceptable activities permitted by the reserve (i.e. no hunting in public parks) and (ii) the limited availability of trophy animals. Since Waters Meeting NR does not have the capacity to host large numbers of guests (100 +), revenue from special events (e.g. weddings) was excluded from the estimation of total annual revenue. These other (minor) potential sources of revenue were excluded from the current and low capital valuations to ensure conservative estimates.

To estimate the total value of annual production for the non-Indalo reserves, the *low capital* scenario assumed that Waters Meeting NR could charge nightly accommodation rates roughly comparable to the average rates per bed (equal to accommodation rate per person for these low capital reserves) reported by the four relatively low capital (public and private) nature reserves that were sampled through the electronic survey. The accommodation revenue for the low capital scenario was thus estimated based on the average accommodation rate per person per night and occupancy rates reported by the non-Indalo reserves. Gross revenue from activity and gate fees for each of these low capital reserves was estimated based on the average fees charged for game viewing assuming that on average 50 % of day visitors took advantage of this activity annually.

Table 5.19 Selected allometric relationships for predicting annual production of ecotourism value on Waters Meeting Nature Reserve

Dependent	Equation	R^2	F	Df	p	SE[1]
Number of beds	y (total no. beds) = 0.0018609*(area(ha)) - 13.61784*(max no. stars) + 13.28773*(no. Big Five) - 1.283894*(no. activities) - 0.0526395*(accommodation fee(R)) + 0.0642658*(rate per bed per night(R)) + 54.44058	0.8823	12.49	(6,10)	0.0004	43.395
Beds per ha	y (no. beds/ha) = 3.20e-07*(area(ha)) + 0.178348*(max no. stars) - 0.0242964*(no. Big Five) - 0.0010979*(no. activities) - 0.0000118*(rate per bed per night(R)) + 0.0705322	0.6546	4.17	(5,11)	0.0227	0.02403
Rand per bed per night	y (no. beds/ha) = 3.20e-07*(area(ha)) + 0.178348*(max no. stars) - 0.0242964*(no. Big Five) - 0.0010979*(no. activities) - 0.0000118*(rate per bed per night(R)) + 0.0705322	0.5993	4.49	(4,12)	0.0190	1,016

[1]Square root of the Mean Square Residual (or Error).

Table 5.20 Model inputs for estimating annual production value of Waters Meeting NR (mean ± SE)

Capital	Accommodation	Bed rate	Occupancy	Gate fee	Game viewing	Stars	Big Five	Activities
Unit	**Rand**	**Rand**	**%**	**Rand**	**Rand**	**#**	**#**	**#**
Current	30	30	14.3	10	0	0	1	3
Low Capital	198 ± 76	198 ± 76	54.3 ± 14.5	61 ± 16	208 ± 27	0	1	3
High Capital	2,714 ± 1,796	1,945 ± 1,188	42.5 ± 3.1[1]	Included	Included	0	1	3

Sources: Electronic survey; Reserve websites; Snowball & Antrobus 2008

[1]Average reported for all Indalo reserves surveyed by Snowball & Antrobus (2008); standard error across values reported for three reserve sizes.

For comparison, annual revenue under the current scenario was calculated based on actual reserve capital assets and estimates of actual gate fees collected, accommodation cost per bed per night and average annual occupancy rate (Powell, pers. comm. 2009). No game viewing activities are currently offered on the reserve. Results of the current valuation exercise should be interpreted with caution given the potential error involved in their calculation. The 'current' valuation results are provided here in order to demonstrate the predicted magnitude of annual revenue foregone based on the reserve's existing resources.

5.3.4.2 Results

As shown in Table 5.21, the results of the standing stock estimation for ecotourism capital on Waters Meeting NR indicate that the value totals R 8,200,990 (US$ 993,247) for the current (low capital) and R 10,455,782 (US$ 1,266,332) for the high capital scenarios, with the high capital scenario valued at roughly 27.5 % higher than the low capital scenario. These translate into per hectare values of R 5,675 (US$ 687) and R 7,236 (US$ 876) for the low and high capital scenarios, respectively. Of the total estimated value for the low capital scenario, land and fencing represent 58.1 %, building and infrastructure 13.7 %, vehicles 9.6 %, and game 18.5 %. For the high capital scenario, land value was incorporated into the buildings & infrastructure estimate according to the estimates reported by Snowball & Antrobus (2008). The combined value of land, fencing, buildings, and other infrastructure under the high capital scenario thus represent 78.3 %, compared with roughly 72 % under the low capital scenario. However, the relative contributions of vehicles (7.2 %) and game (14.5 %) to the total value of standing stock in the high capital scenario are higher than those of the low capital scenario.

Table 5.21 Estimated value of standing stock of ecotourism capital on Waters Meeting NR (Constant 2008 Rand, US$)

Unit	Land & fencing	Buildings & infrastructure	Vehicles	Game	Total	Per hectare
Current = Low Capital Scenario						
R	4,765,478	1,127,130	791,370	1,517,012	8,200,990	5,675
US$	577,162	136,510	95,845	183,730	993,247	687
% Total	58.1	13.7	9.6	18.5	-	
High Capital Scenario						
R		8,183,035	755,735	1,517,012	10,455,782	7,236
US$		991,072	91,529	183,730	1,266,332	876
% Total		78.3	7.2	14.5	-	

Sources: ABSA 2003; Waters Meeting NR electronic survey

As shown below in Table 5.22, the results of the annual production model indicate that the existing natural and built capital on Waters Meeting NR could support a total of 69 ± 192 beds assuming the average accommodation charged by low capital reserves in the area that target primarily domestic leisure tourists, while the reserve could likely support roughly 49 ± 1,234 beds under a high capital scenario comparable to that which characterizes the Indalo member reserves. Using the assumptions outlined above, the low capital scenario indicates that Waters Meeting NR could generate per hectare annual revenues (including gate and activity fees) of R 2,438 ± 368 (US$ 295 ± 45), totaling R 3,522,983 ± 531,288 (US$ 426,679 ± 64,346) across the reserve. Of this total value, 76.6 % is generated by accommodation fees, 8.7 % from gate fees, and 14.7 % from game viewing fees. Assuming a 40 % reduction from the values reported by Indalo reserves, the high capital scenario suggests that Waters Meeting NR could generate per hectare values of R 8,508 ± 2,874 (US$ 1,030 ± 348) totaling R 12,294,666 ± 4,152,612 (US$ 1,489,045 ± 502,936) from all-inclusive accommodation rates that cover gate and game viewing fees. However, if the 80 % discount rate is used under the high capital scenario, the potential annual production values are roughly comparable to those estimated under the low capital scenario: R 2,836 ± 958 (US$ 343 ± 116) per hectare or a total of R 4,098,222 ± 1,384,204 (US$ 496,348 ± 167,645) from all-inclusive accommodation rates.

Table 5.22 Model results of annual production value of Waters Meeting NR (Constant 2008 Rand, US$)

Capital	Accommodation		Gate fees	Game viewing	Total value		Total per ha	
Unit	**# Beds[1]**	**Rand**	**Rand**	**Rand**	**Rand**	**US$**	**Rand**	**US$**
Current	16	25,094	10,000	0	35,094	4,250	24	3
Std Error	Not applicable							
Low	69	2,698,816	305,000	519,167	3,522,983	426,679	2,438	295
Std Error[2]	192	383,287	79,746	68,256	531,288	64,346	368	45
High -40 %	49	12,294,666	Included	Included	12,294,666	1,489,045	8,508	1,030
Std Error[2]	1,234	4,152,612	Included	Included	4,152,612	502,936	2,874	348
High -80 %	49	4,098,222	Included	Included	4,098,222	496,348	2,836	343
Std Error[2]	49	1,384,204	Included	Included	1,384,204	167,645	958	116

[1]Dependent variable

[2]Assuming that reserve characteristics (size, number of Tourism Council of South Africa stars, number of Big Five and activities, and visitors per year) remain constant.

For comparison, it is estimated that Waters Meeting NR currently generates just R 24 (US$ 3) per hectare, or R 35,094 (US$ 4,250) in total annual revenues, of which 71.5 % is generated by accommodation fees and the remaining 28.5 % comes from gate revenues. Although the current value should be interpreted with caution given the various assumptions employed in its estimation, it is nevertheless worth noting that by charging accommodation, gate, and game viewing fees comparable to those of similarly capitalized reserves in the area, Waters Meeting NR could potentially increase annual revenues by nearly 10,000 %! Assuming that Waters Meeting NR could match roughly 60 % of the revenues reported by the luxury Indalo reserves, the reserve could potentially earn nearly 35,000 % more than its current revenues.

5.3.4.3 Discussion

Despite the numerous assumptions required to estimate the total value of ecotourism on Waters Meeting NR, it is evident that current revenues collected on the reserve significantly under value its potential. Given the reserve's accessibility to other popular holiday destinations, including the seaside towns of Port Alfred and Kenton on Sea, and proximity to Port Elizabeth (180 km by road) it is not unreasonable to assume that Waters Meeting NR could take advantage of as yet untapped marketing opportunities in the region. It is worth noting that per hectare annual production value of R 2,438 (US$ 295) predicted by the low capital model is roughly comparable to the average figure reported by nine of the Indalo reserves for the 2007/08 tourism season, while the value predicted by the high capital model assuming an 80 % discount rate is R 2,836 (US$ 343) (Snowball & Antrobus 2008). Thus, while the reserve may never be a luxury international tourist destination home to all members of the Big Five, there is clearly room for increasing revenues from accommodation, activity, and gate fees based on the experience of other low-budget reserves that target primarily domestic tourists (e.g. Assegaai Trails, Belton Hiking Trails, etc.).

For example, the existing number of beds could be increased by roughly 3.3 times from 16 to 69, and there may be further scope for adding additional beds based on the nearly 5-fold difference in bed densities per hectare between Waters Meeting NR and the four non-Indalo reserves surveyed. Likewise, the current hiking fee of R 10 per person could feasibly be raised by roughly six times, while the gate (vehicle) fee could be augmented by 3.5 times. There may also be revenue potential from guided tours through the reserve featuring, for example, the diverse bird life and unique Xeric Kaffrarian Thicket vegetation that have been protected for over a century by Waters Meeting NR, which is possibly the oldest protected area in the Eastern Cape.

Although beyond the purview of this study, it is also worth noting that the reserve's location within the Sub-Tropical Thicket Ecosystem Planning (STEP) domain may offer further revenue potential through reserve expansion if future Mega-Conservancy plans are implemented (Cowling, *et al.* 2003). In fact, the community mapping exercise completed recently by Fabricius, *et al.* (2006) suggests that there are already opportunities for integrating the land uses of neighboring farms into a broader conservancy area, as shown by the golden areas in Figure 5.6. Both the Baviaanskloof Mega-Reserve and Addo Elephant National Park (NP) in the region have successfully expanded their borders through direct acquisitions and harmonization of neighboring land uses, thereby increasing the total protected area as well as creating additional opportunities for revenue generation through, for example, private concessions for ecotourism development (Boshoff 2005; Child, *et al.* 2004). In addition to managing its own accommodation and tourism activities, Addo Elephant NP contains five private concession areas managed by luxury tourism operators (see www.sanparks.org for details on the reserve's tourism infrastructure compared with the facilities offered by private concession operators). Although revenue data were not available, these private concessions undoubtedly contribute significant added value to the site.

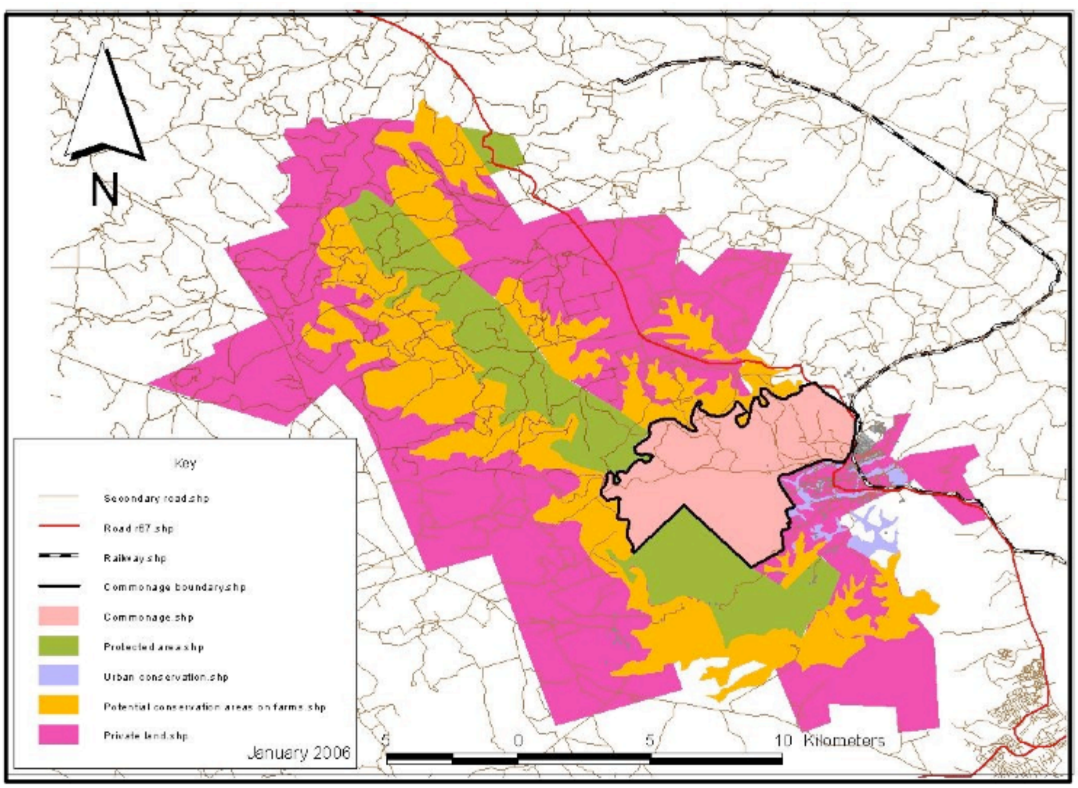

Figure 5.6 Integrated land use planning in the study area

Source: Fabricius, *et al.* 2006

5.4 Conclusion

The results of the field valuation exercise (Tables 5.23 – 5.26) clearly indicate that non-extractive and indirect ecosystem services can potentially provide a significant source of income to support concerted land use planning on the study site. The total annual production value of the study site was estimated to be R 6 million (US$ 731,852) or R 1,363 (US$ 165) per hectare, with R. 8.4 million (US$ 1.02 million) worth of standing natural capital (R 1,891 or US$ 229 per ha). The total annual value of indirect services valued in the high use zone of the commonage (endangered species conservation and avoided deforestation) is R 533,213 (US$ 64,579) or R 399/ha (US$ 48/ha). In the low use zone, the total annual value of indirect (conservation) and non-extractive (honey production) services was R 1.3 million (US$ 162,116), or R 810/ha (US$ 98/ha), with a further R 128,489 (US$ 15,562) in standing natural capital. Indirect and non-extractive services valued for Waters Meeting NR (ecotourism[6], conservation, and honey production[7]) totaled R 4.2 million (US$ 505,158) or R 2,886/ha (US$ 350/ha), with an additional R 8.4 million (US$ 1.02 million) estimated for standing natural capital.

Table 5.23 Non-extractive ESVs attributed to the high use zone (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	EC Rocky conservation	275,872	33,412	206	25	-	-	-	-
2	Leopard conservation	207,380	25,116	155	19	-	-	-	-
3	Avoided deforestation	49,961	6,051	37	5	-	-	-	-
	Total	**533,213**	**64,579**	**399**	**48**	**0**	**0**	**0**	**0**

Table 5.24 Non-extractive ESVs attributed to the low use zone (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	EC Rocky conservation	633,150	76,683	383	46	-	-	-	-
2	Leopard conservation	475,954	57,644	288	35	-	-	-	-
3	Honey[1]	229,444	27,789	139	17	128,489	15,562	78	9
	Total	**1,338,548**	**162,116**	**810**	**98**	**128,489**	**15,562**	**78**	**9**

[6] The figures used to estimate the total ecotourism value are based on the low capital investment scenario.
[7] The figures used to estimate the total honey production value are based on the small-scale sales scenario.

Table 5.25 Non-extractive ESVs attributed to Waters Meeting NR (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Ecotourism[2]	3,522,983	426,679	2,438	295	8,200,990	993,247	5,675	687
2	EC Rocky conservation	314,915	38,140	218	26	-	-	-	-
3	Leopard conservation	236,729	28,671	164	20	324,678	39,323	225	27
4	Honey[2]	96,333	11,667	67	8	53,947	6,534	37	5
	Total	**4,170,960**	**505,158**	**2,886**	**350**	**8,383,425**	**1,015,342**	**5,802**	**703**

Table 5.26 Non-extractive ESVs attributed to the study site (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Ecotourism[2]	3,522,983	426,679	795	96	8,200,990	993,247	5,675	687
2	EC Rocky conservation	1,223,937	148,235	276	33	-	-	-	-
3	Leopard conservation	920,062	111,432	208	25	-	-	-	-
4	Honey[1]	325,778	39,456	73	9	182,436	22,095	41	5
5	Avoided deforestation	49,961	6,051	11	1	-	-	-	-
	Total	**6,042,721**	**731,852**	**1,363**	**165**	**8,383,425**	**1,015,342**	**1,891**	**229**

[1]Under the small-scale sales scenario. [2]Under the low capital scenario.

Unfortunately, it is difficult to compare these results to other ecosystem service valuation literature (e.g. Chen, *et al.* 2009; Grêt-Regamey, *et al.* 2009; Hassan 2003; Ingraham & Foster 2009; Turpie, *et al.* 2008) due to the different services included in the valuation exercises and the different socio-economic and ecological contexts of the study sites. However, it is worth noting that the combined value of all the measured non-extractive revenue streams in the high and low use zones of the commonage (R 1.9 million, US$ 226,695) totals just under 72 % of the annual revenue generated by livestock alone (R 2.6 million, US$ 316,150) and roughly half (53 %) of the total value of direct ecosystem services. This finding is in line with research on the economic implications of managing rangeland ecosystem health for alternative management goals (Teague, *et al.* (2009).

Teague, *et al.* (2009) found that when a semi-arid savanna rangeland was managed to improve range condition rather than maximize profit, total thirty-year profits were estimated to be 78 – 87 % of the stocking rates that would maintain range condition and 67 – 75 % of those that would maximize profit. Also of note, a recent valuation of indirect ecosystem services provided by the U.S. National Wildlife Refuge system found that while forests and shrublands generate values of US$ 344/ha/yr and US$ 223/ha/yr, respectively, grasslands produce just US$ 21/ha/yr (Ingraham & Foster 2008). In light of the fact that grasslands cover approximately 31 % of the commonage, compared with just 5 % of Waters Meeting NR, and the considerable value generated by direct natural resource extraction in the high use zone, in particular, it is unlikely that the commonage will ever support exclusively conservation (non-extractive) land uses.

Even though the scope for compensating commonage resource users through payments for ecosystem services generated by strict conservation regimes may be limited, these results suggest that there are a nevertheless number of opportunities for supplementing local livelihoods while encouraging sustainable land use management on the study site. For example, residents of the Kowie River catchment indicated that they would be willing to pay over R 2.1 million (US$ 259,666) to conserve locally endangered species, including leopard and Eastern Cape Rocky, and improve the health of the Kowie River as a whole. This amount of funding could provide significant stimulus to support, for instance, the creation of temporary employment opportunities to systematically remove alien fish from the river or the reestablishment of a regulated dipping program for cattle on the commonage that would encourage better enforcement of restrictions on herd size per person.

There may also be additional income generation opportunities derived from non-extractive ecosystem services, such as honey production. As mentioned above, small-scale beekeeping on the commonage, with each entrepreneur responsible for maintaining an average of seven hives (G. Cambray, pers. comm. 2008), would generate annual revenues of R 1,458 $\pm$ 545 (US$ 177 $\pm$ 66) for each of roughly 157 beekeepers, with an additional 66 supported by Waters Meeting NR. Considering that 16 % of residents of Ward 5 of Ndlambe local municipality (LM), which includes Bathurst, earned no annual income and a further 16 % earned less than R 4,800 (US$ 603) in 2007, this represents a noteworthy income generation opportunity for local residents (Ndlambe IDP 2007).

Furthermore, it is clear that there is considerable potential for exploiting the ecotourism potential of Waters Meeting NR through the expansion of existing accommodation facilities and enhanced marketing. With potential estimated annual production worth perhaps R 3.5 million (US$ 426,679) and a standing stock of natural and built capital worth roughly R 8.2 million (US$ 993,247), ecotourism accounts for fully 85 % of the measured value generated by the reserve and roughly 58 % of the total economic value of the study site as a whole. The successful exploitation of the full ecotourism value of Waters Meeting NR would no doubt require non-trivial investments in marketing, staffing capacity, and the construction of additional accommodation facilities. Nonetheless, it is worth considering that the estimated annual revenue from Waters Meeting NR would contribute nearly half (48.2 %) of the gross total revenue reported by Eastern Cape Parks for fiscal year 2008 (ECPB 2008). Assuming that one third of this revenue (R 1.2 million or US$ 142,226) was shared with Ndlambe LM, the annual income from ecotourism alone could fund the municipality's entire nature conservation and environmental compliance operating budget (Ndlambe LM 2008).

Thus, although payments for indirect ecosystem services would likely be insufficient to entirely offset the direct use values generated by the commonage, in particular, the results of the field valuation exercise suggest that there are considerable income generation opportunities through the exploitation of indirect and/or non-extractive ecosystem services, such as honey production, ecotourism, avoided deforestation, and voluntary payments for conservation. The final chapter will present the total economic value of the study site as estimated through the benefit transfer and field valuation exercises. It will briefly compare the ESVs estimated through both benefit transfer and field data collection (honey production and endangered species conservation) in order to explore discrepancies between anticipated and actual values and the assumptions underlying these differences. Finally, the conclusion will address the expected trade-offs between direct and indirect values in more detail with a view toward integrating land uses on the study site in a way that maximizes annual revenues without sacrificing long-term sustainability.

6 Chapter Six

Conclusion: Managing the Commons for an Optimal Mix of Services

6.1 Introduction

To date, the performance of commonage management in South Africa has substantially underachieved its promise to facilitate the transition of previously disadvantaged ‘emergent’ farmers into viable commercial agriculturalists (Anderson & Pienaar 2003; Atkinson & Benseler 2004; Buso 2003). Nevertheless, the common property resources available on commonage lands remain central to the subsistence of poor households who often have few alternative livelihood options (Andrew, *et al.* 2003; Cartwright, *et al.* 2002; Davenport 2008a; Ingle 2006; Millennium Assessment 2005; Shackleton, *et al.* 2001). At the same time, it is imperative that commonage lands be managed sustainably to ensure that these resources are available in future and avoid the intensification of poverty that would likely arise in their absence (Ngwenya & Hassan 2005). Given that the provision of direct (e.g. fuel wood, fodder for cattle, and wild foods) and indirect (e.g. climate regulation, nutrient formation, and tourism) ecosystem goods and services (EGS) is dependent on healthy ecosystems (Aylward, *et al.* 2005; Costanza, *et al.* 1997; Rapport, *et al.* 1998; Zurlini & Girardin 2008), it is imperative that commonage management plans be carefully designed, monitored, and implemented with a view toward maximizing short-term revenues from direct natural resource collection without compromising the long-term sustainability of the ecosystems upon which so many subsistence livelihoods depend (Davenport 2008a).

To better understand the trade-offs between different levels of natural resource consumption and the variety and magnitude of ecosystem services provided to the local community, this thesis has measured the ecological health and provision of ecosystem goods and services derived from three zones of varying land use intensity on the Bathurst commonage and adjacent Waters Meeting NR. The next section details the overall results of the valuation exercise, compares the values estimated by the benefit transfer and field valuation exercises, and makes recommendations for balancing value streams derived from both direct and indirect ecosystem goods and services at the study site. Following this are suggestions on the design and implementation of a payments for ecosystem services (PES) project to promote silvo-pastoral practices on the commonage, including opportunities for improving the sustainability of resource management on both the commonage and Waters Meeting NR through enhanced commonage user rights. The chapter closes with an overview of the findings and their implications for future management on the site.

6.2 Total economic value of the study site

Table 6.1 – Table 6.4 display the combined results of the benefit transfer and field valuation exercises. Values estimated for the commonage as a whole in the benefit transfer exercise were allocated to the high and low use zones according to the proportional area covered by each. Where a service was valued using both methods, only the field valuation results are shown below. As such, the value of honey production estimated for the high use zone is not included here to account for the results of the field valuation, which found no *Scutia myrtina* plants in this zone. As in the previous chapter, the ecotourism is valued under the low capital scenario and honey the small-scale sales scenario. The value of bush meat on Waters Meeting NR shown here excludes the value of game animals to avoid double counting with the ecotourism field valuation exercise.

As shown in Table 6.4, the total annual value of the study site was estimated to be R 9.8 million (US$ 1.2 million), or roughly R 2,200 (US$ 266) per hectare. The total value of standing natural capital on the study site was estimated to be R 28 million (US$ 3.4 million), or R 6,323 (US$ 766) per hectare. Of the total annual value, Waters Meeting NR accounts for nearly 45 %, while the high and low use zones of the commonage contribute roughly 22 % and 34 %, respectively. With respect to total standing capital on Waters Meeting NR, the high use zone and low use zone account for approximately 36 %, 28 %, and 36 % of the total value. In terms of the per hectare value of standing natural capital, at R 7,014 (US$ 850) Waters Meeting NR is again valued the highest among the land use zones, being 16.4 % and 18 % higher than the low use and high use zones, respectively, whose per hectare standing stock values are roughly equal: R 6,023 (US$ 730) and R 5,946 (US$ 720), respectively.

Notably, the per hectare annual production value of the low use zone (R 1,996 or US$ 242) is over a quarter (26 %) higher than that of the high use zone (R 1,585 or US$ 192), while the per hectare annual value of Waters Meeting NR (R 23,001 or US$ 363) is 50 % higher than that of the low use zone, due largely to the ecotourism potential. Since the estimation of total economic value is subject to the ecosystem services included in the valuation exercise, it is worth noting that Davenport (2008a) concluded that direct incomes derived from (in decreasing order of importance) wood, livestock, wild foods, medicinal plants, sand and clay, and grass sweepers totaled R 1,578[1] (US$ 191) per hectare. This suggests that the area-weighted average per hectare annual production value of R 1,812 (US$ 220) estimated by this study for the commonage as a whole, which excludes some of these goods and includes indirect services totaling 30 % of the total, underestimates the actual total economic value of the commonage. In fact, the per hectare annual production value of the direct ecosystem services valued by this study (livestock, bush meat, honey, fuel wood, and medicinal plants) was just R 1,263 (US$ 153), roughly 20 % lower than that reported by Davenport (2008a).

Table 6.1 Total economic value of the high use zone (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Livestock	1,168,142	141,477	873	106	5,106,522	618,467	3,817	462
2	Bush meat	403,110	48,822	301	36	1,999,594	242,177	1,494	181
3	EC Rocky conservation	275,872	33,412	206	25	-	-	-	-
4	Leopard conservation	207,380	25,116	155	19	-	-	-	-
5	Avoided deforestation	49,961	6,051	37	5	-	-	-	-
6	Fuel wood	14,062	1,703	11	1	827,169	100,181	618	75
7	Medicinal plants	1,929	260	1	0	22,226	3,001	17	2
	Total	**2,120,455**	**256,842**	**1,585**	**192**	**7,955,512**	**963,825**	**5,946**	**720**

Table 6.2 Total economic value of the low use zone (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Livestock	1,442,231	174,673	873	106	6,304,700	763,582	3,816	462
2	EC Rocky conservation	633,150	76,683	383	46	-	-	-	-
3	Bush meat	497,694	60,277	301	36	2,468,773	299,001	1,494	181
4	Leopard conservation	475,954	57,644	288	35	-	-	-	-
5	Honey production	229,444	27,789	139	17	128,489	15,562	78	9
6	Fuel wood	17,361	2,103	11	1	1,021,253	123,687	618	75
7	Medicinal plants	2,381	322	1	0	27,441	3,705	17	2
	Total	**3,298,216**	**399,490**	**1,996**	**242**	**9,950,657**	**1,205,536**	**6,023**	**730**

[1] NB: Davenport (2008a) used a total commonage area of 2,900 ha, whereas this thesis assumed an area of 2,989 ha based on available maps for the commonage. To account for this discrepancy and ensure comparability across the two methods, the total value reported by Davenport was divided by 2,989 ha.

Table 6.3 Total economic value of Waters Meeting NR (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Ecotourism	3,522,983	426,679	2,438	295	8,200,990	993,247	5,675	687
2	EC Rocky conservation	314,915	38,140	218	26	-	-	-	-
3	Leopard conservation	236,729	28,671	164	20	-	-	-	-
4	Bush meat excl. game	135,251	16,381	94	11	219,278	26,557	152	18
5	Honey	96,333	11,667	67	8	53,947	6,534	37	5
6	Fuel wood	27,778	3,364	19	2	1,634,011	197,900	1,131	137
7	Medicinal plants	2,349	317	2	0	27,068	3,655	19	3
	Total	**4,336,338**	**525,220**	**3,001**	**363**	**10,135,294**	**1,227,893**	**7,014**	**850**

Table 6.4 Total economic value of the study site (Constant 2008 Rand, US$)

Rank	Service	Annual production value				Standing stock value			
		Rand	US$	Rand/ha	US$/ha	Rand	US$	Rand/ha	US$/ha
1	Ecotourism	3,522,983	426,679	794	96	8,200,990	993,247	1,849	224
2	Livestock	2,610,373	316,150	589	71	11,411,223	1,382,048	2,573	312
3	EC Rocky conservation	1,223,937	148,235	276	33	-	-	-	-
4	Bush meat	1,036,056	125,480	234	28	4,687,646	567,735	1,057	128
5	Leopard conservation	920,062	111,432	207	25	-	-	-	-
6	Honey	325,778	39,456	73	9	182,436	22,095	41	5
7	Fuel wood	59,201	7,170	13	2	3,482,434	421,768	785	95
8	Avoided deforestation	49,961	6,051	11	1	-	-	-	-
9	Medicinal plants	6,659	899	2	0	76,735	10,361	17	2
	Total	**9,755,010**	**1,181,552**	**2,200**	**266**	**28,041,463**	**3,397,255**	**6,323**	**766**

6.2.1 Comparing the results from the benefit transfer and field valuations

Only two ecosystem services were valued through both the benefit transfer and field valuation exercises, namely willingness to pay (WTP) for endangered species conservation and honey production. The values estimated by each method are presented in Table 6.5 – Table 6.7. Based on these results, it is clear that not only are local residents' WTP much higher values to protect locally endangered species than those predicted by the benefit transfer exercise, but also that the potential annual value of honey production in the low use zone is considerably higher than that predicted based on small-scale honey production experience elsewhere in South Africa.

Table 6.5 Comparison of values in the high use zone (Constant 2008 Rand, US$)

Service	Annual production value				Standing stock value			
	Field data		Benefit transfer		Field data		Benefit transfer	
	Rand	US$	Rand	US$	Rand	US$	Rand	US$
EC Rocky conservation	275,872	33,412	42,122	5,101	-	-	-	-
Leopard conservation	207,380	25,116	72,460	8,776	-	-	-	-
Honey	0	0	141,631	17,153	0	0	267,769	32,430

Table 6.6 Comparison of values in the low use zone (Constant 2008 Rand, US$)

Service	Annual production value				Standing stock value			
	Field data		Benefit transfer		Field data		Benefit transfer	
	Rand	US$	Rand	US$	Rand	US$	Rand	US$
EC Rocky conservation	633,150	76,683	52,005	6,298	-	-	-	-
Leopard conservation	475,954	57,644	89,462	10,835	-	-	-	-
Honey	229,444	27,789	174,862	21,178	128,489	15,562	330,597	40,040

Table 6.7 Comparison of values on Waters Meeting NR (Constant 2008 Rand, US$)

Service	Annual production value				Standing stock value			
	Field data		Benefit transfer		Field data		Benefit transfer	
	Rand	US$	Rand	US$	Rand	US$	Rand	US$
EC Rocky conservation	314,915	38,140	227,289	27,528	-	-	-	-
Leopard conservation	236,729	28,671	390,994	47,355	-	-	-	-
Honey	96,333	11,667	150,256	18,198	53,947	6,534	284,076	34,405

Table 6.8 shows the percent change between the honey production values estimated through field data collection and those estimated through benefit transfer. Since the allocation of honey production value between land use zones was highly dependent on proportional differences in woody biomass or *Scutia myrtina* density in the benefit transfer and field valuation exercises, respectively, the large discrepancies between the predicted and actual values for the high use zone and Waters Meeting NR may be due in large part to the low density of flowering *S. myrtina* individuals inside the reserve and the lack of any *S. myrtina* plants recorded in the high use zone. At the same time, the lower value of standing stock estimated by the field valuation exercise compared with the benefit transfer estimate reflects the production cost savings made possible by constructing homemade bee hives, rather than purchasing them commercially. Although a single beehive with three supers costs roughly R

400 (US$ 48) to purchase, a local beekeeper produces his for just R 117 (US 14.14) on average, representing a nearly 70 % savings in production costs.

Table 6.8 Comparison of honey production values (% change of field over benefit transfer)

Land use zone	Annual production	Standing stock
High use zone	-100%	-100%
Low use zone	31%	-61%
Waters Meeting NR	-36%	-81%

However, it is worth noting that the predicted value of annual production in the low use zone is nearly one-third higher than that predicted by the benefit transfer exercise. Based on the experience of other beekeepers in the area, this suggests that there may be important income generation opportunities possible through the sustainable exploitation of existing nectar sources on the commonage. Given that the yield estimates used for the field valuation were based on those obtained by a commercial beekeeper in the region, the field estimated annual revenue potential likely represents an upper bound that might be achieved as local beekeepers gain experience.

Nonetheless, the steep terrain and distance from the township makes the low use zone considerably less accessible to local Nolukhanyo residents than the high use zone. To reduce harvesting labor and average costs per kilogram of honey collected, periodic honey collections could be organized as a group activity, which may also facilitate greater knowledge sharing within the group. Novice beekeepers would also have to learn proper harvesting methods that minimize accidental fires and damage to the surrounding vegetation. These implementation issues notwithstanding, the valuation results imply that small-scale honey production could be a viable income-generating enterprise on the commonage with minimal environmental impact for at least 150 local entrepreneurs that can co-exist with other conservation or direct resource collection land uses. Moreover, assuming beekeepers are from different households, the estimated annual revenues of R 1,458 (US$ 545) per beekeeper would augment commonage income across all user households (R 3,828 or US$ 424) by roughly 5 % (Davenport 2008a).

As shown in Table 6.9, the results of the WTP survey administered to residents of the Kowie River catchment strongly suggest that, despite significant income disparities between the study site and the original communities surveyed by international studies sampled for the benefit transfer exercise, local residents would be willing to support initiatives to clean up the

Kowie River and protect locally endangered species, such as leopard and the Eastern Cape Rocky, at a much higher level than that predicted based on the international literature. This is in line with the findings of Turpie (2003), who found that respondents' WTP for biodiversity in the fynbos floral kingdom of South Africa's Western Cape province was comparatively high relative to income levels in the study area that are significantly below average incomes in Europe or North America.

Table 6.9 Comparison of annual household WTP for endangered species (Constant 2008 Rand, US$)

Animal valued	Leopard WTP		Rocky WTP	
	Rand	US$	Rand	US$
Kowie River catchment (actual)	128	15.52	144	17.41
Predicted value (benefit transfer)	16.13	1.95	9.38	1.14
Actual WTP over predicted		694%		1,435%

Based on the field valuation of WTP for both leopard and the Rocky, it appears that residents of the Kowie River catchment would be willing to pay nearly 150 times the value predicted by the benefit transfer to conserve the endemic and endangered Eastern Cape Rocky. Contrary to expectations that local residents would value the charismatic leopard more highly than the relatively unknown Rocky, the results indicate that local households are WTP roughly seven times more for leopard conservation than that predicted by the benefit transfer but still 11 % lower than they are WTP to protect the Rocky. As discussed previously (section 5.3.3.6) this may be a response to the Rocky's heightened vulnerability to habitat loss given its endemism to just three rivers and its supposed role as an indicator of overall river health. Similarly, Turpie (2003) found that educating respondents on the predicted impacts of climate change on South African biomes resulted in a five-fold increase in their WTP to protect national biodiversity, indicating that WTP is strongly influenced by perceived levels of threat.

Since any valuation exercise is likewise dependent on the assumptions included, and the applicability of values estimated through benefit transfer is also subject to the comparability of the original context to the study site (Desvousges, *et al.* 1998; Loomis 1992; Troy & Wilson 2006), it is not possible to extrapolate the discrepancies between the field and benefit transfer values for these two ecosystem services to the other ESVs valued through benefit transfer. However, it is clear from these two comparisons that important predictions about the distribution of natural capital, in particular woody biomass and flowering plants, between the three land use zones differed considerably from the observed allocation across the study site.

As reported in Chapter 3, the standing stock of woody biomass in the low use zone of the commonage may be noticeably higher than that within Waters Meeting NR, with concomitant implications for ESVs.

More importantly, given the explicit reference to the impact of commonage management on the health of the Kowie River catchment as a whole in the household survey, the emphatic response clearly indicates that there is significant potential for designing an appropriate payment for ecosystem services (PES) project in Bathurst. Possible PES projects to improve the health of the catchment might include (i) implementing a cattle-dipping program to increase livestock health and monitor herd size as part of enhanced livestock management on the commonage, which would depend crucially on municipal enforcement of maximum herd size regulations to reduce grazing pressure and allow vegetative regrowth to stablize soils and reduce run-off into the Kowie River; (ii) establishing a woodlot to substitute for natural timber and thus encourage woody plant growth and carbon sequestration; (iii) developing alternative sustainable livelihood options, such as small-scale honey production, that would reduce pressure for direct natural resource collection and facilitate, for example, enhanced biodiversity; or implementing a job creation program to remove invasive alien fish from the Kowie River and protect the endemic Eastern Cape Rocky. Given that 77 % of Nolukhanyo households surveyed by Davenport (2008a) depend on government social welfare grants, including pensions, child grants, and/or disability grants, any PES scheme would need to account for existing social grants in the selection of beneficiaries and also consider how best to integrate PES payments into existing grant distribution mechanisms. Other implications of the valuation results are presented in the next section with a view toward improving land use management on the study site to maximize revenue potential without compromising long-term ecosystem health.

6.2.2 Balancing revenues from direct and indirect ecosystem goods and services

As shown in Table 6.10 (adapted from Davenport 2008a), roughly 17 % of Nolukhanyo households subsist on less than US$ 1/day, 63 % survive on less than US$ 2/day (Ndlambe IDP 2007), and 80 % live below the national indigence line (StatsSA 2007). In this impoverished context, Davenport (2008a) found that, on average, Nolukhanyo residents derive annual revenues of roughly R 3,828 (US$ 464) per household from commonage resources, representing roughly 14 % of total their livelihoods. The remainder of local livelihoods was derived from state welfare grants (47 %), employment (30 %), remittances (5

%) and natural resources derived from small home gardens or other off-commonage sources (3 %) (Davenport 2008a).

Table 6.10 Proportion of households (%) living in poverty with and without commonage resources

Benchmark	Poverty threshold (R)	Including commonage resources	Excluding commonage resources
International poverty line – US$ 1 pppd	2,905 pppa	17	27
International poverty line – US$ 2 pppd	5,811 pppa	63	73
Poverty line (Carter & May 1999)[1]	4,654 pppa[2]	17	23
Poverty line (BMR 2003)[1]	5,806 pppa[2]	33	47
Indigence line (StatsSA 2007)	30,240 phpa	80	93
Mean		**42**	**53**
Standard deviation		29	30

[1]Shackleton 2005; [2]Per adult equivalent.
NB: "pppd" = per person per day; "pppa" = per person per annum; "phpa" = per household per annum.
Here US$ 1 = R 7.96; dated poverty lines were adjusted to 5 % inflation per annum (Davenport 2008a).

Given the official municipal unemployment rate of nearly half (46.5 %, Ndlambe IDP 2007) and lack of arable land for deriving alternative livelihoods, it is unsurprising but nonetheless significant that fully 70 % of local Nolukhanyo households (1,232 households) reportedly access one or more natural resources from the Bathurst commonage annually and a quarter relied extensively on just one commonage resource to support their livelihoods (Davenport 2008a). It is therefore evident that municipal commonage resources constitute an important livelihood source for local households, which, if lost due to unsustainable natural resource use rates, could force an additional 10 – 13 % of local households below the poverty threshold (Davenport 2008a). As such, it is imperative that the Bathurst commonage be managed in a sustainable manner that maximizes income generation opportunities from non-extractive and/or indirect ecosystem goods and services to reduce direct resource collection pressure without compromising local livelihoods.

The aim of this thesis was therefore to value both direct and indirect ecosystem services with a view toward identifying potential trade-offs between different levels of direct resource consumption. In total, nine ecosystem services were valued for the study site as a whole: (% total annual production value, % total standing natural capital value) ecotourism (36.1 %, 29.2 %), livestock production (26.8 %, 40.7 %), Eastern Cape Rocky conservation (12.5 %, n.a.), bush meat production (10.6 %, 16.7 %), leopard conservation (9.4 %, n.a.), honey production (3.3 %, 0.7 %), fuel wood production (0.6 %, 12.4 %), avoided deforestation (0.5 %, n.a.), and medicinal plants (0.1 %, 0.3 %). Overall, annual production values derived from conservation activities (ecotourism, endangered species protection, and avoided deforestation) account for roughly 59 % of the total (Table 6.11).

Table 6.11 Proportional contribution (%) of each ESV to overall value on the study site

Rank	Service	% Total annual production value	% Total standing stock value
1	Ecotourism	36.1	29.2
2	Livestock production	26.8	40.7
3	EC Rocky conservation	12.5	n.a.
4	Bush meat production	10.6	16.7
5	Leopard conservation	9.4	n.a.
6	Honey production	3.3	0.7
7	Fuel wood production	0.6	12.4
8	Avoided deforestation	0.5	n.a.
9	Medicinal plants	0.1	0.3
	Conservation	**58.5**	**29.2**

For comparison, conservation accounts for fully 94 % of the total annual production value of Waters Meeting NR, of which 86.5 % (roughly 81 % of the total annual value) is derived from ecotourism. The high use zone, by contrast, derives just over 25 % of its total annual production value from conservation land uses, while livestock and bush meat production contribute roughly 55 % and 19 %, respectively, to the total annual economic value of the high use zone. Although conservation accounts for a slightly higher proportion of total annual value (roughly 34 %) in the low use zone, livestock production remains the single most important source of revenue in the low use zone (about 44 % of the total value of annual production).

In light of the magnitude of direct compared with indirect ecosystem service values derived from natural capital on the commonage, it is unlikely that local users could obtain comparable incomes solely from indirect ESVs, such as ecotourism or payments for conserving the Eastern Cape Rocky or local leopard. The total annual revenue derived from the five direct provisioning services valued on the commonage was estimated to be roughly R 3.8 million (US$ 457,426), compared with the R 4.7 million (US$ 571,648) estimated for six goods valued by Davenport (2008a). In contrast, the total value of indirect existence and regulating services was estimated to total just R 1.6 million (US$ 198,906). Nonetheless, this valuation exercise suggests a number of possible land use management options that could potentially increase the total value of ecosystem services derived from the study site while improving or at least maintaining current ecosystem health.

To this end, it is significant that the potential value of avoided deforestation in the high use zone is estimated to be nearly 2.6 times the current value of fuel wood harvested from this part of the commonage on a proportional area basis. As the high use zone is more accessible in terms of both distance and terrain to local harvesters in Nolukhanyo, it is likely that the actual proportion of fuel wood harvested from the high as compared to low use zone is actually larger than that reported here, suggesting that the estimated value may under-represent the full extent of fuel wood benefits derived from the high use zone. Nonetheless, the magnitude of the difference between the non-extractive and sustainable revenue from reducing deforestation by just 5 % of current extraction rates strongly suggests that a compensated reduction scheme could offer an important alternative land use option that would at least replace and could potentially surpass current incomes generated from fuel wood collection in the high use zone while contributing to more sustainable future benefit streams derived directly or indirectly from woody plants.

Moreover, as discussed in the previous chapter, there appears to be considerable potential for increasing tourism revenues from Waters Meeting NR. The results indicate that, under a low capital scenario whereby existing accommodation facilities are expanded to cater for 69 budget guests per night, the reserve's existing natural capital (e.g. game) and service level in terms of activities offered and accommodation grading could generate as much as R 3.5 million (US$ 426,679) per year, compared with the estimated R 35,094 (US$ 4,250) currently earned annually. This value is especially noteworthy in light of the estimated revenue potential of the most likely alternative use of the reserve's game resources (e.g. bush meat hunting). Even though the bush meat valuation exercise predicted a higher standing stock value than the field valuation of game animals for ecotourism, the bush meat valuation indicated that these game would produce just R 706,100 (US$ 85,518), only 20 % of the estimated annual revenues from ecotourism (Table 6.12).

Table 6.12 Comparison of ecotourism and game meat values

Service	Annual production		Ecotourism vs. bush meat	Standing stock		Ecotourism vs. bush meat
	Rand	US$	% difference	Rand	US$	% difference
Ecotourism	3,522,983	426,679	399%	1,517,012	183,730	-62%
Game meat[1]	706,098	85,518		3,965,753	480,304	

[1]Here includes only those game animals currently present on Waters Meeting NR (WMNR 2007).

While the expansion of existing accommodation facilities and related staff capacity would no doubt entail considerable initial investment in both built and human capital, the potential development effects of this amount of revenue on the local economy cannot be overlooked. In addition to creating new jobs within the reserve itself, for example additional tour guides and housekeeping staff, a successful ecotourism venture would have substantial multiplier effects on the local economy through the creation and/or augmentation of related enterprises, such as restaurants, arts and crafts, outdoor equipment, and other accommodation venues (ABSA 2003).

In fact, the current Strategic Management Plan (WMNR 2007) for Waters Meeting NR proposes to expand existing overnight and canoe facilities, as well as to add a tea garden at the viewpoint overlooking Horseshoe Bend and an 'up-market leisure' facility in the far southeast corner of the reserve. Thus, with a well-implemented tourism development strategy, it is possible that, even without the Big Five, Waters Meeting NR could achieve considerably greater tourism revenues by introducing more overnight visitors to its unique vegetation, myriad animals, and scenic vistas. This would not only encourage greater appreciation for the historic reserve's natural capital resources, but could also contribute to job creation in an area where the official unemployment rate is nearly 46.5 % and roughly half of local household incomes are derived from state welfare grants (Davenport 2008a; Ndlambe IDP 2007).

6.3 <u>Toward integrated land use planning based on PES and enhanced local user rights</u>

Given that Waters Meeting NR shares a long border with the Bathurst commonage, land use planning on the study site should ideally incorporate incentives to promote conservation-friendly land uses on the commonage while expanding income generation opportunities for local residents on Waters Meeting NR. The former could be accomplished by augmenting revenues generated from non-extractive and indirect ecosystem services while substituting some direct resource collection activities with payments for ecosystem services (PES) programs. In particular, there appears to be considerable opportunity for promoting small-scale honey production in the low use zone of the commonage, with annual revenues of R 229,444 (US$ 27,789) that could potentially support 150 local entrepreneurs with an additional annual income of R 1,458 (US$ 545), a significant sum in the Bathurst context, where few alternative livelihood options outside the commonage are available. In addition, the results of this valuation exercise suggest that local residents could earn an additional R 49,961 (US$ 6,051) per annum (nearly 2.6 times the value currently derived from fuel wood collection attributed to the high use zone) by reducing their wood off-take in the high use zone by just 5 % through a PES scheme that compensates them for preventing the release of carbon that would have occurred under a ‘business as usual’ scenario.

Meanwhile, enhancing the value of Waters Meeting NR for local residents could be achieved through, for example, direct employment on the reserve and/or the establishment of a community resource monitoring campaign on the commonage, which would augment the capacity of the existing game rangers on the reserve (five for the entire 4,421 ha including Waters Meeting II and the Kowie Nature Reserve). In fact, the most recent Strategic Management Plan (SMP) for Waters Meeting NR (WMNR 2007) includes as one of its objectives the reserve’s consolidation within external planning frameworks, including the Ndlambe Municipal Spatial Development Framework. The SMP (WMNR 2007) proposes to accomplish this through consultation with Ndlambe Municipality “regarding the possibility of assigning appropriate zoning for the Bathurst Commonage, which is not in conflict with [Waters Meeting NR].” However, considering the admirable progress that South African National Parks has made toward integrating local communities into reserve planning and management (see, e.g. Child 2004; Hall-Martin & Carruthers 2003), Eastern Cape Parks, which is responsible for managing Waters Meeting NR, would likely do well to incorporate lessons learned at the national level in terms of the improved community-park relations made possible through co-management and/or benefit-sharing arrangements with local communities.

Revenues earned through enhanced ecotourism exploitation of the reserve could be redistributed to Nolukhanyo residents who engage in resource monitoring and enforcement on the commonage side of the fence to augment existing reserve staff capacity and promote sustainable natural resource use on the neighboring commonage. In fact, evidence from thirteen international case studies suggests that involving local individuals and communities in monitoring ecological indicators can offer a relatively inexpensive approach to effective surveillance compared with data collection by professional scientists that “can identify underlying temporal or spatial variation in biological resources” (Danielsen, *et al.* 2005), albeit with some data precision concerns that require attention, such as higher variance and less accuracy identifying more difficult taxa (Brandon, *et al.* 2003; Genet & Sargent 203). Community-based monitoring approaches must also be carefully designed to ensure that, at a minimum, the benefits from participating in a local monitoring system exceed the costs (Hockley, *et al.* 2005; Topp-Jørgensen, *et al.* 2005), and the indicators are locally-relevant, such as the abundance of commonly-harvested species (Danielsen, *et al.* 2005), water supply

(Becker, *et al.* 2005), or aesthetic values (Bennun, *et al.* 2005; Roberts, *et al.* 2005; Topp-Jørgensen, *et al.* 2005).

These challenges notwithstanding, experience suggests that even "relatively simple participatory methods" can be important for tracking long-term trends and "creating a context for community discussion of formal wildlife management" (Noss, *et al.* 2005). Globally, community-based monitoring approaches have been shown to improve cooperation between local and government stakeholders (Andrianandrasana, *et al.* 2005; Topp-Jørgensen, *et al*. 2005; van Rijsoort & Jinfeng 2005) and enhance awareness of and support for sustainable natural resource management among local participants (Kerr, *et al.* 1994; Obura, *et al.* 2002; Ticheler, *et al.* 1998). Moreover, by integrating local monitoring systems into existing land use management structures and/or linking them to the delivery of ecosystem goods and services to the local community, it appears that local resource users and government authorities alike are able to respond to immediate threats promptly and effectively (Becker, *et al.* 2005; Danielsen, *et al.* 2005; NORDECO & DENR 2002; Topp-Jørgensen, *et al.* 2005; Townsend, *et al.* 2005).

In addition to enhanced ecotourism that could underpin (and be supported by) community-based monitoring, Waters Meeting NR could feasibly support not only small-scale (non-extractive) honey production, but also bush meat and fuel wood harvesting that is managed within sustainable limits and contributes to landscape-scale conservation (Ashley, *et al.* 2006; McKean 2001, 2003; Mwalukomo 2007; Shackleton 1990; Traynor 2008). Extractive natural resource collection is not currently addressed by the reserve's conservation mandate based on philosophical, rather than biological, grounds that follow from South Africa's history of protected area conservation which "had more to do with the ideology of politicians living at the time than with the preservation of the environment" (Msimang 2000). However, experience around the world indicates that conservation areas delineated by hard boundaries, such as Waters Meeting NR, essentially displace resource collection activities elsewhere (Griffiths 2007; Kellert, *et al.* 2000; Wunder, *et al.* 2008), in this case to the commonage, and, in so doing, can "serve to isolate patches within an intensified agricultural landscape, or even backfire to degrade the area under protection" (Ashley, *et al.* 2006; Spiteri & Nepal 2008). The results reported here suggest that the natural resources in the high use zone of the Bathurst commonage have been significantly affected by a century of heavy utilization, while the vegetation on Waters Meeting NR has been effectively protected but produced relatively few direct benefits to the local community. In contrast, a growing body of evidence from

South Africa (McKean 2001, 2003; Mwalukomo 2007; Shackleton 1990; Traynor 2008) and beyond (Ashley, *et al.* 2006; Bauer 2003; Gadd 2005; Salafsky & Wollenberg 2000; Schroth, *et al.* 2004) suggests that integrating protected areas with their surrounding landscapes through, for example, carefully-monitored harvesting of thatch grass or bush meat, can contribute to the livelihood needs of local communities while also contributing to the achievement of their conservation goals.

By supporting local access to key livelihood resources within the reserve, such as fuel wood and bush meat, future management plans for Waters Meeting NR could reduce pressure on commonage resources (Hutton & Leader-Williams 2003; Spiteri & Nepal 2006) and thereby facilitate landscape-level conservation through increased connectivity between subtropical thicket protected within the reserve and on the neighboring commonage (van Noordwijk, *et al.* 2001). Global evidence clearly indicates that sharing the benefits of protected areas with local communities, whether through direct resource access or indirect benefits like improved infrastructure, market access, or social and technical capital, reduces the social costs of conservation by offering alternative livelihood options and enhancing community-park relations (Brockington 2002; Griffiths 2007; Igoe 2004; Peskett, *et al.* 2006; Peskett & Harkin 2007). Although it is important to note that approaches to incentivize conservation through local benefit sharing are not a "panacea for conservation and development" (Barrett, *et al.* 2005; Kruger 2005; Spiteri & Nepal 2008), experience suggests that complementing effective protected area surveillance with community participation and equitable benefit sharing can reduce pressure on resources in the surrounding area (Hutton & Leader-Williams 2003; Spiteri & Nepal 2006) and lead to long-term sustainability by encouraging local investment in land and natural resources (Carriere-Buschenschutz 2004; Dkamela 2001; Oestricher, *et al.* 2009; Spiteri & Nepal 2008).

Achieving catchment-wide improvements in ecosystem health would no doubt require coordination with other land uses along the Kowie River, in particular input-intensive commercial pineapple farming around Bathurst and commercial and residential development at the river mouth in Port Alfred. Nonetheless, the results of the contingent valuation survey imply that the relatively wealthy residents of Grahamstown, Bathurst, and Port Alfred would be willing to financially support projects that promote conservation-friendly land uses on the study site to the benefit of locally endangered species and the broader Kowie River catchment. Given that PES payments are typically (i) dependent on the provision of exclusive private goods (e.g. water) or public goods for which a market exists due to regulation (e.g.

carbon) (Pagiola *et al.* 2007b) and (ii) tied to proxy variables that can be observed directly by ecosystem providers (Pagiola & Platais 2007), any PES project aiming to promote biodiversity conservation on the study site would likely need to identify sustainable land use practices that have been demonstrated to improve biodiversity in the catchment which could then be 'bundled' into a PES project that markets quantifiable improvements in overall ecosystem health, rather than biodiversity *per se*. To this end, it is recommended that future commonage management incorporate payments for silvo-pastoral practices that have been shown to not only increase private returns to livestock production but would also create positive externalities for other commonage users (Current, *et al.* 1995; Dagang & Nair 2003; Pagiola, *et al.* 2007b) and enhance biodiversity conservation in the Kowie River catchment ecosystem more broadly (e.g. Daily, *et al.* 2003; Lindell, *et al.* 2004).

6.3.1 PES for integrating silvo-pastoral practices into commonage management

Experience with paying for ecosystem services generated by agroforestry projects generally and silvo-pastoral practices, in particular, across Africa (e.g. Jindal, *et al.* 2006) and Latin America (e.g. Pagiola, *et al.* 2005b, 2007b) offers a number of lessons that could be applied to increase revenues derived from direct and indirect ecosystem services produced by natural capital on the Bathurst commonage. There is considerable evidence that silvo-pastoral practices, which promote integrated rangelands that incorporate both fodder plants, such as grasses and leguminous herbs, and woody plants, such as trees and shrubs, can lead to a number of potential benefits, including enhanced on-site productivity, carbon sequestration, biodiversity conservation (Jindal, *et al.* 2006; Pagiola, *et al.* 2007b). Integrating trees into rangeland systems has been shown to have positive impacts on grass species composition, yield, and soil nutrients, although heavy grazing may override these benefits (Abdallah, *et al.* 2008; Abule, *et al.* 2005). Reforestation more broadly has been shown to improve water quality and reduce soil erosion and sedimentation (Bruijnzeel 2004; Scherr, *et al.* 2004), "though the effect is variable and not always as clear-cut as often supposed" (Pagiola, *et al.* 2007b). Nonetheless, especially in water- or nutrient-poor regions, integrating trees into rangelands has been demonstrated to enhance overall pasture productivity by increasing the provision of not only fodder and shade (Chivaura-Mususa, *et al.* 2000), which is good for milk production, but also of other direct ecosystem goods, such as fruit and fuel wood (Current, *et al.* 1995; Dagang & Nair 2003; Kidanu, *et al.* 2004).

Crucially, silvo-pastoral practices could also lead to significant carbon sequestration in both the soil (Pfaff, *et al.* 2000; Takimoto, *et al.* 2009; Vågene, *et al.* 2005) and standing tree biomass (Lal 2003; Powell 2008; Takimoto, *et al.* 2008) that could be marketed separately or bundled within a single over-arching PES project. Jindal, *et al*. (2006) report on initial findings from 19 carbon sequestration projects across Africa and note that, in addition to donor support from *inter alia* the World Bank, the United States Agency for International Development (USAID), and the European Union, a handful of projects have already begun selling carbon credits directly to United Kingdom-based companies and sharing the carbon revenues with local farmers. Moreover, carbon financing can be used to meet secondary objectives, such as improving local livelihoods (Rosa, *et al.* 2003; Smith & Scherr 2002) and biodiversity conservation (Gutman 2003), important outcomes that are all too often underfunded in Africa (Jindal, *et al.* 2006).

In fact, by enhancing woody cover, silvo-pastoral systems have been shown to support higher biodiversity relative to traditional pastures (Daily, *et al.* 2003; Eshiamwata, *et al.* 2006; Harvey & Haber 1999; Horner-Devine, *et al.* 2003; Lindell, *et al.* 2004; Moguel & Toledo 1999; Ricketts, *et al.* 2001). Biodiversity conservation itself can also be marketed as a global public good through financing institutions like the Global Environment Fund, which funds silvo-pastoral projects in Nicaragua, Colombia, and Costa Rica through the Regional Integrated Silvopastoral Ecosystem Management (RISEM) Project (Pagiola, *et al.* 2005b; 2007b). In addition to offering additional financing options, enhanced biodiversity conservation achieved through implementing silvo-pastoral practices on the commonage could also provide additional wildlife habitat and potentially be used to propagate indigenous thicket species, which could encourage greater connectivity among the remaining isolated thicket clumps in the high use zone (e.g. Saunders & Hobbs 1991) and thereby contribute to broader subtropical thicket restoration efforts (e.g. Powell 2008).

Despite these numerous benefits, a number of technical obstacles have thus far prevented widespread adoption of silvo-pastoral practices, including low profitability from the farmer's perspective, time lags before the system becomes fully productive with concomitant lost opportunity costs, and high initial costs for tree planting in the absence of credit (Dagang & Nair 2003). The RISEM project was designed to overcome some of these challenges by compensating landholders for the provision of biodiversity conservation and carbon sequestration through silvo-pastoral practices (Pagiola, *et al.* 2005b; 2007b). To make payments contingent upon demonstrated improvements in ecosystem health, the project first developed indices of biodiversity and carbon sequestration under different land uses and then aggregated them into a single "Environmental Services Index" (ESI) that was measured across each landholders entire farm to ensure that improvements in one area were not offset be degradation elsewhere on the farm (a.k.a. 'leakage').

By establishing baseline scenarios and revealing possible relationships between different land use intensities and the provision of ecosystem goods and services, this thesis represents a first step toward calibrating a similar index for the Bathurst commonage which could then be used to reward improvements in overall commonage health based on a selection of relatively easily quantified indicators. Since these processes represent a significant share of the relatively high start-up costs typically associated with PES implementation (Wunder, *et al.* 2008, see below), it is anticipated that future efforts to implement PES on the study site could focus on designing cost-effective methods of quantifying these indicators (and/or identifying indicators directly related to the desired ecosystem goods and services derived from silvo-pastoral practices) and refining them into a consolidated index of ecosystem services. As this thesis has reduced the bulk of the start-up costs, the primary task required to begin PES would be negotiating the structure of the PES system, including the financing, governance, and terms under which participants will be paid.

Although difficult to compare across different socio-economic and ecological contexts, start-up costs for other PES systems have been as high as R 630 – 1,520 (US$ 76 – 184) per hectare (Wunder & Albán 2008). Once these start up activities are completed, however, recurrent costs tend to be an order of magnitude lower and may be as low as R 8 – 25 (US$ 1 – 3) per ha per year (Wunder, *et al.* 2008). Regardless of the method employed, international experience suggests that while maintaining healthy ecosystems tends to be expensive, conservation is nevertheless much cheaper than undertaking environmental restoration projects (Wunder, *et al.* 2008).

This is especially the case in the context of subtropical thicket, where restoration efforts have proven to be a complex and expensive endeavor (Powell 2008). For example, three-year trials of thicket restoration through propagation of *Porticularia afra*, a keystone succulent plant species that has been shown to sequester large amounts of carbon relative to other plants in arid regions, achieved only 35 % survivorship (Powell 2008), and in general many thicket plant species tend to grow slowly (Turpie, *et al.* 2003). Although Powell (2008) notes that restoration costs in some environments could be limited to R 1,500 – 2,000 (US$ 182 – 242) per ha, a recent modeling of the economic viability of thicket restoration (Mills, *et al.* 2007) used estimated per hectare costs of R 5,000 (US$ 606) and anticipated that high mortality rates would require re-planting in the two years subsequent to the initial restoration intervention, suggesting that restoration could be prohibitively expensive in the context of South Africa's unique subtropical thicket ecosystem. As such, investments in land uses that protect the existing natural capital on the study site are likely to be much more efficient than restoration efforts (Wunder, *et al.* 2008).

Nonetheless, it is worth noting that despite the many advantages of implementing silvo-pastoral practices on the commonage, the successful realization of payments for ecosystem services (PES) in Africa is often constrained by a number of limitations, including poor governance, weak institutional capacity, high transaction costs, and tenure insecurity (Jindal, *et al.* 2006). PES financing depends fundamentally on credible guarantees to the investor(s), whether public (e.g. government, the World Bank) or private (e.g. corporations, individuals), that resource managers will make long-term improvements to their land use practices that enhance public goods of interest to a wider constituency, such as biodiversity conservation and carbon sequestration (Pagiola, *et al.* 2007b). As such, the shortcomings of municipal commonage management in South Africa generally (e.g. Atkinson 2005; Atkinson & Benseler 2004; Buso 2003) and at the study site in particular (Martens 2008) imply that local governance capacity will have to be augmented through formal (i.e. the local municipality authorities) and/or informal (i.e. the commonage management committee) institutions prior to or as part of PES implementation.

Moreover, as demonstrated by the projects reviewed by Wunder, *et al.* (2008), transaction costs represent a significant obstacle to implementing any PES scheme, and these costs are typically made even higher by the working with a large number of smallholders (Kerr, *et al.* 2006; Pagiola 2008). Many large-scale government-financed programs can achieve better cost efficiency than small-scale projects financed by private companies or individuals (Wunder, *et al.* 2008), as evidenced by Working for Water (Turpie, *et al.* 2008) in South Africa, the National Forest Commission (CONAFOR) in Mexico (Muñoz-Piña, *et al.* 2008), and the National Fund for Forest Financing (FONAFIFO) in Costa Rica (Pagiola 2008). In an attempt to reduce transaction costs through economies of scale, the United Nations Framework Convention on Climate Change (UNFCCC 2007b) recently implemented new guidelines that allow small-scale Clean Development Mechanism (CDM) projects that target low-income communities to be 'bundled' together. Thus, PES implementation on the

Bathurst commonage could be incorporated into the Working for Woodlands mandate to lower transaction costs (Pagiola 2008) and ideally contribute to broader community development.

Finally, and significantly for the Bathurst context, PES implementation is dependent on secure land and resource rights that allow ecosystem goods and services (EGS) suppliers to "make credible commitments to supply" the agreed upon EGS over the term of the contract (Gutman 2003; Jindal 2006). Tenure security has been demonstrated to be particularly important for long-term investments in EGS, such as tree planting (Arifin, *et al.* 2009; Meinzen-Dick, *et al.* 2002; Pagiola & Platais 2007). In fact, a meta-study of factors leading to successful agro-forestry adoption found that tenure variables were significant in nearly three-quarters (72 %) of the studies that measured them (Pattanayak, *et al.* 2003).

In addition to land rights, tenure over trees is equally important to successful agroforestry projects. For example, the results of recent research with participants of a community forestry scheme in the Sumber Jaya watershed in Indonesia "imply that farmer…would be willing to abide by fairly strict limitations on land use, provided that they can be assured of long-term rights to the planted trees (Arifin, *et al.* 2009). Similarly, Barrow (1990) documented how flexible rights to trees that could be used as supplemental fodder governed through the *Ekwar* tenure system in arid central Turkana, Kenya, were critical in allowing herders to spread risk during the dry season.

Although individual private property rights are not necessarily a pre-condition for successful PES implementation, in all but one of the thirteen international PES case studies reviewed by Wunder, *et al.* (2008), namely the Working for Water (WfW) program in South Africa, payments are made to the land holders. In the case of WfW, however, formerly disadvantaged individuals are compensated in the form of a cash wage for removing alien invasive plants from predominantly public lands, including protected areas, and more recently from some private farms, to which these individuals would not normally otherwise have access (Turpie, *et al.* 2008). In contrast, commonage users currently do not have sufficient tenure rights to manage and benefit from commonage resources conserved through a PES project (Fouche, pers. comm. 2008 in Davenport 2008a; Martens 2008). However, recent experience implementing a PES project on communal land in Mozambique suggests that well-defined common property rights could be sufficient to facilitate PES implementation on the Bathurst commonage (Jindal 2004).

6.3.2 Increasing incentives for sustainability via enforceable property rights

Despite the demonstrated ability of PES programs to induce cost-effective ecosystem health improvements by compensating land users for environmentally friendly behaviors to prevent degradation in the first place rather than restore degraded ecosystems (e.g. Forest Trends, *et al.* 2008; Pagiola, *et al.* 2007b; Wunder & Albán 2008), it is important to keep in mind that

> "PES is not a silver bullet that can be used to address any environmental problem, but a tool tailored to address a specific set of problems: those in which ecosystems are mismanaged because many of their benefits are externalities from the perspective of ecosystem managers". (Engel, *et al.* 2008)

As Engel, *et al.* (2008) note in the introductory paper to the special edition on PES in *Ecological Economics*, poor ecosystem management can result from a number of inefficiencies, including information gaps (Bulte & Engel 2006), credit gaps (Engel 2007), and situations where

> "local ecosystem managers [lack] the authority to manage ecosystems, because the ecosystems belong to nobody or to the state (which amounts to the same if the state is unable to enforce management rules) and thus tend to neglect even the on-site impacts of their management decisions (Ostrom 2003). The suitable response in this case would be to ensure that local ecosystem managers have appropriate property rights". (Engel, *et al.* 2008)

Therefore, PES implementation on the Bathurst commonage would in all likelihood require that ecosystem managers (i.e. commonage users) have the authority to monitor and if necessary enforce land use behaviors that lead to improved EGS, such as rotational grazing to augment herbaceous cover and protect the top soil or reduced wood collection to prevent carbon loss (Engel, *et al.* 2008). Municipal commonage is a classic common property resource (CPR) where sustainable management is particularly challenging because goods are both non-exclusive (i.e. it is difficult, if not impossible, to exclude users from accessing commonage resources due to technical deficiencies, such as a lack of proper fencing, and poorly enforced property rights) and subtractive (i.e. one household's use of commonage resources affects the supply of resources for all other households). It is therefore worth considering here the pivotal work of Ostrom (1990, 1994) who lists eight principles that are prerequisites for successful CPR management structures:

(i) "Membership and boundaries are clearly defined.

(ii) Rules that govern the appropriation of the resource and provision of inputs are sensitive to local conditions.

(iii) Collective choice arrangements allow most group members to participate.

(iv) Individuals who monitor the behavior of group members are accountable to the members.

(v) Appropriators who violate rules are likely to be punished according to the seriousness of their offence.

(vi) Resource users and officials have access to low-cost local arenas for the resolution of conflicts.

(vii) The right of resource users to organize is not challenged by external authorities.

(viii) Governance activities are organized in nested layers of enterprise (e.g. property rights granted to a local authority by the State can be further allocated to individual households or groups of resource users by the local authority)". (Swallow & McCarthy 1999)

Unfortunately, the prevailing land use planning and management structures on the commonage largely fall short of these prerequisites (Martens 2008). By law, municipal commonage is owned and managed by the local municipality, even though evidence from across South Africa clearly indicates that municipalities have limited capacity to design or enforce commonage plans (Anderson & Pienaar 2004; Atkinson 2005; Atkinson & Benseler 2004; Buso 2003). Notably, in addition to a 'community-based' commonage management committee (CMC), there is a commonage ranger assigned to monitor a permit system for controlling natural resource harvesting on the Bathurst commonage (Fouche, pers. comm. quoted in Davenport 2008a; Martens 2008).

These efforts constitute a considerable improvement upon other commonage governance structures in the area, which tend to be characterized by entirely open-access management regimes (Davenport 2008a). Nevertheless, the existing CMC is mostly populated and effectively controlled by emerging livestock farmers, who are typically more wealthy and powerful than subsistence commonage users characterized by their heavy reliance on direct natural resource collection, such as for obtaining fuel wood and wild foods, to support their livelihoods (Davenport 2008a; Martens 2008). Moreover, despite the existence of nominal rules governing grazing rights, the absence of effective monitoring and sanctioning for non-compliance has led to a situation where some cattle owners "grossly exceed" the 20 head maximum allocation by as much as 1,000 % (Fabricius, *et al.* 2006).

In general, the results of a recent evaluation of governance on the Bathurst commonage (Martens 2008) and of the ecosystem health evaluation presented here suggest that there is significant room for improving the effectiveness of commonage management by incorporating more of the principles outlined by Ostrom (1990, 1994). In particular, the governance of commonage resources would likely be improved in the short term by (i) expanding participation by all resource users in commonage land use planning and management and (ii) increasing incentives for community monitoring and enforcement. By empowering all users, not just powerful livestock owners, to make decisions about the management of commonage resources, more equitable representation on the CMC could lead to increased recognition of all commonage users' rights to and thus control over commonage resources, which would contribute to the development of appropriate rules to govern access and harvesting. Meanwhile, since individuals who are highly dependent on a given resource typically place greater value on its long-term sustainability than those whose livelihoods are not similarly resource-dependent (Gibson 2001), it follows that enhancing local subsistence users' rights to monitor and enforce locally-designed governance rules should improve compliance and thereby lead to more sustainable long-term commonage management to the benefit of all commonage users (Gibson, *et al.* 2005).

Ideally, commonage governance should promote direct management by local users through not only more equitable representation of all commonage users on the CMC and enhanced monitoring and enforcement rights assigned to subsistence users in the short term, but also by increasing the authority (via the CMC or another user community institution) of local users to participate in decisions on the maintenance and development of the commonage in the long term. This could improve the management of commonage resources by empowering users to draw on their knowledge of local conditions and positive incentives for ensuring the system's future sustainability to design, monitor, and enforce compliance with land use plans (Gibson, *et al.* 2005). This would then set the stage for PES implementation, which is typically contingent upon demonstrated adherence to defined land use practices voluntarily undertaken by land users (Wunder, *et al.* 2008).

Notably, experience with PES in Costa Rica (Miranda, *et al.* 2003) and Bolivia (Robertson & Wunder 2005) has shown that PES contracts actually had a positive influence on tenure security. The same may also be true in Kalimantan, Indonesia, where PES implementation in the context of weakly defined forest property rights is expected to enhance incentives for local communities to enforce their rights against external logging pressure by increasing the value of non-timber forest resources (Engel & Palmer 2008). Crucially, research elsewhere in Indonesia also indicates that enhanced land tenure security leads to greater investment in complex agroforestry systems (Suyanto, *et al.* 2005; Verbist, *et al.* 2005). Thus, successful

PES implementation on the commonage could potentially reinforce local users' rights to manage, access, and sustainably withdraw natural resources from the commonage.

It is important to note, however, that direct user control over natural resources does not automatically result in reduced harvesting pressure and may in some cases increase incentives for individuals to enforce their property rights to common property resources to the detriment of the community as a whole (Engel, *et al.* 2008). Wunder (2005) also warns that PES projects implemented in areas with insecure land tenure can result in land speculation that can effectively displace local land users. To ensure that incentives for conservation outweigh those for direct resource extraction without leading to local displacement, it is imperative that payments are sufficiently high to induce socially-beneficial land use behaviors (Pagiola 2005) and adequately targeted to prevent more powerful stakeholders from benefitting at the expensive of the vulnerable, known as 'elite capture' (Kerr, *et al.* 2006; Landell-Mills & Porras 2002).

Since all Nolukhanyo residents currently have *de facto* access to the commonage, it will also be important to ensure that payments are adequately targeted to ensure that they support actual improvements in land use behaviors on the commonage. As even a small number of 'holdouts' who refuse to participate in the PES project could potentially prevent other users from benefiting, one solution would be to establish an "ecosystem service district" to facilitate cooperation among all (current and potential) commonage users (Goldman, *et al.* 2007). This conservation institution, which usually employs a combination of regulations and incentives to induce environmentally friendly land use behaviors, has been successfully implemented in other agricultural landscapes (Goldman, *et al.* 2007; Heal, *et al.* 2001) and has recently been applied to carbon sequestration project on communal lands in Mozambique, where payments are made on a per hectare basis to a community fund (Jindal 2004; Jindal, *et al.* 2008). Although the creation of an ecosystem service district is typically voluntary, once a majority of residents agree, adherence to agreed upon rules would be mandatory. While applying the ecosystem service district approach to the Bathurst context could potentially overcome the holdouts problem, only careful compliance monitoring and enforcement can combat the emergence of 'free riders' who take advantage of improvements in ecosystem health without contributing to the project (Goldman, *et al.* 2007).

Even though PES projects are typically designed and implemented primarily as a mechanism to promote efficient natural resource management, there is some evidence that PES can have a positive impact on poor households. For example, experience implementing PES for silvo-pastoral practices in Colombia and Nicaragua suggests that poor households are at least as or more able to participate in PES than relatively more wealthy households (Pagiola, *et al.* 2007a, 2008). In fact, despite the fact that the RISEM project did not explicitly target poor households, Pagiola, *et al.* (2007a) found that poor and extremely poor households in Nicaragua contributed a substantial share of the land use changes attributed to the project, including 50 % of the decrease in degraded pasture. Poor households even implemented more complex and expensive silvo-pastoral practices, such as 71 % of fodder banks and 64 % of pastures with high tree density, than the non-poor, who preferred to implement simpler practices, such as low tree density pastures. Overall, quantitative evidence that demonstrates substantial PES benefits to the poor is lacking (Engel, *et al.* 2008). Nevertheless, the body of literature on the relationships between PES and poverty is growing (Grieg-Gran, *et al.* 2005; Kerr 2002; Landell-Mills & Porras 2002; Rosa, *et al.* 2003; Smith & Scherr 2002), and the fact that most PES programs are voluntary implies that participants must be at least no worse off than they were without the program (Pagiola, *et al.* 2005a; Wunder 2008).

The results of this thesis suggest that if commonage users are empowered to participate in the design, monitoring, and enforcement of rules governing the natural resources upon which on average 14 % of their livelihoods depend, the conservation-friendly residents of the Kowie River catchment might be willing to finance a R 2.1 million (US$ 259,666) PES program to support silvo-pastoral practices on the commonage. Payments for ecosystem services are typically made based on the cost of ecosystem service provision (Wunder, *et al.* 2008), which in this case would be the lost opportunity costs from reduced natural resource collection on the commonage. In keeping with the ecosystem service district model, if all Nolukhanyo households (1,760) received a one-time payment equal to one-quarter of the mean annual income per household derived from commonage resources (i.e. payment of R 957 or US$ 116 based on Davenport (2008a) and estimated per hectare start up and recurrent costs[2] were on the order of R 2,000 (US$ 242) and R 110 (US$ 13) per ha, respectively (Wunder, *et al.* 2008), this budget could support a PES project over roughly 218 ha of the commonage, presumably in an area adjacent to Nolukhanyo township that could be carefully monitored (Table 6.13).

[2] Start up costs include the procurement of pre-implementation information, such as linkages between land uses and ecosystem service provision and baseline ecosystem health indicators, as well as program design and negotiation costs. Recurrent implementation costs consist of monitoring compliance, sanctioning for non-compliance, payment administration, etc. (Cacho, *et al.* 2005; Pagiola & Platais 2007).

Though all future payments should be made conditional upon compliance with a sustainable resource collection plan and measured against actual reductions in household resource use (Wunder, *et al.* 2008), rewarding all households with equal benefits in year one reduces perverse incentives for households to overexploit commonage resources in an attempt to either demonstrate dependence (and thereby derive greater benefits for 'lost opportunities') or reduce the baseline ecosystem health scenario such that future payments be more easily obtained for 'reductions' in 'actual' natural resource use that lead to ecosystem health 'improvements' (Pagiola, *et al.* 2007b). Once the initial start-up costs are covered, each Nolukhanyo household could be eligible receive up to an additional R 248 (US$ 30) per year contingent upon their adoption of improved land use practices that have been linked to enhanced EGS provision (Wunder, *et al.* 2008), such as planting trees or shrubs adjacent to existing thicket clumps on the commonage or along the perimeter to reinforce the currently permeable fence line and potentially underpin a rotational grazing system to further improve commonage rangeland management.

Table 6.13 Estimated budget for a PES project on the Bathurst commonage

Budget item	Rand	US$
WTP = Total budget	2,144,000	259,666
Financing for payments @ R 957/hh)	1,684,320	116
Estimated total costs[1] (excl. payments)	459,680	55,673
Estimated start-up costs (R 2,000/ha)	435,786	52,779
Recurring costs (R 110/ha)	23,893	13
Additional payment/hh/yr after start-up	248	30
Area (ha) (Year 1, incl. start up costs)	218	

[1]Costs based on case studies summarized by Wunder, *et al.* (2008)

The obstacles to successful implementation of PES on the Bathurst commonage should not be underestimated. In particular, enhancing subsistence users' rights to define, monitor, and enforce sustainable land use plans will require not only overcoming the existing power inequalities between livestock owners and other commonage users, but also gradually shifting commonage management responsibilities away from the local municipality. Both of these transitions will require careful negotiation with all stakeholders to facilitate the empowerment of impoverished subsistence users without compromising the future cooperation of more powerful stakeholders.

Moreover, although this thesis has made a first attempt to quantify ecosystem goods and services (EGS) of possible interest to a PES project, considerable time and finances must still be devoted to the design and negotiation of a user-financed PES scheme whereby thousands of Kowie River catchment residents compensate Nolukhanyo households for land use practices that have been demonstrated to improve EGS provision on the commonage. Transaction costs could possibly be reduced by sourcing financing from future ecotourism revenues generated by Waters Meeting NR, but augmenting existing ecotourism revenues will itself require considerable expenditures for enhancing built and human capital on the reserve, not to mention marketing these improvements.

Nonetheless, the magnitude of potential annual revenues derived from conservation donations from Kowie River catchment residents (R 2.1 million or US$ 259,666) or ecotourism (R 3.5 million or US$ 426,679) suggest that a PES project on the study site could be financially viable. At the same time, the demonstrated discrepancies in both ecosystem health indicators and annual production values between, in particular, the high use zone of the commonage and Waters Meeting NR, indicate that there is considerable room for enhancing ecosystem service provision on the study site. Especially in the context of Bathurst, where commonage users have few alternative livelihood options, there should thus be substantial incentives for local stakeholders to overcome the challenges outlined above to support sustainable natural resource management on the study site while augmenting or at least maintaining local livelihoods through increased revenues from non-extractive land uses, such as honey collection in the low use zone, and payments for ecosystem services generated by, for example, silvo-pastoral practices in the high use zone.

6.4 Conclusion

In summary, the findings of this thesis may have a number of implications for the way that land uses on the Bathurst commonage and Waters Meeting NR could be managed in the future. The results of the ecosystem health evaluation clearly demonstrate that the ecological health of high use zone of the commonage is not unsurprisingly lower than that of both the low use zone and Waters Meeting NR. A history of intensive natural resource collection in the high use zone is evidenced by discrepancies in specific indicators between the different land use zones on the commonage, including mean grass height in the high use zone, which is less than one-third of that measured in the low use zone, and a 70 % higher cattle dung density in the high compared to low use zone.

A growing body of research suggests that the relationship between rangeland management and environmental 'degradation' is by no means straightforward (Allsopp, *et al.* 2007a; Benjaminsen, *et al.* 2006; Forsyth 2003), and spatial variation of geophysical characteristics, such as soil, vegetation, and microclimates, makes it difficult to draw unambiguous conclusions about the extent to which land use, rather than underlying ecological characteristics, has resulted in the observed differences in the ecological health indicators measured (Blackmore, *et al.* 1990; Schlesinger & Pilmanis 1998). Nevertheless, a number of differences were observed between the high and low use zones of the commonage in several important functional indicators of rangeland health, including soil organic content, moisture accessibility, compaction and fertility, as well as numerous status indicators, such as proportional aerial cover and woody plant layer characteristics. These differences in ecological health indicators measured on the commonage suggest that higher land use intensity in the high use zone, which is subjected to livestock grazing as well as other direct natural resource harvesting activities, has had significant and negative implications on its comparative ecological health. Though long-term monitoring would be required for confirmation, the functional indicators of rangeland health measured imply that the intensity of land use in the high use zone may negatively affect the ecosystem's "resilience", or its ability to withstand and recover from degradation (Pyke, *et al.* 2002; Rapport, *et al.* 1998).

In contrast, despite indications that the low use zone is subjected to higher land use intensity than Waters Meeting NR, such as a significantly shorter mean grass height and higher proportion of grassy biomass removed in the low use zone compared with the protected area, numerous ecological health indicators measured in the two land use zones showed no significant differences. For example, no significant differences were found between the two land use zones for the proportional aerial cover of woody plants, grass, forbs, bare ground, and litter; the mean and maximum basal diameter of woody plant stems, density of dead stems, proportion of sexually mature *Scutia myrtina* stems; or for all but one soil characteristic measured. Overall, these results imply that the intensity of land use in the low use zone has not significantly altered its ecosystem health judged against Waters Meeting NR, which may imply that the local subtropical thicket ecosystem is actually more resilient than would be suggested by recent research on arid landscapes when subjected to relatively low levels of human use (e.g. Holm, *et al.* 2005; Rapport, *et al.* 1998). The interesting observation that both woody plant cover and grassland cover varied significantly across years within the protected Waters Meeting NR, whereas the proportional areas covered by these vegetation classes were historically fairly consistent on the variously utilized commonage, would seem to suggest that the local ecosystem is characterized by a non-equilibrium model (Briske, *et al.* 2003; Fabricius, *et al.* 2006; Illius & O'Connor 1999).

At the same time, the results of the valuation exercise strongly imply that differences in the ecological health of the three land use zones on the study site affect ecosystem service provision and related value generation (Petrosillo, *et al.* 2007). Despite comparable ecosystem health indicators between the low use zone of the commonage and Waters Meeting NR, it was estimated that the standing stock of natural capital in the protected reserve is worth R 7,014 (US$ 850) per hectare, over 16 % higher than the per hectare standing natural capital stock measured in the low use zone. Interestingly, despite significant differences between the low and high use zones in the status of ecosystem health indicators measured by this research, the per hectare standing stock value of natural capital of the two commonage zones is roughly equal: R 6,023 (US$ 730) and R 5,946 (US$ 720) in the low and high use zones, respectively. This may indicate that resource managers on the commonage are actively using natural resources from different parts of the commonage in an attempt to maintain comparable levels of standing stock value across the commonage of those resources that are directly valuable to local livelihoods (Allsopp, *et al.* 2007a; Benjaminsen, *et al.* 2006).

Meanwhile, the annual revenue that could be derived from the standing stock of natural capital in Waters Meeting NR totals approximately R 3,000 (US$ 363) per ha, roughly half again as high as the per hectare annual revenue derived from natural resources in the low use zone (R 1,996 or US$ 242), which is itself 26 % above the value generated by the natural resources measured within the high use zone of the commonage (R 1,585 or US$ 192). This is significant given that 'healthy' ecosystems are so judged for their ability to provide critical ecosystem goods and services that support human life (Rapport, *et al.* 1998). Thus, the results of both the ecological health and valuation exercises suggest that while land use management in the low use zone of the commonage has not yet resulted in "a long-term negative effect on range resources" (Scoones 1994), the intensity of resource extraction in the high use zone, may have resulted in localized ecological changes which, in turn, appear to have affected the magnitude and value of ecosystem goods and services provided to the Nolukhanyo community.

In light of the fact that 17 % of Bathurst residents subsist on less than US$ 1/day and fully 70 % of Nolukhanyo households rely on the commonage for the provision of one or more ecosystem goods due to the limited local availability of alternative livelihood options (Davenport 2008a), it will be imperative to prevent further deterioration of and ideally improve the ecological health of the high use zone and at least maintain the health of the low use zone to ensure that the commonage continues to supply poor local households with sustainable livelihood opportunities (Ngwenya & Hassan 2005). Davenport (2008a) estimated that, should natural resource collection on the commonage become unsustainable such that these resources were no longer available to local households in Nolukhanyo, commonage users would be faced with a shortfall of on average over 14 % of their livelihoods. Thus, the loss of commonage livelihoods could force an additional 10 – 13 % of the community below the poverty threshold, bringing the total proportion of residents living on less than US$ 1/day to 27 % and increasing the share of the Nolukhanyo residents living below the national indigence line to an astounding 93 % (Davenport 2008a).

Since the results reported here indicate that current fuel wood harvesting activities, which support the livelihoods of fully 86 % of commonage users, may be outstripping sustainable re-growth rates, it will be imperative to quantify actual wood collection rates across the commonage and compare these to the annual growth rates of locally preferred woody plants. In light of the considerable revenue potential possible through a compensated reduction scheme, a carefully implemented wood resource management plan could not only reduce pressure on natural timber but also generate nearly three times as much revenue as that currently derived from fuel wood collection on the commonage. Though not quantified here due to insufficient data, Davenport (2008a) also notes that medicinal plant harvesting on the commonage should be monitored more carefully in light of sustainability concerns in South Africa and its contribution (27 %) to local users' commonage-derived incomes. Moreover, even though current levels of resource extraction in the low use zone appear not to have significantly altered ecosystem health, the potential for increasing revenues from honey production in the low use zone of the commonage should be considered in future commonage management. This non-extractive and potentially sustainable land use could generate as much as R 229,444 (US$ 27,789) per year, enough to support some 150 entrepreneurs with an additional annual income of R 1,458 (US$ 545) per beekeeper.

At the same time, it is clear that, despite the considerable value derived from the natural resources protected within Waters Meeting NR, there may be even greater potential for augmenting existing ecotourism revenue by expanding available accommodation within the reserve. Even under a low capital investment scenario that caters to domestic budget travelers, the current estimated annual revenue of roughly R 35,000 (US$ 4,250) could be increased to at least R 3.5 million (US$ 426,679) if an additional 53 beds were added to the reserve's existing 16-bed capacity. Given that ecotourism contributes the vast majority (81 %) of the total estimated annual production value of Waters Meeting NR, compared with a combined direct use value of less than 6 %, the further development of ecotourism could not only increase non-consumptive sustainable revenues derived from the study site, but also potentially lead to considerable job creation on and off the reserve through, for example, related tourism businesses (ABSA 2003) or a local resource monitoring campaign on the commonage.

In addition, sustainable harvesting of key natural resources, such as fuel wood, bush meat, and/or honey, within the reserve could reduce pressure on commonage resources (Hutton & Leader-Williams 2003; Spiteri & Nepal 2006) and contribute to landscape-scale conservation of subtropical thicket on the study site (van Noordwijk, *et al.* 2001). Although careful planning is required to effectively target beneficiaries and avoid elite capture (Timsina 2003; Walpole & Goodwin 2000), evidence from South Africa (McKean 2001, 2003; Mwalukomo 2007; Shackleton 1990; Traynor 2008) and elsewhere (Ashley, *et al.* 2006; Bauer 2003; Gadd 2005; Salafsky & Wollenberg 2000; Schroth, *et al.* 2004) has demonstrated that, by providing incentives for local investments in land and natural resources, sharing the benefits of protected areas with local communities can enhance livelihoods and encourage long-term sustainability. Regulated natural resource harvesting could be combined with a community monitoring program to measure ecological indicators across the study site and create a context for incorporating the Nolukhanyo community into sustainable land use management planning and implementation on the commonage and Waters Meeting NR (Danielsen, *et al.* 2005; Noss, *et al.* 2005).

Finally, the results of this thesis suggest that, despite the small area for which services were valued, there is considerable potential for the implementation of a payments for ecosystem services (PES) project that promotes silvo-pastoral practices on the commonage. By integrating trees and shrubs into pasture lands and encouraging alternative land use practices, such as fodder banks, to increase vegetative cover on the commonage, it is expected that incorporating silvo-pastoral practices into commonage management could lead to not only enhanced carbon sequestration (e.g. Vågen, *et al.* 2005) and biodiversity conservation (e.g. Eshiamwata, *et al.* 2006), but also increased provision of fuel wood, timber, fodder, and shade to increase rangeland productivity (Chivaura-Mususa, *et al.* 2000; Kidanu, *et al.* 2004). Since many of the benefits generated by silvo-pastoral practices accrue to stakeholders outside of Nolukhanyo, a PES approach would be ideal to provide positive incentives for adoption (Engel, *et al.* 2008). Furthermore, experience implementing payments for silvo-pastoral practices in Colombia and Nicaragua indicates that poor and extremely households are equally if not more able to participate in PES than their wealthier neighbors (Pagiola, *et al.* 2007b, 2008).

However, successful PES implementation on the commonage would require that local users have appropriate authority to manage this common property resource (Ostrom 2003). At present, the limited capacity of the single ranger assigned to monitor and manage the daily up-keep of all 2,989 ha of the commonage and overrepresentation of livestock owners, who account for only 21 % of all commonage users, on the commonage management committee (CMC) largely undermine the rights of the majority of commonage users who rely on commonage resources to support subsistence livelihoods (Davenport 2008a; Martens 2008). Evidence from common property resource management structures around the world suggests that subsistence commonage users, whose livelihoods depend directly on the continued availability of commonage resources, would have significant motivation to monitor agreed upon land use plans on the commonage and enforce compliance where necessary if empowered to do so through, at a minimum, equitable representation on the commonage management committee and enhanced rights to design, monitor, and enforce commonage management plans (Gibson, *et al.* 2005; Ostrom 1990, 1994).

Notably, most PES projects and valuations of ecosystem goods and services in South Africa have so far covered entire catchments or even biomes, especially where indirect services are concerned (e.g. Hassan 2003; Turpie 2003; Turpie, *et al*. 2008). In contrast, this thesis has made a first attempt to quantify baseline ecosystem health indicators at a micro-scale (< 5,000 ha) and link measured differences in ecosystem health between three distinct land use intensity zones with the variety and magnitude of local goods and services derived from natural resources under these different land use regimes (Wunder, *et al.* 2008). By quantifying the trade-offs between direct (e.g. fuel wood, bush meat, medicinal plants) and indirect (e.g. ecotourism, carbon sequestration, and endangered species conservation) ecosystem services (IIED 2007; MDTP 2008) at a micro-scale appropriate for informing local resource management institutions, this thesis has laid the foundation for the implementation of a small-scale PES project to promote better local land use management on the study site through the adoption of silvo-pastoral practices that would be underpinned by enhanced commonage user rights and community resource monitoring.

PES implementation on the commonage would still require significant transaction costs to refine the indicators identified here (or quantify new/additional variables) into an aggregate index of ecosystem services that could be used as a basis for making payments to Nolukhanyo households for quantifiable improvements in ecosystem health that have been demonstrated to lead to enhanced provision of the ecosystem goods and services (EGS) of interest to the 'buyers' supporting the project, such as carbon sequestration and biodiversity conservation (Pagiola, *et al.* 2007b). Negotiation with Kowie River catchment residents (the EGS 'buyers') and Nolukhanyo households (the EGS 'sellers') to agree on an appropriate structure for financing and executing payments for silvo-pastoral practices on the commonage will also be a time- and resource-intensive process that must carefully manage existing imbalances in knowledge and power between stakeholders (Kerr, *et al.* 2006; Landell-Mills & Porras 2002; Wunder, *et al.* 2008).

These challenges notwithstanding, the results of the contingent survey reported here indicate that residents of the Kowie River catchment would be willing to contribute as much as R 2.1 million (US$ 259,666) on an annual basis to support the conservation of locally endangered species, including the leopard and Eastern Cape Rocky, and promote enhanced river health throughout the catchment. Using estimated transaction costs based on international PES implementation experience, this sum could support a one-time payment equal to one-quarter of the mean annual income per household derived from commonage resources (i.e. payment of R 957 or US$ 116 based on Davenport 2008a) made to each Nolukhanyo household (1,760) in the first year of the project and an additional payment of up to R 248 (US$ 30) per household contingent upon demonstrated improvements in the ecosystem health of the commonage as a result of silvo-pastoral practices. In a ward where 16 % of residents earned less than R 4,800 (2008 US$ 648) and another 16 % no income at all in 2007 (Ndlambe IDP 2007), this could potentially have a significant impact on local livelihoods through the once-off infusion of additional income in the short term, increased provision of fuel wood and fodder in the medium-term, and enhanced sustainability of commonage resources in the long-term. Given that PES implementation in other developing countries has been found to increase tenure security (Miranda, *et al.* 2003; Robertson & Wunder 2005), the potential impact of the proposed PES project on local capacity building and economic development in Nolukhanyo should not be underestimated.

In conclusion, it has been shown that differences in the prevailing land use intensities on the study site are associated with substantial discrepancies in the ecosystem health of and provision of ecosystem goods and services derived from the three land use zones quantified by this study. In light of the considerable dependence of local livelihoods on commonage resources, it is recommended that future land use plans on the commonage emphasize the sustainable exploitation of non-extractive ecosystem services, such as honey production in the low use zone, while promoting the substitution of some direct natural resource collection, with revenues from, for example, a compensated reduction scheme to facilitate carbon sequestration and ensure the long-term sustainability of fuel wood resources in the high use zone. Moreover, although the natural capital protected within Waters Meeting NR has been found to generate considerably higher annual values than either the low or high use zones of the commonage, there is also significant potential for increasing revenue generation through the expansion of existing accommodation facilities on the reserve.

At the same time, the long border between the commonage and Waters Meeting NR implies that integrated land use management across the study site would improve resource management on both sides of the fence to facilitate the achievement of each parcel's primary objectives, namely direct natural resource provision to support subsistence livelihoods and emergent farmers and conservation, respectively. To this end, a payments for ecosystem services project that could potentially be financed by conservation-friendly residents of the Kowie River catchment and/or increased ecotourism revenues would provide positive incentives for improved land use practices on the commonage (e.g. Pagiola, *et al.* 2007b) and effective community resource monitoring (Danielsen, *et al.* 2005; Noss, *et al.* 2005) to ensure not only that land uses on the commonage at least do not jeopardize and ideally enhance conservation efforts inside Waters Meeting NR. Furthermore, there is ample evidence that allowing carefully designed (Timsina 2003; Walpole & Goodwin 2000) and monitored local access to natural resources within Waters Meeting NR could not only increase community support for conservation, but also reduce pressure on commonage resources (Hutton & Leader-Williams 2003; Schroth, *et al.* 2004; Spiteri & Nepal 2006). In combination, these approaches could lead to a more sustainable subtropical thicket landscape (Ashley, *et al.* 2006; van Noordwijk, *et al.* 2001) and ensure that critical natural resources remain available to support local livelihoods in the long-term (Ngwenya & Hassan 2005).

References

Abdallah, F., Noumi, Z., Touzard, B., Belgacem, A. O., Neffati, M., & Chaieb, M. (2008). The influence of *Acacia tortilis* (Forssk.) subsp. *raddiana* (Savi) and livestock grazing on grass species composition, yield and soil nutrients in arid environments of south Tunisia. *Flora: Morphology, Distribution, Functional Ecology of Plants, 203*(2), 116-125.

ABSA Group Economics Research (ABSA). (2003). *Game ranch profitability in South Africa.* South African Financial Sector Forum.

Abule, E., Smit, G. N., & Snyman, H. A. (2005). The influence of woody plants and livestock grazing on grass species composition, yield and soil nutrients in the Middle Awash valley of Ethiopia. *Journal of Arid Environments, 60*(2), 343-358.

Acocks, J. (1953). *Veld types of South Africa* No. 28). Pretoria: Government Printer.

Adamowicz, W., Beckley, T., MacDonald, D., Just, L., Luckert, M. K., Murray, E., et al. (1998). In search of forest resource values of indigenous peoples: Are nonmarket valuation techniques applicable? *Society and Natural Resources, 11*(1), 51-66.

African National Congress (ANC). (1994). *The reconstruction and development programme*. Johannesburg: Umanyano Publications.

Agarwal, C., & Ferraro, P. (2007). *Draft paper prepared for the Bellagio expert meeting.* Unpublished manuscript.

Allen, B. P., & Loomis, J. B. (2006). Deriving values for the ecological support function of wildlife: An indirect valuation approach. *Ecological Economics, 56*(1), 49-57.

Allsopp, N. (1999). Effects of grazing and cultivation on soil patterns and processes in the Paulshoek area of Namaqualand. *Plant Ecology, 142*(1-2), 179-187.

Allsopp, N., de Lange, W., & Veldtman, R. (2008). Valuing insect pollination services with cost of replacement. *PloS ONE, 3*(9), e3128.

Allsopp, N., Gaika, L., Knight, R., Monakisi, C., & Hoffman, M. T. (2007b). The impact of heavy grazing on an ephemeral river system in the succulent Karoo, South Africa. *Journal of Arid Environments, 71*(1), 82-96.

Allsopp, N., Laurent, C., Debeaudoin, L. M. C., & Igshaan Samuels, M. (2007a). Environmental perceptions and practices of livestock keepers on the Namaqualand commons challenge conventional rangeland management. *Journal of Arid Environments, 70*(4), 740-754.

Ambrose-Oji, B. (2003). The contribution of NTFPs to the livelihoods of the 'forest poor': Evidence from the tropical forest zone of south-west Cameroon. *International Forestry Review, 5*(2), 106-117.

Anderson, J., & Van Wyk, B. (1999). Biomes and hotpots of South Africa. In J. Anderson (Ed.), *Towards Gondwana alive: Promoting biodiversity and stemming the sixth extinction*. Pretoria: National Botanical Institute.

Anderson, M., & Pienaar, K. (2003). *Municipal commonage* No. 5). Cape Town: Institute for Poverty, Land and Agrarian Studies (PLAAS).

Andrew, M., Ainslie, A., & Shackleton, C. M. (2003). *Land use and livelihoods* No. 8). Cape Town: Institute for Poverty, Land and Agrarian Studies (PLAAS).

Andrews, M. (2007). Struggling for a life in dignity. In L. Ntsebeza, & R. Hall (Eds.), *The land question in South Africa: The challenge of transformation and redistribution* (Electronic ed., pp. 202-219). Cape Town: Human Sciences Research Council.

Andrianandrasana, H. T., Randriamahefasoa, J., Durbin, J., Lewis, R. E., & Ratsimbazafy, J. H. (2005). Participatory ecological monitoring of the Alaotra wetlands in Madagascar. *Biodiversity and Conservation, 14*(11), 2757-2774.

Arifin, B., Swallow, B. M., Suyanto, S., & Coe, R. D. (2009). A conjoint analysis of farmer preferences for community forestry contracts in the Sumber Jaya watershed, Indonesia. *Ecological Economics, 68*(7), 2040-2050.

Arrow, K., Solow, R., Portney, P. R., Lebert, T., Radner, R., & Schuman, H. (1993). *Report of the NOAA panel on contingent valuation* No. 1993;58:08/01/07-4601-14). Washington, D.C.: Federal Register.

Ashley, R., Russell, D., & Swallow, B. M. (2006). The policy terrain in protected area landscapes: Challenges for agroforestry in integrated landscape conservation. *Biodiversity and Conservation, 15*(2), 663-689.

Asquith, N. M., Vargas, M. T., & Wunder, S. (2008). Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. *Ecological Economics, 65*(4), 675-684.

Atkinson, D. (2005). *Commons Southern Africa: People-centred environmental management and municipal commonage in the Nama Karoo* No. 11). Cape Town: Institute for Poverty, Land and Agrarian Studies (PLAAS).

Atkinson, D. (2007a). The need for agricultural extension services for municipal commonage farmers. *South African Journal of Agricultural Extension, 36*, 190-202.

Atkinson, D. (2007b). People-centred environmental management and municipal commonage in the Nama Karoo. *Development Southern Africa, 24*(5), 707-724.

Atkinson, D., & Benseler, A. (2004). *Review of municipal commonage programme of the Department of Land Affairs*. Phillipolis: Heartland & Karoo Research Institute.

Atkinson, D., & Buscher, B. (2006). Municipal commonage and implications for land reform: A profile of commonage users in Philipolis, Free State. *Agrekon, 45*, 437-466.

Aune, J. B., Alemu, A. T., & Gautam, K. P. (2005). Carbon sequestration in rural communities: Is it worth the effort? *Journal of Sustainable Forestry, 21*(1), 69-79.

Aylward, B., Bandyopadhyay, J., & Belausteguigotia, J. (2005). Freshwater ecosystem services. In K. Chopra, R. Leemans, P. Kumar & H. Simons (Eds.), *Ecosystems and human well-being: Policy responses, volume 3*. Washington, D.C.: Island Press.

Aylward, B., & Barbier, E. B. (1992). Valuing environmental functions in developing countries. *Biodiversity and Conservation, 1*(1), 34-50.

Bank, L., & Hobson, S. (1993). *Informal settlements in the Eastern Cape, border and the Transkei: A pilot study* Unpublished report to the HNRE). Grahamstown: Institute for Social and Economic Research (ISER), Rhodes University.

Bardhan, P., Bowles, S., & Gintis, H. (2000). Chapter 10 wealth inequality, wealth constraints and economic performance. In A. B. Atkinson, & F. Bourguignon (Eds.), *Handbook of income distribution* (pp. 541-603). Oxford: Elsevier.

Barnes, J., & Jones, B. (2009). Game ranching in Namibia. In H. Suich, B. Child & A. Spenceley (Eds.), *Evolution and innovation in wildlife conservation: Parks and game ranches to transfrontier conservation areas* (pp. 113-126). London: Earthscan.

Barrett, C. B., Lee, D. R., & McPeak, J. G. (2005). Institutional arrangements for rural poverty reduction and resource conservation. *World Development, 33*(2), 193-197.

Barrett, J. C. (1992). *The economic roles of cattle in communal farming systems in Zimbabwe* No. 32b). London: Overseas Development Institute.

Barrow, E. G. C. (1990). Usufruct rights to trees: The role of *ekwar* in dryland central Turkana, Kenya. *Human Ecology, 18*(2), 163-176.

Bauer, H. (2003). Local perceptions of Waza National Park, northern Cameroon. *Environmental Conservation, 30*(2), 175-181.

Becker, C., Agreda, A., Astudillo, E., Constantino, M., & Torres, P. (2005). Community-based surveys of fog capture and biodiversity monitoring at Loma Alta, Ecuador enhance social capital and institutional cooperation. *Biodiversity Conservation, 14*(11), 2695-2707.

Begzsuren, S., Ellis, J. E., Ojima, D. S., Coughenour, M. B., & Chuluun, T. (2004). Livestock responses to droughts and severe winter weather in the Gobi Three Beauty National Park, Mongolia. *Journal of Arid Environments, 59*(4), 785-796.

Behnke, R., & Scoones, I. (1993). Rethinking range ecology: Implications for rangeland management in Africa. In R. Behnke, & I. Scoones (Eds.), *Range ecology at disequilibrium: New models of natural variability and pastoral adaptation in African savannas* (pp. 1-30). London: Overseas Development Institute.

Bell, K., Huppert, D., & Johnson, R. L. (2000). Willingness to pay for local Coho salmon enhancement in coastal communities. *Marine Resource Economics, 18*, 15-31.

Bellassen, V., & Gitz, V. (2008). Reducing emissions from deforestation and degradation in Cameroon — assessing costs and benefits. *Ecological Economics, 68*(1-2), 336-344.

Bellassen, V., & Leguet, B. (2007). *The emergence of voluntary carbon offsetting* No. Mission Climat Research Report). Paris: Caisse des Depots.

Belsky, A. J., Amundson, R. G., Duxbury, J. M., Riha, S. J., Ali, A. R., & Mwonga, S. M. (1989). The effects of trees on their physical, chemical, and biological environments in a semi-arid savanna in Kenya. *Journal of Applied Ecology, 26*(3), 1005-1024.

Bendix, W. (2001). *The fight to save Rocky (the fish).* Retrieved October 15, 2008, from http://www.scienceinAfrica.co.za/2001/august/rocky.htm

Benjaminsen, T. A., Rohde, R., Sjaastad, E., Wisborg, P., & Lebert, T. (2006). Land reform, range ecology, and carrying capacities in Namaqualand, South Africa. *Annals of the Association of American Geographers, 96*(3), 524-540.

Bennun, L., Matiku, P., Mulwa, R., Mwangi, S., & Buckley, P. (2005). Monitoring important bird areas in Africa: Towards a sustainable and scaleable system. *Biodiversity and Conservation, 14*(11), 2575-2590.

Benseler, A. (2003). *Municipal commonage administration in the Northern Cape: Can municipalities promote emergent farming?* Human Sciences Research Council.

Bernard, P. (2008). *Personal communication*. Lecturer, Department of Anthropology, Rhodes University, Grahamstown.

Bertollo, P. (1997). Assessing ecosystem health in governed landscapes: A framework for developing core indicators. *Ecosystem Health, 4*(1), 33-51.

Bhattacharya, P., & Hayat, S. F. (2004). Sustainable NTFP management for rural development: A case from Madhya Pradesh, India. *International Forestry Review, 6*(2), 161-168.

Binns, J. A., Illgner, P. M., & Nel, E. L. (2001). Water shortage, deforestation and development: South Africa's working for water programme. *Land Degradation and Development, 12*(4), 341-355.

Bird, P., & Rapport, D. J. (1986). *State of the environment report for Canada*. Ottawa: Environment Canada and the Canadian Government Publishing Centre.

Blackmore, A. C., Mentis, M. T., & Scholes, R. J. (1990). The origin and extent of nutrient-enriched patches within a nutrient-poor savanna in South Africa. *Journal of Biogeography, 17*(4-5), 463-470.

Blaikie, P., & Brookfield, H. (1987). *Land degradation and society*. London: Methuen.

Boman, M. (2009). To pay or not to pay for biodiversity in forests – what scale determines responses to willingness to pay questions with uncertain response options? *Journal of Forest Economics, 15*(1-2), 79-91.

Boman, M., & Bostedt, G. (1999). Valuing the wolf in Sweden: Are benefits contingent upon the supply? In M. Boman, R. Brännlund & B. Kriström (Eds.), *Topics in environmental economics* (Illustrated ed., pp. 157-174). Dordrecht: Kluwer Academic Press.

Boshoff, A. (2005). *The Baviaanskloof Mega-Reserve: An environmentally, socially and economically sustainable conservation and development initiative*. Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Bostedt, G., Ericsson, G., & Kindberg, J. (2008). Contingent values as implicit contracts: Estimating minimum legal willingness to pay for conservation of large carnivores in Sweden. *Environmental Resource Economics, 39*, 189-198.

Botha, J., Witkowski, E. T. F., & Shackleton, C. M. (2004). Market profiles and trade in medicinal plants in the lowveld, South Africa. *Environmental Conservation, 31*(1), 38-46.

Bothma, J. D. P. (1989). *Game ranch management* (2nd ed.). Pretoria: Van Schaik.

Bothma, J. D. P. (Ed.). (2002). *Game ranch management* (4th ed.). Pretoria: Van Schaik.

Bothma, J. D. P., Suich, H., & Spenceley, A. (2009). Extensive wildlife production on private land in South Africa. In H. Suich, B. Child & A. Spenceley (Eds.), *Evolution & innovation in wildlife conservation: Parks and game ranches to transfrontier conservation areas* (pp. 147-161). London: Earthscan.

Bouyer, O., & Merckx, V. (2007). Communication to the conference. *International Regime, Avoided Deforestation and the Evolution of Public and Private Forest Policies in the South.*

Boyle, K. J., Johnson, F. R., McCollum, D. W., Desvousges, W. H., Dunford, R. W., & Hudson, S. P. (1996). Valuing public goods: Discrete versus continuous contingent-valuation responses. *Land Economics, 72*(3), 381-396.

Brandon, A., Spyreas, G., Molano-Flores, B., Carroll, C., & Ellis, J. (2003). Can volunteers provide reliable data for forest vegetation surveys? *Natural Areas Journal, 23*(3), 254-261.

Briske, D. D., Fuhlendorf, S. D., & Smeins, F. E. (2003). Vegetation dynamics on rangelands: A critique of the current paradigms. *Journal of Applied Ecology, 40*(4), 601-614.

Broberg, T., & Brännlund, R. (2008). On the value of large predators in Sweden: A regional stratified contingent valuation analysis. *Journal of Environmental Management, 88*(4), 1066-1077.

Brockington, D. (2002). *Fortress conservation: The preservation of the Mkomazi Game Reserve, Tanzania*. Bloomington: Indiana University Press.

Brown Jr., G. M., & Shogren, J. F. (1998). Economics of the Endangered Species Act. *Journal of Economic Perspectives, 12*(3), 3-20.

Brown, J. L. (1997). Preserving species: The Endangered Species Act versus ecosystem management regime, ecological and political considerations, and recommendations for reform. *Journal of Environmental Law and Litigation, 12*(2), 151-265.

Brown, K. (1998). The political ecology of biodiversity, conservation and development in Nepal's Terai: Confused meanings, means and ends. *Ecological Economics, 24*(1), 73-87.

Bruijnzeel, L. A. (2004). Hydrological functions of moist tropical forests: Not seeing the soil for the trees? *Agriculture, Ecosystems and Environment, 104*, 185-228.

Bruijnzeel, L. A., & van Noordwijk, M. (2007). *Bellagio tropical land use and hydrology: What do we know and is it enough?* Unpublished manuscript.

Bulte, E., & Engel, S. (2006). Conservation of tropical forests: Addressing market failure. In R. Lopez, J. Stiglitz & M. Toman (Eds.), *Sustainable development: New options and policies* (pp. 412-452). New York: Oxford University Press.

Bureau of Market Research (BMR). (2003). *Research report 3/19 on minimum living levels.* Pretoria: BMR.

Buso, N. (2003). *Municipal commonage administration in the Free State Province: Can municipalities in the current local government dispensation promote emerging farming?* Human Sciences Research Council.

Cacadu District Municipality (Cacadu IDP). (2007). *Cacadu municipality integrated development plan (IDP)*. Port Elizabeth: Cacadu District Municipality.

Cacho, O. J., Marshall, G. R., & Milne, M. (2005). Transaction and abatement costs of carbon-sink projects in developing countries. *Environment and Development Economics, 10*(5), 597-614.

Cairns Jr., J., McCormick, P. V., & Niederlehner, B. R. (1993). A proposed framework for developing indicators of ecosystem health. *Hydrobiologia, 263*(1), 1-44.

Cambray, J. (1996). Threatened fishes of the world: *Sandelia bainsii* castelnau, 1861 (anabantidae). *Environmental Biology of Fishes, 45*(2), 150.

Cambray, J. (2008). *Personal communication*. Curator, Department of Ichthyology, Albany Museum, Grahamstown.

Cambray, G. (2008). *Personal communication*. Founder and Director, Makana Meadery, Grahamstown.

Campbell, B. M., Jeffrey, S., Kozanyi, W., Luckert, M. K., Mutamba, M., & Zindi, C. (2002). *Household livelihoods in semi arid regions: Options and constraints*. Jakarta: Centre for International Forestry Research (CIFOR).

Campbell, B. M., Vermeulen, S. S., Mangono, J. J., & Mabugu, R. (2003). The energy transition in action: Urban domestic fuel choices in a changing Zimbabwe. *Energy Policy, 31*, 553-562.

Carriere-Buchsenschutz, S. (2004). From monocultures to agroforests: Cocoa and *ntumu* swiddening. *9th International Congress of Ethnobiology.*

Carruthers, J. (2008). Wilding the farm or farming the wild? the evolution of game farming in South Africa. *Seminar delivered to the Southern African Historical Society (SAHS)*. Durban: SAHS.

Carter, M. R., & May, J. (1999). Poverty, livelihood and class in rural South Africa. *World Development, 27*(1), 1-20.

Cartwright, A., Harrison, T., & Benseler, A. (2002). *Municipal commonage management.* Pretoria: Department for International Development (DFID) and the Department of Provincial and Local Government.

Cavendish, W. (2000). Empirical regularities in the poverty-environment relationship of rural households: Evidence from Zimbabwe. *World Development, 28*(11), 1979-2003.

Cavendish, W. (2002). Quantitative methods for estimating the economic value of resource use to rural households. In B. M. Campbell, & M. K. Luckert (Eds.), *Uncovering the hidden harvest: Valuation methods for woodland and forest resources* (pp. 17-65). London: Earthscan.

Ceballos-Lascurain, H. (1996). *Tourism, ecotourism, and protected areas*. Gland: The World Conservation Union (IUCN).

Centre for Scientific and Industrial Research (CSIR). (2000). *Tenure status for South Africa: The public land inventory*. Pretoria: Department of Land Affairs (DLA).

Chambers, R. (1997). *Whose reality counts? putting the last first*. Essex: Longman Scientific and Technical.

Chambers, C. M., & Whitehead, J. C. (2003). A contingent valuation estimate of the benefits of wolves in Minnesota. *Environmental and Resource Economics, 26*(2), 249-267.

Chan, Y. K. (2008). Breeding of seed and vegetatively propagated tropical fruits using papaya and pineapple as examples. *Acta Horticulturae, 787*, 69-76.

Charley, J. (1972). The role of shrubs in nutrient cycling. In C. McKell, J. Blaisdell & J. Goodin (Eds.), *Wildland shrubs: Their biology and utilization* (pp. 182-203). Logan: USDA Forest Service.

Chen, N., Li, H., & Wang, L. (2009). A GIS-based approach for mapping direct use value of ecosystem services at a county scale: Management implications. *Ecological Economics, 68*(11), 2768-2776.

Chenevix-Trench, P. (1997). *People and cattle: Agents of ecological change in a dry montane forest, Samburu district, Kenya.* Unpublished PhD Dissertation, University of London, London.

Child, B. (Ed.). (2004). *Parks in transition: Biodiversity, rural development, and the bottom line*. London: Earthscan.

Child, B., Castley, G., Knight, M. H., Gordon, J., Daitz, D., Johnson, S., et al. (2004). Innovations in park management. In B. Child (Ed.), *Parks in transition: Biodiversity, rural development and the bottom line* (pp. 165-188). London: Earthscan.

Chivaura-Mususa, C., Campbell, B. M., & Kenyon, W. (2000). The value of mature trees in arable fields in the smallholder sector, Zimbabwe. *Ecological Economics, 33*(3), 395-400.

Cocks, M. L., & Bangay, L. (2006). The significance of natural resources to urban households: Implications for bio-cultural diversity in South Africa. In M. L. Cocks (Ed.), *Wild resources and cultural practices in rural and urban households in South Africa. implications for bio-cultural diversity conservation* (PhD Dissertation ed., pp. 146-167). Wageningen: Wageningen University.

Cocks, M. L., & Dold, A. P. (2002). The informal trade of *Cassipourea flanaganii* as a cosmetic in South Africa. In J. R. W. Stepp F.S., & R. K. Zarger (Eds.), *Ethnobiology and biocultural diversity: Proceedings of the 7th international congress of ethnobiology* (pp. 412-427). Athens: International Society of Ethnobiology.

Cocks, M. L., & Dold, A. P. (2006). Cultural significance of biodiversity: The role of medicinal plants in urban African cultural practices in the Eastern Cape, South Africa. *Journal of Ethnobiology, 26*, 60-81.

Cocks, M. L., Dold, A. P., & Grundy, I. (2004). The trade in medicinal plants from forests in the Eastern Cape province. In M. Lawes, H. Eeley, C. M. Shackleton & B. S. Geach (Eds.), *Indigenous forests and woodlands in South Africa: Policy, people and practice* (pp. 473-492). Scottsville: University of Natal.

Cole, L. (2008). *Personal communication*. Proprietor, Cole Ridge, Bathurst.

Cook, D. C., Thomas, M. B., Cunningham, S. A., Anderson, D. L., & De Barro, P. J. (2007). Predicting the economic impact of an invasive species on an ecosystem service. *Ecological Applications, 17*, 1832-1840.

Coomes, O. T., Grimard, F., Potvin, C., & Sima, P. (2008). The fate of the tropical forest: Carbon or cattle? *Ecological Economics, 65*(2), 207-212.

Cooper, J. C., & Hanemann, W. M. (1995). *Referendum contingent valuation: How many bounds are enough?*. Washington, D.C.: USDA Economic Research Service, Food and Consumer Economics Division.

Corbera, E., & Brown, K. (2008). *Building institutions to trade ecosystem services: Marketing forest carbon in Mexico*

Costanza, R., D'Arge, R., de Groot, R. S., Farber, S., Grasso, M., Hannon, B., et al. (1997). The value of the world's ecosystem services and natural capital. *Nature, 387*(6630), 253-260.

Cousins, B. (1996). Livestock production and common property struggles in South Africa's agrarian reform. *Journal of Peasant Studies, 23*, 166-208.

Cowling, R. M. (1984). A syntaxonomic and synecological study in the Humansdorp region of the fynbos biome. *Bothalia, 15*, 175-227.

Cowling, R. M. (2000). Challenges to the 'new' rangeland science. *Trends in Ecology and Evolution, 15*(8), 303-304.

Cowling, R. M., & Hilton-Taylor, C. (1994). Patterns of plant diversity and endemism in Southern Africa: An overview. In B. Huntley (Ed.), *Botanical diversity in Southern Africa* (pp. 31-52). Pretoria: National Botanical Institute.

Cowling, R. M., Lombard, A. T., Rouget, M., Kerley, G. I. H., Wolf, T., Sims-Castley, R., et al. (2003). *A conservation assessment for the subtropical thicket biome* No. 43). Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Cowling, R. M., Proch, S., & Vlok, S. (2005). On the origin of Southern African subtropical thicket vegetation. *South African Journal of Botany, 71*(1), 1-23.

Cowlishaw, G., Mendelson, S., & Rowcliffe, J. M. (2005). Evidence for post-depletion sustainability in a mature bushmeat market. *Journal of Applied Ecology, 42*(3), 460-468.

Crocker, T. D., & Tschirhart, J. (1992). Ecosystems, externalities, and economies. *Environmental & Resource Economics, 2*(6), 551-567.

Croitoru, L. (2007). Valuing the non-timber forest products in the Mediterranean region. *Ecological Economics, 63*(4), 768-775.

Cummings, R. G., Ganderton, P. T., & MacGuckin, T. (1994). Substitution effects in CVM values. *American Journal of Agricultural Economics, 76*, 205-214.

Cunningham, A. B. (1997). An Africa-wide overview of medicinal plant harvesting, conservation and health care. *Non-Wood Forest Products, 11*, 116-129.

Current, D., Lutz, E., & Scherr, S. J. (1995). *The costs and benefits of agroforestry to farmers* No. 10). Washington, D.C.: World Bank.

Dagang, A. B. K., & Nair, P. K. R. (2003). Silvopastoral research and adoption in Central America: Recent findings and recommendations for future directions. *Agroforestry Systems, 59*(2), 149-155.

Daily, G. C., Ceballos, G., Pacheco, J., Suzan, G., & Sanchez-Azofeifa, G. A. (2003). Countryside biogeography of neotropical mammals: Conservation opportunities in agricultural landscapes of Costa Rica. *Conservation Biology, 17*(6), 1814-1826.

Daly-Hassen, H., & Ben Mansoura, A. (2005). Tunisia. In M. Merlo, & L. Croitoru (Eds.), *Valuing Mediterranean forests: Towards total economic value* (pp. 105-122). Wallingford: CABI Publishing.

Danielsen, F., Burgess, N. D., & Balmford, A. (2005). Monitoring matters: Examining the potential of locally-based approaches. *Biodiversity and Conservation, 14*(11), 2507-2542.

Davenport, N. (2008). *Personal communication*. MSc Candidate, Department of Environmental Science, Rhodes University, Grahamstown.

Davenport, N. (2008a). *The contribution of municipal commonage to local peoples' livelihoods in small South African towns.* Unpublished MSc Dissertation, Rhodes University, Grahamstown.

Davenport, N. (2008b). *Eco-tourism feasibility.* Unpublished manuscript.

Davenport, N., & Gambiza, J. (2009). Municipal commonage policy and livestock owners: Findings from the Eastern Cape, South Africa. *Land use Policy, 26*(3), 513-520.

de Merode, E., Homewood, K., & Cowlishaw, G. (2004). The value of bushmeat and other wild foods to rural households living in extreme poverty in Democratic Republic of Congo. *Biological Conservation, 118*, 573-581.

De Soto, H. (2000). *The mystery of capital: Why capitalism triumphs in the West and fails everywhere else* (First Paperback Edition ed.). New York: Basic Books.

De Soyza, A. G., Whitford, W. G., Herrick, J. E., Van Zee, J. W., & Havstad, K. M. (1998). Early warning indicators of desertification: Examples of tests in the Chihuahuan Desert. *Journal of Arid Environments, 39*(2), 101-112.

de Waal, A. (1988). Famine early warning systems and the use of socioeconomic data. *Disasters, 12*, 81-91.

Deciduous Fruit Producers Trust (DFPT). (2005). *Key deciduous fruit statistics, 2005*. Paarl: Optimal Agricultural Business Systems.

Dei, G. (1989). Hunting and gathering in a Ghanaian rain forest community. *Ecology of Food and Nutrition, 22*, 225-243.

Deininger, K. (1999). Making negotiated land reform work: Initial experience from Colombia, Brazil and South Africa. *World Development, 27*(4), 651-672.

Deininger, K., Ali, D. A., Holden, S. T., & Zevenbergen, J. (2008). Rural land certification in Ethiopia: Process, initial impact, and implications for other African countries. *World Development, 36*(10), 1786-1812.

Department of Economic Affairs, Environment and Tourism (DEAET). (2004). *Strategic assessment of biodiversity in the Eastern Cape* Draft). Bisho: DEAET.

Department of Economics and Statistics (DES). (2003). *Economic review 2003*. Thiruvananthapuram: Government of Kerala.

Department of Economics and Statistics (DES). (2005). *Season and crop report 2004-2005*. Chennai: Governmnet of Tamilnadu.

White Paper on South African Land Policy, (1997).

Department of Land Affairs (DLA). (2005). *Strategic plan 2005 - 2010: Sustainable land and agrarian reform* A Contribution to Vision 2014). Pretoria: DLA.

Establishment of a Pricing Strategy for Water use Charges in Terms of Section 56(1) of the National Water Act 1998, (2007).

Derner, J. D., Briske, D. D., & Boutton, T. W. (1997). Does grazing mediate soil carbon and nitrogen accumulation beneath C4, perennial grasses along an environmental gradient? *Plant and Soil, 191*(2), 147-156.

Derner, J. D., & Schuman, G. E. (2007). Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. *Journal of Soil and Water Conservation, 62*(2), 77-85.

Desvousges, W. H., Johnson, F., & Spencer Banzhaf, H. (1998). *Environmental policy analysis with limited information: Principles and application of the transfer method*. Cheltenham: Edward Elgar.

Dkamela, G. P. (2001). *Les institutions communautaires de gestion des produits forestiers non-ligneux dans les villages peripheriques de la Reserve de Biosphere du Dja* No. 7). Wageningen: The Tropenbos Foundation.

Dold, A. P., & Cocks, M. L. (2002). The trade in medicinal plants in the Eastern Cape province, South Africa. *South African Journal of Science, 98*, 589-597.

Dormaar, J., Johnston, A., & Smoliak, S. (1977). Seasonal variation in chemical characteristics of soil organic matter of grazed and ungrazed mixed prairie and *Fescue* grassland. *Journal of Range Management, 30*, 195-198.

Dovie, D. B. K., Shackleton, C. M., & Witkowski, E. T. F. (2002). Direct-use values of woodland resources consumed and traded in a South African village. *International Journal of Sustainable Development and World Ecology, 9*(3), 269-283.

Dovie, D. B. K., Shackleton, C. M., & Witkowski, E. T. F. (2006). Valuation of communal area livestock benefits, rural livelihoods and related policy issues. *Land use Policy, 23*(3), 260-271.

Earle, S. (2008). *Personal communication*. MSc Candidate, Department of Environmental Science, Rhodes University, Grahamstown.

East, T., Kümpel, N. F., Milner-Gulland, E. J., & Rowcliffe, J. M. (2005). Determinants of urban bushmeat consumption in Río Muni, Equatorial Guinea. *Biological Conservation, 126*(2), 206-215.

Eastern Cape Business Information Service. (2001). *Profile of the Eastern Cape.* Retrieved April 14, 2008, from http://www.cimec.co.za/ecprofile.htm

Eastern Cape Parks Board (ECPB). (2008). *Annual report 2007/08*. East London: ECPB.

Ebeling, J., & Yasué, M. (2008). Generating carbon finance through avoided deforestation and its potential to create climatic, conservation and human development benefits. *Philosophical Transactions of the Royal Society B: Biological Sciences, 363*(1498), 1917-1924.

Eberhard, A. (1986). *Energy consumption patterns in undeveloped areas in South Africa* No. 151). Cape Town: University of Cape Town.

Edwards, P. J., & Abivardi, C. (1997). The value of biodiversity: Where ecology and economy blend. *Biological Conservation, 83*(3), 239-246.

Ellis, J. E., & Swift, D. (1988). Stability of African pastoral ecosystems: Alternative paradigms and implications for development. *Journal of Range Management, 41*, 450-459.

Engel, S. (2007). *Payments for environmental services: Potentials and caveats* No. 1). Zurich: ETH.

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics, 65*(4), 663-674.

Engel, S., & Palmer, C. (2008). Payments for environmental services as an alternative to logging under weak property rights: The case of Indonesia. *Ecological Economics, 65*(4), 799-809.

Environmental Protection Agency (EPA), U.S. (2000). *Guidelines for preparing economic analyses* No. EPA 240-R-00-003). Washington, D.C.: US EPA.

Ericsson, G., Bostedt, G., & Kindberg, J. (2008). Wolves as a symbol of people's willingness to pay for large carnivore conservation. *Society and Natural Resources, 21*(4), 294-309.

Ericsson, G., & Heberlein, T. A. (2003). Attitudes of hunters, locals, and the general public in Sweden now that the wolves are back. *Biological Conservation, 111*(2), 149-159.

Ericsson, G., Kindberg, J., & Bostedt, G. (2007). Willingness to pay (WTP) for wolverine *Gulo gulo* conservation. *Wildlife Biology, 13*, 2-12.

Eshiamwata, G. W., Berens, D. G., Bleher, B., Dean, W. R. J., & Bohning-Gaese, K. (2006). Bird assemblages in isolated *Ficus* trees in Kenyan farmland. *Journal of Tropical Ecology, 22*(6), 723-726.

Estrada, M. (2007). Views of Latin American countries. *International Roundtable on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries.*

European Climate Exchange (ECX). (2008). *Monthly report July 2008*. London: ECX.

Evans, N., Avis, A. M., & Palmer, A. R. (1998). Changes to the vegetation of the mid-Fish River valley, Eastern Cape, South Africa, in response to land-use, as revealed by direct gradient analysis. *African Journal of Range and Forage Science, 14*, 68-74.

Everard, D. (1987). A classification of the subtropical transitional thicket in the Eastern Cape based on syntaxonomic and structural attributes. *South African Journal of Botany, 53*, 329-349.

Eves, H., & Ruggiero, R. (2000). Socioeconomics and the sustainability of hunting in the forests of northern Congo (Brazzaville). In J. G. Robinson, & E. L. Bennett (Eds.), *Hunting for sustainability in tropical forests* (pp. 427-454). New York: Columbia University Press.

Fa, J. E. (2000). Hunted animals in Bioko Island, West Africa: Sustainability and future. In J. G. Robinson, & E. L. Bennett (Eds.), *Hunting for sustainability in tropical forests* (pp. 168-198). New York: Columbia University Press.

Fa, J. E., Currie, D., & Meeuwig, J. (2003). Bushmeat and food security in the Congo basin: Linkages between wildlife and people's future. *Environmental Conservation, 30*(1), 71-78.

Fa, J. E., & García Yuste, J. E. (2001). Commercial bushmeat hunting in the Monte Mitra forests, Equatorial Guinea: Extent and impact. [Caza comercial en los bosques de Monte Mitra, Guinea Ecuatorial: Alcance e impacto] *Animal Biodiversity and Conservation, 24*(1), 31-52.

Fa, J. E., García Yuste, J. E., & Castelo, R. (2000). Bushmeat markets on Bioko Island as a measure of hunting pressure. *Conservation Biology, 14*(6), 1602-1613.

Fa, J. E., Johnson, P. J., Dupain, J., Lapuente, J., Köster, P., & Macdonald, D. W. (2004). Sampling effort and dynamics of bushmeat markets. *Animal Conservation, 7*(4), 409-416.

Fa, J. E., Peres, C. A., & Meeuwig, J. (2002). Bushmeat exploitation in tropical forests: An intercontinental comparison. *Conservation Biology, 16*(1), 232-237.

Fa, J. E., Ryan, S. F., & Bell, D. J. (2005). Hunting vulnerability, ecological characteristics and harvest rates of bushmeat species in Afrotropical forests. *Biological Conservation, 121*(2), 167-176.

Fa, J. E., Seymour, S., Dupain, J., Amin, R., Albrechtsen, L., & Macdonald, D. W. (2006). Getting to grips with the magnitude of exploitation: Bushmeat in the Cross-Sanaga Rivers region, Nigeria and Cameroon. *Biological Conservation, 129*(4), 497-510.

Fabricius, C. (1997). *The impact of land use on biodiversity in xeric succulent thicket, South Africa.* Unpublished PhD Dissertation, University of Cape Town, Cape Town.

Fabricius, C., Burger, M., & Hockey, P. A. R. (2003). Comparing biodiversity between protected areas and adjacent rangeland in xeric succulent thicket, South Africa: Arthropods and reptiles. *Journal of Applied Ecology, 40*(2), 392-403.

Fabricius, C., Cundill, G., & McGarry, D. (2006). *Bathurst commons community conservancy project: Stakeholder engagement and feasibility study*. Grahamstown: South African National Biodiversity Institute.

Farrow, R. S., Goldburg, C. B., & Small, M. J. (2000). Economic valuation of the environment: A special issue. *Environmental Science and Technology, 34*(8), 1381-1383.

Federative Republic of Brazil. (2007). Brazilian perspective on reducing emissions from deforestation. *United Nations Framework Convention on Climate Change (UNFCCC) Submission*.

Forest Trends, The Katoomba Group, & the United Nations Environment Programme (UNEP). (2008). *Payments for ecosystem services — getting started: A primer*. Nairobi: United Nations.

Forrester, J. (2004). *Ilex mitis.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/planthij/ilexmitis.htm

Forsyth, T. (2003). *Critical political ecology*. London: Routledge.

Fouche, F. (2008). *Personal communication cited in Davenport 2008a*. Director, Conservation Department, Ndlambe local municipality, Port Alfred.

Fox, H. (2002). *A comparative cultural valuation of three rural landscapes in the Eastern Cape, South Africa*. Grahamstown, South Africa: Rhodes University.

Frank, A. B., Tanaka, D. L., Hofmann, L., & Follett, R. F. (1995). Soil carbon and nitrogen of northern Great Plains grasslands as influenced by long-term grazing. *Journal of Range Management, 48*(5), 470-474.

Friderichs, C. (2008). *Personal communication*. Proprietor, Honey Dawn Farm, Port Alfred.

Fynn, R. W. S., & O'Connor, T. G. (2000). Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. *Journal of Applied Ecology, 37*(3), 491-507.

Gadd, M. E. (2005). Conservation outside of parks: Attitudes of local people in Laikipia, Kenya. *Environmental Conservation, 32*(1), 50-63.

Galpin, M. (2007). Flowering plants visited by bees. *Clippings of the Albany Branch of the Botanical Society of South Africa, Winter*

Ganesan, R. (2003). Identification, distribution and conservation of *Phyllanthus indofischeri*, another source of Indian gooseberry. *Current Science, 84*(12), 1515-1518.

Garrod, G. D., & Willis, K. G. (1999). Methodological issues in valuing the benefits of environmentally sensitive areas. *Journal of Rural Studies, 15*(1), 111-117.

Geist, H. J., & Lambin, E. F. (2002). Proximate causes and underlying driving forces of tropical deforestation. *Bioscience, 52*(2), 143-150.

Genet, K. S., & Sargent, L. G. (2003). Evaluation of methods and data quality from a volunteer-based amphibian call survey. *Wildlife Society Bulletin, 31*(3), 703-714.

Georgiou, S., Whittington, D., Pearce, D., & Moran, D. (1997). *Economic values and the environment in the developing world*. Cheltenham: Edward Elgar.

Giannecchini, M., Twine, W., & Vogel, C. (2007). Land-cover change and human-environment interactions in a rural cultural landscape in South Africa. *Geographical Journal, 173*(1), 26-42.

Gibson, C. C. (2001). Forest resources: Institutions for local governance in Guatemala. In J. Burger, E. Ostrom, R. B. Norgaard, D. Policansky & B. D. Goldstein (Eds.), *Protecting the commons: A framework for resource management in the Americas* (pp. 71-89). Washington, D.C.: Island Press.

Gibson, C. C., Williams, J. T., & Ostrom, E. (2005). Local enforcement and better forests. *World Development, 33*(2), 273-284.

Giliomee, H. (2003). The making of the Apartheid plan 1929-1948. *Journal of Southern African Studies, 29*(2), 373-392.

Giraud, K. L., Turcin, B., Loomis, J. B., & Cooper, J. C. (2002). Economic benefit of the protection program for the Steller sea lion. *Marine Policy, 26*(6), 451-458.

Glenday, J. (2008). Carbon storage and emissions offset potential in an African dry forest, the Arabuko-Sokoke Forest, Kenya. *Environmental Monitoring and Assessment, 142*(1-3), 85-95.

Global Insight. (2006). *Regional explorer*. Johannesburg: IHS Global Insight Southern Africa.

Godoy, R., Brokaw, N., & Wilkie, D. S. (1995). The effect of income on the extraction of non-timber tropical forest products: Model, hypotheses, and preliminary findings from the Sumu Indians of Nicaragua. *Human Ecology, 23*, 29-52.

Goldman, R. L., Thompson, B. H., & Daily, G. C. (2007). Institutional incentives for managing the landscape: Inducing cooperation for the production of ecosystem services. *Ecological Economics, 64*(2), 333-343.

Gray, A. N., & Azuma, D. L. (2005). Repeatability and implementation of a forest vegetation indicator. *Ecological Indicators, 5*(1), 57-71.

Greenline Business Management (Pty) Ltd. (GBM). (2007). *Eastern Cape forestry sector profile*. Pretoria: Department of Water Affairs and Forestry (DWAF).

Grêt-Regamey, A., Bebi, P., Bishop, I. D., & Schmid, W. A. (2008). Linking GIS-based models to value ecosystem services in an alpine region. *Journal of Environmental Management, 89*(3), 197-208.

Grieg-Gran, M., Porras, I. T., & Wunder, S. (2005). How can market mechanisms for forest environmental services help the poor? preliminary lessons from Latin America. *World Development, 33*, 1511-1527.

Griffiths, T. (2007). *Seeing 'RED'? 'avoided deforestation' and the rights of indigenous peoples and local communities*. Moreton-in-Marsh: Forest Peoples Program.

Grossman, R., Harms, D., Kingsbury, D., Shaw, R., & Jenkins, A. (2001). Assessment of soil organic carbon using the U.S. soil survey. In R. Lal, J. M. Kimble, R. F. Follett & B. Stewart (Eds.), *Assessment methods for soil carbon* (pp. 87-104). Boca Raton: Lewis Publishers.

Guagnano, G. A., Stern, P. C., & Dietz, T. (1995). Influences on attitude-behavior relationships: A natural experiment with curbside recycling. *Environment and Behavior, 27*(5), 699-718.

Gubbi, S., & MacMillan, D. C. (2008). Can non-timber forest products solve livelihood problems? A case study from Periyar Tiger Reserve, India. *Oryx, 42*(2), 222-228.

Gundimeda, H. (2004). How 'sustainable' is the 'sustainable development objective' of CDM in developing countries like India? *Forest Policy and Economics, 6*(3-4), 329-343.

Gutman, P. (Ed.). (2003). *From goodwill to payments for environmental services: A survey of financing options for sustainable natural resource management in developing countries*. Washington, D.C.: WWF.

Hahn, B. D., Richardson, F. D., Hoffman, M. T., Roberts, R., Todd, S. W., & Carrick, P. J. (2005). A simulation model of long-term climate, livestock and vegetation interactions on communal rangelands in the semi-arid succulent Karoo, Namaqualand, South Africa. *Ecological Modelling, 183*(2-3), 211-230.

Hahndiek, Q. (2009). *Personal communication*. Sub-Region Manager for Department Economic Development and Environment Affairs (DEDEA), Settlers district, Cacadu region, Grahamstown.

Hall, R. (2007). Transforming rural South Africa? taking stock of land reform. In L. Ntsebeza, & R. Hall (Eds.), *The land question in South Africa: The challenge of transformation and redistribution* (Electronic ed., pp. 87-106). Cape Town: Human Sciences Research Council.

Hall, R., Isaacs, M., & Saruchera, M. (2007). *Land and agrarian reform in integrated development plans (IDPs)* No. 23). Cape Town: Institute for Poverty, Land and Agrarian Studies (PLAAS).

Hall, R., Jacobs, P., & Lahiff, E. (2003). *Final report* No. 10). Cape Town: Institute for Poverty, Land and Agrarian Studies (PLAAS).

Hall-Martin, A., & Carruthers, J. (Eds.). (2003). *South African national parks: A celebration. commemorating the Fifth World Parks Congress 2003*. Auckland Park: Horst Klem Publications.

Ham, C. (2000). *The importance of woodlots to local communities, small scale entrepreneurs and indigenous forest conservation: A case study*. London and Pretoria: International Institute for Environment and Development (IIED) and CSIR-Environmentek.

Han, G., Hao, X., Zhao, M., Wang, M., Ellert, B. H., Willms, W., et al. (2008). Effect of grazing intensity on carbon and nitrogen in soil and vegetation in a meadow steppe in Inner Mongolia. *Agriculture, Ecosystems and Environment, 125*(1-4), 21-32.

Hanemann, M., Loomis, J. B., & Kanninen, B. (1991). Statistical efficiency of double-bounded dichotomous choice contingent valuation. *American Journal of Agricultural Economics, 73*(4), 1255-1263.

Hanemann, W. M. (1984). Welfare evaluations in contingent valuation experiments with discrete responses. *American Journal of Agricultural Economics, 66*(3), 332-342.

Harvey, C. A., & Haber, W. A. (1998). Remnant trees and the conservation of biodiversity in Costa Rican pastures. *Agroforestry Systems, 44*(1), 37-68.

Haskell, B., Norton, B., & Costanza, R. (1992). What is ecosystem health and why should we worry about it? In R. Costanza, B. Norton & B. Haskell (Eds.), *Ecosystem health: New goals for environmental management* (pp. 269). Washington, D.C.: Island Press.

Hassan, R. M. (2003). Measuring asset values and flow benefits of non-traded products and ecosystems services of forest and woodland resources in South Africa. *Environment, Development and Sustainability, 5*(3-4), 403-418.

Heal, G., Daily, G. C., Ehrlich, P. R., Salzman, J., Boggs, C., Hellman, J., et al. (2001). Protecting natural capital through ecosystem service districts. *Stanford Environmental Law Journal, 20*, 333-364.

Hein, L., van Koppen, K., de Groot, R. S., & van Ierland, E. C. (2006). Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics, 57*(2), 209-228.

Heinzen, R. R., & Bridges, J. F. P. (2008). Comparison of four contingent valuation methods to estimate the economic value of a pneumococcal vaccine in Bangladesh. *International Journal of Technology Assessment in Health Care, 24*(4), 481-487.

Herman, R. P., Provencio, K. R., Herrera-Matos, J., & Torrez, R. J. (1995). Resource islands predict the distribution of heterotrophic bacteria in Chihuahuan Desert soils. *Applied and Environmental Microbiology, 61*(5), 1816-1821.

Herrick, J. E., Schuman, G. E., & Rango, A. (2006). Monitoring ecological processes for restoration projects. *Journal for Nature Conservation, 14*(3-4), 161-171.

Higginbottom, K., Ntsebeza, L., Kedema, W., & Bank, L. (1995). *The fate of the commons: Urban land reform, squatting and environmental management in the Eastern Cape province* No. 65). Grahamstown: Institute for Social and Economic Research (ISER), Rhodes University.

Hobson, C. (1993). *The vegetation of the Bathurst commonage with implications for future land use*. Grahamstown: Department of Botany, Rhodes University.

Hockley, N. J., Jones, J. P. G., Andriahajaina, F. B., Manica, A., Ranambitsoa, E. H., & Randriamboahary, J. A. (2005). When should communities and conservationists monitor exploited resources? *Biodiversity and Conservation, 14*(11), 2795-2806.

Hoffman, M. T., & Cowling, R. M. (1990). Desertification in the lower Sundays River valley, South Africa. *Journal of Arid Environments, 19*(1), 105-118.

Holling, C. S., Schindler, D., Walker, B., & Roughgarden, J. (1995). Biodiversity in the functioning of ecosystems: An ecological synthesis. In C. Perrings, K. Maler, C. Folke, B. Jansson & C. S. Holling (Eds.), *Biodiversity loss: Economic and ecological issues* (pp. 48-83). New York: Cambridge University Press.

Holm, A. M., Watson, I. W., Speijers, E. J., Allen, R. J., Eliot, G. J., Shackleton, K. R., et al. (2005). Loss of patch-scale heterogeneity on secondary productivity in the arid shrubland of Western Australia. *Journal of Arid Environments, 61*(4), 631-649.

Home, R., Keller, C., Nagel, P. Bauer, N., & Hunziker, M. (2009). Selection criteria for flagship species by conservation organizations. *Environmental Conservation 36*(2), 139-148.

Horner-Devine, M., Daily, G. C., Ehrlich, P. R., & Boggs, C. L. (2003). Countryside biogeography of tropical butterflies. *Conservation Biology, 17*(1), 168-177.

Houghton, R. A. (2005). Tropical deforestation as a source of greenhouse gas emissions. In P. Mutinho, & S. Schwartsman (Eds.), *Tropical deforestation and climate change*. Washington, D.C.: Instituto de Pesquisa Ambiental da Amazonia (IPAM) and Environmental Defense.

Howells, M. I., Alfstad, T., Victor, D. G., Goldstein, G., & Remme, U. (2005). A model of household energy services in a low-income rural African village. *Energy Policy, 33*, 1833-1851.

Hutton, J. M., & Leader-Williams, N. (2003). Sustainable use and incentive-driven conservation: Realigning human and conservation interests. *Oryx, 37*(2), 215-226.

Igoe, J. (2004). *Conservation and globalization: A study of national parks and indigenous communities from East Africa to South Dakota*. Belmont: Wadsworth/Thompson.

Illius, A. W., & O'Connor, T. G. (1999). On the relevance of nonequilibrium concepts to arid and semiarid grazing systems. *Ecological Applications, 9*(3), 798-813.

Imeson, A. C., & Prinsen, H. A. M. (2004). Vegetation patterns as biological indicators for identifying runoff and sediment source and sink areas for semi-arid landscapes in Spain. *Agriculture, Ecosystems and Environment, 104*(2), 333-342.

Ingle, M. (2006). Municipal commonage in South Africa: A public good going bad? *Africa Insight, 36*, 46-55.

Ingraham, M. W., & Foster, S. G. (2008). The value of ecosystem services provided by the U.S. national wildlife refuge system in the contiguous U.S. *Ecological Economics, 67*(4), 608-618.

Intergovernmental Panel on Climate Change (IPCC). (1997). *Revised 1996 IPCC guidelines for national 19 greenhouse gas inventories*. Bracknell: Meira Filho,L.G.; Lim,B.; Tranton,K.; Mamaty,I.; Bonduki,Y.; Griggs,D.J.; Callander,B.A.

International Institute of Environment and Development (IIED). (2007). *Water ecosystem services and poverty reduction under climate change*. London: IIED.

Iovanna, R., & Griffiths, C. (2006). Clean water, ecological benefits and benefits transfer: A work in progress at the U.S. EPA. *Ecological Economics, 60*, 473-482.

James, I., Hoffman, M. T., Cowling, R. M., Roberts, R., Campbell, B. M., Boumans, R., et al. (2005). *Valuing Namaqualand's natural resources: The costs and benefits of communal, commercial and conservation land use practices in Namaqualand incorporating ecological, economic and social values.* Unpublished manuscript.

Jaramillo, V., Boone Kauffman, J., Renteria-Rodriguez, L., Cummings, D., & Ellingson, L. (2003). Biomass, carbon, and nitrogen pools in Mexican tropical dry forest landscapes. *Ecosystems, 6*, 609-629.

Jeltsch, F., Milton, S. J., Dean, W. R. J., & Van Rooyen, N. (1996). Tree spacing and coexistence in semiarid savannas. *Journal of Ecology, 84*(4), 583-595.

Jindal, R. (2004). *Measuring the socio-economic impact of carbon sequestration on local communities: An assessment study with specific reference to the Nhambita pilot project in Mozambique.* Unpublished manuscript.

Jindal, R. (2006). *Carbon sequestration projects in Africa: Potential benefits and challenges to scaling up*. Washington, D.C.: World Resources Institute (WRI).

Jindal, R., Swallow, B. M., & Kerr, J. (2008). Forestry-based carbon sequestration projects in Africa: Potential benefits and challenges. *Natural Resources Forum, 32*(2), 116-130.

Jodamus, N. (2004). *Polygala virgata.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantnop/polyvirg.htm

Johnson, C. F., Cowling, R. M., & Phillipson, P. B. (1999). The flora of the Addo Elephant National Park, South Africa: Are threatened species vulnerable to elephant damage? *Biodiversity and Conservation, 8*(11), 1447-1456.

Jorgensen, B. S., Wilson, M. A., & Heberlein, T. A. (2001). Fairness in the contingent valuation of environmental public goods: Attitude toward paying for environmental improvements at two levels of scope. *Ecological Economics, 36*(1), 133-148.

Kaushal, K. K., & Melkani, V. K. (2005). India: Achieving the Millennium Development Goals through non-timber forest products. *International Forestry Review, 7*(2), 128-134.

Kellert, S. R. (1985). Public perceptions of predators, particularly the wolf and coyote. *Biological Conservation, 31*(2), 167-189.

Kellert, S. R., Black, M., Rush, C. R., & Bath, A. J. (1996). Human culture and large carnivore conservation in North America. *Conservation Biology, 10*(4), 977-990.

Kellert, S. R., Mehta, J. N., Ebbin, S. A., & Lichtenfeld, L. L. (2000). Community natural resource management: Promise, rhetoric, and reality. *Society and Natural Resources, 13*(8), 705-715.

Kerley, G. I. H., Knight, M. H., & De Kock, M. (1995). Desertification of subtropical thicket in the Eastern Cape, South Africa: Are there alternatives? *Environmental Monitoring and Assessment, 37*(1-3), 211-230.

Kerley, G. I. H., Tongway, D. J., & Ludwig, J. A. (1999). Effects of goat and elephant browsing on soil resources in succulent thicket, Eastern Cape, South Africa. *Proceedings of the Sixth International Rangeland Congress, , 1* 116-117.

Kerr, J. (2002). Watershed development, environmental services, and poverty alleviation in India. *World Development, 30*(8), 1387-1400.

Kerr, J., Foley, C., Chung, K., & Jindal, R. (2006). Sustainable development in the clean development mechanism: Constraints and opportunities. *Journal of Sustainable Forestry, 23*(1), 1-18.

Kerr, M., Ely, E., Lee, V., & Mayio, A. (1994). A profile of volunteer environmental monitoring: National survey results. *Lake and Reservoir Management, 9*, 1-4.

Kidanu, S., Mamo, T., & Stroosnijder, L. (2004). *Eucalyptus*-wheat interaction on Ethiopian nitosols. *Agricultural Systems, 80*(2), 151-170.

Knight, A. T., Boshoff, A., Cowling, R. M., & Wilson, S. (2003). *Keeping people on the land in living landscapes: A co-operative strategy for conserving landscapes and enhancing livelihoods in the subtropical thicket biome* No. 46). Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Knight, A. T., & Cowling, R. M. (2003). *The megaconservancy network concept: "keeping people on the land in living landscapes". report for partners of the subtropical thicket ecosystem planning (STEP) project* No. 45). Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Kosoy, N., Corbera, E., & Brown, K. (2008). Participation in payments for ecosystem services: Case studies from the Lacandon Rainforest, Mexico. *Geoforum, 39*(6), 2073-2083.

Kotchen, M. J., & Reiling, S. D. (2000). Environmental attitudes, motivations, and contingent valuation of nonuse values: A case study involving endangered species. *Ecological Economics, 32*(1), 93-107.

Kruger, O. (2005). The role of ecotourism in conservation: Panacea or pandora's box? *Biodiversity and Conservation, 14*(3), 579-600.

Landell-Mills, N., & Porras, I. T. (2002). *Silver bullet or fools' gold? A global review of markets for forest environmental services and their impact on the poor*. London: International Institute for Environment and Development (IIED).

Langford, I. H., Bateman, I. J., & Langford, H. D. (1996). A multilevel modelling approach to triple-bounded dichotomous choice contingent valuation. *Environmental and Resource Economics, 7*(3), 197-211.

Langholz, J. A., & Kerley, G. I. H. (2006). *Combining conservation and development on private lands: An assessment of ecotourism-based private game reserves in the Eastern Cape* No. 56). Port Elizabeth: Center for African Conservation Ecology (ACE), Nelson Mandela Metropolitan University.

Le Houerou, H. N. (1984). Rain use efficiency: A unifying concept in arid-land ecology. *Journal of Arid Environments, 7*(3), 213-247.

Le Roux, L.-N. (2005). *Strychnos spinosa.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantqrs/strychspin.htm

Lechmere-Oertel, R., Cowling, R. M., & Kerley, G. I. H. (2005a). Landscape dysfunction and reduced spatial heterogeneity in soil resources and fertility in semi-arid succulent thicket, South Africa. *Austral Ecology, 30*(6), 615-624.

Lechmere-Oertel, R., Kerley, G. I. H., & Cowling, R. M. (2005b). Patterns and implications of transformation in semi-arid succulent thicket, South Africa. *Journal of Arid Environments, 62*(3), 459-474.

Leibbrandt, M., & Woolard, I. (1999). A comparison of poverty in South Africa's nine provinces. *Development Southern Africa, 16*(1), 37-54.

Leibbrandt, M., & Woolard, I. (2001). The labour market and household income inequality in South Africa: Existing evidence and new panel data. *Journal of International Development, 13*(6), 671-689.

Leopold, A., Callicott, J., & Freyfogle, E. (Eds.). (1999). *For the health of the land*. Washington, D.C.: Island Press.

Lester, A., Nel, E. L., & Binns, T. (2000). *South Africa past, present and future: Gold at the end of the rainbow?*. Longman: Harlow.

Liebig, M. A., Gross, J. R., Kronberg, S. L., Hanson, J. D., Frank, A. B., & Phillips, R. L. (2006). Soil response to long-term grazing in the northern Great Plains of North America. *Agriculture, Ecosystems and Environment, 115*(1-4), 270-276.

Lindell, C. A., Chomentowski, W. H., & Zook, J. R. (2004). Characteristics of bird species using forest and agricultural land covers in southern Costa Rica. *Biodiversity and Conservation, 13*(13), 2419-2441.

Lipper, L., & Cavatassi, R. (2004). Land-use change, carbon sequestration and poverty alleviation. *Environmental Management, 33*(Suppl. 1)

Lloyd, J., van den Berg, E., & Palmer, A. R. (2002). *Patterns of transformation and degradation in the thicket biome, South Africa*. Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Locatelli, B., & Pedroni, L. (2006). Will simplified modalities and procedures make more small-scale forestry projects viable under the clean development mechanism? *Mitigation and Adaptation Strategies for Global Change, 11*(3), 621-643.

Lombard, A. T., Johnson, C. F., Cowling, R. M., & Pressey, R. L. (2001). Protecting plants from elephants: Botanical reserve scenarios within the Addo Elephant National Park, South Africa. *Biological Conservation, 102*(2), 191-203.

Lombard, A. T., Wolf, T., & Cole, N. (2003). *GIS coverages and spatial analyses for the subtropical thicket ecosystem planning (STEP) project* No. 42). Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Loomis, J. B. (1992). The evolution of a more rigorous approach to benefit transfer-benefit function transfer. *Water Resources Research, 28*, 701-705.

Loomis, J. B., & Ekstrand, E. (1997). Economic benefits of critical habitat for the Mexican spotted owl: A scope test using a multiple-bounded contingent valuation survey. *Journal of Agricultural and Resource Economics, 22*(2), 356-366.

Loomis, J. B., & Larson, D. (1994). Total economic values of increasing gray whale populations: Results from a contingent valuation survey of visitors and households. *Marine Resource Economics, 9*, 275-286.

Loomis, J. B., & White, D. S. (1996). Economic benefits of rare and endangered species: Summary and meta-analysis. *Ecological Economics, 18*(3), 197-206.

Loureiro, M., & Ojea, E. (2008). Valuing local endangered species: The role of intra-species substitutes. *Ecological Economics, 68*(1-2), 362-369.

Low, A., & Rebelo, A. (Eds.). (1996). *Vegetation of South Africa, Lesotho and Swaziland*. Pretoria: Department of Environmental Affairs and Tourism (DEAT).

Lubke, R., Everard, D., & Jackson, S. (1986). The biomes of the Eastern Cape with emphasis on their conservation. *Bothalia, 16*, 251-261.

Lucas, R. M., Honzak, M., & Foody, G. M. (1993). Characterizing tropical secondary forests using multi-temporal Landsat sensor imagery. *International Journal of Remote Sensing, 14*(16), 3061-3067.

Lucas, R. M., Xiao, X., Hagen, S., & Frolking, S. (2002). Evaluating TERRA-1 MODIS data for discrimination of tropical secondary forest regeneration stages in the Brazilian legal Amazon. *Geophysical Research Letters, 29*(8)

Ludwig, J. A., & Tongway, D. J. (1995). Spatial organisation of landscapes and its function in semi-arid woodlands, Australia. *Landscape Ecology, 10*(1), 51-63.

Mabbutt, J. A., & Fanning, P. C. (1987). Vegetation banding in arid Western Australia. *Journal of Arid Environments, 12*(1), 41-59.

Macdicken, K. G. (1997). Project specific monitoring and verification: State of the art and challenges. *Mitigation and Adaptation Strategies for Global Change, 2*(2-3), 191-202.

Madubansi, M., & Shackleton, C. M. (2007). Changes in fuelwood use and selection following electrification in the Bushbuckridge lowveld, South Africa. *Journal of Environmental Management, 83*, 416-426.

Maffi, L. (Ed.). (2001). *On biocultural diversity: Linking language, knowledge, and the environment*. Washington, D.C.: Smithsonian Institution Press.

Malan, C., & Notten, A. (2005). *Crassula ovata.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantcd/crassovat.htm

Mali. (2008). *Personal communication*. Traditional healer, Bathurst.

Maloti Drakensberg Transfrontier Project (MDTP). (2008). *Payment for ecosystem services: Developing an ecosystem services trading model for the Mnweni/Cathedral Peak and Eastern Cape Drakensberg areas* No. IR281). South Africa: Development Bank of Southern Africa; Department of Water Affairs and Forestry; Department of Environment Affairs and Tourism; Ezemvelo KZN Wildlife.

Mander, M. (1998). *Marketing of indigenous medicinal plants in South Africa. A case study in KwaZulu-Natal* No. W9195/E). Rome: FAO.

Martens, C. (2008). *The management and governance of commonages in three small towns in the Eastern Cape, South Africa.* Unpublished MA Dissertation, Rhodes University, Grahamstown.

Martin-Lopez, B., Montes, C., & Benayas, J. (2007). The non-economic motives behind the willingness to pay for biodiversity conservation. *Biological Conservation, 139*, 67-82.

Mason, S., & Jury, M. (1997). Climatic variability and change over Southern Africa: A reflection on underlying processes. *Progress in Physical Geography, 21*, 23-50.

May, J., & Roberts, B. (2000). *Monitoring and evaluating the quality of life of land reform beneficiaries: 1998/1999* No. Summary report). Pretoria: Department of Land Affairs.

Mazzarino, M. J., Bertiller, M. B., Sain, C. L., Laos, F., & Coronato, F. R. (1996). Spatial patterns of nitrogen availability, mineralization, and immobilization in northern Patagonia, Argentina. *Arid Soil Research and Rehabilitation, 10*(4), 295-309.

Mbambezeli, G. (2008). *Strychnos decussata.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantqrs/strychnosdecussata.htm

Mbatha. (2008). *Personal communication*. Traditional healer, Bathurst.

Mbumba, M. (2008). *Personal communication*. Traditional healer, Bathurst.

McGarry, D., & Shackleton, C. M. (2009). Is HIV/AIDS jeopardizing biodiversity? *Environmental Conservation, 36*(1), 5-7.

McKean, S. G. (2001). Productivity and sustainable use of *Phragmites* in the Fuyeni reedbed - Hluhluwe-Umfolozi park - management guidelines for harvest. *South African Journal of Botany, 67*(2), 274-280.

McKean, S. G. (2003). Toward sustainable use of palm leaves by a rural community in KwaZulu-Natal, South Africa. *Economic Botany, 57*(1), 65-72.

McMichael, A. J. (1993). Planetary overload: Global environmental change and the health of the human species. *Planetary Overload: Global Environmental Change and the Health of the Human Species,*

McMichael, A. J. (1996). Global environmental change and human health: Impact assessment, population vulnerability, and research priorities. *Ecosystem Health, 3*(4), 200-210.

Meadows, M. E., & Hoffman, M. T. (2002). The nature, extent and causes of land degradation in South Africa: Legacy of the past, lessons for the future? *Area, 34*(4), 428-437.

Meinzen-Dick, R. S., Knox, A., Place, F., & Swallow, B. M. (Eds.). (2002). *Innovation in natural resource management: The role of property rights and collective action in developing countries*. Baltimore: Johns Hopkins University Press for IFPRI.

Mertens, B., & Lambin, E. F. (2000). Land-cover-change trajectories in southern Cameroon. *Annals of the Association of American Geographers, 90*(3), 467-494.

Metrick, A., & Weitzman, M. L. (1996). Patterns of behavior in endangered species preservation. *Land Economics, 72*(1), 1-16.

Metrick, A., & Weitzman, M. L. (1998). Conflicts and choices in biodiversity preservation. *Journal of Economic Perspectives, 12*(3), 21-34.

Midgley, J. (1991). Valley bushveld dynamics and tree euphorbias. *Proceedings of the First Valley Bushveld/Subtropical Thicket Symposium, , Special Publication* 8-9.

Milchunas, D. G., & Lauenroth, W. K. (1993). Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs, 63*(4), 327-366.

Millennium Assessment. (2005). *Ecosystems and human well-being: Synthesis*. Washington, D.C.: Island Press.

Mills, A. J., & Cowling, R. M. (2006). Rate of carbon sequestration at two thicket restoration sites in the Eastern Cape, South Africa. *Restoration Ecology, 14*(1), 38-49.

Mills, A. J., Cowling, R. M., Fey, M. V., Kerley, G. I. H., Donaldson, J. S., Lechmere-Oertel, R., et al. (2005). Effects of goat pastoralism on ecosystem carbon storage in semiarid thicket, Eastern Cape, South Africa. *Austral Ecology, 30*(7), 797-804.

Mills, A. J., Turpie, J. K., Cowling, R. M., Marais, C., Kerley, G. I. H., Lechmere-Oertel, R., et al. (2007). Assessing costs, benefits and feasibility of subtropical thicket restoration in the Eastern Cape, South Africa. In S. J. Milton, & J. Aronson (Eds.), *Restoring natural capital* (pp. 179-187). Washington, D.C.: Island Press.

Milner-Gulland, E. J., Bennett, E. L., Abernethy, K., Bakarr, M., Bennett, E., Bodmer, R., et al. (2003). Wild meat: The bigger picture. *Trends in Ecology and Evolution, 18*(7), 351-357.

Minang, P. A., McCall, M. K., & Bressers, H. T. A. (2007). Community capacity for implementing clean development mechanism projects within community forests in Cameroon. *Environmental Management, 39*(5), 615-630.

Ministry of Agriculture and Land Affairs (MALA). (2001). *Land redistribution for agricultural development: A sub-programme of the land redistribution programme*. Pretoria: Ministry of Agriculture and Land Affairs.

Miranda, M., Porras, I. T., & Moreno, M. L. (2003). *The social impacts of payments for environmental services in Costa Rica: A quantitative field survey and analysis of the Virilla watershed* No. 1). London: International Institute for Environment and Development (IIED).

Mitchell, R. C., & Carson, R. T. (1989). *Using surveys to value public goods: The contingent valuation method* (3rd ed.). Washington, D.C.: Resources for the Future (RFF).

Moguel, P., & Toledo, V. M. (1999). Biodiversity conservation in traditional coffee systems of Mexico. *Conservation Biology, 13*(1), 11-21.

Mohai, P. (1985). Public concern and elite involvement in environmental conservation issues. *Social Science Quarterly (University of Texas Press), 66*(4), 820-839.

Moolman, H. J., & Cowling, R. M. (1994). The impact of elephant and goat grazing on the endemic flora of South African succulent thicket. *Biological Conservation, 68*(1), 53-61.

Moreira, F., Rego, F. C., & Ferreira, P. G. (2001). Temporal (1958-1995) pattern of change in a cultural landscape of northwestern Portugal: Implications for fire occurrence. *Landscape Ecology, 16*(6), 557-567.

Motinyane, T. (2002). *The influence of structure, density and direct use benefits on the harvesting of trees in the rural village of Pikoli, Eastern Cape, South Africa.* Unpublished MSc Dissertation, Rhodes University, Grahamstown.

Moxham, B. (2008). *Personal communication*. Proprietor, BeeMox Honey, Port Alfred.

Msimang, M. (2000). Prologue. In E. Joseph, & D. Parris (Eds.), *Visions of change: Social ecology and South African national parks*. Johannesburg: Development Communications Corporation in association with South African National Parks.

Mudekwe, J. (1997). *Assessing the effect of tree size on the production of harvesting deadwood for fuelwood.* Unpublished MSc Dissertation, University of the Witwatersrand, Johannesburg.

Munn, R. (1993). Monitoring for ecosystem integrity. In S. Woodley, G. Francis & J. Kay (Eds.), *Ecological integrity and the management of ecosystems* (pp. 105-115). Delray Beach: St. Lucie Press.

Munoz-Pina, C., Guevara, A., Torres, J. M., & Brana, J. (2008). Paying for the hydrological services of Mexico's forests: Analysis, negotiations and results. *Ecological Economics, 65*, 726-737.

Mwalukomo, H. (2007). *The role of traditional leaders in environmental governance in the context of decentralisation: A case study of grass utilisation in QwaQwa, eastern Free State.* Unpublished MSc Dissertation, University of the Witwatersrand, Johannesburg.

Mwendera, E. J., & Saleem, M. A. M. (1997). Hydrologic response to cattle grazing in the Ethiopian highlands. *Agriculture, Ecosystems and Environment, 64*(1), 33-41.

Myers, N. (1990). The biodiversity challenge: Expanded hot-spots analysis. *Environmentalist, 10*(4), 243-256.

National Agricultural Marketing Council (NAMC). (2008). *The South African beekeeping industry: A section 7 committee investigation* No. 3rd Draft). Pretoria: NAMC.

Ndlambe Local Municipality (Ndlambe IDP). (2007). *Ndlambe municipality integrated development plan*. Port Alfred: Ndlambe Local Municipality.

Ndlambe Local Municipality (Ndlambe LM). (2008). *Annual budget: 1 July 2008 - 30 June 2009*. Port Alfred: Ndlambe Local Municipality.

Nel, E. L., & Davies, J. (1999). Farming against the odds: An examination of the challenges facing farming and rural development in the Eastern Cape province of South Africa. *Applied Geography, 19*(3), 253-274.

Nel, E. L., & Hill, T. (1997). The role of small urban centres in economic recovery and regional development. In United Nations Centre for Regional Development (UNCRD) (Ed.), *The role of small urban centres in economic recovery and regional development* (pp. 1389). Nairobi: United Nations Centre for Regional Development (UNCRD).

Ngwenya, P., & Hassan, R. M. (2005). An environmental accounting approach to valuing the services of natural forest and woodlands in Swaziland. *Agrekon, 44*, 264-283.

Nordic Agency for Development and Ecology (NORDECO), & Department of Environment and Natural Resources (DENR). (2002). *DENR biodiversity monitoring system: Implementation results from eight protected areas January 1999-June 2001, the Philippines*. Manila: DENR.

Noss, A. J. (2000). Cable nets and snares in the Central African Republic. In J. G. Robinson, & E. L. Bennett (Eds.), *Hunting for sustainability in tropical forests* (pp. 282-304). New York: Columbia University Press.

Noss, A. J., Oetting, I., & Cuellar, R. L. (2005). Hunter self-monitoring by the Isoseño-Guaraní in the Bolivian Chaco. *Biodiversity and Conservation, 14*(11), 2679-2693.

Notten, A. (2004). *Rhoicissus tomentosa.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantqrs/rhoicistom.htm

Ntsebeza, L. (2007). Land redistribution in South Africa: The property clause revisited. In L. Ntsebeza, & R. Hall (Eds.), *The land question in South Africa: The challenge of transformation and redistribution* (Electronic ed., pp. 107-131). Cape Town: Human Sciences Research Council.

Obura, D. O., Wells, S., Church, J., & Horrill, C. (2002). Monitoring of fish and fish catches by local fishermen in Kenya and Tanzania. *Marine and Freshwater Research, 53*(2), 215-222.

Ogutu, J. O., Piepho, H.-P., Dublin, H.T., Bhola, N., & Reid, R. S. (2009). Dynamics of Mara-Serengeti ungulates in relation to land use changes. *Journal of Zoology, 278*(1), 1-14.

Oestreicher, J. S., Benessaiah, K., Ruiz-Jaen, M. C., Sloan, S., Turner, K., Pelletier, J., et al. (2009). Avoiding deforestation in Panamanian protected areas: An analysis of protection effectiveness and implications for reducing emissions from deforestation and forest degradation. *Global Environmental Change, 19*(2), 279-291.

Ostrom, E. (1990). *Governing the commons: The evolution of institutions for collective action*. New York: Cambridge University Press.

Ostrom, E. (1994). Neither market nor state: Governance of common-pool resources in the twenty-first century. *International Food Policy Research Institute* (*IFPRI) Lecture Series 2,*

Ostrom, E. (2003). How types of goods and property rights jointly affect collective action. *Journal of Theoretical Politics, 15*(3), 239-270.

Pagiola, S. (2005). *Assessing the efficiency of payments for environmental services programs: A framework for analysis*. Washington, D.C.: World Bank.

Pagiola, S. (2008). Payments for environmental services in Costa Rica. *Ecological Economics, 65*(4), 712-724.

Pagiola, S., Agostini, P., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., et al. (2005b). Paying for biodiversity conservation services: Experience in Colombia, Costa Rica, and Nicaragua. *Mountain Research and Development, 25*(3), 206-211.

Pagiola, S., Arcenas, A., & Platais, G. (2005a). Can payments for environmental services help reduce poverty? an exploration of the issues and the evidence to date from Latin America. *World Development, 33*(2 SPEC. ISS.), 237-253.

Pagiola, S., Landell-Mills, N., & Bishop, J. (2002). Making market-based mechanisms work for forests and people. In S. Pagiola, J. Bishop & N. Landell-Mills (Eds.), *Selling forest environmental services: Market-based mechanisms for conservation*. London: Earthscan.

Pagiola, S., & Platais, G. (2007). *Payments for environmental services: From theory to practice*. Washington, D.C.: World Bank.

Pagiola, S., Ramírez, E., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., et al. (2007b). Paying for the environmental services of silvopastoral practices in Nicaragua. *Ecological Economics, 64*(2), 374-385.

Pagiola, S., Rios, A. R., & Arcenas, A. (2007a). *Poor household participation in payments for ecosystem services: Lessons from the silvopastoral project in Quindio, Colombia*. Washington, D.C.: World Bank.

Pagiola, S., Rios, A. R., & Arcenas, A. (2008). Can the poor participate in payments for environmental services? lessons from the silvopastoral project in Nicaragua. *Environment and Development Economics, 13*(3), 299-325.

Palmer, A. R. (2005). *Makana LEAP: Livestock and commonage management*. Grahamstown: Agricultural Research Council - Range and Forage Institute.

Pandey, D. (2002). *Fuelwood studies in India: Myth and reality*. Bogor: Center for International Forestry Research (CIFOR).

Pattanayak, S. K., Mercer, D. E., Sills, E., & Yang, J. -. (2003). Taking stock of agroforestry adoption studies. *Agroforestry Systems, 57*(3), 173-186.

Pattanayak, S. K., & Sills, E. (2001). Do tropical forests provide natural insurance? the microeconomics of non-timber forest product collection in the Brazilian Amazon. *Land Economics, 77*, 595-613.

Penzhorn, B. J., Robertse, P., & Olivier, M. (1974). The influence of the African elephant on the vegetation of the Addo Elephant National Park. *Koedoe, 17*, 137-158.

Peres, C. A., & Terborgh, J. (1995). Amazonian nature-reserves: An analysis of the defensibility status of existing conservation units and design criteria for the future. *Conservation Biology, 9*, 34-46.

Peskett, L., Brown, D., & Luttrell, C. (2006). *Can payments for avoided deforestation to tackle climate change also benefit the poor?* No. 12). London: Overseas Development Institute.

Peskett, L., & Harkin, Z. (2007). *Risk and responsibility in reducing emissions from deforestation and degradation* No. 15). London: Overseas Development Institute.

Petrosillo, I., Muller, F., Jones, K., Zurlini, G., Krauze, K., Victorov, S., et al (Eds.). (2007). *Use of landscape sciences for the assessment of environmental security*. The Netherlands: Springer Science and Business Media B.V.

Petty, R. E., Unnava, R. H., & Strathman, A. J. (1992). Theories of attitude change. In T. S. Robertson, & H. H. Kassirjian (Eds.), *Handbook of consumer behavior* (pp. 241-268). Englewood Cliffs: Prentice Hall.

Pfaff, A. S. P., Kerr, S., Hughes, R. F., Liu, S., Sanchez-Azofeifa, G. A., Schimel, D., et al. (2000). The Kyoto protocol and payments for tropical forest: An interdisciplinary method for estimating carbon-offset supply and increasing the feasibility of a carbon market under the CDM. *Ecological Economics, 35*(2), 203-221.

Pfaff, A. S. P., Kerr, S., Lipper, L., Cavatassi, R., Davis, B., Hendy, J., et al. (2007). Will buying tropical forest carbon benefit the poor? evidence from Costa Rica. *Land use Policy, 24*(3), 600-610.

Pienaar, K., & May, H. (2003). SPP/LRC presentation at the HSRC Kimberley commonage workshop. Kimberley.

Pineiro, G., Paruelo, J. M., & Oesterheld, M. (2006). Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of southern South America. *Global Change Biology, 12*(7), 1267-1284.

Pote, J., Shackleton, C. M., Cocks, M. L., & Lubke, R. (2006). Fuelwood harvesting and selection in valley thicket, South Africa. *Journal of Arid Environments, 67*(2), 270-287.

Powell, M. (2008). *Personal communication*. MSc Candidate, Department of Environmental Science, Rhodes University, Grahamstown.

Powell, M. (2008). *Restoration of degraded subtropical thickets in the Baviaanskloof Megareserve, South Africa: The role of carbon stocks and* Portulacaria afra *survivorship.* Unpublished MSc Dissertation, Rhodes University, Grahamstown.

Powell, M. (2009). *Personal communication*. Researcher, Rhodes University, Grahamstown.

Powell, M., Mills, A. J., & Marais, C. (2006). Carbon sequestration and restoration: Challenges and opportunities for subtropical thicket restoration. *Multiple Use Management of Natural Forests and Woodlands: Policy Refinements and Scientific Progress,* 10.

Pretorius, E., De Klerk, G. W., & Van Rensberg, H. C. J. (1993). *The traditional healer in South African health care* No. ASS/BBS-27). Pretoria: Co-operative HSRC Programme.

Pyke, D. A., Herrick, J. E., Shaver, P., & Pellant, M. (2002). Rangeland health attributes and indicators for qualitative assessment. *Journal of Range Management, 55*(6), 584-597.

Rai, N. D., & Uhl, C. F. (2004). Forest product use, conservation and livelihoods: The case of *uppage* fruit harvest in the Western Ghats, India. *Conservation & Society, 2*(289), 313.

Rapport, D. J. (1995a). Ecosystem health: Exploring the territory. *Ecosystem Health, 1*(1), 5-13.

Rapport, D. J. (1995b). Ecosystem health: More than a metaphor? *Environmental Values, 4*(4), 287-309.

Rapport, D. J. (2007). Sustainability science: An ecohealth perspective. *Sustainability Science, 2*(1), 77-84.

Rapport, D. J., Costanza, R., & McMichael, A. J. (1998). Assessing ecosystem health. *Trends in Ecology and Evolution, 13*(10), 397-402.

Ras, B. (2008). *Data output from laboratory analysis of soil samples collected from the study site.* Unpublished manuscript.

Ready, J. (2007). *Helichrysum aureonitens.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/planthij/helichrysaureonit.htm

Reeder, J. D., & Schuman, G. E. (2002). Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environmental Pollution, 116*(3), 457-463.

Reid, R. S., & Ellis, J. E. (1995). Impacts of pastoralists on woodlands in south Turkana, Kenya: Livestock-mediated tree recruitment. *Ecological Applications, 5*(4), 978-992.

Renzhong, W., & Ripley, E. A. (1997). Effects of grazing on a *Leymus chinensis* grassland on the Songnen plain of north-eastern China. *Journal of Arid Environments, 36*(2), 307-318.

Republic of South Africa (RSA). (2003). *Estimates of national expenditures*. Pretoria: National Treasury.

Reuters. (2009). *REG-Mondi PLC: 2008 final dividend euro/sterling exchange rate R.* Retrieved August 23, 2009, from http://www.reuters.com/article/idUS107434+11-May-2009+PRN20090511

Richardson, F. D., Hahn, B. D., & Hoffman, M. T. (2005). On the dynamics of grazing systems in the semi-arid succulent Karoo: The relevance of equilibrium and non-equilibrium concepts to the sustainability of semi-arid pastoral systems. *Ecological Modelling, 187*(4), 491-512.

Richardson, L., & Loomis, J. B. (2009). The total economic value of threatened, endangered and rare species: An updated meta-analysis. *Ecological Economics, 68*(5), 1535-1548.

Ricketts, T. H., Daily, G. C., Ehrlich, P. R., & Fay, J. P. (2001). Countryside biogeography of moths in a fragmented landscape: Biodiversity in native and agricultural habitats. *Conservation Biology, 15*(2), 378-388.

Riethmuller, P. (2003). The social impact of livestock. *Animal Science Journal, 74*, 245-253.

Rietkerk, M., Ketner, P., Burger, J., Hoorens, B., & Olff, H. (2000). Multiscale soil and vegetation patchiness along a gradient of herbivore impact in a semi-arid grazing system in West Africa. *Plant Ecology, 148*(2), 207-224.

Rietkerk, M., Van, D. B., & Koppel, V. D. (1997). Site-specific properties and irreversible vegetation changes in semi-arid grazing systems. *Oikos, 80*(2), 241-252.

Roberston, N., & Wunder, S. (2005). *Fresh tracks in the forest: Assessing incipient payments for environmental services initiatives in Bolivia*. Bogor: Centre for International Forestry Research (CIFOR).

Roberts, R. L., Donald, P. F., & Fisher, I. J. (2005). Wordbirds: Developing a web-based data collection system for the global monitoring of bird distribution and abundance. *Biodiversity and Conservation, 14*(11), 2807-2820.

Roques, K. G., O'Connor, T. G., & Watkinson, A. R. (2001). Dynamics of shrub encroachment in an African savanna: Relative influences of fire, herbivory, rainfall and density dependence. *Journal of Applied Ecology, 38*(2), 268-280.

Rosa, H., Kandel, S., & Dimas, L. (2003). *Compensation for environmental services and rural communities in the Americas: Lessons from the Americas and key issues for strengthening community strategies*. San Salvador: Programa Salvadoreño de Investigación sobre Desarrollo y Medio Ambiente (PRISMA).

Rouget, M., Cowling, R. M., Lombard, A. T., Knight, A. T., & Kerley, G. I. H. (2006). Designing large-scale conservation corridors for pattern and process. *Conservation Biology, 20*(2), 549-561.

Royal Botanic Gardens, Kew. *Clausena anisata.* Retrieved April 29, 2009, from http://apps.kew.org/efloras/namedetail.do?flora=fz&taxon=1251&nameid=3003

Salafsky, N., & Wollenberg, E. (2000). Linking livelihoods and conservation: A conceptual framework and scale for assessing the integration of human needs and biodiversity. *World Development, 28*(8), 1421-1438.

Sankhayan, P. L., & Hofstad, O. (2001). A village-level economic model of land clearing, grazing, and wood harvesting for sub-Saharan Africa: With a case study in southern Senegal. *Ecological Economics, 38*(3), 423-440.

Santilli, M., Moutinho, P., Schwartzman, S., Nepstad, D., Curran, L. M., & Nobre, C. (2005). Tropical deforestation and the Kyoto protocol. *Climatic Change, 71*(3), 267-276.

SAPPI. (2005). *2005 annual report*. Johannesburg: SAPPI.

Saruchera, M. (2004). *The context of land and resource rights struggles in Africa* No. 9). Cape Town: Institute for Poverty, Land and Agrarian Studies (PLAAS).

Saunders, D. A., & Hobbs, R. J. (1991). The role of corridors in conservation: What do we know and where do we go? In D. A. Saunders, & R. J. Hobbs (Eds.), *The role of corridors* (). Surrey: Beaty & Sons.

Savadogo, P., Sawadogo, L., & Tiveau, D. (2007). Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. *Agriculture, Ecosystems and Environment, 118*(1-4), 80-92.

Scarpa, R., & Bateman, I. J. (2000). Efficiency gains afforded by improved bid design versus follow-up valuation questions in discrete-choice CV studies. *Land Economics, 76*(2), 299-311.

Scherr, S. J., White, A., & Khare, A. (2004). *For services rendered: The current status and future potential of markets for the ecosystem services provided by tropical forests*. Yokohama: International Tropical Timber Organization.

Schlesinger, W. H., & Pilmanis, A. M. (1998). Plant-soil interactions in deserts. *Biogeochemistry, 42*(1-2), 169-187.

Schroth, G., Da Fonseca, A., Harvey, C. A., Gascon, C., Vasconcelos, H., & Izac, A. (2004). *Agroforestry and biodiversity conservation in tropical landscapes*. Washington, D.C.: Island Press.

Schulze, P. C., Wilcox, K. J., Swift, A., & Beckert, J. L. (2009). Fast, easy measurements for assessing vital signs of tall grassland. *Ecological Indicators, 9*(3), 445-454.

Scoones, I. (1994). New directions in pastoral development in Africa. In I. Scoones (Ed.), *Living with uncertainty* (pp. 1-36). London: Intermediate Technology Publications.

Scoones, I. (1999). New ecology and the social sciences: What prospects for a fruitful engagement? *Annual Review of Anthropology, 28*, 479-507.

Scoones, I., Melnyk, M., & Pretty, J. (1992). *The hidden harvest: Wild foods and agricultural systems. A literature review and annotated bibliography* (1st ed.). London: International Institute for Environment and Development (IIED).

Seidman, G. (1999). *Is South Africa different? sociological comparisons and theoretical contributions from the land of apartheid*

Shabalala, N., & Mosima, B. (2002). *Report on the survey of large and small scale agriculture*. Pretoria: Statistics South Africa (StatsSA).

Shachak, M., & Pickett, S. (1997). Linking ecological understanding and application: Patchiness in a dryland system. In S. Pickett, R. Ostfeld, M. Shachak & G. Likens (Eds.), *The ecological basis of conservation: Heterogeneity, ecosystems and biodiversity* (pp. 108-119). New York: Chapman-Hall.

Shackleton, C. M. (1993a). Are the communal grazing lands in need of saving? *Development Southern Africa, 10*(1), 65-78.

Shackleton, C. M. (1993b). Fuelwood harvesting and sustainable utilisation in a communal grazing land and protected area of the eastern Transvaal lowveld. *Biological Conservation, 63*(3), 247-254.

Shackleton, C. M. (1993c). *The prediction of woody productivity in the savanna biome, South Africa.* Unpublished PhD Dissertation, University of the Witwatersrand, Johannesburg.

Shackleton, C. M. (1998). Annual production of harvestable deadwood in semi-arid savannas, South Africa. *Forest Ecology and Management, 112*(1-2), 139-144.

Shackleton, C. M., Gambiza, J., & Jones, R. (2007a). Household fuelwood use in small electrified towns of the Makana district, Eastern Cape, South Africa. *Journal of Energy, 18*, 4-10.

Shackleton, C. M., Grundy, I., & Williams, A. (2004). Use of South Africa's woodlands for energy and construction. In M. Lawes, H. Eeley, C. M. Shackleton & B. Geach (Eds.), *Indigenous forest and woodlands in South Africa: Policy, people and practice* (pp. 337-366). Pietermaritzburg: University of KwaZulu-Natal Press.

Shackleton, C. M., Guthrie, G., Keirungi, J., & Stewart, J. (2003). Fuelwood availability and use in the Richtersveld National Park, South Africa. *Koedoe, 46*(2), 1-8.

Shackleton, C. M., & Shackleton, S. E. (2004). The importance of non-timber forest products in rural livelihood security and as safety nets: A review of evidence from South Africa. *South African Journal of Science, 100*(11-12), 658-664.

Shackleton, C. M., Shackleton, S. E., Buiten, E., & Bird, N. (2007b). The importance of dry woodlands and forests in rural livelihoods and poverty alleviation in South Africa. *Forest Policy and Economics, 9*(5), 558-577.

Shackleton, C. M., Shackleton, S. E., & Cousins, B. (2001). The role of land-based strategies in rural livelihoods: The contribution of arable production, animal husbandry and natural resource harvesting in communal areas in South Africa. *Development Southern Africa, 18*(5), 581-604.

Shackleton, C. M., Shackleton, S. E., Netshiluvhi, T. R., & Mathabela, F. R. (2005). The contribution and direct-use value of livestock to rural livelihoods in the Sand River catchment, South Africa. *African Journal of Range and Forage Science, 22*(2), 127-140.

Shackleton, C. M., Shackleton, S. E., Ntshudu, M., & Ntzebeza, J. (2002). The role and value of savanna non-timber forest products to rural households in the Kat River valley, South Africa. *Journal of Tropical Forest Products, 8*, 45-65.

Shackleton, S. E. (1990). Socio-economic importance of *Cymbopogon validus* in Mkambati Game Reserve, Transkei. *South African Journal of Botany, 56*, 675-682.

Shackleton, S. E. (2000). *Generic criteria and indicators for assessing the sustainability of common property/community based natural resource management systems* No. ENV-P-1-2000-042). Pretoria: Council for Scientific and Industrial Research.

Shackleton, S. E. (2005). *The significance of the local trade in natural resource products for livelihoods and poverty alleviation in South Africa.* Unpublished PhD Dissertation, Rhodes University, Grahamstown.

Shackleton, S. E., Campbell, B. M., Lotz-Sisitka, H., & Shackleton, C. M. (2008). Links between the local trade in natural products, livelihoods and poverty alleviation in a semi-arid region of South Africa. *World Development, 36*(3), 505-526.

Shackleton, S. E., & Shackleton, C. M. (2005). The contribution of Marula (*Sclerocarya birrea*) fruit and fruit products to rural livelihoods in the Bushbuckridge district, South Africa: Balancing domestic needs and commercialisation. *Forests Trees and Livelihoods, 15*(1), 3-24.

Shackleton, S. E., Shackleton, C. M., & Cousins, B. (2000). *Re-valuing the communal lands of Southern Africa: New understandings of rural livelihoods* No. 62). London: Overseas Development Institute (ODI).

Shahabuddin, G., & Prasad, S. (2004). Assessing ecological sustainability of non-timber forest produce extraction: The Indian scenario. *Conservation & Society, 2*, 235-250.

Shultz, S. D., Pinazzo, J., & Cifuentes, M. (1998). Opportunities and limitations of contingent valuation surveys to determine national park entrance fees: Evidence from Costa Rica. *Environment and Development Economics, 3*(1), 131-149.

Simberloff, D. (1998). Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? *Biological Conservation 83*(3), 247-257.

Simbi, T., & Aliber, M. (2000). *Agricultural employment crisis in South Africa* No. 3:1). Pretoria: Trade and Industrial Policy Secretariat.

Sims-Castley, R. (2002). *A preliminary review of gross financial incomes generated by industries dependent on thicket vegetation* No. 37). Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Smit, L., & Wiseman, K. (2001). Environmental resource economics as a tool for environmental management in the city of Cape Town. *Forum for Economics and Environment - First Conference Proceedings.*

Smith, J., & Scherr, S. J. (2002). *Forest carbon and local livelihoods: Assessment of opportunities and policy recommendations* No. 37). Jakarta and Washington, D.C.: Centre for International Forestry Research (CIFOR) and Forest Trends.

Smith, J., & Scherr, S. J. (2003). Capturing the value of forest carbon for local livelihoods. *World Development, 31*(12), 2143-2160.

Smith, A. M., & Sutton, S. G. (2008). The role of flagship species in the formation of conservation intentions. *Human Dimensions of Wildlife, 13*, 127-140.

Smithers, R. (1983). *The mammals of the Southern Africa subregion*. Pretoria: University of Pretoria.

Snowball, J., & Antrobus, G. (2008). *Ecotourism and socio-economic development: The impact of the conservation, economic, and social activities of private game reserves in the Eastern Cape*. Grahamstown: Indalo.

Snyman, H. A., & Du Preez, C. C. (2005). Rangeland degradation in a semi-arid South Africa - II: Influence on soil quality. *Journal of Arid Environments, 60*(3), 483-507.

Snyman, H. A., & Fouche, H. (1991). Production and water-use efficiency of semi-arid grasslands of South Africa as affected by veld condition and rainfall. *Water South Africa, 17*, 263-268.

Söderström, B., Kiema, S., & Reid, R. S. (2003). Intensified agricultural land-use and bird conservation in Burkina Faso. *Agriculture, Ecosystems and Environment, 99*(1-3), 113-124.

South African National Biodiversity Institute (SANBI), South African Medical Research Council (SAMRC) & University of the Western Cape. *Cissampelos capensis herba.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/medmonographs/cissampcap.pdf

South African Revenue Service (SARS). (2008). *SARS average exchange rates: Average exchange rates for purposes of the income tax act.* Retrieved February 16, 2009, from http://www.sars.gov.za/home.asp?pid=324

South African Weather Service (SAWS). (2005). *Air temperature and precipitation data, climate of South Africa (WB 42).*

Spiteri, A., & Nepal, S. K. (2006). Incentive-based conservation programs in developing countries: A review of some key issues and suggestions for improvements. *Environmental Management, 37*(1), 1-14.

Spiteri, A., & Nepal, S. K. (2008). Evaluating local benefits from conservation in Nepal's Annapurna Conservation Area. *Environmental Management, 42*(3), 391-401.

Stanley, D. (2005). Local perception of public goods: Recent assessments of willingness to pay for endangered species. *Contemporary Economic Policy, 23*(2), 165-179.

Statistics South Africa (StatsSA). (1996). *The land area of South Africa.* Retrieved August 23, 2009, from http://www.statssa.gov.za/census01/Census96/HTML/CIB/Land.htm

Statistics South Africa (StatsSA). (1998). *The people of South Africa, population census 1996.* Retrieved October 8, 2008, from http://www.statssa.gov/za/census01/census96/HTML/default.htm

Statistics South Africa (StatsSA). (2001). *Census 2001.* Retrieved January 25, 2009, from http://www.statssa.gov.za/census01/html/C2001DPLG.asp

Statistics South Africa (StatsSA). (2003). *Census 2001: Census in brief*. Pretoria: StatsSA.

Statistics South Africa (StatsSA). (2005). *Data supplied to national treasury for conversion of census 2001 data in line with the preliminary municipal boundaries as at December 2005.* Retrieved January 25, 2009, from http://www.statssa.gov.za/census01/html/C2001Interactive.asp

Statistics South Africa (StatsSA). (2007). *A national poverty line for South Africa.* Retrieved September 2, 2008, from http://www.treasury.gov.za

Statistics South Africa (StatsSA). (2009). *Consumer price index (CPI).* Retrieved February 16, 2009, from http://www.statssa.gov.za/keyindicators/cpi.asp

Statistics Sweden. (2009). *Statistical database.* Retrieved January 31, 2009, from http://www.ssd.scb.se/databaser/makro/start.asp?lang=2

Steffens, M., Kolbl, A., Totsche, K. U., & Kogel-Knabner, I. (2008). Grazing effects on soil chemical and physical properties in a semiarid steppe of Inner Mongolia (P.R. China). *Geoderma, 143*(1-2), 63-72.

Stiglitz, J. (2005). *Cleaning up economic growth.* Unpublished manuscript.

Stroosnijder, L. (1996). Modelling the effect of grazing on infiltration, runoff and primary production in the Sahel. *Ecological Modelling, 92*(1), 79-88.

Stuart-Hill, G. (1991). Adaptation of *Portulacaria afra* to defoliation. In P. Zacharias, G. Stuart-Hill & J. Midgley (Eds.). Howick: Grassland Society of Southern Africa.

Stuart-Hill, G. (1992). Effects of elephants and goats on the Kaffrarian succulent thicket of the Eastern Cape, South Africa. *Journal of Applied Ecology, 29*(3), 699-710.

Stuart-Hill, G., & Aucamp, A. (1993). Carrying capacity of the succulent valley bushveld of the Eastern Cape. *African Journal of Range and Forage Science, 10*, 1-10.

Stuart-Hill, G., & Danckwerts, J. (1988). Influence of domestic and wild animals on the future of succulent valley bushveld. *Pelea, 7*, 45-56.

Sullivan, S. (1999). The impacts of people and livestock on topographically diverse open wood- and shrub-lands in arid north-west Namibia. *Global Ecology and Biogeography, 8*(3-4), 257-277.

Sullivan, S., & Rohde, R. (2002). On non-equilibrium in arid and semi-arid grazing systems. *Journal of Biogeography, 29*(12), 1595-1618.

Surplus People Project. (1983). *Forced removals in South Africa* No. Vol. 1). Cape Town: Surplus People Project.

Suyanto, S., Pandu Permana, R., Khususiyah, N., & Joshi, L. (2005). Land tenure, agroforestry adoption, and reduction of fire hazard in a forest zone: A case study from Lampung, Sumatra, Indonesia. *Agroforestry Systems, 65*(1), 1-11.

Swallow, B. M., & McCarthy, N. (1999). Property rights, risks and livestock development in Africa: Issues and project approach. In N. McCarthy, B. M. Swallow, M. Kirk & P. Hazell (Eds.), *Property rights, risks and livestock development in Africa* (pp. 1-20). Washington, D.C. and Nairobi: International Food Policy Research Institute (IFPRI) and International Livestock Research Institute (ILRI).

Takimoto, A., Nair, P. K. R., & Nair, V. D. (2008). Carbon stock and sequestration potential of traditional and improved agroforestry systems in the West African Sahel. *Agriculture, Ecosystems and Environment, 125*(1-4), 159-166.

Takimoto, A., Nair, V. D., & Nair, P. K. R. (2009). Contribution of trees to soil carbon sequestration under agroforestry systems in the West African Sahel. *Agroforestry Systems, 76*(1), 11-25.

Teague, W. R., Kreuter, U. P., Grant, W. E., Diaz-Solis, H., & Kothmann, M. M. (2009). Economic implications of maintaining rangeland ecosystem health in a semi-arid savanna. *Ecological Economics, 68*(5), 1417-1429.

Thwala, W. (2003). *Land and agrarian reform in South Africa*. Johannesburg: Land Research Action Network (LRAN).

Ticheler, H. J., Kolding, J., & Chanda, B. (1998). Participation of local fishermen in scientific fisheries data collection: A case study from the Bangweulu Swamps, Zambia. *Fisheries Management and Ecology, 5*(1), 81-92.

Tiffen, M., Mortimore, M. J., & Gichuki, F. (1994). *More people, less erosion: Environmental recovery in Kenya*. Chilchester: John Wiley & Sons.

Timmermans, H. (2005). *Case studies on beekeeping and honey production in South Africa*. Grahamstown: Department of Environmental Science, Rhodes University.

Timsina, N. (2003). Prompting social justice and conserving montane forest environments: A case study of Nepal's community forestry programme. *The Geographical Journal, 169*, 236-242.

Tongway, D. J., & Ludwig, J. A. (1994). Small-scale resource heterogeneity in semi-arid landscapes. *Pacific Conservation Biology, 1*, 201-208.

Topp-Jørgensen, E., Poulsen, M. K., Lund, J. F., & Massao, J. F. (2005). Community-based monitoring of natural resource use and forest quality in montane forests and miombo woodlands of Tanzania. *Biodiversity and Conservation, 14*(11), 2653-2677.

Toulmin, C. (1986). *Access to food: Dry season strategies and household size among the Bambara of central Mali* No. 17). Brighton: Institute for Development Studies (IDS).

Townsend, W. R., Borman, A. R., Yiyoguaje, E., & Mendua, L. (2005). Cofan Indians' monitoring of freshwater turtles in Zabalo, Ecuador. *Biodiversity and Conservation, 14*(11), 2743-2755.

Traynor, C. H. (2008). *Juncus kraussii* harvesting in Umlalazi Nature Reserve, KwaZulu-Natal, South Africa: Socio-economic aspects and sustainability. *African Journal of Aquatic Science, 33*(1), 27-36.

Troy, A., & Wilson, M. A. (2006). Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. *Ecological Economics, 60*(2), 435-449.

Tseng, W. C., & Chen, C. C. (2008). Valuing the potential economic impact of climate change on the Taiwan trout. *Ecological Economics, 65*, 282-291.

Turner, S. (1997). *Mid-term review of the land reform pilot programme*. Pretoria: Department of Land Affairs in association with DANIDA, the European Commission, and UK-DFID.

Turpie, J. K. (2003). The existence value of biodiversity in South Africa: How interest, experience, knowledge, income and perceived level of threat influence local willingness to pay. *Ecological Economics, 46*(2), 199-216.

Turpie, J. K., Heydenrych, B., & Hassan, R. M. (2001). Accounting for fynbos: A preliminary assessment of the status and economic value of fynbos vegetation in the Western Cape. In R. M. Hassan (Ed.), *Accounting for stock and flow values of wooded land resources: Methods and results from South Africa* (pp. 73-116). Pretoria: Centre for Environmental Economics and Policy in Africa (CEEPA), University of Pretoria.

Turpie, J. K., Lechmere-Oertel, R., Sigwela, A. M., Antrobus, G., Donaldson, J. S., Robertson, H., et al. (2003). The ecological and economic implications of conversion to game farming in the xeric succulent thicket of the Eastern Cape, South Africa. In J. K. Turpie (Ed.), *An ecological-economic appraisal of conservation on commercial farmland in four areas of South Africa* (Final report of the ecological-economics research component of the Conservation Farming Project ed.). Cape Town: National Botanical Institute.

Turpie, J. K., Marais, C., & Blignaut, J. N. (2008). The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics, 65*(4), 788-798.

Twine, W., Moshe, D., Netshiluvhi, T. R., & Siphugu, V. (2003a). Consumption and direct-use values of savanna bio-resources used by rural households in Mametja, a semi-arid area of Limpopo province, South Africa. *South African Journal of Science, 99*(9-10), 467-473.

Twine, W., Siphugu, V., & Moshe, D. (2003b). Harvesting of communal resources by 'outsiders' in rural South Africa: A case of xenophobia or a real threat to sustainability? *International Journal of Sustainable Development and World Ecology, 10*(3), 263-274.

Ullsten, O. (2003). The politics of the environment. In D. J. Rapport, W. Lasley, D. Rolston, N. Nielsen, C. Qualset & A. Damania (Eds.), *Managing for healthy ecosystems* (pp. 11-14). Boca Raton: CRC Press.

United Nations Environment Programme (UNEP)-Risoe. (2008). *Clean development mechanism pipeline overview.* Nairobi: UNEP.

United Nations Food and Agriculture Organization (FAO). (2001). *Livestock keeping in urban areas: A review of traditional technologies based on literature and field experience* No. 151). Rome: FAO.

United Nations Food and Agriculture Organization (FAO). (2005). *Global forest resources assessment 2005*. Rome: FAO.

United Nations Framework Convention on Climate Change (UNFCCC). (2007a). *Decision 1/CMP.3 (Bali Action Plan)*.

United Nations Framework Convention on Climate Change (UNFCCC). (2007b). *Guidelines for completing the form for submission of bundled small-scale CDM project activities. version 1* No. F-CDM-SSC-BUNDLE). Bonn: UNFCCC.

United States Bureau of Labor Statistics. (2008). *Consumer price inflation index.* Retrieved January 24, 2009, from http://www.bls.gov

United States Census Bureau. (2007). *Current population survey. 2007 annual social and economic supplement*. Washington, D.C.: United States Census Bureau.

United States Census Bureau. (2008a). *Census 2000 urban and rural classification.* Retrieved January 31, 2009, from http://www.census.gov/geo/www/ua/ua_2k.html

United States Census Bureau. (2008b). *Historical income tables - Households* . Retrieved January 24, 2009, from http://www.census.gov/hhes/www/income/histinc/h08.html

United States Census Bureau. (2009). *Census 2000 gateway.* Retrieved January 31, 2009, from http://www.census.gov/main/www/cen2000.html

United States Central Intelligence Agency (CIA). (2009). *The world factbook.* Retrieved January 24, 2009, from https://www.cia.gov/library/publications/the-world-factbook/

Vågen, T.-G., Lal, R., & Singh, B. R. (2005). Soil carbon sequestration in sub-Saharan Africa: A review. *Land Degradation and Development, 16*(1), 53-71.

van der Walt, L. (2003). *Helichrysum splendidum.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/planthij/helichrysumsplend.htm

van der Walt, L. (2003). *Scabiosa incisa.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantqrs/scabincisa.htm

van der Walt, L. (2004). *Artemisia afra.* Retrieved April 29, 2009, from http://www.plantzAfrica.com/plantab/artemisafra.htm

Van Noordwijk, M., Susswein, P., Palm, C., Izac, A., & Tomich, T. (2001). *Problem definition for integrated natural resource management in forest margins of the humid tropics: Characterization and diagnosis of land use practices* No. 1). Bogor: World Agroforestry Center (ICRAF).

Van Rijsoort, J., & Jinfeng, Z. (2005). Participatory resource monitoring as a means for promoting social change in Yunnan, China. *Biodiversity and Conservation, 14*(11), 2543-2573.

Van Wilgen, B. W., Le Maitre, D. C., & Cowling, R. M. (1998). Ecosystem services, efficiency, sustainability and equity: South Africa's working for water programme. *Trends in Ecology and Evolution, 13*(9), 378.

van Wyk, A., & Smith, G. (2001). *Regions of floristic endemism in Southern Africa: A review with emphasis on succulents*. Pretoria: Umdaus Press.

Verbist, B., Putra, A. E. D., & Budidarsono, S. (2005). Factors driving land use change: Effects on watershed functions in a coffee agroforestry system in Lampung, Sumatra. *Agricultural Systems, 85*(3), 254-270.

Versveld, D., Le Maitre, D. C., & Chapman, R. (1998). *Alien invading plants and water resources in South Africa: A preliminary assessment*. Stellenbosch: Environment and Forestry Technology.

Vlok, J. H. J., & Euston-Brown, D. (2002). *Subtropical thicket ecosystem planning (STEP) project: Biological survey report (plant and birds)* No. 40). Port Elizabeth: Terrestrial Ecology Research Unit (TERU), University of Port Elizabeth.

Vlok, J. H. J., Euston-Brown, D., & Cowling, R. M. (2003). Acocks' valley bushveld 50 years on: New perspectives on the delimitation, characterisation and origin of subtropical thicket vegetation. *South African Journal of Botany, 69*(1), 27-51.

Walker, C. (2005). The limits to land reform: Rethinking 'the land question'. *Journal of Southern African Studies, 31*(4), 805-824.

Walker, C. (2007). Redistributive land reform: For what and for whom? In L. Ntsebeza, & R. Hall (Eds.), *The land question in South Africa: The challenge of transformation and redistribution* (Electronic ed., pp. 132-151). Cape Town: Human Sciences Research Council.

Walpole, M. J., & Goodwin, H. J. (2000). Local economic impacts of dragon tourism in Indonesia. *Annals of Tourism Research, 27*(3), 559-576.

Ward, C., & Parker, C. (2008). *The use and appreciation of urban green space: The example of botanical gardens in South Africa.* Unpublished Honours Dissertation, Rhodes University, Grahamstown.

Waters Meeting Nature Reserve (WMNR). (2007). *Integrated reserve management plan: Strategic management plan (SMP)* No. SMP/WMNR/01). East London: Eastern Cape Parks Board.

Wattage, P., & Mardle, S. (2008). Total economic value of wetland conservation in Sri Lanka identifying use and non-use values. *Wetlands Ecology and Management, 16*(5), 359-369.

White, A., & Martin, A. (2002). *Who owns the world's forests? forest tenure and public forests in transition*. Washington, D.C.: Forest Trends.

Whynes, D. K., Wolstenholme, J. L., & Frew, E. (2004). Evidence of range bias in contingent valuation payment scales. *Health Economics, 13*(2), 183-190.

Wickramasinghe, A., Ruiz Perez, M., & Blockhus, J. (1996). Nontimber forest product gathering in Ritigala Forest (Sri Lanka): Household strategies and community differentiation. *Human Ecology, 24*, 493-519.

Wiersum, K. F., Dold, A. P., Husselman, M., & Cocks, M. L. (2006). Cultivation of medicinal plants as a tool for biodiversity conservation and poverty alleviation in the Amatola region, South Africa. In R. J. Bogers, L. E. Craker & D. Lange (Eds.), *Medicinal and aromatic plants* (pp. 43-57). London: Springer.

Wilkie, D. S., & Carpenter, J. F. (1999). Bushmeat hunting in the Congo basin: An assessment of impacts and options for mitigation. *Biodiversity and Conservation, 8*, 927-955.

Wilkie, D. S., Carpenter, J. F., & Zhang, Q. (2001). The under-financing of protected areas in the Congo basin: So many parks and so little willingness-to-pay. *Biodiversity and Conservation, 10*(5), 691-709.

Wilkie, D. S., Starkey, M., & Abernethy, K. (2005). Role of prices and wealth in consumer demand for bushmeat in Gabon, Central Africa. *Conservation Biology, 19*, 268-274.

Williams, C. K., Ericsson, G., & Heberlein, T. A. (2002). A quantitative summary of attitudes toward wolves and their reintroduction (1972-2000). *Wildlife Society Bulletin, 30*(2), 575-584.

Working for Water Programme (WFWP). (1998). *The environmental impacts of invading alien plants in South Africa.* Unpublished manuscript.

Working for Wetlands (WfWet). (Undated). *Managing rivers wisely.* Unpublished manuscript.

World Wide Fund for Nature (WWF). (2007). *Water neutrality: The WWF water neutral scheme. proposal to assist South African breweries limited in achieving operational "water neutrality" at its Newlands and Ibhayi breweries* (November 6, 2007 ed.). South Africa: WWF.

Wotshela, L. (2004). Territorial manipulation in Apartheid South Africa: Resettlement, tribal politics and the making of the northern Ciskei, 1975-1990. *Journal of Southern African Studies, 30*(2), 317-337.

Wunder, S. (2005). *Payments for environmental services: Some nuts and bolts* No. 42). Bogor: Center for International Forestry Research (CIFOR).

Wunder, S., & Albán, M. (2008). Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. *Ecological Economics, 65*(4), 685-698.

Wunder, S., Engel, S., & Pagiola, S. (2008). Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics, 65*(4), 834-852.

Young, H. (1992). *Food security and famine: Assessment and response* No. 7). London: Oxfam Publications.

Zhang, Q., Devers, D., Desch, A., Justice, C. O., & Townshend, J. (2005). Mapping tropical deforestation in Central Africa. *Environmental Monitoring and Assessment, 101*(1-3), 69-83.

Zhao, Y., Peth, S., Krummelbein, J., Horn, R., Wang, Z., Steffens, M., et al. (2007). Spatial variability of soil properties affected by grazing intensity in Inner Mongolia grassland. *Ecological Modelling, 205*(1-2), 241-254.

Zurlini, G., & Girardin, P. (2008). Introduction to the special issue on "ecological indicators at multiple scales". *Ecological Indicators, 8*(6), 781-782.

APPENDICES

APPENDIX 1: KOWIE RIVER ENDANGERED SPECIES CV SURVEY

	Grahamstown Port Alfred Bathurst	I, the respondent have been informed of my rights as a respondent and grant my permission to take part in this survey. ☐				**START TIME**:
A	**Question**					
1)	Years you have lived in [town]?					
2)	What is the highest level of education you have completed?					
3)	What is your profession/occupation?					
4)	What is your age?					
5)	Gender?					
6)	What is your primary language?					
B	**Wildlife appreciation**					
7)	Do you **spend time at** or **use** the Kowie?	YES		NO		
8)	**Where** did you last visit the Kowie R.?					
9)	**How often** do you visit or use the river?	First time – very seldom	Occasionally over holidays	A weekend per month	2 - 3 weekends per month	Either every weekend or more often
10)	What are your **main reasons** for visit(s)?	Hiking	Fishing	Game viewing	Canoe/kayak/ water sports	Other: Please list: ________
11)	Have you ever **stayed overnight** in one of the local nature reserves (Thomas Baines or Waters Meeting Nature Reserve)?	No, never	Yes, once in my life	Yes, more than 1x in past 5 years	Yes, multiple times a year	I have been there but for day visits only.
12)	What do you **appreciate the most** about the river – what attracts you to it?	Please answer freely (no more than 3 items please), but examples could include, Wildlife, Aesthetic beauty, Locality				
13)	Do you feel there is anything about the river that **needs attention**?	Please answer freely (no more than 3 items please), but examples could include, Water quality, Wildlife, or Nothing at all				
14)	Have you ever seen **evidence of leopard** in the Kowie River valley?	Heard about it recently (<5 yr)	Read about it recently (<5 yrs)	Seen evidence recently (<5 yr)	Yes, but long ago (>5 yrs)	NO, never

15)	Have you or anyone you know **lost livestock to leopard** in the *past 5 years*?	No, I don't know anyone	Yes, I have lost my livestock	Yes, I know someone else	Yes, I've heard of it happening recently	Yes, I've heard of it but it was long ago
16)	Are you **familiar with the EC Rocky**, a critically endangered fresh water fish only found in the Kowie river?	No, I've never heard of it	Yes, I've heard of it but I haven't seen	Yes, I've seen one in a museum	Yes, I've seen the EC Rocky in the Kowie	

		1	2	3	4	5
C	**Please use the categories at right to describe your reaction to the following:**	**Totally disagree**	**Disagree slightly**	**Indifferent**	**Agree slightly**	**Totally agree**
17)	It is important to conserve local wildlife.					
18)	Leopards are a serious threat to livestock.					
19)	Species that only exist in this area, such as the EC Rocky, should be protected.					
20)	The quality of the Kowie River and its wildlife have improved over time.					
21)	The quality of the Kowie has no impact on my enjoyment of my visit to the river.					
22)	The quality of the Kowie does not affect my use of the river (business/recreation).					
23)	Residents should **contribute financially** to the protection of the Kowie River and endangered wildlife that live in the area.					

D The paragraph below presents a **completely hypothetical** scenario for research purposes only. Please consider the questions that follow from the perspective of your **entire household**, keeping in mind the money you have to spend on all your HH expenses.

The leopard is one of the only major predators left in the Kowie River valley, and it plays an important role in regulating natural food chains. Although leopards used to roam freely along the Kowie River, they have been eliminated from many areas due to a number of factors, including illegal hunting and incompatible land uses that have reduced its territory size. Suppose a fund was created to ensure the future survival of leopard in the Kowie River valley. Possible interventions might include monitoring the movement of leopard in the valley to alert farmers of their presence and/or increasing available habitat.

24)	Would your **household contribute anything on an annual basis** to ensure the continued survival of local leopards?	YES	NO	Why (not)?

25)	What items would you **donate annually** to a leopard conservation fund, such as money, time, labor, supplies or others?	Not willing	Money (Rands)	Time (hrs)	Labor (hrs)	Supplies: specify	Other: specify
26)	How much **money** would your **household** donate annually to a leopard conservation fund? (*respondent bids up to WTP*)	0 20 40 80 100 120 140 160 180 200 220 240 260 280 300 Other: (please specify)					
27)	How much **time** would you donate to the collection/**administration** of such a fund?	Not willing	2 hrs/yr	2 hrs/mo	2 hrs/wk	>2 hrs/wk	Other: specify

28)	How many **hours of labor** would you donate, such as to remove leopard snares?	Not willing	2 hrs/yr	2 hrs/mo	2 hrs/wk	>2 hrs/wk	Other: specify
29)	What kind of **supplies** would you donate to the leopard conservation fund?	Please answer freely (no more than 3 items please), but examples could include, Tracking equipment, Land or a Bakkie					
E	Again, the paragraphs below present a **completely hypothetical** scenario for research purposes only. Please consider the questions that follow from the perspective of your **entire household**, keeping in mind the money you have to spend on all your household expenses, including food, electricity, transportation, and donations to other conservation-related causes. The Eastern Cape Rocky is a small fish that grows to about 30 cm long and only exists in tributaries of three rivers in the Eastern Cape, one of which is the Kowie River. According to a local expert, the Rocky may go extinct in the next ten years unless management actions are taken to protect it from local threats, such as loss of habitat, invasive alien fish like bass and catfish, and sedimentation in the Kowie. Overgrazing on the Bathurst commonage and poor crop management on farms adjacent to the river can lead to soil erosion and silt up the Kowie. Because the Rocky is so specially adapted to its habitat, it acts as an indicator of the health of the Kowie River as an ecosystem. Therefore, the sharp decline in Rocky numbers observed over the past thirty years in the Kowie River may be an indication of ecosystem decline. Suppose that an organization was formed to clean up the Kowie River and protect the Eastern Cape Rocky. Possible interventions might include encouraging farmers along the river to adopt erosion control practices and reduce their use of harmful chemicals and/or more careful regulation of cattle on the commonage.						
30)	Would your **household contribute on an annual basis** to improve the health of the Kowie River and ensure the continued survival of the EC Rocky?	YES	NO	Why (not)?			
31)	What **items would you donate annually** to clean up the Kowie River, such as money, time, labor, supplies or others?	Not willing	Money (Rands)	Time (hrs)	Labor (hrs)	Supplies: specify	Other: specify
32)	How much **money** would your household donate annually to clean up the Kowie?	0 20 40 80 100 120 140 160 180 200 220 240 260 280 300 Other: (please specify)					
33)	How much **time** would you donate to the **administration** of such an organization?	Not willing	2 hrs/yr	2 hrs/mo	2 hrs/wk	>2 hrs/wk	Other: specify
34)	How many **hours of labor** would you donate to the project, such as to remove invasive fish or meet with landowners?	Not willing	2 hrs/yr	2 hrs/mo	2 hrs/wk	>2 hrs/wk	Other: specify
35)	What kind of **supplies** would your household donate to clean up the Kowie?	Please answer freely (no more than 3 items please), but examples could include, Fishing equipment or a Bakkie					
	END TIME:						

Additional comments: Please feel free to expand on the answers you've shared or add anything else we haven't already discussed.

THANK YOU FOR TAKING THE TIME TO COMPLETE THIS SURVEY.

APPENDIX 2: PRIVATE GAME RESERVE ECOTOURISM SURVEY

Thank you for taking the time to complete this survey. We appreciate that the information shared below is privileged and your answers will be treated accordingly. No responses will be identified individually, but your answers may be included as points in a graph showing the distribution of responses from various regional nature reserves. Should you have any questions or concerns, please feel free to contact Mercedes at mmstickler@gmail.com.

	Business:______________________ I, the respondent, have been informed of my rights and grant my permission to take part in this survey. ☐						FOR OFFICE USE:
A	**General information**						
1)	How many years old is your business?						
2)	How many hectares is your reserve?						
3)	When did you purchase the land?						
4)	What was the primary land use before this one?	Game farm	Cattle ranch	Crops	Homestead	Other: please list: __________________	
5)	What is the ownership structure? (Please circle)	Family-run business			Private business with shareholders		
6)	Tourism Council Grading (Stars)?	Not graded	1	2	3	4	5
B	**Natural capital**						
7)	What kinds of **vegetation** are found within your reserve?	Subtropical thicket	Savanna or Grassland	Forest	Fynbos	Other: please list: __________________	
8)	**How many of the "Big Five"** are represented on your reserve?	None	1	2	3	4	5
9)	Roughly **how many game animals** are located on your reserve?						
10)	What would you say are the **primary visitor attractions** to your reserve?	Please answer freely, but examples could include: Wildlife viewing, natural landscape, outdoor sporting activities, etc.					
11)	What are most **unique or special natural features** of your reserve?	Please answer freely, but examples could include: Riverine ecosystem, dense natural vegetation, abundant wildlife, etc.					

C	Visitor attractions offered	Wildlife viewing	Hiking trails	Outdoor sports (e.g. cycling, climbing, horse riding)	Water sports (e.g. canoeing, fishing)	Other: please list: ________________
13)	Which of these attractions are offered at your reserve? (please check all that apply)					
14)	What **tariffs/fees** (if any) are charged for these activities? (please list all that apply)					
15)	How many **visitors per year** experience these attractions on your reserve? (please list for each one that applies)					
16)	Average **annual** number of **day** visitors (total per year)					

D	Capital investments	Please circle the appropriate range or fill in where appropriate.
17)	Current value of land in your area?	**Please list in rand/hectare:**
18)	Total square meters of all service buildings?	Please list **area of all service buildings** or the # of buildings and approximate size pe building:
19)	Total square meters of all tourist buildings?	Please list **area of all tourist buildings** (accommodation, education center, etc.) or the of buildings and approximate size per building:
20)	Tariffs/fees (if any) charged to use/rent any non-accommodation buildings?	Please list **hourly/daily tariffs** for renting any **conference/education centers**, etc.
21)	How many visitors per year access each kind of building?	Please list **approximate number of paying guests** that access each **non-accommodatio** building.

E	Accommodation type	Lodge	Guest house or cottage	Dormitory	Camping	Other: please list: ____________
22)	Which of the overnight accommodation options listed above does your reserve offer? (please check all that apply)					
23)	How many **"beds" per night** are available? (please fill in for each category that applies to your reserve)					
24)	What **rates** are charged per night? (fill in for each applicable category)					
25)	What is the average **annual** percent occupancy for each accommodation? (fill in for each applicable category)					

26)	Number or extent (i.e. length) of **infrastructure you have built** since taking over the reserve	Waterhole or Dam	Paved Rd	Dirt Rd	Parking	Bridges	Drainage	Other: please list: ____________
27)	Number or extent (i.e. length) of **infrastructure you have inherited** from the previous owner	Waterhole or Dam	Paved Rd	Dirt Rd	Parking	Bridges	Drainage	Other: please list: ____________

28)	**Number** and approximate **age** of all heavy machinery, equipment and vehicles	Service vehicles	Visitor vehicles	Tractors/large machinery	Animal care	Other: please list: ____________

29)	Game animals	Please list the **approximate current value of all game animals:**
30)	Rehabilitation of ecosystem	Please list **actions taken including the extent** (area):

31)	**Other capital costs**: Please list: ____________	None	<R500,000	R500,000 – R1 million	R1-2 million	R2-5 million	>R5 millio

F	Operations							
32)	How many **salaried staff** does your reserve employ in each of the categories at right?	Grounds	Housekeeping	Guides	Catering	Marketing	Management	Other: pleas list: ___________
33)	**Percentage of marketing costs** directed at different markets	Eastern Cape	South Africa	UK	EU excl UK		USA	Other: plea list: ___________

34)	Approximate **annual gross revenue** (Please check a box)	R0-1/2 million	R1/2-1 million	R1-5 million	R5-10 million	R10-20 million	R20-50 million	>R50 millior
	High							
	Low							

G	Please use the categories at right to describe your reaction to the following statements:	Totally disagree	Disagree slightly	Indifferent	Agree slightly	Totally agree
35)	Intact natural vegetation *improves* the visitor's experience of ecotourism.					
36)	Natural features are *less* important than the services/attractions offered in visitors' choice of reserve.					
37)	Too many tourists at one time *reduce* each visitor's overall satisfaction with their experience at the reserve.					
38)	Existing businesses will *not* be able to keep pace with future demand for ecotourism in the Eastern Cape.					
39)	**This reserve** plans to *expand* its operations in the next five years.					
40)	Ecotourism growth in the Eastern Cape will *peak* in 5-10 years.					
41)	What constraints might hinder the future development of nature-based tourism in the Eastern Cape?	Please answer freely, but examples could include: Government policy, alien/invasive species, water availability and quality, demand				

THANK YOU FOR TAKING THE TIME TO COMPLETE THIS SURVEY.
Please return this survey to mmstickler@gmail.com or fax it to the Rhodes Department of Environmental Science (Attention: Charlie Shackleton) at 046 622 9319.