Journal of Environmental Management 151 (2015) 472–485

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Reorienting land degradation towards sustainable land management: Linking sustainable livelihoods with ecosystem services in rangeland systems

M.S. Reed [a,*], L.C. Stringer [b], A.J. Dougill [b], J.S. Perkins [c], J.R. Atlhopheng [c], K. Mulale [c], N. Favretto [b]

[a] Knowledge ExCHANGE Research Centre of Excellence, Birmingham School of the Built Environment, Birmingham City University, Millennium Point, Curzon Street, Birmingham B4 7XG, UK
[b] Sustainability Research Institute, School of Earth & Environment, University of Leeds, West Yorkshire LS2 9JT, UK
[c] Department of Environmental Science, University of Botswana, Private Bag, 00704 Gaborone, Botswana

ARTICLE INFO

Article history:
Received 24 April 2014
Received in revised form
4 November 2014
Accepted 7 November 2014
Available online 21 January 2015

Keywords:
Land degradation
Desertification
Bush encroachment
Botswana
Land policy
Payments for ecosystem services
Economics of land degradation

ABSTRACT

This paper identifies new ways of moving from land degradation towards sustainable land management through the development of economic mechanisms. It identifies new mechanisms to tackle land degradation based on retaining critical levels of natural capital whilst basing livelihoods on a wider range of ecosystem services. This is achieved through a case study analysis of the Kalahari rangelands in southwest Botswana. The paper first describes the socio-economic and ecological characteristics of the Kalahari rangelands and the types of land degradation taking place. It then focuses on bush encroachment as a way of exploring new economic instruments (e.g. Payments for Ecosystem Services) designed to enhance the flow of ecosystem services that support livelihoods in rangeland systems. It does this by evaluating the likely impacts of bush encroachment, one of the key forms of rangeland degradation, on a range of ecosystem services in three land tenure types (private fenced ranches, communal grazing areas and Wildlife Management Areas), before considering options for more sustainable land management in these systems. We argue that with adequate policy support, economic mechanisms could help reorient degraded rangelands towards more sustainable land management.

1. Introduction

Drylands occupy approximately 41% of the world's land area and support the livelihoods of around 2 billion people (Middleton et al., 2011). Between 10 and 20% of the world's drylands are considered degraded (medium certainty) (Millennium Ecosystem Assessment, 2005a). However, land degradation is subject to a range of different definitions and measurements. Conceptualisations range from those that focus more on biophysical functions and changes (e.g. Holling, 1986; Dean et al., 1995; IPCC, 2001), to those based primarily on changes in the productive potential of the land for human use (e.g. UNEP, 1992, 1997; Kasperson et al., 1995; ELD Initiative, 2013). The former tends to emphasise biophysical assessments of natural capital stocks (e.g. using ecological and soil-based approaches and remote sensing) and approaches to tackling land degradation based on techniques such as soil stabilization and re-vegetation. The latter focuses more on assessing flows of ecosystem services,[1] and considers the perceptions of local communities and economic indicators (e.g. productivity trends based on livestock census data). Increasingly, assessments are combining biophysical and socio-economic approaches to provide a more holistic and contextualised picture of dryland degradation (e.g. Milton et al., 2003; Katjiua and Ward, 2007; Klintenberg et al.,

* Corresponding author.
E-mail address: Mark.Reed@bcu.ac.uk (M.S. Reed).

[1] Ecosystem services are the benefits that humans derive from the natural environment. They are typically grouped as: supporting services (necessary for the production of other ecosystem services e.g. soil formation, photosynthesis and nutrient cycling); provisioning services (ecosystem products e.g. food, fibre and water); regulating services (including processes such as climate stabilisation, erosion regulation and pollination); and cultural services (non-material benefits from ecosystems e.g. spiritual fulfilment, cognitive development and recreation) (Millennium Ecosystem Assessment, 2003).

http://dx.doi.org/10.1016/j.jenvman.2014.11.010
0301-4797/

2007; Reynolds et al., 2007, 2011; Reed et al., 2008). Such combined assessments are essential for policy development, enabling decision-makers to assess the likely socio-economic and environmental impacts of different policy options (Arntzen, 2001), and identify policy mechanisms that can realistically be implemented to tackle land degradation and help shift practices towards sustainability (Schwilch et al., 2011).

This paper presents new ways of moving from land degradation towards sustainable land management (SLM)[2] through the use of economic mechanisms. The paper fuses insights from the ecosystem services framework with Sustainable Livelihoods Analysis, to identify economic opportunities arising from the ecosystem services provided by SLM. This is explored using a case study of bush encroachment[3] in Botswana's Kalahari sandveld, with focus on livestock producing areas of Kgalagadi District. This is an area in which land tenure is varied, providing a context from which insights may be applied across a range of international situations. On a global scale, bush encroachment has been described as "the most widespread problem on dryland pastures" (Warren and Agnew, 1988: 6). It reduces the availability and heterogeneity of forage resources (Scoones, 1995) by reducing grass production whilst increasing the abundance of invasive species that are unpalatable to cattle (Perkins and Thomas, 1993). Such land degradation can therefore lead to a reduction in economic outputs in cattle-based livelihood systems (Quan et al. 1994; Scholes, 2003).

The paper first describes the socio-economic and ecological characteristics of the Kalahari in southwest Botswana and a range of perspectives on the land degradation taking place. It then focuses on bush encroachment to explore how a focus on the flows of ecosystem services supporting livelihoods, could inform the development of new economic measures. It does this by evaluating the likely impacts of bush encroachment on ecosystem services in three land tenure types (including communal rangelands, privately owned rangelands and Wildlife Management Areas (WMAs)), before considering opportunities for more sustainable land management that would improve the provision of ecosystem services. By exploring the extent to which SLM is able to enhance ecosystem service provision, it is possible to systematically consider some of the costs, benefits and trade-offs associated with different ways of tackling land degradation. In particular, the paper assesses the potential for Payments for Ecosystem Services (PES) schemes to enhance livelihoods in semi-arid rangeland systems, to help pay for land degradation remediation and incentivise more sustainable land management.

2. Case study area: Kalahari rangelands

Our analysis focuses on the Kalahari rangelands, particularly Kgalagadi District in Southwestern Botswana (Fig. 1). We present a critical evaluation of ongoing integrated analyses from private game ranches (Tshabong and Bokspits areas), karakul sheep ranches (Bokspits area), communal livestock grazing (unfenced cattle posts) (Tshabong area) and the Kgalagadi Transfrontier Park and its surrounding WMAs. Information about each area is summarised in Table 1.

2.1. Existing perspectives on land degradation and sustainable rangeland management in the Kalahari

Botswana has been described as "one of the most desertified countries in sub-Saharan Africa" (Barrow, 1991: 191). However, assessments of land degradation severity and extent can vary significantly, depending on the methods and scales under consideration. This section briefly reviews the different approaches that have been used to date, underscoring the need for a more integrated approach.

Biophysical conceptualisations of land degradation tend to focus on ecological and abiotic functions and their changes over time, to assess changes in stocks of natural capital. This typically relies on ecological, soil-based and remote sensing methods to assess biophysical indicators of land degradation. For example, bush encroachment around water points has been observed in numerous ecological studies throughout the Kalahari (e.g. Perkins and Thomas, 1993; Moleele and Perkins, 1998; Dougill et al., 1999; Moleele et al., 2002). These zones generally occur across areas of between 1 and 4 km, but can extend much further. For example, in parts of south Kgalagadi district (e.g. between Tshabong and Omaweneno; Fig. 1), bush encroached zones around water points are coalescing, where they can extend up to 9 km from individual boreholes, resulting in impenetrable stretches of bush for tens of kilometres (Reed, 2005). Ecological studies have also documented the retreat of grass cover up to 18 km around the Matsheng villages (Moleele and Mainah, 2003; Moleele and Chanda, 2003; Chanda et al., 2003). The exact causes of bush encroachment remain contested, with dynamic ecological models assigning differing importance to variables such as grazing levels (Dougill et al., 1999), changing fire regimes (Joubert et al., 2012a,b) and the effects of CO_2 fertilisation (Bond and Midgley, 2000). The interaction of these driving forces and factors affecting ecological tipping points remains poorly understood (Sietz et al., 2011; Lohmann et al., 2012).

Using remotely sensed data, Tanser and Palmer (1999) noted significantly lower standing biomass, lower basal cover, and more bare soil in intensively grazed communal rangeland in comparison to WMAs and National Parks. However, it is not possible to determine vegetation height using satellite data alone, so it is difficult to distinguish between bush encroachment and natural tree cover. "Rangeland condition" maps based on Normalised Difference Vegetation Index readings are used by policy-makers to identify degradation extent and assess the extent of droughts and wildfire impacts (Reed, 2005). Although time series analysis of such data can shed some light on degradation issues, any interpretation should be based on a detailed understanding of how land use and land management has varied over time within the study area.

Socio-economic conceptualisations of land degradation tend to focus on changes in the productive potential of the land for human use. These generally rely on economic indicators, and opinions of local communities and other stakeholders. Most of the evidence based on this approach in the Kalahari has focused on changes in provisioning services, with a focus on trends in livestock populations, as the main source of livelihoods and Botswana's main agricultural export (e.g. Reed, 2005; Reed et al., 2007; Dougill et al., 2010).

Local perceptions are often at variance with published assessments of land degradation and SLM. For example, Thomas and Twyman (2004) and Reed et al. (2007) found that land managers (mainly owning goats and sheep) in southwest Botswana regarded the encroacher *Rhigozum trichotomum* as an important forage resource and windbreak. This was contrary to views in South African literature that bush encroachment by this species was a major problem in this region for communities whose livelihoods depend on cattle ownership (van Rooyen, 1998). Reed et al. (2008)

[2] SLM is defined as the use of land resources, including soils, water, animals and plants, for the production of goods to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions (WOCAT, 2010).

[3] Defined as "a directional increase in the cover of indigenous woody species in savanna" (O'Connor et al., 2014).

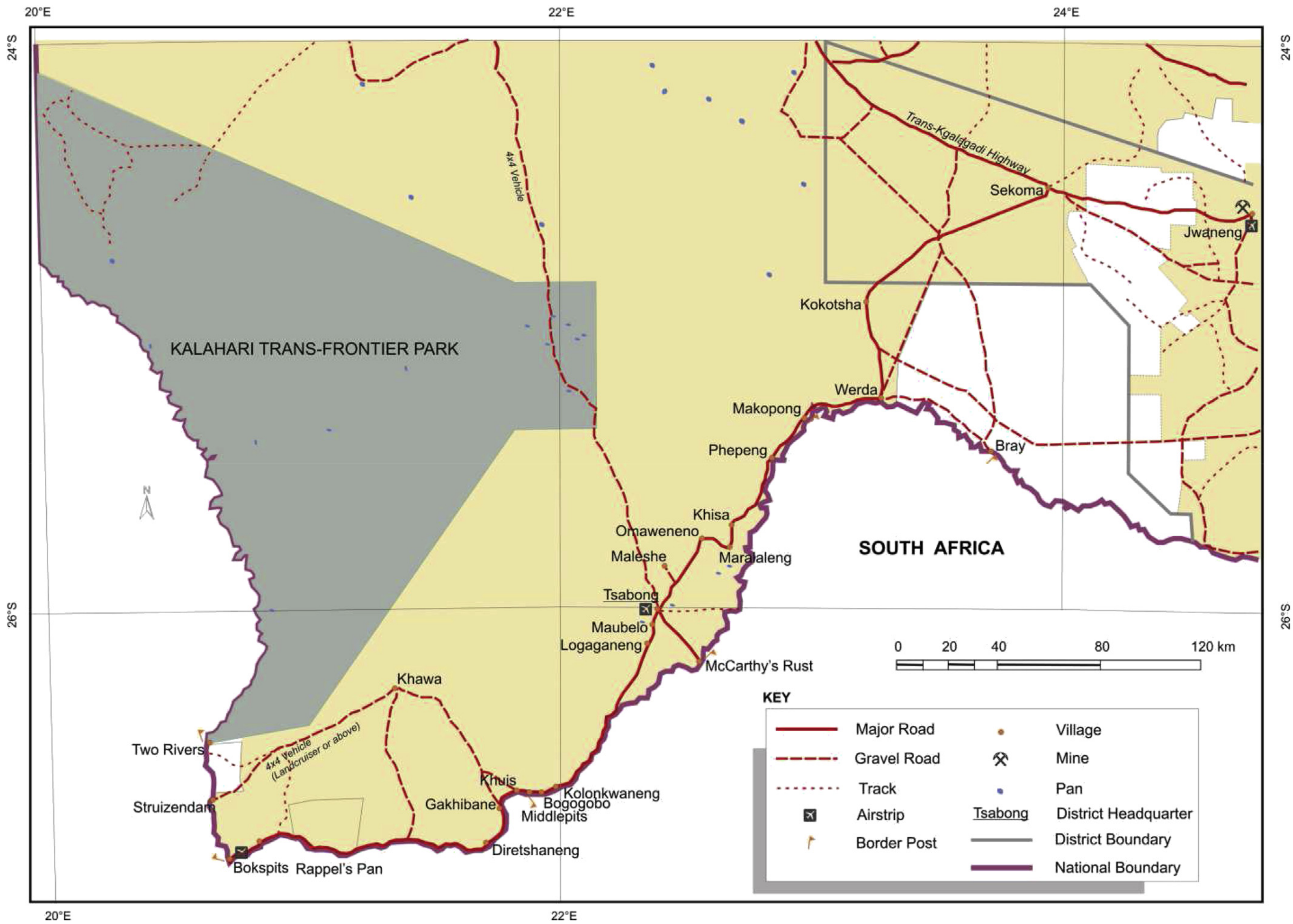

Fig. 1. Map of southwest Botswana (*map drawn courtesy of Mr G. Koorutwe*).

conducted participatory mapping with pastoralists (corroborated with ecological sampling) around Bokspits and Struizendam (Fig. 1) and found that degradation extended just 3–4 km around settlements, and 1–2 km elsewhere along the Molopo riverbed (where most boreholes are located). Within 6–8 km of water, depending on proximity to villages, vegetation was dominated by perennial, palatable grasses. Evidence collected by Reed (2005)[4] suggested that the majority of livestock owners were expanding their herds, despite regular droughts. However, Sallu et al. (2009) noted that within these trends it was possible to see significant discrepancies in livestock ownership and access to rangeland resources between rural elites and the poorest in society. Indeed, elsewhere Reed and Dougill (2002) interviewed pastoralists between Tshabong and Omaweneno (Fig. 1), and found that many of them had noticed a steady decline in the condition of their rangeland since the Government improved access to ground water in the 1970s. More than half of those interviewed said their livelihoods were constrained by the condition of the rangeland, mainly due to thorny bush encroachment.

Different approaches to assessing land degradation at different scales can yield different results, making comparisons difficult. This complicates matters for policymakers and communities who are keen to shift away from land degradation towards more sustainable land management. Local communities rarely have tools that they can easily and cost-effectively use to identify the early signs of land degradation, in order to adapt their management. Reed and Dougill (2010) developed rangeland assessment guides that attempted to provide communities with land degradation indicators based on a combination of local and scientific knowledge, linked to management options designed to fit the needs and resources of local people. The approach was designed to include biophysical as well as socio-economic indicators, to provide as holistic a picture as possible of rangeland health. These guides would benefit from further mainstreaming into extension service advice across the Kalahari, and on-going discussions and trial over longer timeframes are required to evaluate their effectiveness.

Countries affected by desertification who are party to the world's major agreement on land degradation, the United Nations Convention to Combat Desertification (UNCCD), must develop National Action Programmes (NAP) that: a) outline the major causes of degradation; and b) propose strategies to address national challenges of desertification and drought (Stringer et al., 2007a, b). Botswana's NAP identifies a number of priority areas for preventing and remediating land degradation (GOB, 2006: 4): (i) poverty alleviation and community empowerment; (ii) partnership and capacity building amongst stakeholders and researchers; (iii) sustainable natural resource management; and (iv) developing mechanisms to fund and resource these activities. The NAP

[4] Graphs showing trends in livestock populations drawn by pastoralists during semi-structured interviews, corroborated by sub-regional livestock census figures provided by Ministry of Agriculture.

Table 1
Comparison of different areas of southwest Botswana, showing the considerable ecological and socio-economic heterogeneity of this part of the Kalahari.

	Tshabong area	Bokspits area	Kgalagadi Transfrontier Park & WMAs
Land tenure	Private ranches and communal rangeland	- Private ranches, many created under the Tribal Grazing Land Policy in the 1970s and 80s - Owners often retain communal grazing rights, keeping fenced land in reserve for drought (Perkins, 1996)	- Aims to conserve wildlife, as a joint effort with stakeholders (local, regional, international) for the benefit of present/future generations - The Park is fenced land, for exclusive use by wildlife, and falls under state control - Wildlife Management Areas (WMAs) occur at the edge of the Park, for communities exclusive use rights under hunting quotas from wildlife department
Climate	Average annual rainfall 315 mm during summer (October–April) (Meteorological Services data) with inter-annual variability of around 45% (Bhalotra, 1987)	Driest part of Botswana, receiving on average 150–200 mm rainfall per year (Thomas and Leason, 2005), with inter-annual variability exceeding 50%	As per Bokspits
Vegetation	- Southern Kalahari bush savanna with perennial tufted grasses and sparse woody vegetation - Due to borehole-driven expansion of the livestock sector in this District, there is now an increased abundance of annual grasses (especially *S. kalahariensis*) and thorn bushes, in particular *A. mellifera*.	- Arid bush savanna with perennial tufted grasses and sparse woody vegetation	As per Bokspits
Geomorphology and soils	Kalahari sands (Arenosols) have extremely poor fertility (Skarpe and Bergstrom, 1986), contain low amounts of organic matter (typically < 0.01%) and have low water-holding capacity (Dougill et al., 1998).	Landscape is dominated by fossil linear dunes composed of deep Kalahari sands, around 5–25 m high, with areas of unvegetated, active dunes around settlements.	- Plains with vegetation-stabilized relict dunes (linear) - Fossil valleys used for salt-licks, drinking points and grazing - Calcrete rimmed fossil valleys
Population characteristics	- Predominantly Batswana inhabit the area. They first arrived in the area at the beginning of the 19th century - A number of other much smaller ethnic groups including Eurasian whites and Basarwa (also known as San or "Bushmen") also inhabit the area	- The vast majority of people living in the study area are mixed race or coloured. They migrated from South Africa in the first two decades of the 20th century - The population of the area is around 1600 (IVP, 2003)	- Transfrontier Park is used by a range of groups Khoikhoi, San and coloureds, who have adopted livelihoods based on hunting and gathering
Livestock	- Livestock are the main source of livelihoods in the area, and cattle ownership is an important status symbol, particularly for the dominant Batswana group.	- Livestock production is the main income source in the area with small-stock production also vitally important. - Karakul sheep (bred for their lambs' pelts) were an important source of income until the market declined in the 1980s.	- Wildlife only, hosting large mammals including large cats (e.g. lions, cheetahs, leopards), ungulates (e.g. gemsbok, hartebeest, elands, wildebeests, kudu, springboks, duiker, impala), and birds (e.g. ostrich). Declines in populations of wildebeest, hartebeest and eland have been reported (Arntzen, 2001).

identifies three types of degradation, which it wants to address: "bare soil degraded areas"; "partially degraded areas" and "bush encroached areas" (Stringer et al., 2009a,b).

To date, policies and projects focussing on land degradation and SLM have mainly been based on biophysical assessments of degradation, predominantly at the farm or ranch scale. For example Botswana's 1991 Agricultural Policy aims to increase production in the livestock sector via the privatization and fencing of land, based on livestock carrying capacities. This evidence was based upon conventional views of range degradation (Abel and Blaikie, 1989), Clementsian succession (Clements, 1916; Westoby et al., 1989) and experimental ranches (Rennie et al., 1977), with traditional cattle posts only marginally less productive than commercial ranches on an output per beast basis (Hubbard, 1982), and twice as productive as ranches on an output per hectare basis (de Ridder and Wagenaar, 1984). Subsequently, there have been claims that this policy may be further worsening land degradation and deepening already pronounced social and economic inequalities, by compromising the possibility of the multiple uses of rangeland (i.e domestic livestock and wildlife) (e.g. Perkins, 1996; Perkins and Ringrose, 1996; Thomas et al., 2000; Adams et al., 2002). According to Arntzen (2001), several challenges persist, some of which include dual grazing rights (where private ranch owners graze communal land until there is insufficient forage, before moving their herd to their fenced grazing reserves), and challenges of sustaining displaced communities within a water scarce environment. Several analysts have observed a degree of political lock-in to ideas supporting the fencing and privatisation of rangeland areas. They argue that this is having an important impact on land degradation and the costs it brings due to the constraints it places on mobility (Perkins, 1996; Adams et al., 2002; Malope and Batisani, 2008; Perkins et al., 2013) and rangeland management decision-making (Reed and Dougill, 2010).

The Government of Botswana has pursued several projects to meet its NAP commitments. For example, the Ministry of Agriculture has initiated experimental projects to remove thorny bushes, and fencing projects that exclude livestock to stabilise sand dunes. However, these initiatives have typically been short-lived, with fences not maintained after funding runs out, allowing livestock to graze at intensities high enough to remobilize dunes and re-initiate bush encroachment. The Government has also provided livestock to many of the rural poor, alongside loans for others to purchase livestock through its Citizen Entrepreneurial Development Agency. Similarly, the Government's Livestock Management and Infrastructure Development programme covers projects on livestock water development, animal husbandry, poultry, chicken and small stock. This programme has encouraged borehole drilling or reticulation of water to keep livestock closer to the Transfrontier Kgalagadi Park and its WMAs. However, it is now possible to observe degradation around boreholes in these locations (bare ground near boreholes, and bush encroachment at intermediate distances from water) (Sallu et al., 2009, 2010), and pastoralists have increasingly had to be compensated for losses of livestock to predators. Livestock encroachment into the Transfrontier Kgalagadi Park from adjacent boreholes led to the erection of the so-called ‘lion-proof fence’, which runs for 100 km along the south-eastern Transfrontier Kgalagadi Park boundary. The fence is not effective in keeping predators inside the Park and also disrupts the movement of wild ungulates, which can experience pronounced die-offs when they move against it (Perkins, pers obs). In contrast to these initiatives, an alternative narrative to reduce land degradation based on sustaining livelihoods in common property land tenure regimes led to the initiation of the Indigenous Vegetation Project,[5] which attempted to provide an alternative to privatisation based on Community-Based Natural Resource Management for rangelands. However, these approaches have not yet been mainstreamed (cf. Akhtar-Schuster et al., 2011) despite successes noted in Namibia with their Community Conservancy programme (Barnes et al., 2002: Hoole and Berkes, 2010).

In the next section, we consider bush encroachment as one of the key forms of land degradation being tackled in Botswana's NAP that is compromising the livelihoods of many communities in southwest Botswana. The goal is to explore a more integrated approach to tackling land degradation and incentivising SLM by focussing on the effects and opportunities of bush encroachment for both livelihoods and ecosystem services.

2.2. Bush encroachment in Botswana: process and potential solutions

For predominantly cattle-based systems in the Kalahari, bush encroachment is viewed as a major form of land degradation by many communities, because of its direct impacts on livelihoods (Sporton and Thomas, 2002; Chanda et al., 2003; Dougill et al., 2010). It is anticipated that the impacts of bush encroachment may continue to intensify as increasing atmospheric CO_2 concentrations exacerbate local factors (such as changes in grazing or burning regimes) to drive further bush encroachment in future (Higgins and Scheiter, 2012; Moncrieff et al., 2014; O'Connor et al., 2014). Apart from the loss of wildlife and grazing resources, natural capital remains intact in terms of the soil resource and total biomass. However, there are a number of effects on the types of flows of ecosystem services from bush encroached systems.

Traditionally, the Kalahari was a wildlife-dominated system, including both browsers and grazers at low densities, with hunting and gathering activities occurring throughout. These activities were critical in terms of the sustenance they provided to local communities in times of drought. Borehole based cattle keeping changed this, and wildlife declines due to the establishment of veterinary cordon fences in the early 1980s led to the loss of half a million wildebeest and hartebeest. Selective grazing by cattle-dominated herds, combined with changes in the frequency and intensity of wildfires,[6] perhaps aided by the effects of CO_2 fertilisation, eventually led to a shift towards less palatable annual grasses and bush encroachment (Dougill et al., 1999; Bond and Midgley, 2000; Joubert et al., 2012a,b). Encroachment of unpalatable bushes into productive rangeland can become problematic when bushes dominate large enough areas of rangeland that they limit forage availability and reduce mobility for livestock and herders (e.g. Reed et al., 2008; Joubert et al., 2012a,b).

Much of the ecological evidence for land degradation in Botswana focuses on the extent (and potential reversibility) of bush encroachment (Behnke and Scoones, 1993; Illius and O'Connor, 1999; Thomas et al., 2001). There has nevertheless been some debate over the extent to which localised bush encroachment can be considered land degradation. *Limited* bush presence may enhance the resilience of Kalahari rangelands, providing drought forage for cattle from fallen pods and leaves, and protecting palatable grass seed sources which can facilitate rapid recovery of rangeland after drought (Perkins and Thomas, 1993; Dougill et al., 1999). Soil hydrochemical research has suggested that bush encroachment could be reversible (Dougill et al., 1998), though other analyses have countered this supposition (Berkeley et al., 2005), indicating that without intervention, enhanced N availability under *Acacia mellifera* canopies can lead to more rapid rates of bush encroachment (Hagos and Smit, 2005). Ecological modelling simulations suggest bush encroachment is only reversible over relatively short periods with mechanical or chemical removal, grass re-seeding and sufficient rain (Joubert et al., 2013). This is exacerbated by the potential for atmospheric CO_2 enrichment to favour C_3 bush species ahead of C_4 grass species (Bond and Midgley, 2000; Wigley et al., 2010). Bush encroachment therefore remains “effectively permanent” for land users who lack sufficient resources to remove bushes and exclude grazing to allow recovery (cf. Abel and Blaikie, 1989).

Given the processes that lead to bush encroachment (grazing intensity by cattle, reduced wildfire intensity and CO_2 fertilisation), approaches to managing bush encroached systems typically focus on reducing grazing intensity whilst removing bushes (using fire, herbicide or mechanical cutting/uprooting). An alternative approach replaces grazing by cattle with browsing by goats, but this adaptation is usually combined with some level of bush control. Table 2 summarises the key types of bush control and adaptation.

Mechanical and chemical control methods such as cutting, uprooting, and herbicide applications are most effective but expensive (Burgess, 2003), and rarely provide a return on investment within an adequate time-frame (Buss and Nuppenau, 2003). They may even give negative returns on investment (Quan et al., 1994) and require considerable expertise and equipment. Cutting and burning are cheaper, require less expertise and equipment, but are less effective unless regularly repeated. Browsing is only effective in combination with other techniques. Consequently, some pastoralists opt to adapt to bush encroachment by shifting from cattle to small-stock production, particularly goats, in order to utilise bushes as a browse resource. As an alternative adaptive strategy, some encroacher species such as *C. mopane* and *A. mellifera* have been shown to be appropriate for charcoal production (Cunningham, 1998). However, Quan et al. (1994) warn that income generation from charcoal production may be constrained by a lack of charcoal markets, and that the sandy soils of the Kalahari are not well suited to traditional charcoal production techniques (Tabor, 1994).

In some areas, game farming may be an adaptation option to bush encroachment, as game need less water per head than cattle, and, providing they are not kept at high densities, are less likely to damage rangeland vegetation and cause further bush encroachment (Cooke, 1985). If the majority of game species kept are browsers, this makes game farming suitable in bush encroached areas (although these are less well suited to game viewing). It may nevertheless be possible to supplement game farming for meat with photographic tourism and the sale of hunting licences. For these reasons, it is seen by some Botswana Government sources (e.g. DHV, 1980) as a suitable way to enhance the livelihoods of some of the poorest people in the Kalahari.

[5] http://www.mewt.gov.bw/DFRR/article.php?id_mnu=133.

[6] Due to a ban on managed burning, which increased fuel-loads in all but bush encroached areas.

Table 2
Methods of bush encroachment control.

Method	Advantages	Disadvantages	Considerations
Cutting (mechanical control)	Cheap; fast;	Rarely effective: most bushes resprout vigorously after cutting unless repeated cutting is performed (Tainton, 1999; Smit et al., 1999)	Pastoralists suggest this technique could be more effective if: stem-cutting is followed up with intense small-stock browsing; cut stems are treated with herbicide; or the ground is hollowed out and stems cut underground (Reed, 2005)
Uprooting (mechanical control)	Effective	Highly labour intensive unless mechanised;	Soil disturbance may favour re-establishment of less palatable annual grasses (Smit et al., 1999).
Prescribed fires	Cheaper than chemical and mechanical control; stem-burning of individual bushes when in leaf is potentially more effective in heavily encroached systems (Smit et al., 1999; Reed, 2005)	High opportunity costs due to lost grazing (Trollope, 1992; Buss and Nuppenau, 2003); stem-burning is labour intensive	Bush mortality is typically 1–10%, so other methods are necessary to prevent re-growth or coppicing of bushes after fire (such as browsing) (Trollope, 1974). Browsing effectively prevents establishment of bush seedlings and bush regrowth but in communal systems in Botswana it may be difficult to achieve sufficient browsing intensity without fences or careful shepherding (Reed, 2005)
Herbicide applications (chemical control)	Effective - can suppress bush seedling growth for up to 4 or 5 years (Reed, 2005)	Expensive; typically only used by ranchers; herbicides that can be applied to the soil (rather than to the plant itself) are usually least capital and labour intensive	Does not work for all bush species (Smit et al., 1999)

DHV (1980) recognized that the mobility of wild ungulates in response to highly spatially and temporally variable rainfall, fires and wild melons (that provide animals with water) was the key to their conservation. They also emphasized that enhanced game use was the best way to improve the livelihoods of the largest number of people in the Kalahari, who were then, and still are, amongst the poorest in the whole country. However, DHV's (1980) vision has been substituted by domestic stock, namely cattle and goats, which have fragmented the rangeland resource, compromised mobility and further contributed towards degradation.

Some Molopo and southern Kalahari cattle ranches have converted to game and built up numbers through the provision of artificial water points. This allows trophy hunting, as well as photographic tourism. However, many game farms suffer from the same problem as cattle ranches (see for example Grossman and Holden, 2005), with fenced private game ranches offering considerable benefits to a few, typically wealthy individuals, but not much prospect of livelihood improvement for local communities who rely on communal areas.

Nature conservancies and game ranches managed by community groups have been profitable elsewhere in the Kalahari (van Rooyen, 1998; Jones and Murphree, 2001; Jones, 2003). However, there have also been instances where such programmes provided conservation benefits but failed deliver the socio-economic gains promised to local communities (Taylor, 2003; Dougill et al., 2012) or to private game ranch enterprises (Grossman and Holden, 2005). In particular, such schemes may not be economically viable in remote areas with poor infrastructure that are rarely visited by tourists. In these areas, livestock development subsidies remain a more attractive option (Jones, 2003).

2.3. Ecosystem services from bush encroached systems

Removing bushes to combat bush encroachment is typically an attempt to secure continued provision of key ecosystem services (principally provision of grazing resources for cattle production) (Archer et al., 1995). However, by adapting to bush encroachment, pastoralists typically use different ecosystem services to support their livelihoods (e.g. energy via charcoal production) in a dynamic manner (Allsopp, 2013; Vetter, 2013). Table 3 summarises the ecosystem service benefits and costs of bush encroachment, based on an analysis of the literature. It shows how these differ between the three most common land tenure types found in the Kalahari. The table demonstrates that bush encroachment has negative effects across all types of ecosystem service, not just provisioning services such as cattle production. However it also shows that there are a number of ecosystem service benefits from bush encroachment, which may in some cases offset some of the negative impacts. Notably, bush encroachment compromises cattle production, which may be important culturally, but offers forage for goat production and camels, and may provide other provisioning services via resins, fuelwood, charcoal and materials for fencing and livestock corrals. It is important to recognise such trade-offs when considering bush removal or management. For example, Shackelton and Gambiza (2008) argued that a Payment for Ecosystem Service scheme based on removal of the shrub *Euryops floribundus* from communal areas in South Africa to improve livestock production, underestimated the amount of use local people made of the species for fuel and timber and also demonstrated higher plant species richness in invaded compared to non-invaded areas. Such trade-offs will vary from species to species however. For example bush species that commonly invade Kalahari rangelands e.g. *Senegalia* (formerly *Acacia*) *mellifera* timber is valued for its termite resistant qualities and can be used as fuelwood (Smith et al., 1996), however *Grewia flava* has limited use as timber or fuelwood but is important culturally for its berries, which are consumed directly and used in local beer-making (Mainah, 2001, 2006).

It is important to note that predicted increases in atmospheric CO_2 are expected to increase the yield of C3 plants, such as woody encroachers like *Senegalia*, *Vachellia* and *Senegalia* spp. (up to 20–35% under a doubling of CO_2 concentrations), more than C4 grasses (which are anticipated to show a 10% increase in yield under the same scenario) (Wolfe and Erickson, 1993; Midgley et al., 1999; Bond, 2008; Bond and Midgley, 2012). Indeed, there is evidence that increasing atmospheric CO_2 may already account for some bush encroachment in southern Africa (Wigley et al., 2010; Buitenwerf et al., 2012; Russell and Ward, 2014; Ward et al., 2014). In addition, *Senegalia*, *Vachellia* and *Senegalia* species are likely to have more carbon available to invest in the production of tannins, better protecting them against grazing (Rohner and Ward, 1997; Ward and

Table 3
Likely effects of bush encroachment on ecosystem services in three land tenure types in Botswana.

Ecosystem service	Communal	Private	Wildlife management area
Provisioning (grazing resources for livestock or wildlife)	Potentially significant reductions in cattle production may be offset by increased goat and sheep production (Reed et al., 2007). Bushes may be used as an energy source or for charcoal production (Cunningham, 1998; Smit, 2004; Reed et al., 2007). Resins can be eaten and have medicinal uses, and resins from some species may have an economic value (Reed, 2005; Abdeta, 2011; Worku et al., 2011). Wood from bushes may provide materials for fencing and livestock corrals (Ellis et al., 1984; Jensen, 1984)	Grazing intensities are typically lower and cattle production may be maintained using herbicide or mechanical operations in large-scale ranches, so less impact of bush encroachment on cattle production (Archer and Smeins, 1991; Reed et al., 2007; McGranahan, 2008). Focus on cattle production means energy, food and medicinal uses more likely to be overlooked	These areas are not used intensively for livestock production, so few impacts of bush encroachment on provisioning services
Supporting (seed dispersal and nutrient cycling)	Protection of perennial grasses from grazing under juvenile bushes may provide a seed source from which surrounding rangeland can be recolonized in future (Dougill et al., 1999). Some evidence of higher concentrations of soil nutrients under bushes due to formation of nebkha dunes around base of bushes and formation of biological soil crusts (Dougill and Thomas, 2002; Dougill and Thomas, 2004; Thomas and Dougill, 2007). Reduced ground cover under bushes may increase soil erosion rates (Smit, 2004), however there is evidence that bush encroachment can reduce erosion, particularly on hillslopes, and can lead to the retreat of soil gullies where this increases vegetation cover (Grellier et al., 2012; Caviezel et al., 2014).	Same as for communal areas, though effects likely to be less pronounced assuming less encroachment or more bush removal	Bushes unlikely to reach densities that would have major impacts on seed dispersal or nutrient cycling in WMAs
Regulating (climate regulation)	Bush encroached areas have greater biomass (including long tap roots) than grass-dominated areas, so they sequester and store more carbon from the atmosphere in their vegetation (Wiegland et al., 2005; Ward, 2005; O'Connor et al., 2014). Soil carbon sequestration and storage typically increases under bush encroachment but then declines if bush densities become so high as to inhibit understorey growth (Hudak et al., 2003) but this is may be offset by removal of trees for fuelwood in communal areas (Reed, 2005).	Assuming trees remain intact, bush encroached ranches will store more carbon than grass-dominated ranches. However this carbon is unlikely to meet additionality[a] criteria for carbon to be traded on the voluntary carbon market	Same as for private rangeland
Cultural (recreational activities)	Where bush encroachment prevents cattle production, this compromises cultural services because cattle are a powerful status symbol in Batswana culture (Reed, 2005). However, if game are introduced, bush encroached areas may be able to support wildlife whilst providing recreational benefits for tourists (Perkins et al., 2002; Reed et al., 2007). Bush encroachment has been shown to reduce plant species richness in a number of studies (Reed et al., 2008; Ratajczak et al., 2012), although increases in plant species richness have been associated with particular encroacher species (e.g. Shackelton and Gambiza, 2008; Belay et al., 2013). There is evidence that decreased diversity of habitat structure in bush encroached systems reduces the diversity of lizard species in Namibia (Meik et al., 2002)	Same as for communal areas	Same as for communal areas

[a] Additionality criteria assess whether changes in management that enhance carbon sequestration and storage would have happened in the absence of carbon finance.

Young, 2002). These two mechanisms may increase the likelihood of bush encroachment in future. However, predicted decreases in rainfall due to climate change in this part of Africa are likely to reduce rates of bush encroachment, as woody species typically establish during wetter years (or after a succession of rainfall events), and are less likely to be able to compete with grasses and become established in drier years (Ward, 2005; Wiegand et al., 2005; Kraaij and Ward, 2006). This means it is difficult to predict how climate change is likely to affect bush encroachment dynamics. It is also difficult to determine the likely relative role of climate change versus other more local drivers of bush encroachment in future. However, no matter how important atmospheric

CO_2 becomes as a driver, it is likely that a range of management actions, notably linked to grazing levels and fire, will be able to moderate climate-induced bush encroachment (O'Connor et al., 2014).

By systematically considering the effects of bush encroachment on each type of ecosystem services (Table 3), it may be possible to devise strategies for tackling this form of degradation based on optimising the provision of ecosystem services, which in turn, sustain a range of local livelihoods. To do this, Table 4 considers how two different scenarios for tackling bush encroachment may improve ecosystem service provision, thereby benefiting local communities. In the first 'basic restoration' scenario, all bushes are simply removed from the system (whether by cutting, uprooting, burning or herbicide – see Section 2.2). In the second 'ecosystem service optimisation' scenario, a package of measures is designed to optimise as many ecosystem services as possible whilst reducing bush cover, increasing overall resilience of the rangeland system and the livelihoods it supports.

Table 4 summarises some of the ecosystem service costs and benefits that may arise from the 'basic restoration' scenario, where bushes are removed completely. Removal is particularly important to reduce the clonal spread of bushes from mature individuals, as this enables new clonal bushes to access deeper soil water than would normally be possible for bush seedlings, increasing their competitive advantage over grasses (Ratajczak et al., 2011). However, although bush clearance should increase the productivity of rangeland for cattle (assuming palatable grass cover can be effectively re-established) (de Ridder and Breman, 1993; Quan et al. 1994; Dean and MacDonald, 1994), complete removal of bushes would reduce browse availability for goats and fuelwood, timber and medicines from bushes (Abdeta, 2011; Worku et al., 2011; O'Connor et al., 2014). The significance of these trade-offs will depend on the importance of goat production to local land users, and whether alternative, better sources of timber, fuelwood and medicines are available. Reed and Dougill (2002) found that bush encroacher species in Botswana's Kalahari were not being used for these purposes by the majority of those they interviewed, who instead mostly viewed bush encroachment as a constraint to their livelihoods. Although bush clearance can increase the economic productivity of rangeland for cattle, grazing exclusion in currently unfenced rangeland is expensive. Furthermore, bush encroachment may either increase or reduce soil erosion rates, depending on the steepness of slope, and whether encroachment has proceeded to the point where there is very little vegetation cover under the bush canopy or whether it represents a net increase in vegetation cover (e.g. replacing over-grazed rangeland with significant amounts of bare ground) (Smit, 2004; Grellier et al., 2012; Caviezel et al., 2014).

For this reason, in the second, 'ecosystem service optimisation' scenario (Table 4), when clearing bushes from rangeland, Kalahari pastoralists have suggested leaving strips of bush cover, arranged along contours or against prevailing winds (Reed et al., 2007). Leaving bushes in this way would continue to provide timber, fuelwood and medicines to the small proportion of the population who still rely on bushes for these products, and where palatable perennial grasses are protected under juvenile bushes, these strips may provide a source of seeds to recolonize strips of rangeland where bushes have been cleared (c.f. Dougill et al., 1999). This better allows for multiple rangeland uses. Based on local knowledge from Kalahari pastoralists, Reed et al. (2007) suggest that removed bushes may be broken up and branches laid on the surface of the ground to reduce wind erosion, retain nutrients on the site and deter grazing in recovering grass areas. Furthermore, deep-rooted trees could be planted at wide spacing to create a silvopastoral system. Kalahari pastoralists have emphasised the importance of maintaining multi-purpose species for enhancing resilience to drought (Reed et al., 2007). Particular reference was made to *Boscia albitrunca*, known as 'Shepherd's Tree' because it is evergreen and palatable, providing a valuable source of forage for livestock during drought (Reed et al., 2007). For this reason, there is a local taboo preventing *B. albitrunca* being cut for timber or fuelwood, which has led to a relative increase in the abundance of this species compared to other trees within the vicinity of settlements (Reed et al., 2008). For these reasons, the hypothetical 'ecosystem service optimisation' scenario in Table 4 considers cultivation of *B. albitrunca* as part of a silvopastoralism system after selective bush clearance, to provide shade and forage for livestock during dry seasons and drought. *B. albitrunca* is slow-growing, but by allowing bushes to persist in strips, it may still be possible to benefit from shade from quickly maturing bushes in the intervening period. Although this scenario has not been tried in practice, and many Kalahari trees are difficult to cultivate, there is now evidence that cultivation of trees with mycorrhizal fungi (alone or in combination with organic amendments) and tree shelters is effective for restoring degraded drylands across the world (Pinero et al., 2013). Although the multi-functionality of other tree species may not be as strong as for *B. albitrunca*, other trees could be used in this scenario, such as acacias. Deep-rooting trees such as these also have significant carbon storage potential, and it may be possible to trade this on international voluntary carbon markets. Given the other ecosystem service benefits associated with such a mixed-landscape scenario, it may be possible to "bundle" the carbon storage benefits with the benefits of a silvopastoral system for biodiversity and local people, so that it becomes possible to charge more for the carbon (e.g. Pagiola et al., 2007; Ibrahim et al., 2010; Root-Bernstein and Jaksic, 2013). In this way, it may be possible to pay for bush removal and the establishment of a silvopastoral system, regaining a productive system that can support livestock whilst potentially being eligible to receive ongoing payments for the climate change mitigation and other co-benefits of SLM.

The second scenario attempts to operationalise a shift away from land degradation towards SLM, by focussing on optimising the provision of a range of ecosystem services to support livelihoods. The package of measures remains hypothetical, but illustrates how the development of new integrated policy interventions may be able to tackle land degradation and facilitate SLM. Policy development needs to consider the context in which such interventions may be applied, in particular the land tenure arrangements. In the Kalahari, most land is still managed communally, although privatisation of rangelands is increasing. Communal systems do not typically involve fencing, enabling livestock to forage over wide distances during drought, and permitting landscape-scale movement to track rainfall and thus good grasses via the local '*mafisa*' livestock movement system.[7] Given the importance of livestock movement to sustain livestock through drought, and increasingly in future, provide resilience to climate change, it is important to consider approaches to SLM that are compatible with current tenure systems. For example, the second 'ecosystem service optimisation' scenario could work effectively with 'borehole syndicates' (groups of pastoralists with communal rights to borehole water and surrounding grazing), with Kalahari pastoralists suggesting 'borehole resting' to allow rangeland to recover after (selective) bush clearance (Reed et al., 2007). The mafisa system is a traditional

[7] Using the mafisa system, livestock are moved (sometimes substantial) distances to friends or family (or in some cases San or minority groups who do not own cattle) living in locations not affected by drought, who look after them in return for using their milk (and meat if animals die), sometimes in return for a payment in calves (Hastrup and Hervik, 2003).

Table 4
Two alternative scenarios for enhancing the provision of ecosystem services from currently bush encroached land in three land tenure types in Botswana, based on simply removing bushes ("basic restoration") versus introducing a package of measures aimed at optimising ecosystem services (described in Section 2.3).

Ecosystem service	Communal		Private		Wildlife management area	
	Basic restoration	*Optimising ecosystem services*	*Basic restoration*	*Optimising ecosystem services*	*Basic restoration*	*Optimising ecosystem services*
Provisioning (grazing resources for livestock or wildlife)	Increased cattle production, but decreased forage for browser and provision of fuelwood, timber and medicines from bushes	Decrease of cattle production in the short term to better match grazing resources availability. A more significant increase in cattle production over the long-term due better re-establishment of grasses (due to protection from grazing and erosion and retention of nutrients from decomposing bush biomass)	Same as for communal areas	Same as for communal areas, though less benefit from laying bushes on the soil as grazing can be controlled by paddock system	Little benefit for livestock as this area is not used significantly for livestock production, but improvements in forage resources for wildlife	*B. albitrunca* forage could be harvested during drought to further increase resilience of herds, but protection from wildfire would be necessary during their the establishment phase in the WMA
Supporting (seed dispersal and nutrient cycling)	Possible loss of soil nutrients, exported in bush biomass removed from the site and due to increase in erosion losses and loss of biological soil crusts due to increased disturbance by livestock.	Retention of soil nutrients by retaining bush biomass on site, controlling erosion and limiting disturbance by livestock during initial years after bush removal	Same as for communal areas, but more control over disturbance by livestock due to fencing. Greater risk of wildfire if under-grazed due to increased fuel-load in grass-dominated sites post-restoration	Same as for communal areas. May be possible to create fire-breaks by leaving wide strips of bush encroached land between paddocks	Where bush encroachment has occurred in localised areas, basic restoration would have similar effects to communal areas, but there would be less disturbance by livestock	Same as for communal areas
Regulating (climate regulation)	Reduced carbon storage	Potential reduced loss of carbon and net increase in carbon storage due to planting deep-rooting trees, with carbon credits possibly sold on international carbon markets	Same as for communal areas	Same as for communal areas	Same as for communal areas	Same as for communal areas
Cultural	Cultural benefit for groups who value cattle ownership as a status symbol, assuming restoration leads to increased cattle production (at the expense of those who value browsers). Improved biodiversity over time, providing traditional rangeland products e.g. vegetables and medicines. Few recreational benefits.	Same as previous column, but with added benefit of increasing the abundance of the trees which may have cultural significance (e.g. the sacred tree *B. albitrunca*).	Same as for communal areas	Same as for communal areas	Same as for communal areas, but less benefits linked to cattle as this area is not used much by cattle	Same as for communal areas

practice that may be in line with the type of cattle-keeping sharing required by such an approach. In this scheme, a number of syndicates would agree to pool livestock across a number of boreholes, leaving one borehole without livestock until the rangeland is recovered, before allowing the rangeland around another borehole in the scheme to rest. As such, it is as important to consider ways of incentivising more sustainable forms of livestock management, as it is to devise ways of reducing bush encroachment, to avoid bush cover expanding again in future, and to prevent encroachment in currently grass-dominated rangelands. Solutions need to address both the drivers and symptoms of bush encroachment. There is a range of policy instruments that may facilitate SLM, ranging from state land ownership and regulatory mechanisms (e.g. enforced stocking rates or exclusion zones) to more incentive-based approaches, including financial instruments (e.g. subsidies or tax breaks) and the creation of new markets (e.g. Payments for Ecosystem Services). Given the focus of this paper on the ecosystem services of rangeland management, the next section considers these instruments, with a focus on Payments for Ecosystem Services.

3. Discussion: reorienting land degradation towards sustainable land management in the Kalahari rangelands via economic mechanisms

Part of the challenge of effectively assessing and responding to land degradation is the multi-faceted nature of the problem. Land degradation affects many different biophysical systems and subsystems, whilst having a range of effects on the livelihoods of those who depend upon rangeland resources. Tackling land degradation therefore requires the integration of definitions and measurements based on both the biophysical and socio-economic components of the systems affected. Integrated conceptualisations of land degradation need to be able to account for changes in each of these system components, and for changes in both stocks of (natural) capital and flows of (ecosystem) services arising from changes in land management. There is a need to better capture the economic benefits of sustainably managed land, and the costs of land degradation, so that decisions about land use and management benefit (and avoid costs to) wider society, as well as those who own and manage land (ELD Initiative, 2013).

Tackling land degradation may be as much about incentivising SLM and restoration (or providing alternative livelihoods), as it is about designing new land management regimes or developing and enforcing new regulatory regimes (Global Mechanism, 2012; ELD Initiative, 2013). For example, the tax system may be used to reward or discourage activities that may affect the sustainability of land management (e.g. increasing the price of agricultural inputs that are not deemed to be sustainable, while making more sustainable alternatives more attractive to land managers). Incentives that currently encourage over-grazing by cattle may be redirected towards public grant aid for activities that reverse degradation and encourage SLM (e.g. schemes to encourage small stock ownership, or to clear bush encroached areas and reduce grazing by cattle).

Alternatively, it may be possible to create or harness existing markets (e.g. REDD+) that can pay for land management activities that provide certain ecosystem services. This is the basis for many publically funded agri-environment schemes around the world, and is increasingly facilitating private investment in SLM. Linked to this, eco-labelling and certification schemes (e.g. FairTrade, Forest Stewardship Council) can help pay indirectly for the provision of ecosystem services by exploiting premium niche markets for products and services arising from SLM (Global Mechanisms, 2012). Through mechanisms such as these, new markets are being created around the world to pay for the provision of clean water, biodiversity and carbon storage for climate change mitigation. Although doubts have been expressed about the capacity for such schemes to contribute towards poverty alleviation (e.g. Zbinden and Lee (2005) and Porras (2010) commenting on Costa Rica's national payment for ecosystem service programme; see Fisher et al. (2013) for a recent global review), there is evidence that such schemes can provide strong incentives for SLM (de Koning et al., 2007). With careful governance, it may be possible to provide benefits to the poorest in society alongside landowners (Dempsey and Robertson, 2012; Fisher, 2012; Adhikari and Boag, 2013; ELD Initiative, 2013).

Integrating both biophysical and socio-economic aspects of land degradation allows systematic consideration of the multi-faceted effects of land degradation on biophysical systems and livelihoods. It enables both stocks and flows (i.e. ecosystem services) of natural capital to be considered, and can capture perspectives from land users themselves. Rather than presenting contradictory evidence in parallel, it may be possible to define and measure land degradation as part of an integrated and interdependent socio-economic and ecological system. In this way, more holistic approaches can be derived to tackle the effects of land degradation on natural capital, ecosystem services and the livelihoods of those who depend upon the land.

There is no linear relationship between most ecosystem services and natural capital. It is typically possible to erode natural capital to a certain extent and retain the provision of ecosystem services, until a threshold is crossed, beyond which ecosystem service provision declines. For example, the salinity of water for dryland irrigation or livestock consumption can continue increasing to a certain threshold before it is unfit for use. Depending on the ecosystem and abiotic processes and structures involved, the decline of an ecosystem service or services may be more or less easy to reverse once this threshold has been crossed.

Similarly, dryland vegetation communities typically follow quite rapid transitions across thresholds to new states that may be more or less productive for livestock. In some cases, for example bush encroachment, there is little evidence for significant changes in other forms of natural capital during such transitions (e.g. Dougill et al. (1999) found no evidence for soil hydro-chemical changes (supporting service) along gradients from bush encroached to grass-dominated sites), but there can be a very significant effect on availability of grazing resources (provisioning service) if the land was used primarily for cattle production.

Where natural capital is retained, it may be possible for those using the system to adapt their use of natural capital to benefit from alternative ecosystem services, e.g. based on charcoal production or game ranching. In the case of bush encroachment, switching from cattle to small-stock production can theoretically sustain livelihoods, due to the ability of goats and sheep to access the nutrition in thorny bushes. In this example, the natural capital is still intact (in the soil and the nutrients held in the bush biomass), but by adapting to keeping small-stock, an alternative provisioning service is derived. However, in practice, cultural and economic barriers may prevent such an adaptation from occurring. For example, cattle (not goats or sheep) are a status symbol in Botswana culture and due to the export market, cattle production remains significantly more profitable than small-stock production (Harvey, 1992; Tsie, 1996; Acemoglu et al., 2003). The importance of cultural and economic barriers should therefore not be underestimated if more sustainable land management options are to be adopted. It may also be necessary to remove perverse incentives that increase cattle populations via Government subsidies and loans (see Section 2.1), so that incentives can be more tightly coupled to the provision of ecosystem services, and so reward sustainable land management.

It is clear from the case study outlined in the previous section that any conceptualization of land degradation cannot divorce biophysical and socio-economic approaches, given the interdependency of socio-economic and biophysical systems for the livelihoods of people living in the Kalahari rangelands. Viewing land degradation as the loss of sustainable livelihoods resulting from an effectively permanent reduction in the provision of ecosystem services from land (including any of provisioning, supporting, regulating or cultural services), due to the undervaluing and consequent loss of natural capital beyond critical thresholds may therefore be useful. The converse of this is to consider SLM as practices that sustain livelihoods through the continued provision of ecosystem services from land, based upon natural capital that has been sufficiently valued to be maintained above critical thresholds.

Re-orienting the relationship between land degradation and SLM means it becomes possible to identify ways of tackling land degradation that focus on protecting as much natural capital as is necessary to prevent critical thresholds from being crossed (i.e. accepting the loss of some natural capital where necessary), and developing adaptations that can use natural capital in different ways to harness new ecosystem services that can sustain livelihoods (e.g. the hypothetical example described in Section 2.3 and Table 4 where bush encroached systems are replaced with silvopastoral systems). An economic value could theoretically be placed on each of the costs and benefits of bush encroachment (Table 3) and restoration (Table 4), aggregating individually expressed values in monetary terms, to compare each of the scenarios presented in Table 4 (e.g. Mersmann et al., 2010; Noel and Soussan, 2010; Global Mechanism, 2012). However, in practice, these approaches are often unable to fully capture collective meanings and significance ascribed to dryland environments, and may miss important, shared dimensions of value. As such, traditional economic analyses can fail to capture the shared, cultural and plural values of SLM, given the range of ecosystem services that may be affected by land degradation. In particular, deeply-held, cultural values and beliefs that may be shared across a community, and widely divergent preferences that may be placed on the same ecosystem state by different communities (cf. bush encroachment for cattle versus goat farmers), can easily be omitted (Kenter et al., 2014).

By considering the shared, cultural and plural value of different SLM options through deliberation, landscape scale changes in land management may occur, that are consistent with actual preferences and more deeply held values of local communities. However, given the complex tenure arrangements that exist in many rangeland areas (for example the three tenure types outlined in the Kalahari case), there remains a danger that economic policy instruments, for example PES schemes, may lead to unintended social justice concerns (Stringer et al., 2012a), in a similar way to the privatisation of communal land since the 1970s in Botswana (see also Dougill et al., 2012). Privatisation conceives that compared to historic communal ownership arrangements, private owners will be more highly motivated and able to prevent natural capital from crossing thresholds that may endanger the flow of ecosystem services from their land (principally provisioning services on which their livelihoods are based). However, in the same way that privatisation can focus on maximising one ecosystem service (cattle production), often at the expense of others (e.g. climate regulation, habitats for wildlife and water quality), mechanisms such as PES can favour the supply of services for which there are markets (e.g. climate regulation via carbon markets) over those for which there is no market (e.g. habitats for wildlife). In the same way that privatisation concentrated natural capital in the hands of the rich at the expense of poorer communal pastoralists, there is a danger that PES schemes will be more accessible with lower transaction costs for richer, landowning pastoralists, who may capture the market before it is possible for communal pastoralists to organise themselves sufficiently to enter the market-place (Stringer et al., 2012b).

It is therefore imperative that the creation of new markets for ecosystem services is seen as one of a number of policy instruments available to governments (see ELD Initiative, 2013 for others), and that these markets are carefully regulated to prevent the provision of certain services at the expense of others. There may also be a role for government to promote social capital among groups of communal pastoralists, to enable them to access PES or agri-environment schemes and compete effectively with private landowners.

4. Conclusions

Land degradation can undermine livelihoods as a result of an effectively permanent reduction in the provision of ecosystem services from land (including provisioning, supporting, regulating and/or cultural services). Often, this is due to the undervaluing and consequent loss of natural capital beyond critical thresholds. To reorient rangeland management pathways away from degradation and towards SLM requires natural capital to be sufficiently valued to be maintained above critical thresholds.

This paper has identified new ways of tackling land degradation and sustaining livelihoods based on retaining critical levels of natural capital and finding ways of providing ecosystem services from that asset base (Section 2.3; Table 4). We used the case of the Kalahari rangelands in Botswana, with a particular focus on bush encroachment, to consider the use of economic instruments to tackle land degradation and sustain livelihoods. These instruments present challenges as well as opportunities for those living in rangelands. It is vital to move beyond purely monetary valuation of ecosystem services, and to learn from the mistakes of similar measures introduced elsewhere. In this way, it may be possible to develop new mechanisms based on addressing the economic root causes of land degradation that can benefit both the rich and the poor, and which can sustain livelihoods through the continued provision of ecosystem services across a range of types of land tenure.

Acknowledgements

This research was funded by the Economics of Land Degradation (ELD) Initiative (Contract Number: 81163498). Thanks to Susannah Sallu for constructive feedback on an early draft of this paper. Thanks to Jasper Kenter for input to material on shared values. Thanks to Emmanuelle Quillerou and Richard Thomas for constructive feedback on the manuscript prior to submission. Thanks also to two anonymous reviewers for constructive feedback.

References

Root-Bernstein, M., Jaksic, F., 2013. The Chilean Espinal: restoration for a sustainable silvopastoral system. Restor. Ecol. 21, 409–414.
Abdeta, A., 2011. Impact of *Acacia drepanolobium* (an invasive woody species) on gum-resin resources and local livelihood in Borana, sourthern Ethiopia. Trop. Subtrop. Agroecosyst. 14, 1063–1074.
Abel, N.O.J., Blaikie, P.M., 1989. Land degradation, stocking rates and conservation policies in the communal rangelands of Botswana and Zimbabwe. Land Degrad. Rehabil. 1, 101–123.
Acemoglu, D., Johnson, S., Robinson, J., 2003. An african Success Story: Botswana. In: Rodrik, D. (Ed.), In Search of Prosperity: Analytic Narratives on Economic Growth. Princeton University Press, Princeton, N.J, pp. 80–119.
Adams, M., White, R., Raditloaneng, N., Aliber, M., Stracey, G., McVey, C., Kalabamu, F., McAuslan, P., Kgengwenyane, N., Sharp, C., Egner, B., 2002. National Land Policy: Issues Report for Republic of Botswana Ministry of Lands, Housing and Environment. Department of Lands, Natural Resource Services (Pty) Ltd, Gaborone.
Adhikari, B., Boag, G., 2013. Designing payments for ecosystem services schemes: some considerations. Curr. Opin. Environ. Sustain. 5, 72–77.
Akhtar-Schuster, M., Thomas, R., Stringer, L.C., Chasek, P., Seely, M., 2011. Improving the enabling environment to combat land degradation: institutional, financial, legal and science-policy challenges and solutions. Land Degrad. Dev. 22, 299–312.
Allsopp, N., 2013. Adaptive management for complex communal rangelands in South Africa. Afr. J. Range Forage Sci. 30, 65–69.
Archer, S., Smeins, F.E., 1991. Ecosystem-level processes. In: Grazing Management: an Ecological Perspective. Timber Press, Portland, OR, pp. 109–139.
Arntzen, J., 2001. The Impacts of Government Policies on Rangelands Conditions and Rural Livelihoods. Working Paper No. 4 of Global Change and subsistence rangelands in Southern Africa (Based on a research project funded by: European Commission as INCO-DC ERBIC 18CT970162, with supplementary funding from NORAD (Project ref: 822.1 Raf 0023). Centre for Applied Research, Gaborone.
Barnes, J.I., McGregor, J., Weaver, L.C., 2002. Economic efficiency and incentives for change within Namibia's community wildlife use initiatives. World Dev. 30, 667–681.
Barrow, C.J., 1991. Land Degradation: Development and Breakdown of Terrestrial Environments. Cambridge University Press, Cambridge.
Behnke, R.H.J., Scoones, I. (Eds.), 1993. Range Ecology at Disequilibrium: New Models of Natural Vulnerability and Pastoral Adaptation in African Savannas. Overseas Development Institute, London.
Belay, T.A., Totland, Ø., Moe, S.R., 2013. Ecosystem responses to woody plant encroachment in a semiarid savanna rangeland. Plant Ecol. 214 (10), 1211–1222.
Berkeley, A., Thomas, A.D., Dougill, A.J., 2005. Spatial dynamics of biological soil crusts: bush canopies, litter and burial in Kalahari rangelands. Afr. J. Ecol. 43, 137–145.

Bond, W.J., 2008. What limits trees in C4 grasslands and savannas? Annu. Rev. Ecol. Evol. Syst. 39, 641–659.

Bond, W.J., Midgley, G.F., 2000. A proposed CO_2-controlled mechanism of woody plant invasion in grasslands and savannas. Glob. Change Biol. 6, 865–869.

Burgess, J., 2003. Country Pasture/Forage Resource Profiles: Botswana. United Nations Food and Agriculture Organisation, Rome [Online]. [Accessed 29/12/04]. Available on World Wide Web at: <http://www.fao.org/WAICENT/FAOINFO/AGRICULT/AGP/AGPC/doc/Counprof/Botswana/botswana.htm> (accessed 03.12.13.).

Buss, H.J., Nuppenau, E.A., 2003. Sustainable range use in semi-arid eco-systems through adapted management: a case study on Namibian farms based on bio-economic models. In: Allsopp, N., Palmer, A.R., Milton, S.J., Kirkman, K.P., Kerley, G.I.H., Hurt, C.R., Brown, C.J. (Eds.), Proceedings of the VIIth International Rangelands Congress 26th July - 1st August, Durban, South Africa, pp. 1949–1952.

Caviezel, C., Hunziker, M., Schaffner, M., Kuhn, N.J., 2014. Soil–vegetation interaction on slopes with bush encroachment in the central Alps–adapting slope stability measurements to shifting process domains. Earth Surf. Process. Landforms 39 (4), 509–521.

Chanda, R., Totolo, O., Moleele, N.M., Setshogo, M., Mosweu, S., 2003. Prospects for subsistence livelihood and environmental sustainability along the Kalahari Transect: the case of Matsheng in Botswana's Kalahari rangelands. J. Arid Environ. 54, 425–445.

Clements, F.E., 1916. Plant Succession: an Analysis of the Development of Vegetation. Carnegie Institute, Washington DC.

Cooke, H.J., 1985. The Kalahari today: a case of conflict over resource use. Geogr. J. 151, 75–85.

Cunningham, P.L., 1998. Potential wood biomass suitable for charcoal production in Namibia. Direct. Agri. Res. Training Namib. Agri-Info 4, 4–7.

de Koning, G.H.J., Benitez, P.C., Munoz, F., Olschewski, R., 2007. Modelling the impacts of payments for biodiversity conservation on regional land-use patterns. Landsc. Urban Plan. 83, 255–267.

de Ridder, N., Wagenaar, K.T., 1984. A comparison between the productivity of traditional livestock systems and ranching in eastern Botswana. ILCA Newsl. 3, 5–7.

Dean, W.R.J., Hoffman, M.T., Meadows, M.E., Milton, S.J., 1995. Desertification in the semiarid Karoo. S. Afr. - Rev. Reassess. J. Arid Environ. 30, 247–264.

Dempsey, J., Robertson, M., 2012. Ecosystem services: tensions and developments within neoliberal environmentalism. Prog. Hum. Geogr. 36 (6), 758–779.

DHV, 1980. Countrywide Animal and Range Assessment Project, vol. 7. European Development Fund and Ministry of Commerce and Industry, Gaborone, 357 pp.

Dougill, A.J., Thomas, A.D., 2002. Nebkha dunes in the Molopo Basin, South Africa and Botswana: formation controls and their validity as indicators of soil degradation. J. Arid Environ. 50 (3), 413–428.

Dougill, A.J., Thomas, A.D., 2004. Kalahari sand soils: spatial heterogeneity, biological soil crusts and land degradation. Land Degrad. Dev. 15 (3), 233–242.

Dougill, A.J., Heathwaite, A.L., Thomas, D.S.G., 1998. Soil water movement and nutrient cycling in semiarid rangeland: vegetation change and system resilience. Hydrol. Process. 12, 443–459.

Dougill, A.J., Thomas, D.S.G., Heathwaite, A.L., 1999. Environmental change in the Kalahari: integrated land degradation studies for Noneqilibrium dryland environments. Ann. Assoc. Am. Geogr. 89, 420–442.

Dougill, A.J., Fraser, E.D.G., Reed, M.S., 2010. Anticipating vulnerability in food systems to climate variability and change: challenges of using dynamic systems approaches and the case of pastoral systems in Botswana. Ecol. Soc. 15 (2), 17.

Dougill, A.J., Stringer, L.C., Leventon, J., Riddell, M., Rueff, H., 2012. Lessons from community-based payment for ecosystem service schemes: from Forests to rangelands. Philos. Trans. R. Soc. B 367, 3178–3190.

Ellis, J.E., Coppock, D.L., McCabe, J.T., Galvin, K., Wienpahl, J., 1984. Aspects of Energy Consumption in a Pastoral Ecosystem: Wood Use by the South Turkana. Wood, Energy and Households: Perspectives on Rural Kenya. Beijer Institute and the Scandinavian Institute of African Studies. Beijer Institute and the Scandinavian Institute of African Studies, 22 pp.

Fisher, J., 2012. No pay, no care? A case study exploring motivations for participation in payments for ecosystem services in Uganda. Oryx 46, 45–54.

Fisher, J.A., Patenaude, G., Meir, P., Nightingale, A.J., Rounsevell, M.D.A., Williams, M., Woodhouse, I.H., 2013. Strengthening conceptual foundations: analysing frameworks for ecosystem services and poverty alleviation research. Glob. Environ. Change 23 (5), 1098–1111.

Global Mechanism, 2012. Incentive and Market-based Mechanisms to Promote Sustainable Land Management: Framework and Tool to Assess Applicability. Global Mechanism of the United Nations Convention to Combat Desertification, Rome.

GOB (Government of Botswana), 2006. Botswana National Action Programme to Combat Desertification (NAP). EAD, Gaborone, Botswana.

Grellier, S., Kemp, J., Janeau, J.L., Florsch, N., Ward, D., Barot, S., Valentin, C., 2012. The indirect impact of encroaching trees on gully extension: a 64year study in a sub-humid grassland of South Africa. Catena 98, 110–119.

Grossman, D., Holden, P.L., 2005. An Overview of Game Ranching in Southern Africa. Occasional Research Paper for Ministry of Agriculture, Botswana.

Hagos, M.G., Smit, G.N., 2005. Soil enrichment by Acacia mellifera subsp. detinens on nutrient poor sandy soil in a semi-arid southern African savanna. J. Arid Environ. 61, 47–59.

Harvey, Charles, 1992. Botswana: is the economic miracle over? J. Afr. Econ. 1.3, 335–368.

Hastrup, K., Hervik, P. (Eds.), 2003. Social Experience and Anthropological Knowledge. Routledge.

Higgins, S.I., Scheiter, S., 2012. Atmospheric CO_2 forces abrupt vegetation shifts locally, but not globally. Nature 488, 209–213.

Holling, C.S., 1986. The resilience of terrestrial ecosystems, local surprise and global change. In: Clark, W.C., Mtmn, R.E. (Eds.), Sustainable Development of the Biosphere. Cambridge University Press, Cambridge, pp. 292–317.

Hoole, A., Berkes, F., 2010. Breaking down fences: recoupling social-ecological systems for biodiversity conservation in Namibia. Geoforum 41, 304–317.

Hubbard, M., 1982. Stock Limitation — Any Economic Alternatives for Botswana?. Pastoralist Development Network Paper, 14c Agricultural Administration Unit, ODI, London, 8 pp.

Hudak, A.T., Wessman, C.A., Seastedt, T.R., 2003. Woody overstorey effects on soil carbon and nitrogen pools in South African savanna. Austral Ecol. 28 (2), 173–181.

Ibrahim, M., Guerra, L., Casasola, F., Neely, C., 2010. Importance of silvopastoral systems for mitigation of climate change and harnessing of environmental benefits. Grassl. Carbon Seq. Manag. Policy Econ. 11, 189–196.

Illius, A.W., O'Connor, T.G., 1999. On the relevance of nonequilibrium concepts to arid and semiarid grazing systems. Ecol. Appl. 9, 798–813.

ELD Initiative, 2013. The Rewards of Investing in Sustainable Land Management. Interim Report for the Economics of Land Degradation Initiative: A global strategy for sustainable land management. Available from: www.eld-initiative.org.

IPCC (International Panel of Climate Change), 2001. Climate Change 2001: Impacts, Adaptation, and Vulnerability. Cambridge University Press, Cambridge.

Jensen, C.L., 1984. Wood use by the Amboseli Maasai. In: Barnes, C., Ensminger, J., O'Keefe, Ph (Eds.), Wood, Energy and Households Perspectives on Rural Kenya, Energy, Environment and Development in Africa Series No. 6. The Beijer Institute/Scandinavian Institute of African Studies, Uppsala, pp. 188–204.

Jones, B.T.B., 2003. Using wildlife and tourism for the diversification of rangeland use in Namibia. In: Allsopp, N., Palmer, A.R., Milton, S.J., Kirkman, K.P., Kerley, G.I.H., Hurt, C.R., Brown, C.J. (Eds.), Proceedings of the VIIth International Rangelands Congress 26th July - 1st August, Durban, South Africa, pp. 1635–1643.

Jones, B., Murphree, M., 2001. The evolution of policy on community conservation in Namibia and Zimbabwe. In: Hulme, D., Murphree, M.W. (Eds.), African Wildlife and African Livelihoods: the Promise and Performance of Community Conservation. James Currey, Oxford.

Joubert, D.F., Smit, G.N., Hoffman, M.T., 2012a. The influence of rainfall, competition and predation on seed production, germination and establishment of an encroaching Acacia in an arid Namibian Savanna. J. Arid Environ. 91, 7–13.

Joubert, D.F., Smit, G.N., Hoffman, M.T., 2012b. The role of fire in preventing transitions from a grass dominated state to a bushthickened state in arid savannas. J. Arid Environ. 87, 1–7.

Joubert, D.F., Smit, G.N., Hoffman, M.T., 2013. The influence of rainfall, competition and predation on seed production, germination and establishment of an encroaching Acacia in an arid Namibian savanna. J. Arid Environ. 91, 7–13.

Kasperson, J., Kasperson, R., Turner, B., 1995. Regions at Risk. United Nations University Press [Online]. [Accessed 29/12/04]. Available on World Wide Web at: http://www.unu.edu/unupress/unupbooks/uu14re/uu14re00.htm.

Katjiua, M., Ward, D., 2007. Pastoralists' perception and realities of vegetation change and browse consumption in the northern Kalahari, Namibia. J. Arid Environ. 69, 716–730.

Kenter, J.O., Reed, M.S., Irvine, K.N., O'Brien, E., Brady, E., Bryce, R., Christie, M., Church, A., Cooper, N., Davies, A., Hockley, N., Fazey, I., Jobstvogt, N., Molloy, C., Orchard-Webb, J., Ravenscroft, N., Ryan, M., Watson, V., 2014. UK National Ecosystem Assessment Follow-on Phase, Technical Report: Shared, Plural and Cultural Values of Ecosystems. UNEP-WCMC, Cambridge.

Klintenberg, P., Seely, M.K., Christiansson, C., 2007. Local and national perceptions of environmental change in central northern Namibia: do they correspond? J. Arid Environ. 69, 506–525.

Kraaij, T., Ward, D., 2006. Effects of rain, nitrogen, fire and grazing on tree recruitment and early survival in bush-encroached savanna, South Africa. Plant Ecol. 186 (2), 235–246.

Lohmann, D., Tietjen, B., Blaum, N., Joubert, D.F., Jeltsch, F., 2012. Shifting thresholds and changing degradation patterns: climate change effects on the simulated long-term response of a semi- arid savanna to grazing. J. Appl. Ecol. 49, 814–823.

Mainah, J., 2001. The distribution and association of Grewia flava with other species in the Kalahari environment. Botsw. Notes Rec. 115–127.

Mainah, J., 2006. Socio and economic importance of grewia flava to the people of Western parts of Kweneng district and factors affecting its spatial distribution in Botswana. Botsw. Notes Rec. 11–19.

Malope, P., Batisani, N., 2008. Land reforms that exclude the poor: the case of Botswana. Dev. South. Afr. 25 (4), 383–397.

McGranahan, D.A., 2008. Managing private, commercial rangelands for agricultural production and wildlife diversity in Namibia and Zambia. Biodivers. conserv. 17, 1965–1977.

Meik, J.M., Jeo, R.M., Mendelson Iii, J.R., Jenks, K.E., 2002. Effects of bush encroachment on an assemblage of diurnal lizard species in central Namibia. Biol. Conserv. 106 (1), 29–36.

Mersmann, C., Quatrini, S., Buric , B., Schild, J., Asen, A., 2010. Economic Valuation of Land: an Innovative and Scalable Approach to Advance Sustainable Land Management. GM of the UNCCD, Rome, Italy.
Middleton, N., Stringer, L.C., Goudie, A., Thomas, D.S.G., 2011. The Forgotten Billion: MDG Achievement in the Drylands. UNCCD-UNDP, New York and Nairobi.
Midgley, G.F., Bond, W.J., Wand, S.J.E., Roberts, R., 1999. Will Gullivers Travel? Potential Causes of Changes in Savanna Tree Success Due to Rising Atmospheric CO2. In: Towards Sustainable Management in the Kalahari Region—some Essential Background and Critical Issues. University of Botswana, Gaberone, Botswana.
Millennium Ecosystem Assessment, 2003. Ecosystems and Human Well-being: a Framework for Assessment. World Resources Institute, Washington, DC.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being. World Health Organization, Geneva, Switzerland.
Milton, S.J., Dean, W.R.J., Richardson, D.M., 2003. Economic incentives for restoring natural capital in southern African rangelands. Front. Ecol. Environ. 1, 247–254.
Moleele, N.M., Chanda, R., 2003. The impact of different land use management systems on the vegetation and wildlife communities of the Matsheng area, Kgalagadi North, Botswana. In: Allsopp, N., Palmer, A.R., Milton, S.J., Kirkman, K.P., Kerley, G.I.H., Hurt, C.R., Brown, C.J. (Eds.), Proceedings of the VIIth International Rangelands Congress 26th July - 1st August, Durban, South Africa, pp. 436–445.
Moleele, N.M., Mainah, J., 2003. Resource use conflicts: the future of the Kalahari ecosystem. J. Arid Environ. 54, 405–423.
Moleele, N.M., Perkins, J.S., 1998. Encroaching woody plant species and boreholes: is cattle density the main driving factor in the Olifants Drift communal grazing lands, south-eastern Botswana? J. Arid Environ. 40, 245–253.
Moleele, N.M., Ringrose, S., Matheson, W., Vanderpost, C., 2002. More woody plants? the status of bush encroachment in Botswana's grazing areas. J. Environ. Manag. 64, 3–11.
Moncrieff, G.R., Scheiter, S., Bond, W.J., Higgins, S.I., 2014. Increasing atmospheric CO2 overrides the historical legacy of multiple stable biome states in Africa. New. Phytol. 201, 908–915.
Noel, S., Soussan, J., 2010. Economics of Land Degradation: Supporting Evidence-base Decision Making. In: Methodology for Assessing Costs of Degradation and Benefits of Sustainable Land Management. Paper commissioned by the Global Mechanism of the UNCCD to the Stockholm Environment Institute (SEI). September 2010.
O'Connor, T.G., Puttick, J.R., Hoffman, M.T., 2014. Bush encroachment in southern Africa: changes and causes. Afr. J. Range Forage Sci. 31 (2), 67–88.
Pagiola, S., Ramirez, E., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., Ruiz, J.P., 2007. Paying for the environmental services of silvopastoral practices in Nicaragua. Ecol. Econ. 64, 374–385.
Perkins, J.S., 1996. Botswana: fencing out the equity issue. Cattle posts and cattle ranching in the Kalahari Desert. J. Arid Environ. 33, 503–517.
Perkins, J.S., Thomas, D.S.G., 1993. Spreading deserts or spatially confined environmental impacts - land degradation and cattle ranching in the Kalahari desert of Botswana. Land Degrad. Rehabil. 4, 179–194.
Perkins, J.S., Stuart-Hill, G., Kgabung, B., 2002. The impact of cattle-keeping on the wildlife, vegetation, and Veld products of the kalahari. In: Sporton, D., Thomas, D.S.G. (Eds.), Sustainable Livelihoods in Kalahari Environments. Oxford University Press, Oxford, pp. 66–79.
Perkins, J.S., Reed, M.S., Akanyang, L., Atlhopheng, J.R., Chanda, R., Magole, L., Mphinyane, W., Mulale, K., Sebego, R.J., Fleskens, L., Irvine, B., 2013. Making land management more sustainable: experience implementing a new methodological framework in Botswana. Land Degrad. Dev. 24, 463–477.
Piñero, J., Maestre, Fernando T., Bartolomé, Lorenzo, Valdecantos, Alejandro, 2013. Ecotechnology as a tool for restoring degraded drylands: a meta-analysis of field experiments. Ecol. Eng. 61, 133–144.
Porras, I., 2010. Fair and Green? Social Impacts of Payments for Environmental Services in Costa Rica. iied, London, 37 pp.
Quan, J., Barton, D., Conroy, C., 1994. A Preliminary Assessment of the Economic Impact of Desertification in Namibia. DEA Research Discussion Paper 3. Directorate of Environmental Affairs, Ministry of Agriculture, Water and Rural Development, Namibia.
Ratajczak, Z., Nippert, J.B., Hartman, J.C., Ocheltree, T.W., 2011. Positive feedbacks amplify rates of woody encroachment in mesic tallgrass prairie. Ecosphere 2 (11), art121.
Ratajczak, Z., Nippert, J.B., Collins, S.L., 2012. Woody encroachment decreases diversity across North American grasslands and savannas. Ecology 93 (4), 697–703.
Reed, M.S., 2005. Participatory Rangeland Monitoring and Management in the Kalahari (PhD Thesis). School of Earth & Environment, University of Leeds, Botswana.
Reed, M.S., Dougill, A.J., 2002. Participatory selection process for indicators of rangeland condition in the Kalahari. Geogr. J. 168, 224–234.
Reed, M.S., Dougill, A.J., 2010. Linking degradation assessment to sustainable land management: a decision support system for Kalahari pastoralists. J. Arid Environ. 74, 149–155.
Reed, M.S., Dougill, A.J., Taylor, M.J., 2007. Integrating local and scientific knowledge for adaptation to land degradation: Kalahari rangeland management options. Land Degrad. Dev. 18, 249–268.
Reed, M.S., Dougill, A.J., Baker, T., 2008. Participatory indicator development: what can ecologists and local communities learn from each other? Ecol. Appl. 18, 1253–1269.
Rennie, T.W., Light, D., Rutherford, A., Miller, A., Fisher, A., Pratchett, D., Capper, B., Buck, N., Trail, J., 1977. Beef cattle productivity under traditional and improved management in Botswana. Trop. Anim. Health Prod. 9, 1–6.
Reynolds, J.F., Grainger, A., Stafford Smith, D.M., Bastin, G., Garcia-Barrios, L., Fernández, R.J., Janssen, M.A., J ürgens, N., Scholes, R.J., Veldkamp, A., Verstraete, M.M., von Maltitz, G., Zdruli, P., 2011. Scientific concepts for an integrated analysis of desertification. Land Degrad. Develop. 22, 166–183.
Reynolds, J.F., Smith, Stafford, Lambin, D.K., Turner II, E.F., Mortimore, B.L., Batterbury, M., Dowing, S.P.J., Th Dowlatabadi, E.H., Ferna ndez, R.J., Herrick, J.E., Huber Sannwald, E., Jiang, H., Leemans, R., Lynam, T., Maestre, F.T., Ayarza, M., Walker, B., 2007. Global desertification: building a science for dryland development. Science 316, 847–851.
Sallu, S.M., Twyman, C., Thomas, D.S.G., 2009. The multidimensional nature of biodiversity and social dynamics and implications for contemporary rural livelihoods in remote Kalahari settlements, Botswana. Afr. J. Ecol. 47, 110–118.
Sallu, S.M., Twyman, C., Stringer, L.C., 2010. Resilient or vulnerable livelihoods? Assessing livelihood dynamics and trajectories in rural Botswana. Ecol. Soc. 15.
Scholes, R.J., 2003. Convex relationships in ecosystems containing mixtures of trees and grass. Environ. Resour. Econ. 26, 559–574.
Schwilch, G., Bestelmeyer, B., Bunning, S., Critchley, W., Herrick, J., Kellner, K., Liniger, H.P., Nachtergaele, F., Ritsema, C.J., Schuster, B., Tabo, R., van Lynden, G., Winslow, M., 2011. Experiences in monitoring and assessment of sustainable land management. Land Degrad. Dev. 22, 214–225.
Scoones, I., 1995. New directions in pastoral development in Africa. In: Scoones, I. (Ed.), Living with Uncertainty: New Directions in Pastoral Development in Africa. Intermediate Technology Publications, London, pp. 1–36.
Sietz, D., Lüdeke, Matthias K.B., Walther, Carsten, 2011. Categorisation of typical vulnerability patterns in global drylands. Glob. Environ. Change 21, 431–440.
Smit, G.N., 2004. An approach to tree thinning to structure southern African savannas for long-term restoration from bush encroachment. J. Environ. Manag. 71 (2), 179–191.
Stringer, L.C., Thomas, D.S.G., Twyman, C., 2007a. From global politics to local land users: applying the united nations convention to combat desertification in Swaziland. Geogr. J. 173 (2), 129–142.
Stringer, L.C., Reed, M.S., Dougill, A., Seely, M.K., Rokitzki, M., 2007b. Implementing the UNCCD: participatory challenges. Nat. Resour. Forum 31, 198–211.
Stringer, L.C., Dyer, J., Reed, M., Dougill, A., Twyman, C., Mkwambisi, D., 2009a. Adaptations to climate change, drought and desertification: insights to enhance policy in southern Africa. Environ. Sci. Policy 12 (7), 748–765.
Stringer, L.C., Scrieciu, S.S., Reed, M., 2009b. Biodiversity, land degradation and climate change: participatory planning in Romania. Appl. Geogr. 29, 77–90.
Stringer, L.C., Dougill, A.J., Thomas, A.D., Spracklen, D.V., Chesterman, S., Ifejika Speranza, C., Rueff, H., Riddell, M., Williams, M., Beedy, T., Abson, D., Klintenberg, P., Syampungani, S., Powell, P., Palmer, A.R., Seely, M.K., Mkwambisi, D.D., Falcao, M., Sitoe, A., Ross, S., Kopolo, G., 2012a. Challenges and opportunities in linking carbon sequestration, livelihoods and ecosystem service provision in drylands. Environ. Sci. Policy 19–20, 121–135.
Stringer, L.C., Dougill, A.J., Mkwambisi, D.D., Dyer, J.C., Kalaba, F., Mngoli, M., 2012b. Challenges and opportunities for carbon management in Malawi and Zambia. Carbon Manag. 2 (3), 159–173.
Tabor, R., 1994. Traditional Woodland Crafts. Batsford, London.
Tanser, F.C., Palmer, A.R., 1999. The application of a remotely-sensed diversity index to monitor degradation patterns in a semi-arid, heterogeneous, South African landscape. J. Arid Environ. 43, 477–484.
Taylor, M., 2003. Trajectories in community based natural resource management in southern africa. In: Proceedings of the Arid Climate Adaptation and Cultural Innovation in Africa Conference. University of Cologne, 1–3 October.
Thomas, A.D., Dougill, A.J., 2007. Spatial and temporal distribution of cyanobacterial soil crusts in the Kalahari: implications for soil surface properties. Geomorphology 85 (1), 17–29.
Thomas, D.S.G., Twyman, C., 2004. Good or bad rangeland? Hybrid knowledge, science, and local understandings of vegetation dynamics in the Kalahari. Land Degrad. Dev. 15, 215–231.
Thomas, D.S.G., Sporton, D., Perkins, J.S., 2000. The environmental impact of livestock ranches in the Kalahari, Botswana: natural resource use, ecological change and human response in a dynamic dryland system. Land Degrad. Dev. 11, 327–341.
Thomas, D.S.G., Sporton, D., Twyman, C., 2001. Policy Intervention, Livelihood Sustainability and the Environment: the FAP PANRUSA Briefing Note No. 10B. University of Sheffield.
Tsie, Balefi, 1996. The political context of Botswana's development performance. J. South. Afr. Stud. 22.4, 599–616.
UNEP, 1992. World Atlas of Desertification, first ed. Edward Arnold, London.
UNEP, 1997. In: Middleton, N., Thomas, D.S.G. (Eds.), World Atlas of Desertification, second ed. Edward Arnold, London.
van Rooyen, A.F., 1998. Combating desertification in the southern Kalahari: connecting science with community action in South Africa. J. Arid Environ. 39, 285–297.
Vetter, S., 2013. Development and sustainable management of rangeland commons – aligning policy with the realities of South Africa's rural landscape. Afr. J. Range Foroage Sci. 30, 1–9.
Ward, D., 2005. Do we understand the causes of bush encroachment in African savannas? Afr. J. Range Forage Sci. 22 (2), 101–105.
Warren, A., Agnew, C., 1988. An Assessment of Desertification and Land Degradation in Arid and Semi-arid Areas. International Institute for Environment and

Development, Drylands Programme, Paper 2. International Institute for Environment and Development, London.
Wigley, B.J., Bond, W.J., Hoffman, M.T., 2010. Thicket expansion in a South African savanna under divergent land use: local vs. global drivers? Glob. Change Biol. 16, 964–976.
WOCAT, 2010. Sustainable Land Management [online]. Available at: https://www.wocat.net/en/about-wocat/vision-mission/sustainable-land-management.html (accessed 06.12.13.).
Worku, A., Lemenih, M., Fetene, M., Teketay, D., 2011. Socio-economic importance of gum and resin resources in the dry woodlands of Borana, southern Ethiopia. For. Trees Livelihoods 20 (2–3), 137–155.
Zbinden, S., Lee, D.R., 2005. Paying for environmental services: an analysis of participation in Costa Rica's PSA Program. World Dev. 33 (2).