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Perspective

Ten lessons for the conservation of African savannah ecosystems

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ABSTRACT

Knowledge of the success or otherwise of conservation interventions is often locked within localised networks, resulting in mistakes being replicated unnecessarily. The savannahs of eastern and southern Africa are home to spectacular ecosystems with similar ecology yet markedly different conservation practices between the two regions. Pressures on east African ecosystems are rising in ways similar to those of southern Africa several decades ago. Conservation practitioners and researchers from southern and eastern Africa came together for a 5-day workshop to identify by consensus a short list of 10 most important lessons for management of savannah habitats learnt from the southern experience. The lessons identified concerned (1) protected area design, (2) community relationships, (3) buffer zones, (4) the importance of migrations and corridors, (5) river catchment management, (6) law enforcement, (7) invasive plants, (8) road planning, (9) loss of heterogeneity, and (10) communication between researchers and practitioners. The lessons learnt from southern Africa can prevent many mistakes being made in east African protected area management, providing they are implemented on the ground.

Contents

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1. Introduction

When conservation managers make decisions they draw on a combination of personal experience, ecological understanding and political expediency. Often, the outcomes of management decisions are noted only informally or in internal reports and are not widely available to others. Recently, there has been a push to record and share management interventions and their consequences, enabling learning from experiences elsewhere that may have relevance to local situations (Sutherland et al., 2004). Evidence-based conservation, with centralised databases such as conservationevidence.com provide a valuable contribution to this process (Sutherland et al., 2004). However, in most of the world, the problem persists that knowledge of successful or unsuccessful conservation actions remains localised and tacit within the memories of experienced conservation practitioners and ecologists. We brought together a number of experienced ecologists and conservation practitioners from Southern and Eastern Africa to explore lessons that could be learnt from the contrasting experiences of the two regions.

The savannah systems of Eastern and Southern Africa are similar in their ecological processes, function and structure and share many animal and plant taxa (Fritz and Duncan, 1994; Deshmukh, 2008). However, conservation paradigms are famously different: traditionally, Southern African protected areas (encompassing South Africa, Zimbabwe and Namibia) are perceived as highly managed systems often with fences, artificial water holes, strict fire regimes and culling programmes (Pienaar, 1983), whereas East African (here meaning Kenya, Tanzania, Uganda, Rwanda and Burundi) protected areas are mostly unenclosed and have traditionally had a 'hands-off' management policy (Newmark, 2008).

In South Africa, the Kruger National Park had its western boundary completely fenced by 1961, as a veterinary cordon to restrict the spread of disease from wildlife to cattle (Joubert, 2007), while Hluhluwe-iMfolozi was completely fenced by 1965, to prevent wildlife from spreading into the densely settled surroundings (Brooks and Macdonald, 1983). Unfortunately, hard boundaries around protected areas led to an array of cascading ecological issues that required further interventions to reverse population declines and habitat degradation: e.g. fences blocked the wildebeest migration in the west, leading to the culling of the wildebeest to contain the perceived overgrazing that resulted within the park, plus widespread provision of artifical waterholes inside the park with ramifying consequences (Pienaar, 1983; Whyte and Joubert, 1988; Owen-Smith, 1996; Smit et al., 2007). With the benefit of hindsight, some of these earlier interventions and the philosophies behind them are now being reviewed and reversed: fences have been removed, water holes closed (Smit and Grant, 2009) and fire regimes altered from rotational block burning (van Wilgen et al., 2008). With similar population and land-use pressures growing in Eastern Africa to those that resulted in an interventionist prerogative in South Africa several decades ago, many of the same 'solutions' (with unintended negative consequences) are likely to occur unless planners make use of lessons learnt elsewhere. As important as unwelcome interventions can be the lack of timely intervention where threats are not realised: lack of action on invasive plants is a classic example (de Lange and van Wilgen 2010).

Already, Kenyan national parks such as Nakuru and the Aberdares have been fenced both to protect wildlife within the parks from humans, and to protect people and crops from wild animals (Gross, 2009). Even without fences many boundaries are becoming harder for animals to cross as large, settled, agricultural populations replace nomadic lifestyles in the surrounding areas (Newmark, 2008; Western et al., 2009). Typical of the increase of agriculture is the expansion of cultivated land around Kenya's Masai Mara Reserve from 4,875 ha in the mid-1970s, to over 50,000 ha 20 years later (Serneels et al., 2001), associated with substantial declines in large mammal populations (Ogutu et al., 2011). There is also growing human-wildlife conflict: in Tanzania, lion attacks on humans have increased dramatically since 1990 (at least 563 people were killed between 1990 and 2004), particularly in areas where natural prey is scarce (Packer et al., 2005). Poaching within and around protected areas is a serious threat and understood to be the primary cause of several reduced and declining mammal populations (Hilborn et al., 2006; Stoner et al., 2007; Newmark, 2008). These problems are replicated in savannahs globally: Brazil's Cerrado is heavily converted for agriculture and under pressuce from invasive species (Klink and Machado, 2005); India's savannah reserves are increasingly isolated (Vidya et al,. 2004) and an interventionist strategy is increasingly applied (Madhusudan and Shankar Raman, 2003); Australian savannahs are suffering biodiversity losses from lost heterogeneity (Bird et al., 2008).

Despite the fact that some 20% of East African lands are officially protected (IUCN and UNEP, 2009), most of East Africa's wildlife depends at least partly on land outside protected areas: many migratory ungulate populations (including the Serengeti migration) spend periods outside protected areas, and significant animal populations utilise village lands year round (Caro et al., 2009). Compared with southern Africa, fewer significant corridors and dispersal areas between protected areas have been completely lost (Jones et al., 2009), with important benefits for animal populations: e.g. an estimated 2000 wildebeest and 3000 zebra recolonised Kenya's Amboseli NP following a catastrophic drought in 2009 (Worden et al., 2010). In East Africa, therefore, there remains a (shrinking) opportunity to maintain fully functional ecosystems containing diverse large mammal communities for the long term. To do so, however, requires not just commitment and funding, but also understanding of how savannah ecosystems function, and how losses of functioning can be prevented, minimized or mitigated through informed management and through turning it into an economic benefit for local people (Mangel et al., 1996). Many of the problems that have occurred in southern Africa are now developing in East Africa and savannahs globally; shared experiences should benefit all areas.

At a workshop in the Serengeti in November 2010, eight conservation managers and researchers working in southern and eastern Africa came together to identify ten conservation activities (or inactivities) that occurred in southern Africa but which, with the benefit of hindsight, should not be repeated or should be adapted to avoid negative consequences for biodiversity. Invitations were sent to a range of prominent researchers and conservation managers active in eastern and southern Africa (the list of authors identifies those organisations and individuals who responded). The primary focus was on how to ensure the ecological integrity of "open" systems as human pressure increases within these systems. A range of topics identified by participants were consolidated into 18 lessons, combining variations on the same issue where possible. Each topic was then discussed to clarify its impact, before participants rated each issue by priority: high (having the potential to result in ecosystem collapse), medium (risking serious damage to the ecosystem) or low (important, but not as potentially serious). Brief discussion usually resulted in unanimous ranking. After all 18 lessons were rated discussions focussed on ranking the medium importance issues in order to identify two to include with the eight high ranking lessons to generate a final list of 10. A further discussions provided unanimous agreement on the top 10 and consensus decisions on relative ranking within these. We recognise that some interventions now perceived as errors were themselves only carried out in an effort to correct the consequences of earlier decisions (i.e. there are cascading consequences): if the earlier errors can be avoided, subsequent problems may never arise. Thus, we attempt to rank the lessons identified into those that deal with the root of the problem, and those that deal with consequences further down the management line. Lessons were ranked without regard to cost or political expediency, yet real-world conservation and landscape planning must balance a wide range of competing demands (Braunisch et al., 2012). This ensures that total reversal of all changes threatening East African ecosystems is impractical, but identifying and prioritising lessons is still important if conservationists are to engage effectively within land-use debates. With a clear view of the problems and optimal solutions that faces biodiversity in the savannah it is much easier to identify acceptable compromises than when such debates are poorly informed (Colyvan et al., 2011).

In this paper we briefly describe each lesson in turn, giving key references identifying the problem, examples and solutions. Here, we do not attempt full discussion of these complex lessons, aiming instead to provide an overview of a wide range of issues with sufficient pointers to enable to interested readers to access further information on specific lessons. Each lesson description takes the form of a brief description of the lesson followed by some examples (culled from our own experiences and knowledge of examples in the literature) of the problems caused in southern Africa and concludes with a description of the current situation in East Africa. We provide an additional summary of the lessons in Table 1.

2. The lessons

2.1. Wrong boundaries

As with protected areas in many parts of the word, the boundaries of southern African reserves were designated on the basis of pragmatism under increasing pressure for land: political boundaries, existing land use, tsetse fly presence, disease and opportunities presented by lack of settlement were more important considerations than ecological integrity. Indeed, at the time of gazetting, information about ecological boundaries of the protected systems was often lacking. Even South Africa's largest two parks, Kruger (19,500 km^2) and Kalahari (5000 km^2), do not represent entire ecosystems (Fynn and Bonyongo, 2011). For instance, the wetter western section of the Sabi Game Reserve, providing the best habitats for rarer antelope and a high-rainfall dry-season range for other ungulates (similar to the high-rainfall regions of the Serengeti), was removed to enable an equivalent amount of land to be acquired to join Sabi to the Shingwedzi Game Reserve, creating the present boundaries of Kruger. However, as a consequence of subsequent human settlement and activities in the adjoining regions it has become impractical to adjust these boundaries in ecologically appropriate ways, except where privately owned land is given over to conservation (Nelson, 2008; Mbaiwa and Stronza, 2011).

The ecological consequences of fixed boundaries not delimiting a fully functioning ecological system are considerable, and are the ultimate cause of many later problems (Pienaar, 1983). Notably, the seasonal routes of migrant mammals are cut, usually leading to smaller resident populations and vegetation change generated by year-round grazing pressure and disruption of fire patterns (Shrader et al., 2010). Infrequent, but vital, movements in response to extreme conditions (e.g. to rarely used drought refugia) become impossible, leading to increased variation in annual survival (Shrader et al., 2010). Similarly, some protected areas excluded topographic features particularly important for plant diversity – for example, steep slopes offer an important refuge from herbivory for many plant species but were initially not included in parks such as Addo Elephant NP (Edkins et al., 2008; Cowling et al., 2009).

In East Africa, many protected areas were designated based upon the experience of hunters, who were mainly aware of dry (hunting) season aggregations of animals. Even Serengeti NP was gazetted without knowledge of the migratory routes: it is only happy coincidence that the migration remains within protected areas for much of the year (Sinclair, 2012). Thus many protected areas in East Africa protect only dry season refugia and, for the ecosystem to remain functional, wildlife must be protected during the wet season as well (Newmark, 1996). In addition buffer areas utilised during extreme conditions need to be identified and secured. Before decisions about realigning protected areas or alternative protection strategies can be proposed, detailed maps of key wildlife habitats inside and outside of protected areas are needed, preferably involving calculation of minimum viable areas, such as those undertaken around Amboseli (Western and Gichohi, 1993; Western, 2007). It is likely that most important wildlife areas with low current protection occur in landscapes with low land-use intensity, meaning adequate protection for these areas remains a realistic possibility. Once ecosystem boundaries have been identified, it is then necessary to identify the most appropriate methods of protecting wildlife in the areas not currently given adequate

Table 1
Summary of the 10 lessons in approximate rank order of importance.

Problem	Solutions
(1) Wrong boundaries	Following mapping of biodiversity resources over several years, revise boundaries enlarging, adjusting and identifying new areas as necessary
(2) Poor public relations	Ensure access for local communities is feasible and inexpensive (including car access); consider sustainable harvests in and around protected areas where appropriate; allocate gate-fees equitably to central and local people; consider options for co-ownership with local communities
(3) Lack of buffer zones	Develop biodiversity maps that enable sensible zonation of land around protected areas, identify (with communities) land uses for these areas that encourage sustainability
(4) Loss of migratory corridors	Identify current and recently used corridors; establish land-use guidelines that enable continued use of corridors by wildlife
(5) Inadequate protection of river catchments	Establish catchment-based land-use plans to balance requirements of water users and wildlife, plus ensuring water retention and provisioning services are prioritised
(6) Inadequate law enforcement	Train, resource and motivate rangers adequately. Ensure arrests lead to convictions, and punishment is a deterrent
(7) Delayed response to invasive plants	Identify likely invasive species, assess current status of existing invasive species. Prioritise activities, implement management plans, resource and monitor effectiveness of management
(8) Inappropriate road planning	The impact of roads in and around protected areas should be thoroughly assessed both for current traffic levels, and for likely future levels of use. Current roads can be rerouted to avoid sensitive areas or to increase game viewing opportunities and encourage slow traffic. Surfacing should be considered where the impact can be reduced through this
(9) Allowing loss of functional heterogeneity	Functional heterogeneity should be identified and mapped at appropriate spatial scales. Land-use and management plans should explicitly consider heterogeneity, and all planned management interventions should consider impacts on heterogeneity in addition to direct influence on biodiversity
(10) Inadequate practitioner-science interaction	Structures should be put in place to improve relationships between researchers and practitioners, from encouraging researchers to contribute to formal management plans and implementation, to researchers offering accessible summaries of their research findings to rangers when in the field

protection: large areas of new National Parks are unlikely, but appropriate community controlled projects can be successful (e.g. Mbaiwa and Stronza, 2011).

2.2. Poor public relations

Poor relationships with both immediate neighbouring communities and the wider public can cause serious problems for conservation. Poor relationships with immediate neighbours can often be traced back to the exclusion of people from traditional lands when strict protected areas were gazetted (e.g. around Kruger NP), and results in hostility towards the protected area and wildlife in general, leading to increases in illegal activity and more negative perceptions of people-wildlife conflict (Lynn, 2010). Similarly, poor relationships with the wider public (in particular through making access to protected areas difficult and/or expensive) can lead to conservation being viewed politically as an irrelevance, or of concern only to the rich. This attitude could easily lead to a lack of public support for conservation and ultimately a lack of will to continue protecting ecosystems (Nelson 2008; Reid et al., 2009; Mbaiwa and Stronza 2011).

Recently, work in southern Africa has aimed at improving public relations. School visits are encouraged, entrance fees are reasonable and accommodations available for a variety of income groups (SANParks 2012). It is notable that when ownership of the northernmost portion of Kruger was returned to the Makuleke tribe, they declined to resettle the land, choosing instead to contract the park authority to manage ecotourism (Robins and van der Waal, 2008). Meanwhile, the Ezemvelo KZ-N Wildlife authority has also reduced antagonism to their parks by allowing people to sustainably harvest plant resources (thatching grass, firewood, reeds and medicinal plants) within certain parks, allocating a proportion of gate fees to community projects, through co-management agreements and poverty alleviation projects and education programmes. In Botswana's Okavango Delta, widespread antipathy to wildlife and tourism among neighbouring communities in the 1990s was reversed following implementation of a community-based natural resource management programme with revenues from tourism reaching community members (often through employment opportunities): poaching decreased dramatically and several mammal populations stabilised or increased by 2004 (Mbaiwa and Stronza 2011).

In East Africa the attitude of neighbouring communities towards protected areas differs regionally. Park fees for East African citizens tend to be low, but access requires a vehicle not available to most citizens. Importantly, very little revenue from park fees and relatively little from other tourism sources filters to local communities, which, combined with the costs of wildlife damage to crops, competition for grazing and losses of livestock to predation, means that these communities have little incentive to conserve wildlife in their seasonal ranges outside parks (Reid et al., 2009). Unless the costs of conserving wildlife on community lands are outweighed by financial returns from wildlife, protected areas are destined to become islands in a sea of non-wildlife-related land uses, as in South Africa: conservation must pay (Mangel et al., 1996). Strategies need to be developed to improve relationships and ensure that local communities see wildlife as an economically valuable resource, with lessons learnt from successful projects elsewhere (Nelson 2008; Reid et al., 2009).

2.3. Not having buffer zones of compatible land use

Land surrounding many southern African protected areas is managed without regard for the conservation of the protected area. For South Africa this means that many protected areas cannot be further expanded, despite a desire to do so. In areas where land use in neighbouring areas was serendipitously compatible (e.g. private ranches around Addo and some of Kruger), wildlife has recently benefited from land converted to private nature reserves providing a buffer for Kruger, and recent expansion of Addo. Moreover, we now suspect that some areas surrounding wildlife hotspots previously acted as population sinks, stabilising populations of at least some species within the core area by allowing density dependent dispersal and relieving pressure on core habitats (Dias 1996), an understanding that underpins the establishment of artificial population sinks for white rhino *Ceratotherium simum* within the Hluhulwe-iMfolozi Park (Owen-Smith 1983). Rhinos dispersing to artificial sink areas are captured and exported, a practice that has stabilised the population but would probably be unnecessary were animals free to disperse to buffer areas where natural population sinks may be found (Owen-Smith 1983). Buffer zones also have the beneficial result of reducing edge effects (Woodroffe and Ginsberg, 1998; DeFries et al., 2010) and enabling corridors between strictly protected areas (Caro et al., 2009; Jones et al., 2009).

There are currently few land-use plans in regions surrounding East African protected areas (Campbell et al., 2000). Traditionally, most parks protecting savannah ecosystems have been surrounded by pastoralist communities: a lifestyle that is relatively compatible with wildlife (Western et al., 2009). Increasingly, however, land is converted to crops, causing problems for conservation and pastoralist communities (Newmark 2008). In addition, governments are pushing ahead with establishment of fenced ranches and individual ownership of parcels of land (as opposed to large scale mobile pastoralism: Campbell et al., 2000), which together with attempts to sedentarize pastoralists is inappropriate in buffer zones around parks. A debate is currently under way in Tanzania about how pastoralist and agricultural lifestyles can be reconciled, with a recognition that land use plans need to be developed which satisfy the need of all groups. It would be prudent to ensure that, during this planning phase, buffer zones for protected areas be considered: an area where considerable benefit may be possible for both wildlife and pastoralist populations.

2.4. Failing to conserve migratory and dispersal corridors

Southern Africa formerly had seasonal or sporadic mass migrations of ungulates, most spectacularly of huge numbers of springbok (*Antidorcas marsupialis*) moving between the north-western Karoo and Namaqualand (Skinner 1993; Roche 2004). Black wildebeest (*Connochaetes gnou*) and blesbok (*Damaliscus pygargus*) probably migrated seasonally between the western Free State plains and the Maluti mountains adjoining Lesotho. In Botswana's central Kalahari, a wildebeest population of c. 200,000 animals collapsed when their movement was blocked by a veterinary cordon fence during extreme drought conditions (Spinage 1992; Fynn and Bonyongo 2011). Today, few significant migrations survive in southern Africa: in Kruger, a few thousand blue wildebeest moving c. 80 km persist from what might have been more substantial movements westwards (Whyte and Joubert 1988; Fynn and Bonyongo 2011). Only in northern Botswana, where no fences exist, do substantial herbivore movements persist (Chase and Griffin 2003; Cushman et al., 2010; Bartlam-Brooks et al., 2011). In addition to intrinsic value, migrations have important influences on vegetation dynamics that cannot be maintained by resident populations, with year-round grazing and the resultant suppression of fires commonly leading to woody plant invasion (Hudak 1999; van Auken 2000).

In East Africa, migratory movements still persist though several are depleted and all extend beyond protected areas (Newmark 2008; Harris et al., 2009). In Tanzania, for example, there is still thought to be movement of individual elephants between all the major populations, though many routes used are highly threatened, primarily by increased agriculture and human settlements (Jones et al., 2009; Mduma et al., 2011). Meanwhile, the former migration of wildebeest between Nairobi NP and the Athi-Kaputei plains to the south has effectively collapsed, largely as a result of land use changes in the dispersal area (Ogutu et al., 2011). However, large mammals in particular may rapidly resume use of a reopened corridor even several years after it was initially closed off (Bartlam-Brooks et al., 2011). This lesson is obviously related to lesson one (wrong boundaries), but is treated separately here because migration corridors are often relatively narrow and used by animals only briefly (Jones et al., 2009). This means effective conservation does not necessarily require strict preservation within formal protected areas: animals can and do migrate through transformed land if their path is not physically blocked (Mduma et al., 2011). Seasonal management of activities and prevention of physical barriers may be sufficient to adequately maintain corridors without creating formal protected areas, in contrast to the problem identified for primary wildlife areas in lesson one.

2.5. Inadequately protecting river catchments

Park boundaries often exclude the upper catchments of the rivers flowing through them, and hence are vulnerable to the consequences of land use upstream of their boundaries altering flow and water quality. For example, in South Africa, the Black Umfolozi River became shallower following sediment accumulation during the 1960s (Vincent 1970), and lost most riparian woodland following a flood in 1984, whilst abstraction, inappropriate damming and pollution from mining and agricultural activities upstream has compromised the health of a number of rivers. Poor water quality is considered a major contributing factor to the mass die off of crocodiles in the Oliphant's river, Kruger NP (Ashton 2010).

East Africa is already experiencing similar problems, as highlighted by the extinction in the wild of the Kihansi Spray Toad *Nectophrynoides asperginis* following upstream damming of the Kihansi river (Channing et al., 2006), whilst the future of Nakuru NP is threatened by land use changes around Nakuru affecting the lake hydrology (Ogutu et al., in press). Meanwhile, Kenya has started work to protect the Mau escarpment forest following concern that perennial water flow in the Mara River has declined. Deforestation and unregulated water extraction for irrigation have resulted in a loss of water for wildlife and riparian vegetation in dry periods. Flooding events have increased, resulting in increased erosion and widening of the river in northern Serengeti and the Maasai Mara (Sinclair et al., 2008; Mango et al., 2011; Mati et al., 2008). Another positive example is the expansion of Tanzania's Ruaha NP in 2008 to include the Usangu Flats of the upper Ruaha drainage basin. Adequate protection and management of watershed components is a priority for both conservation and humanitarian reasons. If protected area boundaries are adjusted, water catchments must be a key consideration.

2.6. Inadequate law enforcement

To many, South African NPs epitomize 'fortress conservation' but while boundary fences, thorough patrolling and rapid responses have restricted subsistence hunting, these measures have proved ineffective for tackling commercial hunting for high value commodities involving poachers from far afield with helicopters and sophisticated technology. Military-type operations can temporarily inhibit these incursions, but ultimately do not prevent them from recurring (>300 rhino were poached in South Africa each year since 2010). The ecological impacts of uncontrolled hunting can be greater than the direct loss of animals, with cascading impacts on vegetation potentially affecting the ecosystem further (Waldram et al., 2008). For example, only following recent increases in white rhino *C. simum* populations in Hluhluwe-iMfolozi and Kruger have the ecosystem impacts of this species been understood: at sufficiently high density these animals create grazing lawns, adding considerably to the functional heterogeneity of the ecosystems with wider biodiversity benefits (Waldram et al., 2008). Faced with such challenges, political pressure to soften law enforcement and impose low penalties (Byrne et al., 2009) in response to human rights concerns was considered ill-advised, though to successfully restrict illegal hunting, law enforcement needs to be backed by effective partnerships with local communities (lesson 2 above). This is most clearly demonstrated in Namibia, where poaching of elephants and rhinos outside protected areas was almost eliminated initially by the employment of community game guards, and ultimately through the establishment of conservancies controlled by representative local councils (Owen-Smith 2011).

In East Africa, poaching remains a major concern both inside and outside the protected area network. When law enforcement declined in Tanzania from 1977 to 1993, populations of resident wildlife in northern Serengeti declined dramatically, with annual mortality rates of 58% for black rhino *Diceros bicornis*, 30% for African elephant *Loxodonta africana* and 15% for African buffalo *Syncerus caffer* (Hilborn et al., 2006; Metzger et al., 2010). Recent improvements in financing of Serengeti NP have enabled enforcement efforts to be increased, and whilst poaching is still a concern in this area, wildlife populations have stabilised and some are now increasing (Hilborn et al., 2006). Laws can be enforced, but only if political will to do so exists and resources and training are provided to rangers.

2.7. Delayed response to invasive plants

If left uncontrolled, invasive plants have the potential to radically transform habitats, change the fire regime, restrict grazing and browsing potential and impede animal movement. Around 180 alien species are established over a total of 10 M ha of South Africa, with e.g. around 30 k ha of Hluhluwe-iMfolozi park infested by *Chromolaena odorata* by 2001 (Howison 2009). *C. odorata* was first identified in 1961 in Hluhluwe-iMfolozi Park (Macdonald and Frame 1988). Despite large-scale control programmes from 1978 to 1982, by 2001 over 30 k ha were infested. Park management and ecologists worked with government to develop and implement a successful alien plant control program, but at a much greater cost then would have been necessary when the issue was first identified (Howison 2009). The long term success of such programs depends on continued support for follow up operations and a proactive early detection rapid response program to tackle emerging alien plants continually threatening the system. The impacts of invasive species are not, however, limited to wildlife as some invasive plants can have devastating impacts on agriculture and human health: the economic cost of alien plants to South Africa alone is estimated at $646 million per year ($5 billion without current control measures: de Lange and van Wilgen 2010). If, however, potential invasive species are identified early, before the infestation is substantial, the costs of removal are significantly lower, especially if appropriate biocontrol agents can be identified (Volchansky et al., 1999). To be effective programmes must be prioritised, with goals identified, plans made, evaluation and adaptive management implemented (Hulme 2006; van Wilgen et al., 2012).

In East Africa some invasive species are already established and will prove costly to eradicate: *Lantana* and *Maesopsis* species are widespread in some forest habitats (Sheil 1994). Other species, such as *Parthenium hysterophorus* are only arriving now, first recorded in Arusha in 2009 and first in Kenya's Masai Mara Reserve in 2010 (pers. obs.). *Chromolaena* has been detected on the periphery of the Serengeti NP (pers. obs.). Ensuring that emerging weeds are immediately recognised and dealt with would minimise potential costs and consequences (Hulme 2006). Adequate control methods (including manual, chemical and biological control where available, plus adequate biosecurity measures at borders) should be identified in advance of likely invasive threats and early detection rapid responses for known emerging weeds should be implemented.

2.8. Inappropriate road planning

Roads, both for game viewing within protected areas, and public roads around (or through) protected areas can cause problems: improvements to a public road through Hluhluwe-iMfolozi Park in 2002 resulted in a well-constructed tarred road, with increasing volumes of traffic and excessive vehicle speeds. Problems associated with the road include human injury (colliding with game), vehicle damage, increased threat of poaching, and animal mortalities, despite numerous speed calming measures. Whilst wild dog are most affected (Woodroffe and Ginsberg, 2002), other species killed include leopard, lion, buffalo, rhino, and numerous antelopes (pers. obs.). Proposals to construct a road through Kruger NP to Mozambique were thwarted by public opposition. Nevertheless, the fencing of numerous public roads west of Kruger has blocked links with private reserves forming the ''Kruger to Canyonlands'' biosphere reserve. Inappropriately placed game viewing roads can also be problematic, impeding hydrological drainage, allowing woody plants to invade seepage zones and acting as firebreaks, making ''natural'' fire management regimes harder to implement, a factor that has contributed to woody thickening in Hluhluwe-iMfoloziPark (Skowno et al., 1998). Tourist roads through Kruger NP and Hluhluwe-iMfolozi Park have acted as major conduits for the arrival of invasive plant species (Foxcroft et al., 2007).

International trunk roads in protected areas are currently a hot topic in East Africa; much debate has centred on a proposed trunk road through Serengeti which could prevent the wildebeest migration reaching its dry season range with a consequent collapse of the population and change of ecosystem function (Holdo et al., 2011; Dobson et al., 2010). New game viewing roads are also being planned to open up remote parts of protected areas to tourism. As decisions on developments are made the lessons from Southern Africa should be considered: design roads to slow traffic and ensure placement is such to minimise unintended impacts. Strict controls on speed and driving times should be enforced.

2.9. Allowing loss of functional heterogeneity

A common theme of recent savannah research has been the recognition that heterogeneity, both spatial and temporal, is critical to the functioning of savannah ecosystems (du Toit et al., 2003). Ecological processes that formerly operated over large spatial scales may no longer be effective within restricted boundaries of protected areas. Management practices in South African protected areas have had the unintended consequence of reducing the effective heterogeneity of ecosystems (Rogers 2003). For example, widespread provision of waterholes in Kruger NP led to an increase in zebra populations, particularly in areas previously far from water. This, in turn, led to an increase in lion predation in the same area, leaving few areas of the park with low predation risk (Owen-Smith and Mills 2006). Thus, spatial variation in predation risk was lost: with consequent declines in populations of predation sensitive ungulates, like roan antelope *Hippotragus equinus*, sable antelope *H. niger* and tsessebe *Damaliscus lunatus* (Harrington et al., 1999; Owen-Smith and Mills 2006). The extensive availability of artificial waterholes also promotes a widened spread of elephant impacts on vegetation, including in areas remote from water that may formerly have escaped impact during the dry season when elephants feed mainly on woody plant parts (Smit and Ferreira 2010). Of critical importance is maintaining heterogeneity of functional wet and dry-season resources which serve to buffer the effects of environmental variability on herbivore population dynamics (Illius and O'Connor 2000; Owen-Smith 2004; Fryxell et al., 2005; Wang et al., 2006). For example, high densities of waterholes in private nature reserves adjacent to Kruger led to the virtual elimination of nearby forage during a severe drought year, and consequent 80–90% mortality of wildebeest and other grazers, whereas in nearby Kruger, with fewer artificial waterholes, mortality was much lower (Walker et al., 1987). Other management strategies also resulted in lower heterogeneity: a tendency toward uniform fire regimes has resulted in a more homogeneous vegetation structure in Kruger NP, with small trees and shrubs expanding at the expense of taller savannah woodlands (van Wilgen et al., 2003). By contrast, patchy fire regimes driven by the interaction between grazing and fire once characterized large-scale grazing ecosystems and favoured high biodiversity (Fuhlendorf et al., 2009) The 900 km^2 Hluhlwe-iMfolozi Park has been unusually resilient to large mammal losses despite its small extent, a feature that seems related to high local heterogeneity in rainfall, topography and soils (Brooks and Macdonald 1983). Such patterns of heterogeneity that influence the functioning of the ecosystem (functional heterogeneity) should be a priority for conservation.

In East Africa functional heterogeneity is poorly understood by park managers, except for the seasonal dependency of migratory populations on spatially separated areas (Fryxell et al., 2005). Meaningful heterogeneity must be identified and monitored if it is not to be lost. In the interior plateau regions where many East African parks are located, the spatial scales of gradients in rainfall, topography and geological substrates are larger than in Hluhluwe-iMfolozi (unpublished data), meaning that a much larger area may be needed to retain similar habitat heterogeneity.

2.10. Inadequate practitioner-science interaction

In Kruger NP comprehensive monitoring of a growing number of ecological aspects has been undertaken since 1978. Large herbivore numbers, distribution and population structure, culled animals, surface water, herbaceous biomass, grass species composition, woody plants and more are now regularly monitored and reported annually. Initially the data were not used for decision making, beyond setting culling quotas for elephant, buffalo and hippo. Annual counts initially exposed the collapse of the roan antelope population during the late 1980s, and subsequent census analysis identified likely causes and solutions, but the declines and solutions were largely dismissed by managers until they became so serious that viability of the population is uncertain (Harrington et al., 1999; McLoughlin and Owen-Smith, 2003; Ogutu and Owen-Smith 2003). The reluctance of managers to take scientific research seriously may have reflected a perceived lack of relevance of many scientists' research agendas and general distrust between the scientific and practitioner communities. Recognising that the failure to respond promptly to observed declines reflected an institutional problem, a programme of adaptive management based on ''thresholds of concern'' was developed and implemented to ensure active cooperation between the two communities (van Wilgen and Biggs 2011). Relationships between managers and scientists improved and much of the monitoring in Kruger is now undertaken by section rangers and their staff. In Hluhluwe–iMfolozi Park this issue was addressed though the development of a processed based management paradigm entrenched within the management plan.

Research and monitoring within East African protected areas varies greatly regionally. Relationships between those involved in monitoring and park management authorities are not always strong, especially in areas where research programmes are focused on questions perceived as too esoteric to assist managers, even in parks where monitoring is developing. Consequently, relevant expertise of researchers is frequently ignored when developing management plans. Undertaking baseline biodiversity surveys must still be considered a research priority for many protected areas, and protocols for feeding questions to researchers and results to management should be developed.

3. Discussion and conclusions

Our workshop identified ten critical lessons concerning the conservation of functional ecosystems that can be learnt from the experience of Southern African protected area management. Several lessons (poor public relations, failing to conserve migrations, poaching problems, etc.) are related to one another through the human dimension: the problems would be smaller if local communities valued wildlife more and were more involved in natural resource management decisions. Consequently, the future of wildlife in Africa may well be influenced more by socio-economic issues outside the parks than ecological issues within the parks: there is growing evidence that poverty causes poaching, for example (Nielsen, 2006). In East Africa, many interventions deemed necessary, which in hindsight were mistakes in Southern Africa, can still be avoided if policy makers take the challenges seriously and act soon, before opportunities are lost. There is space for public, private and non-governmental organisations to all assist in protecting East Africa's savannah ecosystems, but there is a crucial need for policy and management decisions to result in concrete and effective action on the ground. Too often are new management plans produced and ignored or not implemented for lack of funding and expertise, though conservation success is directly correlated with basic management activities (Bruner et al., 2001). Developing on the ground capacity and recognising that investment in wildlife conservation is necessary to ensure sustainability is fundamental to success.

Whilst discussions at the workshop focused on African savannah ecosystems, many lessons identified here are relevant for conservation planners wherever protected areas are embedded within a matrix of relatively untransformed landscapes. Similar lessons will be of direct relevance in other savannah ecosystems worldwide where equivalent pressures are being felt (savannah ecosystems cover around 20% of the terrestrial land mass, particularly in Africa, Asia, South America and Australia: Sankaran et al., 2005), but the concepts involved transcend biomes. It is noteworthy that several lessons highlighted here are the same as those identified as important in considering protected area planning during the current period of climate change: larger areas, with topographic heterogeneity and maintenance of connections between protected areas, etc. (Hannah et al., 2005). Addressing these lessons in East Africa whilst there is still time is therefore likely to have multiple benefits for conservation. How long the window of opportunity for action remains open is uncertain, with, e.g., important corridors for elephant, buffalo and lion movement between Udzungwa NP and the Selous Game Reserve having been lost in the last 5 years (Jones et al., 2012). As changes can happen with surprising rapidity, the time for action is now. The formal establishment of Transfrontier Conservation Areas extending across international boundaries, currently being implemented in southern Africa, has much promise for alleviating some of these problems (Hanks 2003).

The actions discussed here, arising from the experiences of southern Africa have been proposed largely without consideration for the current political climate in East Africa. This naïve viewpoint is useful to highlight ideal actions from the perspective of conservation but is, of course, unrealistic. For instance, whilst adjustments to park boundaries are possible and have occurred recently around, e.g., Ruaha NP, large-scale changes in strict protected area networks are unlikely. It is important, therefore, that whilst keeping the ideals identified here in mind, planning and conservation action must aim to work within the bounds of realism. Novel solutions to protecting ecosystems and wildlife inside and outside strict protected areas need to be developed and must involve all sectors of society: perhaps mobile pastoralism should be encouraged in buffer areas with parks taking the lead in encouraging better husbandry (Western et al., 2009), or market-driven payments for ecosystem services may be appropriate (Jack et al., 2008). It seems prudent that the actions of highest priority should be those that maintain future options as long as possible: habitat protection. Even if the current institutional and political situation is such that wildlife will take second place to other goals in much of East Africa, the lessons of southern Africa suggest that these values can change as the perceived value of wildlife increases (income from nature-based tourism in South Africa in 2000 was nearly equal to income from agriculture, forestry and fisheries combined, but was growing more rapidly along with employment opportunities: Scholes and Biggs, 2004). Action now to protect habitats and prevent the closure of future options is likely to be strategic: if there is one lesson to be learned from the southern African experience it is that implementing the suggestions above today could both secure functioning ecosystems for future generations and avoid many, more costly interventions in the future. All these actions, however, require the identification and mapping of important ecological processes and natural capital assets across the East African landscape, with the aim of developing balanced conservation and development opportunity plans, which must be considered a priority for conservation in East Africa.

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