PATTERNS OF WILDLIFE EXPLOITATION IN THE UGALLA ECOSYSTEM OF WESTERN TANZANIA

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DEDICATION

This thesis is dedicated to my uncle - Moses Paulo Msuya - for his love, encouragement and support through the early stages of my life.

ABSTRACT

Unsustainable use of wildlife is a global conservation challenge. Understanding ecosystem specific patterns of wildlife exploitation is key to addressing this challenge. This thesis explores the nature of wildlife exploitation in and around Ugalla Game Reserve in western Tanzania. The reserve is divided into Ugalla east and Ugalla west tourist hunting blocks. First, I assessed the status of wildlife in the hunting blocks. Overall, estimates of wildlife population parameters suggested that Ugalla west was somewhat more exploited than Ugalla east. Second, I looked at the degree to which the hunting blocks experienced illegal wildlife hunting (poaching) and factors behind this. The spatial distribution of poaching signs and household interviews revealed that poaching was widespread, more so in Ugalla west than Ugalla east. Proximity to the reserve encouraged poaching, although bushmeat consumption increased with distance from the reserve. A wide range of bushmeat species was favoured, but the common species were impala *Aepyceros melampus*, dik-dik *Madoqua kirkii* and common duiker *Sylvicapra grimmia*. Availability of alternative sources of animal protein, agricultural production and income had significant influences on poaching. Different forms of poaching were specialist activities largely independent of each other. To address poaching, the main focus of attention has been on creating wildlife management areas (WMAs) along with allowing legal subsistence hunting by the communities around the reserve. Third, I assessed the impact of legal subsistence hunting on the wildlife species, and showed that it is not well managed and wildlife populations are contracting. This leaves WMAs as a potentially viable option for the conservation of Ugalla. Therefore, lastly, I identified and recounted some options for promoting the sustainability of WMAs. This thesis presents the first detailed assessment of wildlife exploitation in Ugalla, thus contributing to the existing body of knowledge on tackling the bushmeat crisis in Africa.

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DECLARATION

The work presented in this thesis is my own except where specifically indicated.

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CHAPTER 1: GENERAL INTRODUCTION

This study explores the nature and extent of wildlife exploitation and the resultant implications for ecology and conservation in western Tanzania. It is acknowledged that wildlife resources in many ecosystems face an increasing array of pressures including over exploitation (Taylor & Dunstone, 1996) and habitat loss (Mills, 2007). Consequently, conservation and ecological sciences explore a broad range of issues relating to aspects of game (or wildlife) hunting and other land uses, local livelihoods, and biology/ecology of exploited species (Prins *et al.*, 2000; Kideghesho, 2006; Coad, 2007; Milner-Gulland & Rowcliffe, 2007). Here below I expand on these factors in detail by first looking at the nature of wildlife exploitation, especially bushmeat hunting, concentrating on African ecosystems. Secondly, I throw some light on bushmeat hunting in Tanzania and western Tanzania in particular.

Wildlife exploitation

Wildlife exploitation involves both non-consumptive and consumptive uses of wildlife resources (Boyle & Samson, 1985; Roth & Merz, 1997; Inamdar *et al.*, 1999). To a large extent non-consumptive use relates to ecotourism and/or game viewing (Boyle & Samson, 1985; Dunstone & O' Sullivan, 1996). It is economically viable, but generally associated with some shortcomings; for example: (1) unauthorised feeding of animals by tourists which alters natural dietary requirements, feeding patterns and overall behaviour of animals; (2) animals becoming habituated to human presence which endangers their survival through increased vulnerability; (3) habitat destruction, fragmentation and wildlife loss through increased human trampling or off-road driving, throwing out food leftovers, bottles, cans and other plastic stuff into the habitat as well as rapid development of environmentally unfriendly tourist facilities such as hotels, lodges and camps sites; and (4) noise pollution – increased stress levels among animals (Boyle & Samson, 1985; McNeilage, 1996; Reynolds & Braithwaite, 2001; Mundia & Murayama, 2009).

Consumptive utilisation relates to bushmeat hunting (Davies & Brown, 2007) and tourist/trophy/sport hunting (Prins *et al.*, 2000). Tourist hunting is a selective form of wildlife offtake, which, through effective management is considered to be economically, ecologically and culturally sound (Caro *et al.*, 1998). Considering the number of visitors, infrastructure required and other environmental issues related to wildlife tourism; tourist hunting is said to be far less destructive to the wildlife habitat than non-consumptive tourism. In addition (when the systems are well managed) the number, size and other biological attributes of the wildlife removed guarantee sustainable utilisation (Damm, 2008). Trophy hunting is an important conservation tool (Caro *et al.*, 1998) since, apart from promoting the economy both nationally and at the local community level, it can substantially pay for conservation (Baldus, 2008). Nevertheless, there are several requirements for a successful hunting industry, but the main ones are: significant reduction of wildlife poaching in the hunted areas (Caro *et al.*,1998; Zeppel, 2006; Grimm, 2008), hunting should generate tangible benefits for local communities, unhunted areas should be connected to hunted area to provide refuge for severely exploited species, regular monitoring to assess impacts of hunting (Grimm, 2008), active involvement of local communities in conservation activities (Zeppel, 2006) and the income generated should be substantially and truthfully directed to the conservation of the hunted areas (Baldus, 2008).

Bushmeat hunting is predominantly hunting for food or protein (Bennett, *et al.*, 2006; Nasi *et al.*, 2008); although it can involve income generation (Brashares, *et al.*, 2004; Kalternborn *et al.*, 2005) on both small and large scale. Large-scale (commercial) bushmeat hunting is characterised by a well organised bushmeat network comprising of hunters, middlemen and end-users (Cowlishaw *et al.*, 2004; Caro & Andimile, 2009). Consumers are normally wealthier people dwelling either immediately near or far from the hunted areas (Caro & Scholte, 2007; Coad, 2007). Hunters and middlemen are motivated by increased profit and not necessarily their own meat consumption (Fa & Garcia-Yuste, 2001; Olupot *et al.*, 2009). Assessing the extent of commercial bushmeat hunting and quantifying its impacts on wildlife is a huge challenge because consumers can hardly be identified and accessed; especially those residing far from wildlife areas (Bowen-Jones *et al.*, 2002; Caro & Andimile, 2009).

Unsustainable subsistence bushmeat hunting is a widespread problem in many ecosystems (Davies & Brown, 2007). It occurs in Africa (Milner-Gulland *et al.*, 2003), Asia (Corlett, 2007; Rao *et al.*, 2010), Central America (Smith, 2008), Australia (Bennett & Whitten, 2003) and South America (Peres & Nascimento, 2006). The problem seems more common in African ecosystems than elsewhere. Bennett, *et al.* (2006) defined bushmeat as "an African term that includes all wildlife species used for food, from cane rats to elephants". Much attention is paid to central/western Africa, where it is believed to have reached a crisis level and the wildlife populations cannot sustainably support current levels of offtake (Noss, 1998; Oates *et al.*, 2000; Milner-Gulland *et al.*, 2003; Wright & Priston, 2010).

In eastern and southern Africa, bushmeat hunting is steadily becoming a veritable minefield of conservation and ecological concerns (Baldus, 2002; Fusari & Carpaneto, 2006; Lindsey *et al.*, 2011). In east Africa, getting a grip on the problem from a research perspective is usually associated with some challenges and controversy, largely stemming from its secretive nature and poor cooperation from hunters. Thus a combination of approaches are used; for instance, interviews with local communities, field observations, dietary recalls, making use of local hunting and law enforcement records and assessing the status of wildlife populations (FitzGibbon *et al.*, 1996; Ndibalema & Songorwa, 2007; Caro, 2008; Knapp *et al.*, 2010; Wilfred & MacColl, 2010).

Bushmeat hunting can be carried out legally or illegally depending on wildlife laws. Hunting activities taking place against conservation laws and policies of a particular protected area are commonly referred to as wildlife poaching (Hofer *et al.*, 1996; Holmern *et al.*, 2007). For example, subsistence bushmeat hunting in the national parks (IUCN category II) and game reserves (IUCN category IV) can be described as a poaching activity as most wildlife laws governing these areas do not support it (Majamba, 2001). Since such strictly protected areas are richer in wildlife (Stoner *et al.*, 2007), they will certainly continue to attract the attention of illegal hunters (Weladji & Tchamba, 2003).

Impacts of consumptive wildlife use

Consumptive utilisation is not without its undesirable effects on wildlife populations, but tourist hunting may have less of an impact on prey populations than uncontrolled bushmeat hunting (Caro *et al.*, 1998). Tourist hunting is selective according to species, density, sex and age of the animals to remove (Coltman *et al.*, 2003). Its selective nature is said to induce a biased sex ratio among individuals of some ungulate species such as impala (Setsaas *et al.*,

2007) and saiga antelope *Saiga tatarica tatarica* (Milner-Gulland, *et al*., 2003). This, as a result, affects species productivity and overall populations performance (Milner *et al*., 2006). It also targets older individuals (Packer *et al*., 2010) by looking at the sizes of the trophies such as horn length, skull length and body length. But, for other species reliance on trophy sizes may lead to accidentally removal of younger animals as described for buffalo *Syncerus Caffer* (Taylor, 2007) and bighorn rams *Ovis canadensis* (Festa-Bianchet *et al*., 2004). There are also those individuals of different species which do not meet the specified quarry attributes, yet are deliberately (in fact, illegally) shot by tourists and professional hunters; not to mention species removed over and above their quotas (Caro & Andimile, 2009).

The impacts of bushmeat hunting (whether legal or illegal) are substantial (Bennett, *et al*., 2006; Nyahongo *et al*., 2009). Illegal bushmeat hunting (or wildlife poaching) is the most problematic form of consumptive utilisation (Milner-Gulland & Akçakaya, 2001). It can be unsustainable since it can cause rapid population decline to very low numbers (Ginsberg & Milner-Gulland, 1994; Coad, 2007; Caro, 2008). Unfortunately, all types of protected areas are vulnerable to wildlife poaching (Arcese *et al*., 1995; Newmark, 2008). The magnitude of poaching differs depending on the effectiveness of the anti-poaching measures (Hilborn *et al*., 2006). Areas adjacent to strictly protected areas, which are also used for tourist hunting, are the most heavily poached (Wittemyer *et al*., 2008) because of the poor law enforcement (Holmern *et al*., 2007) and human population pressures. Caro *et al.* (1998) argued that the combined impact of trophy hunting and poaching on wildlife populations is considerably larger than the perceived impact of a single form of hunting. Thus dealing with bushmeat has become an unprecedentedly important element in the field of conservation (Sinclair & Arcese, 1995; Taylor & Dunstone, 1996; Blom *et al*., 2005; Fa *et al*., 2006; Rist *et al*., 2008; Desbiez *et al*., 2011).

Conservation approaches

Generally, conservation has relied on the creation of protected areas (Aspinall, 1996; Kideghesho, 2006; Rutagarama & Martin, 2006; Kaimowitz & Sheil, 2007). The International Union for Conservation of Nature (and Natural Resources) (IUCN) defined a protected area as "a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values". A protected area may either be "terrestrial", "inland water" or "marine" (Dudley, 2008). The primary objective of establishing a protected area is to conserve biodiversity (Eichbaum *et al.*, 1996; Abuzinada, 2003) and provide economic, cultural and aesthetic values to human beings (Margules & Pressey, 2000).

The increase in human populations presents one of the major challenges in the management of protected areas (Jachmann, 2008*a*). The greatest worry is considered to be the rapidly increasing human populations and their settlements near protected areas (Hofer *et al.*, 1996; Wittemyer *et al.*, 2008). Although Joppa *et al.* (2009) questioned the claim that there are overwhelming increases in the human populations near protected areas, it is generally accepted that human pressure on the edges of protected areas is massive. In terrestrial protected areas, human pressure is mainly through the use of natural resources (such as forest and wildlife) and habitat devastation (Linkie *et al.*, 2003); influenced to a greater extent by the factors explained in the following paragraphs.

Agriculture: This is one of the major issues in the effort to curb escalating pressures on protected areas. For example, in Serengeti there has been a close relationship between agricultural production and wildlife poaching (Barret & Arcese, 1998). Johannesen (2005) found that an increase in the sizes of some food and commercial crop farms lessened illegal hunting in Serengeti, but warned that livestock predation and crop raiding by wild animals could perpetuate poaching in the area. A study of bushmeat and food security in the Congo Basin also acknowledged the role of agriculture in halting wildlife exploitation intensity (Fa *et al.*, 2003). On the other hand, agriculture is said to alter wildlife habitats (Laurance, 2008). Farm encroachments in the areas around Kilombero Game Controlled Area in Tanzania is a good example (Haule *et al.*, 2002). Unsustainable agriculture is one of the land use activities jeopardizing the health of the wildlife habitat in Europe (Stavrinidis & Anayiotos, 2006). Mechanized agriculture coupled with high usage of chemical fertilizers has been responsible for wildlife habitat degradation in both Africa (Kideghesho *et al.*, 2006) and Europe (Young *et al.*, 2005). Adoption of low-input agriculture, characterised by good management of chemical fertilizers (Weinberg, 1990; Mkpado & Onuoha, 2008) and firm integration of indigenous knowledge (Mkpado & Onuoha, 2008), would be the best way forward for improving agricultural yield and conserving habitats in the areas of conservation importance. Low-input agricultural systems are an important ingredient of eco-agriculture. Eco-agriculture can be defined as "a fully integrated approach to agriculture, conservation and rural livelihoods, within a landscape or ecosystem context" (Scherr & McNeely, 2008). But, promoting eco-agriculture, especially in Africa, is a function of agricultural extension and education (McNeely & Scherr, 2001). Unfortunately, extension services in Africa are confronted with a number of challenges including few and inexperienced extension officers, financial constraints, and lack of incentives to do extension work (Gebremedhin *et al.*, 2006); therefore, appropriate interventions are needed (Omamo *et al.*, 2002).

Animal protein: The conservation literature highlights the importance of alternatives to bushmeat in reducing wildlife poaching (Wilkie & Carpenter, 1999; Fa *et al.*, 2000; Mbete *et al.*, 2011). The often mentioned alternatives are fish and other livestock (Brashares *et al.*, 2004; Rowcliffe *et al.*, 2005; Ndibalema & Songorwa, 2007), provided any challenges or problems facing livestock keeping are adequately addressed (Brashares, 2004; Rowcliffe, 2005). Such challenges may differ from locality to locality; bringing about some variations in the livestock species accepted as viable alternatives to the bushmeat problem. For example, in northern Cameroon, domestication of guineafowl is recommended among the options for reducing bushmeat hunting (Njiforti, 1996). Poultry-keeping and fish farming are important activities for meeting animal protein demands in Brazzaville, the Republic of the Congo (Mbete *et al.*, 2011). Feral pig *Sus scrofa* is a potentially significant livestock species in reducing pressure on wildlife in the Brazilian Pantanal (Desbiez *et al.*, 2011). Therefore, the role of livestock as a source of animal protein alternative to bushmeat should be assessed on an ecosystem basis.

A regulated local hunting of some wildlife species can also be used as a supplemental source of animal protein. This has been the case in the areas adjacent to state-protected areas in Tanzania (URT, 1974). In west Africa, local hunter associations have been useful institutions through which subsistence hunting takes place (Bassett, 2005). All the same, sustainability of any legal subsistence hunting is a paramount ingredient for successful conservation (Baldus & Caudwell, 2004).

Income sources: Rural economies in Africa are becoming more diverse (Smith *et al*., 2001) with a range of income sources such as crop farming, livestock keeping, formal employment, small business and natural resources (Mooko, 2005; Narain *et al*., 2005; Carletto *et al*., 2007). Of these, agriculture (crop farming and livestock keeping) remains a predominant livelihood activity (Yaro, 2006). In a situation where agriculture provides low economic returns, natural resources become the second main source of income (Butler, 2006; TNRF, 2008). For example, forest offers timber, fuel wood, construction poles, medicinal plants and other non-wood products such as honey (Sunderlin *et al*., 2005; Giliba *et al*., 2010). Wildlife is also valued as a source of household income (Loibooki *et al*., 2002; Coad, 2007; Willcox & Nambu, 2007). As living standards rise, demand for wildlife as a source of livelihoods decimates populations of different species (Caro & Scholte, 2007), and conservationists put much emphasis on promoting non-wildlife sources of livelihoods (Johannesen, 2005; Olupot *et al*., 2009; Mfunda & Røskaft, 2010; Wright & Priston, 2010). Nevertheless, we need to understand the relative importance of such income sources and how they influence wildlife poaching or bushmeat hunting. This would, undoubtedly, help us to identify key income sources for communities around protected areas and focus our efforts towards promoting these.

Law enforcement: Law enforcement has become a common approach to ensure adherence to conservation norms (Forsyth, 2008). The aim is to bring law breakers to justice and deter illegal activities (Milner-Gulland & Rowcliffe, 2007; Jachmann, 2008*b*). The idea of law enforcement bears close relationship with the American Yellowstone model also known as "fortress conservation" or "fences and fines" conservation approach (Norgrove & Hulme, 2006) that despises natural resources related needs and interests of people, particularly those near conservation areas (Pimbert & Pretty, 1995). While some conservationists defend fortress conservation and the top-down approach to wildlife law enforcement (for example, Fischer, 2008), the other school of thought emphasises a more participatory law enforcement where local communities are actively involved in the protection of buffer zones, wildlife corridors, game controlled areas, open areas and other lower category protected areas (hereinafter collectively referred to as partially protected areas) (Mesterton-Gibbons & Milner-Gulland, 1998; Baldus *et al*., 2003; Kafle & Balla, 2005). Since resources (financial resources, trained personnel etc.) for law enforcement are always scarce (Hilborn *et al*., 2006), monitoring and community based conservation are the preferred supplements to traditional anti-poaching measures (Songorwa, 1999; Kaimowitz & Sheil, 2007; Milner-Gulland & Rowcliffe, 2007).

Monitoring: As a means of understanding wildlife exploitation impacts, monitoring seems to be indirectly linked with conservation. It encompasses a range of activities or tasks to understand how, why and to what degree wildlife populations are influenced by their habitats and human actions. It also offers recommendations on priority areas for anti-poaching survey or law enforcement, and conservation (see Kahindi *et al.*, 2009). Depending on the objective, a monitoring exercise may involve several activities such as distance sampling techniques, namely, line transects (Thomas *et al.*, 2007) and point counts (Buckland, 2001).

Monitoring the exploitation intensity or status, of say small mammals, can make use of various traps; for example, box and pitfall traps (Barnett & Dutton, 1995). Indirect approaches such as the use of indices can also be helpful in the monitoring of exploited species (Witmer, 2005). Indices are often used in assessing the spatial and temporal extent of wildlife exploitation. For example, Reyna-Hurtado & Tanner (2007) made use of tracks when estimating abundance of ungulate species in hunted and non-hunted sites in Calakmul Forest, southern Mexico. Wright *et al.* (2000) used parameters like poachers' tracks, poacher sightings, shot-gun shells and poachers' camps, in assessing the impact of poaching on the abundance of some mammal species in the Neotropical forests. In Serengeti, Setsaas *et al.* (2007) used flight initiation distances, age and sex composition, and vigilance when assessing the impact of exploitation on impala populations. The use of harvest record is also not uncommon (Msoffe *et al.*, 2007; Kahindi *et al.*, 2009; Jachmann, 2008*b*). It is an informative index that can help in examining trends in the status of the exploited species (Milner-Gulland & Rowcliffe, 2007).

Participatory conservation: This aims at creating or raising conservation awareness amongst local communities (Wilfred *et al.*, 2007); which is significant in promoting wildlife as a valuable land resource (Emerton & Mfunda, 1999). The strategy was adopted to address problems associated with local resentment towards conservation triggered by the isolation of people from the very natural resources on which they depend (Songorwa, 1999; Chatty & Colchester, 2002). Examples of human-conservation conflicts include poor relationship between communities and the conservation of Machalilla National Park in Ecuador (Fiallo & Jacobson, 1995). A number of human-wildlife conflicts in the Serengeti ecosystem have also been highlighted by Kideghesho (2006). The relocation of people from Dwesa and Cwebe Nature Reserves in South Africa in the 1920s and 1930s not only created negative attitudes towards conservation but also resulted in the accelerated loss of species and their habitats (Fabricius & de Wet, 2002). Many countries, especially those in Africa, have instigated different participatory conservation projects (community-based wildlife management projects) in which communities around protected areas are important stakeholders (Songorwa, 1999; Wilfred, 2010). Among the often cited examples is the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe. The project ensures sustainable use of wildlife resources while improving people's livelihoods. Illegal killings of elephant and other wildlife species have been substantially reduced; because through the realisation of tangible benefits, as a result of CAMPFIRE projects, a majority of local communities have appeared to support anti-poaching activities (see Child, 1996). To reverse the trend of wildlife populations' declines mainly through poaching and loss of habitats, the government of Namibia initiated participatory conservation projects called "conservancies" in 1996 (Weaver & Skyer, 2003). These are "legally recognized, geographically defined areas that have been formed by communities who have united to manage and benefit from wildlife and other natural resources" (Weaver & Petersen, 2008). Therefore, the management of wildlife utilisation activities in conservancies is brought down to the grassroots level, with tangible benefits trickling down to local communities (Weaver & Petersen, 2008). Another good example of participatory conservation is the Community Conservation for Uganda Wildlife Authority Project (CCUWA) in Uganda. CCUWA is actively involved in the community development projects such as those related to health and educational services. It has been effective in Lake Mburo National Park where neighbouring communities realise conservation benefits and their support for conservation has increased as a result (Emerton, 1999). In Tanzania, the contemporary approach to participatory wildlife conservation has been the establishment of Wildlife Management Areas (WMAs). Although there are some challenges in their administration, some of them; for example, Ipole and Uyumbu WMAs in Tabora Region, have been somewhat successful. The WMAs offer a potentially very useful platform for addressing people's wildlife-based livelihood needs while ensuring sustainable conservation of wildlife resources (IRA, 2007; Nelson, 2007; Wilfred, 2010).

Community-based wildlife management (CWM) is normally practiced in partially protected areas (Gaidet *et al.*, 2003). Since most of the partially protected areas are linked to core protected areas (Dudley, 2008), their management does stabilize populations of different species through connectivity conservation (Bennett & Mulongoy, 2006). Regrettably, in many countries CWM is heavily dependent on donors (DeGeorges & Reilly, 2009; Hoole, 2010). This has some significant disadvantages. Firstly, it is a great way of attracting people near protected areas at the expense of wildlife and their habitats (Wittemyer *et al.*, 2008). Secondly, donor aids may be perceived as normal humanitarian or development aids that have nothing to do with conservation efforts. Nelson (2000) noted this in some Maasai lands in northern Tanzania where a growing dependence on donor aids has hindered the recognition of the economic value and livelihood contributions of wildlife resources amongst local people. Consequently, the communities seldom support wildlife conservation and in most cases they are neither concerned with poaching activities nor are they bothered by the presence of poachers in their villages. Thirdly, when donors pull out the survival of the CWM projects becomes questionable (DeGeorges & Reilly, 2009). On balance, to ensure long-term sustainability of CWM projects, it is important that we rigorously understand hopes and fears associated with their envisaged objectives and be able to develop workable recommendations (Songorwa, 1999; Nelson, 2007; Wilfred, 2010).

Species conservation: Wild animals live in home ranges of varying sizes (Knight *et al.*, 2009), and the majority of the time they are found outside core protected areas (Thirgood *et al.*, 2004). Some of the factors determining species home ranges include the quality and quantity of food resources, reproductive characteristics (van Beest *et al.*, 2011) and species' migration patterns (Boone *et al.*, 2006; Mpanduji & Ngomello, 2007). Since human land use changes around wildlife areas occur concomitantly with habitat manipulation (Mundia & Murayama, 2009), different species in and outside core protected areas become increasingly vulnerable (Newmark, 2008). However, severity of vulnerabilities may differ among species depending on the degree of the species-specific connectivity between habitats (Al-jabber, 2003). Species connectivity in turn depends on the availability of species-specific habitat requirements (Merriam, 1991; Taylor *et al.*, 1993), and the pattern of the habitat isolates on a landscape level (Crooks & Sanjayan, 2006).

The nature of connectivity between two wildlife sites with different exploitation intensities or protection status influences source-sink system (Bennett, 1999) where animals move to and fro between sources (non-hunted or slightly hunted areas) and sinks (commonly hunted areas) (Novaro *et al.*, 2005; Naranjo & Bodmer, 2007). Therefore, what happens to a species in the sink, as a result of exploitation, can be the best indicator of the status of the same species in the source. For example, species offtake trend in the hunted areas adjacent to non-hunted or slightly hunted areas can act as a proxy indicator of its population performance in the latter.

Conservation & illegal game hunting in Tanzania

The conservation of Tanzania's wildlife resources can be traced as far back as the 1800s. Since then, fortress conservation, dominated by the creation of core protected areas alongside the relocation of the people living in them, has been the main approach ensuring that our and subsequent generations benefit from wildlife (Chatty & Colchester, 2002; Kideghesho, 2006). Now, Tanzania is among the African countries rich in protected areas (Severe, 2003). Twenty eight percent of about 900,000 sq. km. of land area of the Tanzanian mainland is occupied by protected areas set aside for wildlife conservation. The network of protected areas includes the Ngorongoro Conservation Area (1% of the total area under protection), 15 national parks (4%), 33 game reserves (15%) and 43 partially protected areas (10%) (Leader-Williams, 2000; Shivji, 2002, see also Fig. 1.1). The national parks are the areas rich in biodiversity and contain high quality wildlife habitats. They are aimed at preserving Tanzania's rich natural heritage and conserving representative habitats and wildlife resources. Consumptive use is strictly prohibited; the only activities permitted are non-consumptive tourism, education and research. Wildlife conservation in the national parks is administered by the Tanzania National Parks Authority (TANAPA). Game reserves on the other hand, constitute the largest proportion of the land under conservation. The main activity in the game reserves is trophy hunting although non-consumptive tourism, research and education are also encouraged. Their management is administered by the Wildlife Division of Tanzania. Ngorongoro Conservation Area, which borders Serengeti national park to the north and west, is a United Nations Educational, Scientific and Cultural Organisation (UNESCO) World Heritage Site. It was established as a pilot project for integrated land use, encompassing activities such as pastoralism, wildlife conservation, photographic tourism, research and education. Ngorongoro is managed by the Ngorongoro Conservation Area Authority. The partially protected areas provide a buffer zone for core protected areas. They contain more land use activities than any other category of protected areas (Shivji, 2002). Apart from having all types of land-uses present in other protected areas, legal subsistence hunting, fishing, beekeeping as well as restricted human settlements may also be permissible in the partially protected areas.

In the 1990s Tanzania's wildlife, particularly in the partially protected areas, began to disappear at a worrying rate because of poaching. The killing of elephants for ivory and rhinos for their horns became increasingly out of control not to mention hunting of countless other animals for both food and the bushmeat trade (WSRTF, 1995). In the mid 1990s the government started to emphasise the need for making people an integral part of the conservation issues. This was done to try to address people's needs for food, shelter, water, energy and income through conservation while also aiming to make them aware of the importance of wild animals and their habitats (Songorwa, 1999; Nelson, 2007). To match their words with action, the government, in collaboration with some non-governmental organisations (NGOs), went to great lengths to initiate WMAs. These are a new category of protected areas gradually replacing partially protected areas where wildlife comes into contact with farmers, pastoralists, and other rural dwellers (URT, 1998*a*). There are three key WMA stakeholders: the government, local/village governments and NGOs. It is envisaged that through WMAs local communities can become formally empowered to manage and benefit from local wildlife and other resources. The initiative has gained wide popularity in the country with about 16 WMAs established in the past 10 years and hopefully many more will follow in the near future (Nelson, 2006; Wilfred, 2010). The future survival of wildlife in Tanzania will apparently rely on the effective administration of the WMAs (URT, 1998*b*).

Like other countries, Tanzania is, at the moment, struggling to deal with a burgeoning problem of illegal bushmeat hunting (Baldus, 2002; Caro & Andimile, 2009). While illegal hunting is extensive, wildlife law enforcement is more effective in the national parks than the game reserves, and far less effective in the partially protected areas (Holmern *et al.*, 2007; Stoner *et al.*, 2007). Inadequate human and financial resources cripple anti-poaching efforts in the partially protected areas (Caro, 1999*a*; Holmern *et al.*, 2007). Thus, the country is losing more wildlife to poaching than might be conceived (Caro & Andimile, 2009). The extent of the problem is not well known across the country. To date, the amount of research on bushmeat is still relatively small, and the majority of it (80%) is done in the northern part of the country especially in the Serengeti ecosystem (70%) (Table 1.1). A small number of studies (12%) has been conducted in other parts of the country; for example, south-central Tanzania. Only 6% of the studies in Table 1.1 has been conducted in western Tanzania. This is regrettable since capturing unbiased information on illegal consumption of wildlife resources in Tanzania is crucial in order to come up with practical recommendations for dealing with the problem across different parts of the country.

A wide range of reasons as well as solutions to the bushmeat hunting problem have been put forward (Table 1.1). Generally, need for animal protein and income is considered to be the key driver of the problem. However, we should know that bushmeat hunting does not take place in a vacuum. There are all sorts of factors to consider which can enable us to address the problem comprehensively. For example, we need to understand the place of legal subsistence hunting and other poaching types such as illegal timber harvesting, fishing etc. in the bushmeat system. Issues like anti-poaching efforts and wildlife status can also determine patterns and intensity of bushmeat hunting.

Western Tanzania: an important bushmeat hotspot

Western Tanzania (Fig. 1.1) is an area dominated by miombo woodlands (Abdallah & Monela, 2007), containing species like *Brachystegia spiciformis*, *B. microphylla*, *B. bussei*, *Isoberlinia globiflora*, *Acacia kirkii*, *Cassia abbreviata*, *Burkea africana*, *Cymbopogon giganteus*, *Julbernadia globiflora*, *Grewia bicolor*, *Ozoroa reticulata*, *Sesbania sesban*, *Ximenia Americana* and *Pterocarpus angolensis*. The area has a diverse range of wildlife species including impala *Aepyceros melampus*, hippopotamus *Hippopotamus amphibius*, roan antelope *Hippotragus equinus*, nile crocodile *Crocodylus niloticus*, African elephant *Loxodonta africana*, bohor reedbuck *Redunca redunca*, oribi *Ourebia ourebi*, topi *Damaliscus korrigum*, waterbuck *Kobus defassa*, common warthog *Phacochoerus africanus* and African wild dog *Lycaon pictus*. Common bird species in the area are helmeted guineafowl *Numida meleagris*, southern ground-hornbill *Bucorvus cafer*, Egyptian goose *Alopochen aegyptiaca*, marabou stork *Leptoptilos crumeniferus*, ostrich *Struthio camelus*, shoebill *Balaeniceps rex*, spur-winged goose *Plectropterus gambensis*, African fish-eagle *Haliaeetus vocifer* and white-backed vulture *Gyps africanus* (Thomas, 1961; URT, 1998*a*, UGR, 2006). Much of the wildlife is now found within the protected areas, namely, Ugalla Game Reserve (UGR, 2006), Katavi National Park and Rukwa-Lukwati Game Reserve (Caro, 1999*b*; Waltert *et al.*, 2008). Wildlife outside the protected areas is found in the partially protected areas (Caro, 1999*a*; Waltert, *et al.*, 2008); which, on the other hand, provide a buffer to protected areas. It is within the partially protected areas where a mixture of conservation measures takes place. This involves legal subsistence hunting, fishing, beekeeping and extraction of fuel wood and building poles (URT, 1998*a*; UGR, 2006). There is currently effort to replace partially protected areas with WMAs. Two WMAs, Uyumbu and Ipole, are in an advanced stage with the facilitation of the international organisation called Africare (Nelson, 2007). The WMAs would provide a venue for- and empower the local communities to administer the management and utilisation of the natural resources in the areas near the protected areas.

Western Tanzania is one of the problematic bushmeat hunting sites in the country (Caro, 2008; Wilfred & MacColl, 2010). In fact, game hunting existed in the area even before the colonial era (Roberts, 1968), and has thus become part of the people's subsistence activities (URT, 1998*a*, Figs. 1.2 & 1.3). At present the most worrying thing is a rapid increase in the human population density (see NBS, 2002) coupled with an intensified poverty. This tends to push more people into hunting for the bushmeat trade. Some other activities indirectly linked to bushmeat exploitation include tobacco cultivation which, through destruction of habitats, increases the vulnerability of wild animals. The flourishing tobacco market encourages people to encroach partially protected areas with destructive tobacco farms, and also extract large amounts of wood for curing tobacco (Waluye, 1994; URT, 1998*a*; Yanda, 2010); creating not only empty forests but also empty landscapes (Fig. 1.4).

Table 1.1 Selected bushmeat studies carried out across Tanzania.

Author & year of publication	Region research was conducted	Main reason for wildlife exploitation	Proposed general solution
Hoffer *et al.* (1996)	Serengeti, northern Tanzania	Need for protein and income	Effective law enforcement, Awareness creation, Community conservation services, Viable alternatives to bushmeat
Campbell & Loibooki (2000)	Serengeti, northern Tanzania	Need for income	Rural livelihoods improvement
Carpaneto & Fusari (2000)	Urumwa Forest Reserve, western Tanzania	Need for protein and income	Effective wildlife management, Human population monitoring, Further researches on wildlife conservation
Campbell *et al.* (2001)	Serengeti, northern Tanzania	Need for protein and income	Effective poverty alleviation strategies, Diversified sources of cash income
Holmern *et al.* (2002)	Serengeti, northern Tanzania	Need for protein and income	Do away with game cropping activities and encourage other income generating ventures
Loibooki *et al.* (2002)	Serengeti, northern Tanzania	Need for protein and income, Low agricultural yield (crops and livestock)	Agricultural development especially "improving small livestock such as goats and sheep", Effective poverty alleviation strategies
Holmern *et al.* (2004)	Serengeti, northern Tanzania	Need for cash income and other "subsistence needs"	Improved agricultural production, Improved local agricultural based income generating activities
Johannesen (2005)	Serengeti, northern Tanzania	Poor participation in the "Serengeti Regional Conservation Project" activities, Low agricultural production (especially maize and cotton), Crop damage and Livestock predation by wildlife	Enhanced cotton and maize production, Reduced crop and livestock loss to wildlife
Kaltenborn *et al.* (2005)	Serengeti, northern Tanzania	Need for protein and income, For cultural and traditional purposes	Sustainable wildlife based rural development
Nyahongo *et al.* (2005)	Serengeti, northern Tanzania	Travel costs to and from hunting destinations	-
Nielsen (2006)	Udzungwa, south-central Tanzania	Need for protein and income	Improved livestock production for low income families, increased economic openings especially in villages adjacent to forest reserves, Monitored human populations near forest reserves
Jambiya *et al.* (2007)	Refugees camps, north-western Tanzania	Need for protein and income, Sophisticated hunting gears such as guns, Living close to protected areas	Considerations of bushmeat trade legalization, Sustainable legal provision of livestock meat and bushmeat, Viable sources of income

Ndibalema & Songorwa (2007)	Serengeti, northern Tanzania	Preference as a result of Consumers' localities and ethnic backgrounds, Meat taste and price, and Species' availability	Effective conservation of all species with extra attention paid to most preferred ones
Knueppel *et al.* (2009)	Ruaha, central Tanzania	Need for income	Promoting income generating activities
Magige *et al.* (2009)	Serengeti, northern Tanzania	Human pressure on wildlife areas	Awareness creation and Agricultural development
Nyahongo *et al.* (2009)	Serengeti, northern Tanzania	Village location distance from protected areas, Need for protein and income	Promoting income generating activities, and Encouraging sustainable local supply of fish
Mfunda & Røskaft (2010)	Serengeti, northern Tanzania	Need for protein and income	Effective community based conservation, Agricultural development, Effective law enforcement, Livelihood improvement through viable and sustainable conservation benefits

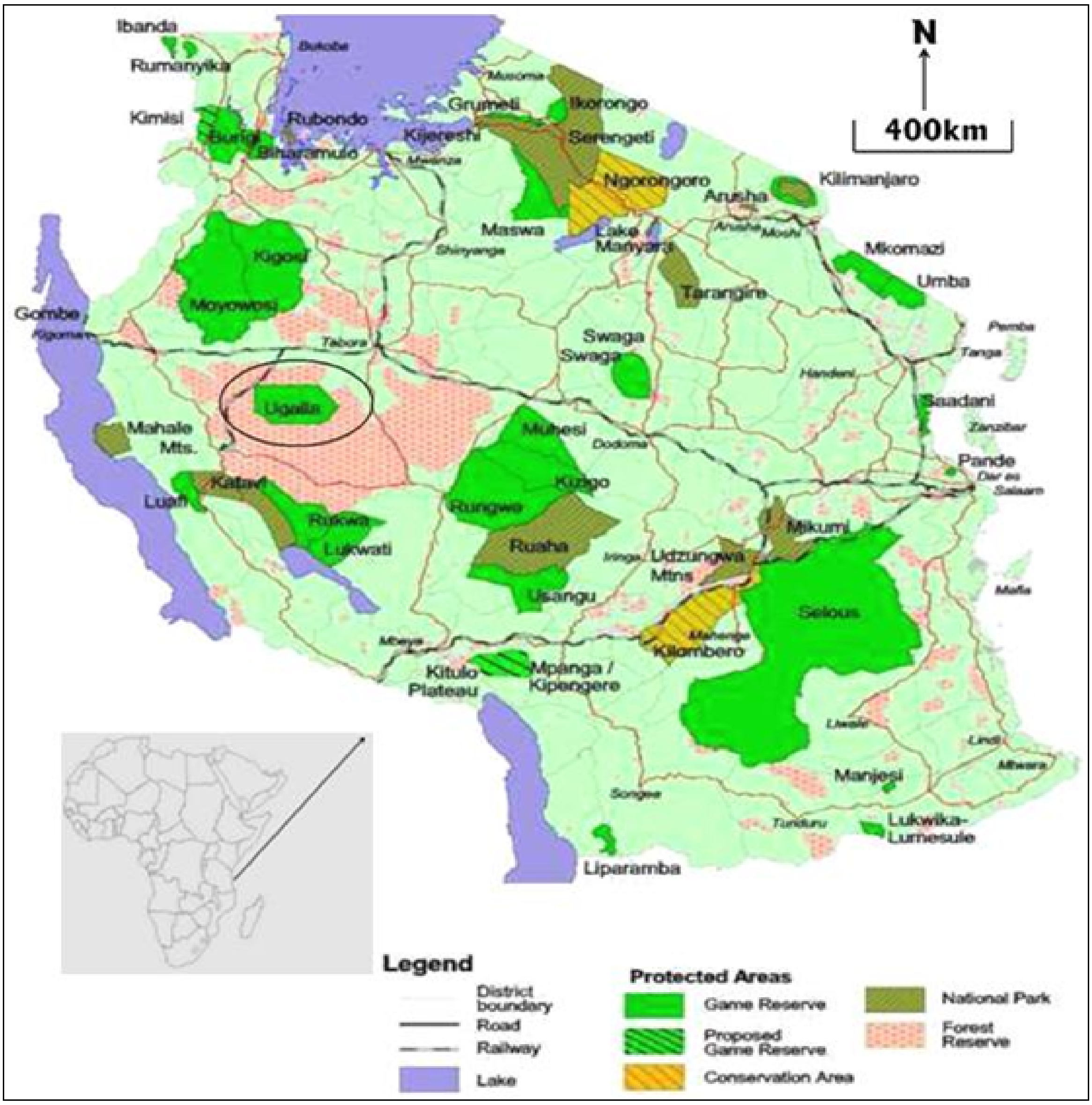

Figure 1.1 Map of Tanzania showing the distribution of protected areas. An inset map of Africa shows the location of Tanzania. Approximate location of the Ugalla ecosystem is encircled. Adapted from URT (2006).

Figure 1.2 Bushmeat on a drying rack in central-western Tanzania. The pile comprised of meat from giraffe, elephant, topi and impala. Photo by Paulo, taken in 2009.

Fig. 1.3 Photograph showing male impala (*Aepyceros melampus)* killed by poachers with pictured muzzle loaders. Next to it is a heap of meat from another impala. Photo by Paulo, taken in 2009 in miombo woodlands, Urambo District, western Tanzania.

Figure 1.4 human settlements and agricultural farms located ad hoc in the partially protected areas of the Ugalla ecosystem. Slash and burn agriculture has been a characteristic in the area to create more space, especially for tobacco farms. Photo by Ugalla Game Reserve, taken in 2009.

The two main ecosystems in western Tanzania, Katavi-Rukwa and Ugalla, are widely separated by a matrix of human settlements and other unsustainable land use activities. The Katavi-Rukwa ecosystem enjoys a stricter protection because of the Katavi National Park (see Fig. 1.1). The national park is immediately adjoined by Rukwa-Lukwati Game Reserve and a number of partially protected areas. A work on conservation and bushmeat is already taking place in this ecosystem (UC-Davis, 2010). The Ugalla ecosystem, on the other hand, suffers lack of a non-hunted source area. Additionally, although Ugalla and Katavi share similar habitat characteristics, and probably plant and animal species, the local communities who are the key source of the bushmeat problem differ in terms of economic and cultural backgrounds. Therefore, recommendations for tackling the bushmeat problem in western Tanzania must be ecosystem-specific.

This study

Here, I present the first study carried out both extensively and intensively to understand the patterns of bushmeat exploitation in and around Ugalla Game Reserve, and explore the ecological implications of these in the context of the present conservation measures. The study puts forth recommendations for effective conservation of Ugalla for present and future generations.

Ugalla Game Reserve is the only key component of the Ugalla ecosystem (URT, 1998*a*). Protection of the reserve started in the 19th century when the Wagalla people were forced to leave the areas around the Ugalla river because of health reasons mainly due to the spread of the sleeping sickness epidemic. In the same vein, the value of the area as an important natural resources management site was gradually realised amongst conservationists. In 1954, the area was gazetted as the Ugalla River Game Controlled Area (Fisher, 2002). This was meant to reduce people's access to natural resources therein.

Owing to their centuries old dependence on natural resources for fishing, hunting, honey gathering, and other livelihood activities (Smith, 1960; Roberts, 1968), as well as political reasons, people continuously forced their way back into Ugalla and natural resources exploitation pressure grew to unacceptable levels. Consequently, in 1965, the government upgraded the conservation status of Ugalla into a game reserve to further tighten restrictions on human activities in the area (Fisher, 2002). This caused another problem of wildlife poaching which is posing real challenges on the conservation efforts also because of the problematic refugees from nearby Katumba camp (UGR, 2006; Zonal Commander of western Tanzania Anti-Poaching Unit Mr. Mwombeki, F. pers.comm) as well as rapid development of Tabora Region (Carpaneto & Fusari, 2000).

Unlike Katavi, Ugalla is seriously lacking adequate scientific information on wildlife conservation and ecology, and bushmeat exploitation. There have been anecdotal reports about wildlife poaching. Indeed, Carpaneto & Fusari (2000) is the only study I have come across which attempted to address bushmeat consumption issues in western Tanzania, but again in some villages surrounding Urumwa Forest Reserve (a small forest fairly far from Ugalla). But, management approaches, conservation status and level of protection differ considerably between Ugalla Game Reserve and Urumwa Forest Reserve. Therefore, nature of bushmeat hunting (hunting gears, preferred species, hunting frequencies etc.) may differ among communities neighbouring the two reserves. For example, poaching in Ugalla Game Reserve may involve walking long distances and a very high risk of being caught. As a result, it may not be possible for poachers to kill more than 200 animals in a period of approximately 2 months as Carpaneto & Fusari (2000) found in Urumwa Forest Reserve. Also due to the need for profit maximization, most poachers may not prefer small mammals such as African pygmy hedgehog *Atelerix albiventris* and hares (*Lepus sp.*); thus the use of traps and dogs in Ugalla is expected to be uncommon compared to Urumwa.

The following chapters are the result of eight months of fieldwork undertaken in Ugalla in 2009. They are written in the form of independent research articles which are logically linked to describe the central theme of the study.

Chapter 2: *Exploited wildlife in Ugalla Game Reserve, western Tanzania*

I carried out a comparative analysis of exploited wildlife in adjacent hunting blocks in Ugalla. The wildlife division of Tanzania is committed to control offtakes at the level of hunting blocks. The assumption is that when a source-sink scenario exists between the hunting sites, as a result of different exploitation intensities, there would be a rapid decline in the wildlife populations in the reserve. Therefore, this chapter sought to establish the following: whether offtake impacts differ greatly between the hunting blocks in the Ugalla Game Reserve using species sex ratios, density estimates and group sizes; whether these population parameters vary among species; the nature of the difference in the abundance between Ugalla and a more protected Katavi-Rukwa ecosystem.

Chapter 3: *Local perspectives on factors influencing the extent of wildlife poaching* & *bushmeat consumption in Ugalla*

I carried out a bushmeat survey around Ugalla Game Reserve to investigate the influence of the following on illegal game hunting at both village and household levels: wildlife species poached; village location distance from the reserve boundary with respect to hunting blocks; fish consumption, livestock, food crops, climate patterns, human population and indicators of household wealth (household size, income, labour and assets).

Chapter 4: *Income sources* & *their relation to wildlife poaching*

Here, I analysed different economic activities of the household to determine whether: their mean contributions to the household income vary across study villages in different distance categories from the reserve boundary; they can actually contribute to the reduction of wildlife poaching.

Chapter 5: *Poaching activities in Ugalla Game Reserve*

I made use of seven years poaching data and intensive field surveys to establish the following: whether different types of poaching in Ugalla are specialist activities that are independent of each other; the intensity of poaching both in terms of the number of poachers and the spatial distribution of poaching activities; the degree of the impact suffered by different natural resources (forest, fish and other wildlife); the place of illegal game hunting in a poaching system along with the relative importance of different types of poaching activities.

Chapter 6: *Subsistence hunting in the Ugalla ecosystem*

I assessed wildlife offtakes from legal subsistence hunting to examine: variations in the hunting licences and animals removed between concession areas under different administration authorities near Ugalla Game Reserve; mismatches in the species and hunting quotas between the wildlife division of Tanzania and the local hunting authorities; variations in the hunting trends over time and success rate across species and hunting authorities; the relationship between legal and illegal game offtakes.

Chapter 7: *Towards sustainable wildlife management areas in Tanzania*

From a theoretical perspective, this chapter presents the challenges faced by the WMAs in Tanzania and suggests possible opportunities to ensure WMAs sustainability. Since lasting solutions to the bushmeat problems in Tanzania will apparently depend on the effectiveness of the WMAs, this article is very timely.

Chapter 8: *General discussion*

In this chapter, I synthesize the main findings of each chapter while describing their conservation and ecological implications. I also identify some study limitations and the main priority areas for further research on bushmeat in Ugalla and, indeed, elsewhere in Tanzania.

CHAPTER 2: EXPLOITED WILDLIFE IN UGALLA GAME RESERVE, WESTERN TANZANIA

Abstract

In Tanzania, tourist hunting is the principal legal form of wildlife utilisation in the Game Reserves which are divided into hunting blocks for this purpose. From an ecological perspective, one of the reasons behind having the hunting blocks is to ensure that hunting activities in the reserves are not clumped around certain areas but are fairly evenly distributed, and well harmonized with wildlife population performance. In this study, I carried out road transect surveys to estimate density and other demographic parameters of exploited wildlife in Ugalla Game Reserve, western Tanzania, to find out whether wildlife population densities suggest that utilisation intensity might be biased between Ugalla east and Ugalla west hunting blocks. Overall, estimates of density, group size and sex ratio across different species were higher in Ugalla east than Ugalla west. Of the individual species, helmeted guineafowl had the highest density, followed by impala and topi; whereas waterbuck had the lowest density. Sex ratios differed significantly between species although they were generally skewed towards females. The extent of legal and illegal off-takes with respect to species and hunting blocks should be put in proper perspective in the future.

Introduction

Wild animals are extensively utilised for food and commercial purposes, cultural beliefs and medicinal purposes, and for controlled population management (Festa-Bianchet, 2003; Davies & Brown, 2007; Smith, 2008). Human beings continue to exploit wildlife both for current benefit and because they have had a historical relationship with wild animals as a valuable natural resource (Mills, 2007). Wildlife exploitation has been widely reported to have an undesirable influence on wildlife populations (Taylor & Dunstone, 1996; Fa *et al.*, 2006; Setsaas *et al.*, 2007; Caro, 2008). However, not all species are affected in the same manner; some are affected more than others, while some are only affected indirectly (Mills, 2007). Contemporary wildlife conservation science has been paying much attention to how, why, and to what extent populations of different wildlife species are affected by human exploitation (Ginsberg & Milner-Gulland, 1994; Swenson *et al.*, 1997; Weber, 2000; Festa-Bianchet, 2003; Ryena-Hurtado & Turner, 2007; Caro *et al.*, 2009).

In developing countries, the main forms of wildlife exploitation are subsistence hunting (hunting for food and small-scale bushmeat trade) and tourist/sport hunting. In most instances, bushmeat hunting is non-selective and carried out illegally. This has raised concern specifically because so much of it is clearly unsustainable and takes no consideration of such ecological aspects as habitat, species-specific impacts, age, sex and density (Ginsberg & Milner-Gulland, 1994; Caro, 2008). Tourist hunting is said to be economically sensible, but it targets valued individuals of different species through selective trophy hunting in many wildlife areas (Ginberg & Milner-Gulland, 1994; Solberg *et al.*, 1999). Some factors guiding trophy hunting are sex and age structure of the hunted populations (Milner *et al.*, 2007). Sport/selective tourist hunting has also become a source of contention in conservation circles, and its adverse effects have been reported in some wildlife studies (for example, Swenson *et al.*, 1997; Weber, 2000; Coltman *et al.*, 2003).

In assessing the impact of exploitation on wildlife populations, a number of studies compare areas where subsistence/tourist hunting is legally allowed (for example, Game Reserves, Game Controlled Areas, and Open Areas) with areas where hunting is not allowed (for example, National Parks). In most cases, such comparative studies are carried out in areas with some sort of ecological connection where individuals of different wildlife species are able to move between hunted and non-hunted sites (Caro, 1999*a,c*; Reyna-Hurtado & Tanner, 2007; Setsaas *et al.*, 2007; Waltert *et al.*, 2008). In areas with ecological connectivity, wildlife populations are said to be stabilized by source-sink metapopulation dynamics in which non-hunted areas are considered as sources and adjacent hunted areas as sinks (Pulliam, 1988; Pulliam & Danielson, 1991; Novaro *et al.*, 2000; Begon *et al.*, 2006). Therefore, we should be careful about extrapolating recommendations from hunted areas that are connected to non-hunted areas to those hunted areas that are discrete or isolated.

In this study, I assess the status of exploited wildlife populations in a tourist hunting site not directly connected to any non-hunted area, but surrounded by a sea of humanity exerting pressure on wildlife resources mainly through illegal bushmeat hunting. I used density, group size and sex ratio to assess the status of exploited wildlife resources in the Ugalla Game Reserve while comparing Ugalla east and Ugalla west hunting blocks. This analysis has two roles: (1) it serves as an indication of whether wildlife use intensity is unbiased between these two hunting blocks; and (2) it serves as a baseline that may inform decisions about tourist hunting in terms of quota allocations and quarry attributes, for sustainable utilisation. Density and sex ratio have been employed elsewhere in Tanzania as indicators of the performance of wildlife populations (Caro, 1999*a,b,c,d*; Setsaas *et al.*, 2007; Caro, 2008; Waltert *et al.*, 2008).

Methods

Study area

The study was carried out in Ugalla Game Reserve (Fig. 2.1), in the regions of Tabora and Rukwa in western Tanzania. The reserve lies between 5º – 6º South and 31º – 32º East, and covers approximately 5000 km². Ugalla experiences a tropical climate defined by a distinct wet season from December – May and dry season from June – November. The rainfall varies between 700-1000 mm per year, and mean maximum and minimum temperatures between 28 – 30 ºC and 15 – 21 ºC respectively (Mbwambo, 2003; Hazelhurst & Milner, 2007). The vegetation is dominated by miombo woodland, containing species such as *Brachystegia speciformis*, *B. microphylla, B. bussei*, and *Isoberlinia globiflora.*

The original occupants of the reserve in the 1950s were the Wagalla people who were hunters, fishermen and honey gatherers. These people were allowed to carry out their natural resource-based livelihood activities until 1965 when the area was gazetted as a Game Reserve. Owing to increased pressure on natural resources, all unauthorised activities involving the use of resources were prohibited in the reserve, and local people were compelled to move out and occupy adjacent areas (UGR, 2006) where they resorted to other livelihood activities such as agriculture. However, due to their increasingly unsustainable ways of living which resulted in extensive deforestation (URT, 1998*a*), the reserve became ecologically isolated from other protected areas of higher categories.

The main legal activity in the reserve is tourist hunting which is carried-out in two hunting blocks, Ugalla east and Ugalla west. Hunting quotas (recommended numbers of individuals of different species that can be killed) are allocated annually to species under the hunting scheme, but in specific seasons (normally from July – December). Tourist hunting data are recorded as coming from each of the hunting blocks, but to no greater spatial accuracy. Additionally, only adult males of ungulate species are hunted. However, Tanzania is determined to ensure that wildlife utilisation is well managed at the level of hunting blocks (Caro *et al*., 2009). If this ambition is fulfilled, then according to (UGR, 2009), the following will be achieved: sustainable conservation of wildlife populations, increased revenue generation from wildlife resources, allocation of hunting quotas to different wildlife species such that hunting pressure remains within acceptable levels, tourist hunting adhering to statutory requirements regulating the use of wildlife in the country; for example, Wildlife Conservation Act of 1974 and Tourist Hunting Regulation (revised edition) of 2002, there will be no conflict between hunting activities and other resource utilisation activities (for example, fishing and beekeeping) in the game reserves.

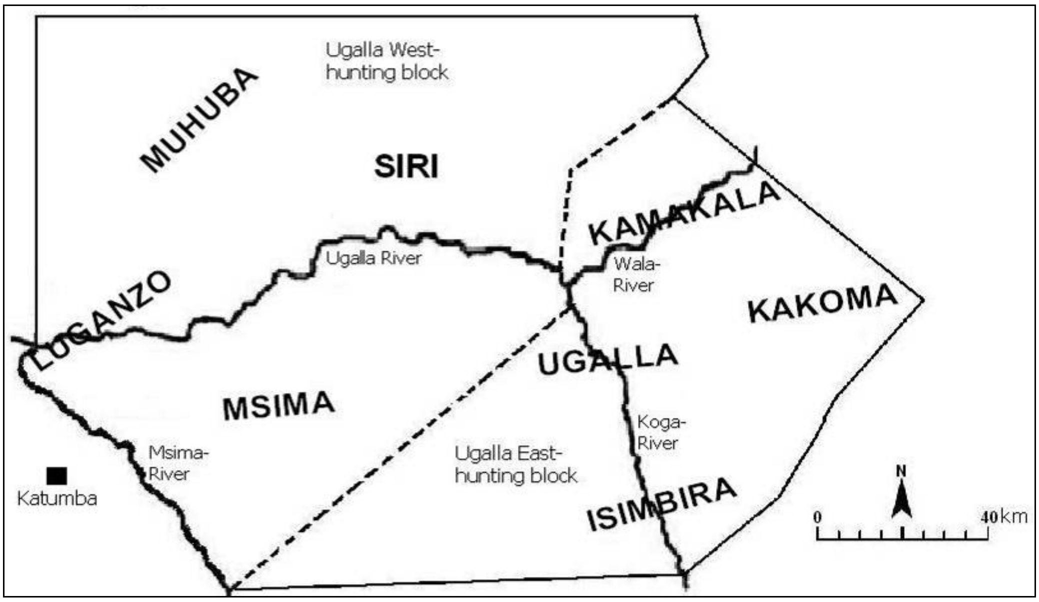

Figure 2.1 Ugalla Game Reserve. Names show approximate locations of the anti-poaching units. Thick line denotes the boundary. The dotted line demarcates the hunting blocks. Meandered lines show the main rivers. Katumba area in which the refugee camps (mentioned in the text) are located is also shown.

Road transect survey

The survey was designed and executed following the principles of distance sampling theory (Buckland *et al.*, 1993; 2001; 2004; Mills, 2007). Road transects were carried out in the dry season (June – September) in 2009. The sampling units were the areas designated as anti-poaching units (Figure 2.1) for patrolling purposes by Ugalla Game Reserve rangers. There were eight anti-poaching units (four in each hunting block). In Ugalla east these were: Isimbira (approximately 55,000 ha), Kakoma (95,000 ha), Kamakala (30,000 ha), and Ugalla (50,000 ha); and in Ugalla west: Muhuba (60,000 ha), Luganzo (70,000 ha), Siri (40,000 ha) and Msima (including Msima-chini and Msima-juu, 80,000 ha). In each month, the hunting blocks and anti-poaching units were sampled in a random order. A total of 36 driven transects were established, 24 in Ugalla east and 12 in Ugalla west. At least three transects were established in each anti-poaching unit. Transects covered a total distance of 782 km. The survey was undertaken with three personnel in an open vehicle driven at a constant speed of about 20 km per h along roads used for hunting and anti-poaching activities. Road transects have increasingly become used by wildlife biologists in surveying wildlife populations (Pollard *et al.*, 2002).

Road transects were driven during the morning hours from 7 – 12 am. Observers searched for mammal and large bird species on both sides of each randomly selected road in the anti-poaching units. Sighting distances and angles were measured using a rangefinder and a compass respectively (Buckland *et al.*, 2001). When groups were spotted, the sighting distance was measured from the road to the approximate geometric centre of the group. Group sizes (number of individuals per sighting/observation), and the number of males and females for adult and sub-adult individuals were recorded.

Statistical analysis

Densities were estimated using Distance software version 5.0. Detection function histograms and goodness of fit statistics were used as criteria for the selection of appropriate models and to assess the presence of outliers (Thomas, *et al.*, 2006). Half-normal and hazard rate models were fitted to data from the 11 species which had >15 observations. Species for which density was estimated included birds and mammals; the majority of which were in the tourist hunting scheme. The bird species analysed were: helmeted guineafowl (*Numida meleagris)* and southern ground-hornbill (*Bucorvus cafer*). Mammals were: impala *(Aepyceros melampus)*, warthog *(Phacochoerus africanus)*, topi *(Damaliscus korrigum)*, oribi *(Ourebia ourebi)*, bohor reedbuck *(Redunca redunca)*, waterbuck *(Kobus defassa)*, hippopotamus *(Hippopotamus amphibius)*, giraffe (*Giraffa camelopardalis*) and hartebeest *(Alcelaphus buselaphus)*. The giraffe is not included in the tourist hunting scheme, but is among the priority species in illegal bushmeat hunting (Baran *et al.*, 2008). Likewise, the southern ground hornbill is not included in the tourist hunting scheme, but it is occasionally targeted by poachers for traditional purposes. Other species with total observations <15 were just involved in the exploration of group sizes. Sex ratio analyses were carried out only for the above mentioned mammal species. However, the hippopotamus was omitted from sex-ratio analyses because it was not easy to distinguish male and female individuals, especially when they utilised most of their day time submerged in water (Caro *et al.*, 2009).

Other statistical analyses and comparisons were completed with the statistical package GenStat® 10 (Payne *et al.*, 2007), and SPSS 15 for Windows. A non-parametric test, namely, the Wilcoxon matched pairs test was used in the comparisons of sightings, detection probabilities, and group sizes between the hunting blocks. Pearson correlation was used to determine the association between detection probability and animal sightings. Mann-Whitney U test was used to test for differences in density estimates, and group sizes for individual species between the hunting blocks. Generalised linear models (with appropriate error distributions) were used to assess the influence of species, hunting blocks, group sizes and observations/sightings on species density and sex ratios. All statistical tests were two-tailed, and the significance level (α) was set at 0.05.

Results

Observation, group size & *density*

Table 2.1 presents 44 species of mammals and birds seen during the survey. A total of 535 sightings of different species was obtained. There were more sightings in Ugalla east (275) than Ugalla west (260) (Wilcoxon matched pairs test adjusted for ties, $n = 44$ species, $z = -2.460$, $p = 0.014$, Table 2.1). Of the species seen, 7 and 11 had >30 and >15 sightings respectively. Groups of impala were encountered more frequently in each of the hunting blocks than any other mammal species. Among birds, helmeted guinea fowl and southern ground-hornbill were observed most frequently. Some species of birds and mammals were not observed in one of the blocks; for example, plains zebra was not seen in Ugalla east. Likewise, African wild dog, greater kudu, and African savanna hare were not seen in Ugalla west. Of the birds, six species, namely, African fish eagle, hammerkop, little egret, shoebill, spur-winged goose, and glossy ibis were not seen in Ugalla west.

Histograms of the distance data for different species (Appendix) indicate that observations tended to decline with distance from the centre of the road transect. However, there were some variations in the detectability among species especially within 50 m from the transect. With the exception of giraffe, detection probability in the first distance category for all other species was below 80%. This is an indication of road avoidance behaviour, and was common in medium-sized ungulates such as reedbuck, impala, and warthog (ref. Appendix

1). Detection probability had a significant positive correlation with number of animals observed ($n = 11$ species with density estimates, $r = 0.7082$, $p = 0.0147$). Overall, detection probability across species in Ugalla east (0.4827 ± 0.0439) was higher than Ugalla west (0.3455 ± 0.0481) (Wilcoxon matched pairs test adjusted for ties, $n = 11$ species, $p = 0.052$).

Mean group sizes across species in Ugalla east exceeded those in Ugalla west (Wilcoxon matched pairs test adjusted for ties, $n = 44$ species, $z = -2.563$, $p = 0.010$). At the level of individual species, differences in group sizes between the hunting blocks were only tested for those species whose numbers of observations in each of the blocks were ≥ 3. With the exception of topi and bushbuck, which were observed in somewhat smaller groups in Ugalla west and Ugalla east respectively, there were no significant differences in group sizes of other species between the hunting blocks (Table 2.1).

A generalised linear model with normal errors was used to determine the relationship between density and the following factors: species, hunting block, group size and number of observations. Only species and hunting block were the best predictors of density (Table 2.2). Density estimates differed among species. Guineafowl had the highest density followed by impala, whereas waterbuck had the lowest density (Table 2.1). Similarly, density estimates differed between the hunting blocks, with Ugalla east having higher animal density than Ugalla west (Tables 2.1 & 2.2). For individual species, density estimates of warthog, topi, guineafowl and hippopotamus were significantly greater in Ugalla east than Ugalla west; whereas for other species, densities did not differ significantly between the hunting blocks (Table 2.1).

Total mammal densities in Ugalla Game Reserve were also compared to findings from a previous study carried out by Caro (1999*a*), using similar methodology, in the Katavi National Park of western Tanzania where consumptive utilisation is not allowed. With the exceptions of hartebeest and reedbuck, which had low densities in both places, densities seemed to be lower in Ugalla than in Katavi National Park (Figure 2.2). The oribi was not plotted due to lack of data for Katavi.

Table 2.1 Species sighted, total number of observations (N), mean group size (MGS, number of observations in brackets), density estimates [D (individuals km^{-2}) ± standard error (s.e.)]. Z-values and probabilities-p based on Mann-Whitney U-tests are also presented to test for significant differences in group sizes and densities between Ugalla east (East) and Ugalla west (West) hunting blocks**.** Species are listed in decreasing total number of observations. Where species have the same total number of observations, alphabetical order is followed.

		East	West	For group size		East	West	Total	For density	
Species	N	MGS	MGS	Z-value	p	D ± s.e.	D ± s.e.	D ± s.e.	Z-value	p
Impala *(Aepyceros melampus)*	86	10.21(39)	10.91(47)	-0.203	0.839	2.19±0.44	1.75±0.55	1.97±0.23	-0.642	0.521
Common warthog *(Phacochoerus africanus)*	51	2.20(25)	2.46(26)	-0.355	0.722	0.66±0.20	0.24±0.17	0.45±0.11	-2.934	0.003
Topi *(Damaliscus korrigum)*	50	4.73(37)	6.92(13)	-1.928	0.054	1.72±0.20	0.91±0.62	1.30±0.16	-2.605	0.009
Oribi *(Ourebia ourebi)*	38	2.14(21)	2.17(17)	-0.140	0.888	0.90±0.45	0.58±0.37	0.76±0.48	-1.734	0.083
Helmeted guineafowl *(Numida meleagris)*	34	11.54(16)	8.62(18)	-0.611	0.541	6.75±2.06	3.39±1.88	5.07±1.77	-2.605	0.009
Bohor reedbuck *(Redunca redunca)*	33	2.32(19)	2.50(14)	-0.444	0.657	0.33±0.38	0.39±0.49	0.33±0.46	-0.863	0.388
Waterbuck *(Kobus defassa)*	33	3.90(10)	4.00(23)	-0.040	0.968	0.20±0.15	0.19±0.05	0.21±0.06	-0.928	0.353
Southern ground-hornbill (*Bucorvus cafer*)	26	3.73(15)	3.72(11)	-0.370	0.711	0.51±0.24	0.32±0.18	0.42±0.22	-1.359	0.174
Hippopotamus *(Hippopotamus amphibius)*	21	6.13(16)	9.40(5)	-0.247	0.805	1.56±2.82	0.43±1.32	0.99±1.42	-2.464	0.006
Giraffe (*Giraffa camelopardalis*)	19	4.40(10)	2.22(9)	-1.629	0.103	0.60±0.79	0.49±0.61	0.55±0.73	-0.825	0.409
Hartebeest *(Alcelaphus buselaphus)*	17	6.09(11)	6.17(6)	-0.051	0.960	0.49±0.75	0.46±0.88	0.48±0.84	-1.622	0.095
African elephant *(Loxodonta africana)*	13	4.00(10)	1.67(3)	-1.159	0.246					
Olive baboon *(Papio anubis)*	9	7.83(6)	4.33(3)	-0.651	0.515					
Sable antelope *(Hippotragus niger)*	9	5.33(4)	4.33(5)	-0.664	0.507					
Bushbuck *(Tragelaphus scriptus)*	7	1.01(4)	2.10(3)	-1.936	0.053					
Roan antelope *(Hippotragus equinus)*	7	4.00(4)	4.33(3)	0.000	1.000					
African buffalo *(Syncerus caffer)*	6	9.50(3)	17.00(3)	-0.775	0.439					
Egyptian goose *(Alopochen aegyptiaca)*	6	9.75(4)	2.33(2)							
Greater kudu *(Tragelaphus strepsiceros)*	6	2.48(6)	0.00(0)							
Nile crocodile *(Crocodylus niloticus)*	5	7.20(3)	4.60(2)							
Vervet monkey (*Chlorocebus pygerythrus*)	5	3.62(3)	2.30(2)							

Common Duiker *(Sylvicapra grimmia)*	4	1.00(1)	1.00(3)
Marabou Stork *(Leptoptilos crumeniferus)*	4	1.85(3)	1.00(1)
Eland *(Turotragus oryx)*	3	3.50(2)	1.00(1)
Kirk's dik-dik (*Madoqua kirkii*)	3	2.00(1)	2.00(2)
Open bill stork *(Anastomus lamelligerus)*	3	2.00(2)	1.00(1)
Plains zebra *(Equus burchelli)*	3	0.00(0)	6.59(3)
Spotted hyena *(Crocuta crocuta)*	3	3.50(2)	2.00(1)
White-backed vulture *(Gyps africanus)*	3	18.00(1)	15.00(2)
African fish-eagle *(Haliaeetus vocifer)*	2	1.50(2)	0.00(0)
African wild dog *(Lycaon pictus)*	2	6.50(2)	0.00(0)
Bushpig *(Potamochoerus larvatus)*	2	3.00(1)	2.00(1)
Great egret (*Ardea alba*)	2	1.00(1)	3.00(1)
Grey heron *(Ardea cinerea)*	2	2.00(2)	1.00(1)
Hadada ibis (*Bostrychia hagedash)*	2	1.00(1)	1.00(1)
Hamerkop (*Scopus umbretta*)	2	1.00(2)	0.00(0)
Little egret *(Egretta garzetta)*	2	5.00(2)	0.00(0)
Ostrich *(Struthio camelus)*	2	1.00(1)	1.00(1)
Saddle-billed stork *(Ephippiorhynchus senegalensis)*	2	3.00(1)	2.00(1)
Shoebill *(Balaeniceps rex)*	2	1.00(1)	0.00(0)
Spur-winged goose (*Plectropterus gambensis*)	2	12.38(2)	0.00(0)
Yellow-billed stork *(Mycteria ibis)*	2	9.00(1)	7.00(1)
African savanna hare (*Lepus microtis*)	1	1.00(1)	0.00(0)
glossy ibis *(Plegadis falcinellus)*	1	1.00(1)	0.00(0)

Table 2.2 General linear model output showing the terms associated with animal density estimates in Ugalla Game Reserve.

	Estimate ± s.e.	d.f. (change, residual)	F-value	Probability
Species		10,20	16.80	<0.001
Hartebeest	-0.01 ± 0.20			
Guineafowl	1.14 ± 0.36			
Hippopotamus	0.33 ± 0.25			
Hornbill	-0.11 ± 0.16			
Impala	0.61 ± 0.53			
Oribi	0.06 ± 0.16			
Reedbuck	-0.27 ± 0.18			
Topi	0.34 ± 0.26			
Warthog	-0.23 ± 0.18			
Waterbuck	-0.43 ± 0.18			
Hunting block (West)	-0.26 ± 0.25	1,11	11.44	0.007
Group size		1,10	1.48	0.254
Observation		1,9	0.32	0.589

Hunting block reference: East block; Species reference: Giraffe

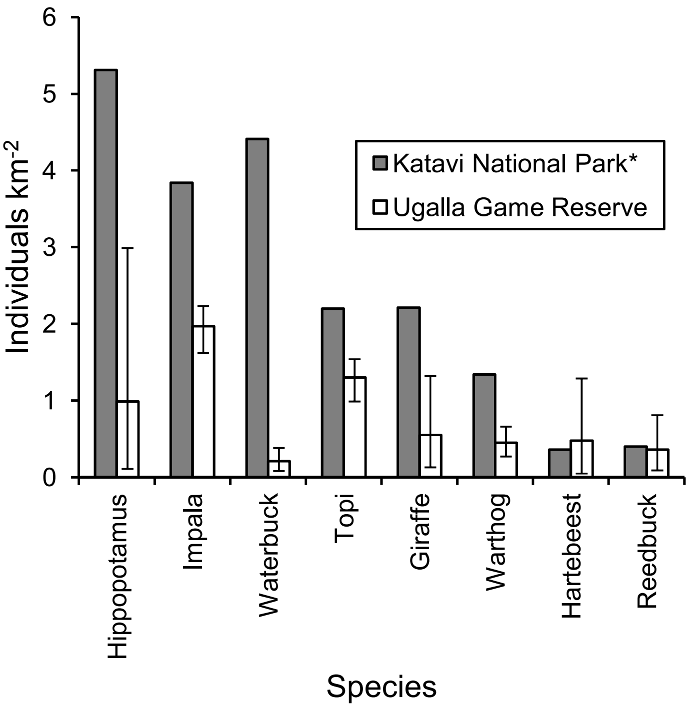

Figure 2.2 A comparison of species' densities between Katavi National Park and Ugalla Game Reserve (*adapted from Caro, 1999*a*). The error bars are 95% confidence intervals.

Sex ratios

Sex ratio was considered to be a proportion of male individuals of a species. Sex ratios were determined only for 8 out of 11 species with density estimates. A generalised linear model with binomial errors was used to identify the best predictors of sex ratio among the following factors: group size, species, density and hunting block. Of these, only group size and species were significant predictors of sex ratio (Table 2.3). Sex ratios differed significantly among species; for example, impala had the lowest sex ratio whereas giraffe had a relatively high sex ratio. Sex ratios were more skewed towards females for species with relatively big group sizes.

Table 2.3 General linear model output showing the terms associated with sex ratios in Ugalla Game Reserve.

	Estimate ± s.e.	d.f. (change, residual)	Deviance	Probability
Group size	-0.34 ± 0.27	1,6	6.39	0.011
Species		7,12	2.94	0.024
Hartebeest	-0.43 ± 0.56			
Impala	-2.01 ± 2.68			
Oribi	-0.90 ± 0.45			
Reedbuck	-0.10 ± 0.70			
Topi	-0.18 ± 1.06			
Warthog	-0.14 ± 0.55			
Waterbuck	-0.32 ± 0.42			
Density		1,5	0.34	0.558
Hunting block (West)		1,4	0.002	0.888

Block reference: East block; Species reference: Giraffe

Discussion

This study has revealed that with the exception of sex ratios, other population parameters of the exploited wildlife in Ugalla Game Reserve differ remarkably between the hunting blocks. In general, estimates of these parameters across species were greater in Ugalla east than Ugalla west. This suggests that wildlife species in Ugalla west are more affected by exploitation than Ugalla east since density and group size can provide useful information about exploitation intensity of the wildlife species (Caro 1999*ac*; Milner *et al.*, 2007; Setsaas *et al.*, 2007). I accept the fact that different habitat types within the study site may also have influenced some of the parameters, such as density (for example, Caro, 1999*d*; Waltert *et al.*, 2009), but this needs to be addressed through a rigorous ecological study.

From the point of view of individual species: density, group size, and sex ratio differed across species. Some species had comparatively high total density estimates (for example, impala and guineafowl) and others had low densities (for example, waterbuck). Species-specific population parameters also varied between the hunting blocks. Most of the ungulates had density estimates lower than their counterparts in the national parks (where tourist hunting is not allowed) for example, Katavi National Park (Caro, 1999*a*). Although these differences could be attributed to levels of hunting, illegal hunting (poaching) could be more to blame (Loibooki *et al*., 2002; Kaltenborn *et al*., 2005; Stoner *et al*., 2006; Wilfred, 2010) than tourist hunting (Loveridge *et al*., 2006). In Tanzania in particular, levels of tourist hunting off take are said to be sustainable (Leader-Williams *et al*., 1996; Caro *et al*., 1998) and in most cases their advantages outweigh disadvantages (Ndolanga, 1996).

I now shed some light on species-specific differences while making comparisons with some other studies especially those carried out in the Katavi-Rukwa ecosystem of western Tanzania (in habitats similar to those of Ugalla Game Reserve).

Impala: Of the wild ungulates, impala had the lowest sex ratio. Impala were widely distributed in groups of different sizes in both Ugalla east and Ugalla west hunting blocks. It was possible to spot individuals and groups of impala even at a truncation distance of >300m (see also Caro 1999*d*). Number of female individuals far exceeded male individuals in most of the larger groups. A highly biased sex ratio in impala was also reported by Setsaas *et al.* (2007) in the hunted areas of Serengeti. The highest density of impala in Ugalla Game Reserve reflects their ability to sustain hunting pressure. Like other ungulates, during the survey, impala showed a high degree of roads avoidance (at least in the presence of the survey vehicle). This kind of evasiveness could be a good indication of "anti-predatory behaviour" (see Setsaas *et al*., 2007), which might have been an important factor in their survival strategy (see Durant, 2000).

Topi: The overall density of topi was also fairly high. As in the case of impala, predator avoidance behaviour might be among the factors maintaining the population of topi. Topi are among wild ungulates with "high vigilance" (Schaller, 1972), and "highly synchronized birth" (Sinclair *et al*., 2000). The results suggest that the topi's density in Ugalla west is lower than Ugalla east, possibly indicating high off-take rate in Ugalla west. The female biased sex ratio of topi, like many other ungulates in Ugalla, might be intensified by the selective nature of tourist hunting. Although for most ungulate populations, sex ratios are generally biased towards females (Ginberg & Milner-Gulland, 1994), also because of natural predation.

Oribi: Little has been documented on the density and demography of oribi in the miombo woodlands of western Tanzania. The total density in Ugalla Game Reserve was far lower than the estimate in Serengeti National Park (Mduma, 1995). In addition, sex ratio was skewed towards females, but both density and group size were not significantly different between the hunting blocks. The low total density of oribi in the reserve may not be immediately linked with exploitation intensity. Probable factors affecting density and other demographic parameters for small antelopes such as oribi are climate (temperature and rain fall) and food quality (Mduma, 1995), as well as predation (Sinclair *et al.*, 2000). However, owing to their small size and hiding behaviour (Sinclair *et al.*, 2000), I cannot discard the possibility that some of the demographic parameters of oribi in Ugalla may have been underestimated.

Waterbuck: Waterbuck had the lowest density and sex ratio was highly biased towards females. Since the survey was conducted during the dry season, environmental factors might have contributed to lower estimates of the population parameters of waterbuck in the reserve. Because, firstly, during dry seasons waterbuck tend to congregate in riverine vegetation (Sinclair & Arcese, 1995) since they are enormously dependent on water (Estes, 1991). This may determine their distribution (Redfern *et al.*, 2003) thus influencing their density and sex ratio estimates. Secondly, as a result of their habitat preference during the dry seasons, they might have become easy target for hunters because of the less cover available along most of the main rivers in Ugalla.

Hippopotamus: Of the large sized ungulates, hippopotamus face the greatest human and environmental pressures. The total density estimate of hippopotamus in Ugalla was higher than the estimates in Rukwa-Lukwati Game Reserve (Caro, 1999*a*; Waltert *et al.*, 2008). This difference in density could possibly be due to the terrain constraints which make it difficult to survey "water courses" in the latter (see Caro, 1999*a*). The survival of hippopotamus in Ugalla Game Reserve is threatened by anthropogenic factors (J. Lymo, pers.comm.). Notorious hippopotamus poachers come from the closest refugee camps (IRF, 2007). A number of individuals armed with sophisticated firearms sneak out of the camps and get into the game reserves, normally in groups of at least 10 people, to poach mainly hippopotamus and elephant for both bushmeat and commercial purposes (F. Mwombeki, pers. comm.). Ugalla west, especially Msima area, has been frequently invaded by such poachers. Nevertheless, as in the case of waterbuck, dry season estimations of the hippopotamus density might be susceptible to environmental factors. Normally during the prolonged dry seasons, the main rivers in the reserve; namely, Ugalla, Walla, Msima and Koga, shrink and form chains of pools in which hippopotamus get more crowded (J. Lymo, pers. comm.) and this might result in hippopotamus "die-offs" (Caro, 2008).

Giraffe: Unlike other ungulates, groups of Giraffe were often seen browsing near the roads (in a distance <100 m from the centre of the road). Giraffe is a national symbol of Tanzania (Kaltenborn *et al.*, 2003), and thus it is not part of the tourist hunting scheme. Since tourist hunting clients shot animals from cars and off-road driving in search of the quarry was not encouraged, giraffe seemed to lose their fear of vehicles and roads. The total density of giraffe was less than other unhunted areas; for example, Katavi National Park (see Caro, 1999*a*). This may be attributed to illegal hunting, which normally takes place on foot in the interior areas of the reserve far from roads to avoid game rangers. Giraffe is a large-bodied species preferred by poachers because it provides substantial amount of meat (J. Lymo, pers. comm.). During the survey, remnants of giraffe killed by poachers were encountered in both Ugalla east and Ugalla west hunting blocks.

Other ungulates included reedbuck, hartebeest and warthog. The density estimate of warthog in Ugalla west was lower than Ugalla east, suggesting high off-take rates in Ugalla west. Overall density of reedbuck was relatively low and was also lower than the estimates elsewhere in Africa; for example, in Bale Mountains National Park in Ethiopia (Afework, *et al.*, 2010). Caro *et al.* (1998) pointed out that reedbuck is among species which may not sustain long term off-takes through tourist hunting, especially when levels of poaching are also high. Although estimates of demographic parameters of hartebeest (for example, group size and density) were balanced between the hunting blocks, the total Ugalla density was well below estimates reported elsewhere in foot surveys of the Katavi-Rukwa ecosystem (Waltert *et al.*, 2008).

Helmeted guineafowl: I also attempted to draw attention to the status of the large bird species in Ugalla Game Reserve. I obtained sufficient number of observations for helmeted guineafowl and southern-ground hornbill. Guineafowl is one of the large savanna birds that are not only preferred in trophy/commercial hunting (Lamprey *et al.*, 2003), but also harvested by local communities in the quest for bushmeat (Bassett, 2005; Thiollay, 2006; Magige *et al.*, 2009). Guineafowl had the highest density among all the surveyed species. Although the density estimate for guineafowl might still be lower than it would be in the absence of hunting, it gives an indication that hunting was possibly biased towards mammal species especially the highly valued wild ungulates (see Ginsberg & Milner-Gulland, 1994; Solberg, *et al.*, 1999; Milner, *et al.*, 2007). Considering the cost involved (in terms of energy, time and poaching gear) and the risk of being caught involved in illegally entering the reserve, most poachers would primarily target mammal species instead of birds to maximise their profits.

Southern-ground hornbill: During the survey, s.ground hornbills were frequently sighted in the open grasslands in both Ugalla east and Ugalla west. There was no significant difference in density of s.ground hornbill between the hunting blocks. Notwithstanding the fact that it has been categorized as a "vulnerable" species (BI, 2010) due to harvesting by people (Thiollay, 2006; Trial, 2007) and more importantly habitat loss as a result of unsustainable human activities (Morrison, *et al.*, 2005; Trial, 2007), the observed total density of s.ground hornbill in the reserve seemed fairly high. This is because protected areas such as game reserves and national parks are known to be strongholds for s.ground hornbills (Thiollay, 2006; van Essen, 2006). According to Ugalla Reports, Ugalla Game Reserve offers prime habitat for savanna birds such as s.ground hornbill (UGR, 2006). Nonetheless, the survey was carried out during the dry season when most reserve patches with tall grasses (especially the ones along tourist hunting roads) were burnt to attract grazing herbivores and to increase visibility during the tourist hunting. Since I often encountered small groups (of at least 3 individuals) scattered in these burnt patches of open grasslands, presumably because s.ground hornbill prefer "open grasslands" (CARNIVORA, 2008), there is every possibility that burning (since it gets rid of tall grasses and increases openness) might have influenced the density estimate. Because of the international appeal for its conservation, especially in southern Africa, empirical studies for estimating the abundance and distribution of s.ground hornbill in Ugalla and indeed across the country are of chief importance.

Conclusions

The intention of this study has been to inform and to estimate some parameters that might positively influence the conservation of the exploited wildlife in Ugalla Game Reserve and western Tanzania as a whole. The analyses of density and other demographic parameters suggest that the distributions of the impacts of hunting activities are not well balanced between the hunting blocks. Ugalla west hunting block seems to be more affected by wildlife exploitation than Ugalla east hunting block. This prompts the need for the in depth assessment of the extent of legal and illegal exploitation of wildlife in the context of the hunting blocks' difference.

The observed differences in density and demography across species suggest that different species in the reserve respond differently to the intensity of utilisation, which has of course been the case in previous studies carried out elsewhere (Caro, 1999*ac*; Naranjo & Bodmer, 2007; Reyna-Hurtado & Tarner, 2007; Caro, 2008). Therefore, reliable and up-to-date species-specific density and other population parameters should be carefully considered when deciding hunting quotas (Caro, 1998).

Wildlife areas surrounding Ugalla Game Reserve (wildlife management areas, game controlled areas and open areas) face severe human pressure since their conservation status and level of protection are lower than the reserve. As a result, populations of different species in the reserve are closed in terms of immigration and emigration (UGR, 2006). Prolonged removal of high value trophy males of different species should be very closely monitored (see Loveridge *et al.*, 2006) since in the long run it might adversely affect species birth rates (see Caro, 1998; Ndibalema, 2009).

CHAPTER 3: LOCAL PERSPECTIVES ON FACTORS INFLUENCING THE EXTENT OF WILDLIFE POACHING & BUSHMEAT CONSUMPTION IN UGALLA

Abstract

Illegal exploitation of wildlife for bushmeat (wildlife poaching) is a widespread problem affecting many ecosystems especially in the Tropics. Understanding the factors associated with such exploitation may help in the management of the problem by conservationists. Although there is a substantial problem of wildlife poaching in east Africa, the factors that affect its occurrence at a local level are poorly explored. I interviewed the heads of households in villages around Ugalla Game Reserve in western Tanzania to obtain data on wildlife exploitation. The results showed that proximity to the reserve encouraged both wildlife poaching and bushmeat consumption on the northern side of the reserve (among communities adjacent to Ugalla west hunting block). Conversely, consumption increased with distance on the eastern side of Ugalla (among communities adjacent to Ugalla east hunting block). Most poaching activities were carried out in the rainy seasons. Both large- and medium-sized wild ungulates especially impala, dik-dik and common duiker were favoured bushmeat species. While households with high fish consumption and adequate food stocks had low bushmeat consumption frequencies, those who were wealthier consumed bushmeat quite often. Problems related to anti-poaching efforts particularly during the rainy seasons should be taken more seriously. Crop farming along with fish production should also be promoted. Further research on the nature of bushmeat exploitation trade is needed.

Introduction

It is generally accepted that human pressure on wildlife protected areas is increasing (Jachmann, 2008*a*; Wittemyer *et al.*, 2008). One of the critical challenges in wildlife conservation has been bushmeat hunting/wildlife poaching (Hofer *et al.*, 1996; Cowlishaw *et al.*, 2004; Coad, 2007). Bushmeat can be defined as "any non-domesticated terrestrial mammals, birds, reptiles and amphibians harvested for food" (Nasi *et al.*, 2008). Some other definitions pay special attention to Africa where bushmeat hunting is believed to be problematic. For example, bushmeat has also been defined as "an African term that includes all wildlife species used for food, from cane rats to elephants" (Bennett, *et al.*, 2006). Bushmeat hunting is often referred to as wildlife poaching because it is pervasively carried out regardless of whether or not the wildlife laws permit it. Poaching is a problem especially in Africa where bushmeat hunting is valued both as a source of income and a source of protein (Brashares, *et al.*, 2004; Kalternborn *et al.*, 2005; Bennett *et al.*, 2006). Since it is deeply blended with other livelihood activities into the socio-economic fabric of people's lives, attempts to tackle it should also explore other related livelihood-based factors.

A number of factors influencing wildlife poaching have been highlighted in the conservation literature. For example, preference for different wildlife species is considered to be an important cause of differences in poaching pressure among species (Njiforti, 1996; Ndibalema & Songorwa, 2007). In northern Cameroon, most people prefer North African porcupine *Hystrix cristata* and guineafowl, as a consequence these species are heavily exploited (Njiforti, 1996). In some areas of the Serengeti ecosystem, species such as buffalo, eland and topi are consumed by a large proportion of the local communities (Ndibalema & Songorwa, 2007).

Agricultural production (crop farming and livestock keeping) is another factor affecting poaching (Barrett & Arcese, 1998; Johannesen, 2005). Most wildlife areas (especially in Africa) are found in rural locations where local communities are constantly struggling to get out of poverty (Roe *et al.*, 2010). It is widely accepted that the majority of the rural poor (about 80%) depend on agriculture as a source of both food and income (Rweyemamu, 2003; Amani, 2005; Davis *et al.*, 2007; Ndobi, 2008). From a wildlife conservation standpoint, agriculture has the potential for ensuring food security, and thereby reducing wildlife exploitation (Fa *et al.*, 2003).

The presence of alternative sources of protein can help to lessen demand for bushmeat (Hofer *et al.*, 1996). The widely documented (and seemingly viable) alternatives to game meat are livestock and fish (Brashares *et al.*, 2004; East *et al.*, 2005; Rowcliffe *et al.*, 2005; Ndibalema & Songorwa, 2007). However, there has been a growing debate on how they can serve as effective substitutes for bushmeat. For example, Brashares *et al.* (2004) found that increased fish supply reduced both bushmeat hunting and consumption in Ghana. In contrast, Rowcliffe *et al.* (2005) doubted this generalization based on the fact that under well controlled bushmeat consumption and considerably reduced fish stock, fish may not provide a viable alternative to bushmeat. The same authors argued further that livestock is "the most important potential substitute" of bushmeat, provided the challenges confronting the livestock industry are given due attention. Notwithstanding such arguments, fish and livestock have been recommended by different bushmeat researchers depending on the local environment of a particular area. In Serengeti, for example, both fish (Nyahongo *et al.*, 2009) and livestock (Loibooki *et al.*, 2002) are potentially important in solving the bushmeat problem. Coad (2007) also acknowledged the potential of livestock as an alternative to bushmeat hunting in Gabon. Njiforti (1996) recommended introduction of wildlife domestication projects for preferred species as bushmeat alternatives in northern Cameroon. Therefore, the availability and importance of a protein alternative to bushmeat differ from place to place, and the treatment of one place is not necessarily as effective as in another place.

Wildlife utilisation is also related to villager proximity to protected areas. For example, in the Serengeti ecosystem wildlife hunting intensity is associated with human settlements at different distances from protected areas (Hofer *et al.*, 1996). Villages closer to the protected areas have high human population densities coupled with undesirable conservation consequences. The effects of village location distances in Serengeti were also reported by Nyahongo *et al.* (2009), who found that rates of meat consumption were quite high in the villages near protected areas. Other studies have attempted to relate distances of human settlements from hunting areas with biomass/number of wildlife caught/trapped (Coad, 2007; Smith, 2008). In these studies, wildlife exploitation intensities decreased with increasing distances from settlements. Generally, there is no single blueprint for dealing with bushmeat hunting because factors associated with the problem can vary widely across countries, regions, and often across ecosystems (Caro & Andimile, 2009).

Despite the fact that Tanzania is among the countries experiencing bushmeat hunting problems in east Africa (Baldus, 2002; Caro & Andimile, 2009), bushmeat studies have paid little attention to the western part of the country particularly in the Ugalla ecosystem where wildlife poaching is problematic (Wilfred & MacColl, 2010). This is regrettable because in order to tackle or slow down wildlife poaching across Tanzania, we need to understand ecosystem-specific drivers of the problem. One of the principal priorities of the Wildlife Division of Tanzania is to deal comprehensively with poaching activities across all sorts of protected areas in the Country. Therefore, this study is timely as it seeks to address factors behind wildlife poaching among local communities around the Ugalla Game Reserve (the central part of the Ugalla ecosystem). More specifically, the study presents wildlife species poached and the frequency with which they are poached (as estimated by villagers). Then it explores the associations between the frequency of wildlife poaching and bushmeat consumption with some livelihood factors such as alternatives to bushmeat (fish and livestock meat) and agricultural production while controlling for village location distances from the reserve. According to the Ugalla Game Reserve Management Team, rainfall and climate seasons are among the factors worthy of consideration when fighting wildlife poaching in Ugalla. The present study, therefore, throws some light on these aspects as well, to try to understand their influence on poaching activities. Vitally, the study identifies indicators of wildlife poaching hotspots for a more effective and results-orientated approach to the problem.

Methods

Study area

Ugalla Game Reserve (5,000 km^2) (Fig. 3.1) is found in the Tabora and Rukwa regions of western Tanzania. A considerable portion of the reserve is found within Sikonge (21,000 km^2) and Urambo (21,299 km^2) districts in the Tabora region. The present study was conducted in these districts. The region is located at $4^0 - 7^0$ S and $31^0 - 34^0$ E, and its average elevation is 1200 m above sea level. The climate is defined by a dry season (June – November) and a wet season (December – May), and annual rainfall ranges between 700 – 1,000 mm. The period of rain spans between the months from November – May with an extension of occasional showers of varying magnitudes until mid of June. The maximum and minimum temperatures lie between 28 – 30 ^{0}C and 15 – 21 ^{0}C respectively. The 2002 human populations (growth rates) of Sikonge and Urambo were 132,733 (3.7%) and 369,329 (4.8%) respectively (NBS, 2002). The main livelihood activity in the region is subsistence farming of both food crops (for example, maize, groundnut, cassava potato) and cash crops (tobacco). Small-scale income generating activities are also present as supplementary sources of livelihoods. The area has a diverse range of natural resources such as fish, wildlife, forests, and wetlands. Thus, natural resources based livelihood activities including fishing, hunting, lumbering, beekeeping/honey gathering are widespread.

Data collection

A questionnaire survey (with heads of households) was conducted in the period from March – October, 2009 in the villages neighbouring Ugalla Game Reserve. Firstly, a sample of 19 study villages was randomly selected from a total of 122 villages in Sikonge and Urambo districts representing a sampling intensity of 15%. Since Urambo and Sikonge border Ugalla west and Ugalla east hunting blocks respectively, study villages were consequently adjacent to either of the two hunting blocks (Fig. 3.1 & Table 3.1). Villages bordering Ugalla west hunting block were north of the reserve. Thereafter, at least 5% of the households from each of the study villages were randomly selected from village registers, giving a total of 573 survey households (out of about 11,000 households in all the study villages). Questionnaires containing both open- and close- ended questions were administered in Swahili (a language familiar to villagers) in order to accommodate a wide range of responses about wildlife exploitation. Questions were asked in order of their sensitivity, beginning with respondents' characteristics and continuing up to bushmeat consumption. For instance, the first part of the questionnaire contained questions about respondents and household characteristics such as age, education, tribe, household members, number of livestock owned and crop yield in kgs. In the second part, respondents were asked to estimate household incomes from different sources, namely: small businesses, formal employment, remittances, and crop and livestock sales in the 6 months prior to the dates of the interview. Recall of dietary protein intake was done separately for different sources of protein in different stages of the interview to avoid any potential bias in the responses. Villagers mentioned the number of times they ate livestock meat and/or fish in the previous week/month/6 months (whichever was easier for them to remember). They were then asked to state whether or not poaching incidents had occurred in their villages in the previous 6 months. This was considered as an indication of poaching frequency in the study village. In the same vein, they were asked which months they thought most poaching activities took place. For the purpose of the present analysis, responses to this question were considered to be monthly poaching frequency. Direct questions about involvement in poaching were avoided because such questions normally receive considerably less cooperation from respondents (see Knapp *et al.*, 2010). The last portion of the interview dealt entirely with bushmeat consumption. As in the case of poaching frequency, villagers were asked whether they had consumed bushmeat in the last 6 months prior to survey. During a reconnaissance survey, it was established that asking about bushmeat consumption in this way was not only a proper approach in the study villages, but also helped interviewers to win respondents' confidence and cooperation.

In order to minimize challenges associated with the bushmeat survey (for example, respondents not saying the truth) arising from the illegal nature of wildlife poaching (Ndibalema & Songorwa, 2007; Knapp *et al.*, 2010), questions were preceded by a brief introduction about the purpose of the survey and the fate of the information gathered as well as requesting the respondent's participation. Also on arriving in each study village, the research team spent the first few days establishing rapport with villagers and their leaders prior to embarking on the survey. Personal observations (see DeWalt & DeWalt, 2002) were carried out to verify the responses. In this case, the survey team monitored closely people's daily activities along with collecting anecdotal information about bushmeat trade in a village before moving on to another study village.

Distance from the centre of the village (as agreed with the village chairman) to a closest point on the Ugalla Game Reserve boundary was estimated using a handheld global positioning system unit (Garmin GPSMAP® 60Cx). Rainfall data, recorded in different months, for 38 years (1970 – 2008) were obtained from Tabora Metrological Station.

Table 3.1 Study villages in Sikonge and Urambo districts, and their locations with respect to Ugalla Game Reserve (UGR).

District	Village	Population	Nearest UGR hunting block	Location distance (km)	Estimate of GPS location	
					Southing	Easting
Sikonge	Igalula	726	East	21	-5.63964	32.56006
Sikonge	Ipole	2638	East	35	-5.59757	32.68147
Sikonge	Mitowo	3037	East	19	-5.31702	32.28944
Sikonge	Mitwigu	1232	East	22	-5.73704	32.60934
Sikonge	Mole	4975	East	19	-5.27862	32.32281
Sikonge	Usanganya	4638	East	18	-5.20245	32.34206
Urambo	Izengabatogilwe	1300	East	16	-5.27812	32.25097
Urambo	Izimbili	3115	East	29	-5.29567	32.43158
Urambo	Nsogolo	1680	East	27	-5.30783	32.32788
Urambo	Isongwa	3295	West	19	-5.22683	32.25082
Urambo	Itebulanda	4471	West	17	-5.23858	32.12459
Urambo	Kangeme	2892	West	6	-5.42424	31.53757
Urambo	Kasisi	3183	West	21	-5.18467	32.17736
Urambo	Lumbe	3255	West	3	-5.50244	31.49984
Urambo	Nsenda	3527	West	18	-5.28276	32.14348
Urambo	Ukumbi siganga	3078	West	2.5	-5.49519	31.51862
Urambo	Usinga	816	West	4	-5.66954	31.28305
Urambo	Wema	3230	West	20	-5.20559	32.15693
Urambo	Zugimlole	8323	West	5.5	-5.35903	31.66463

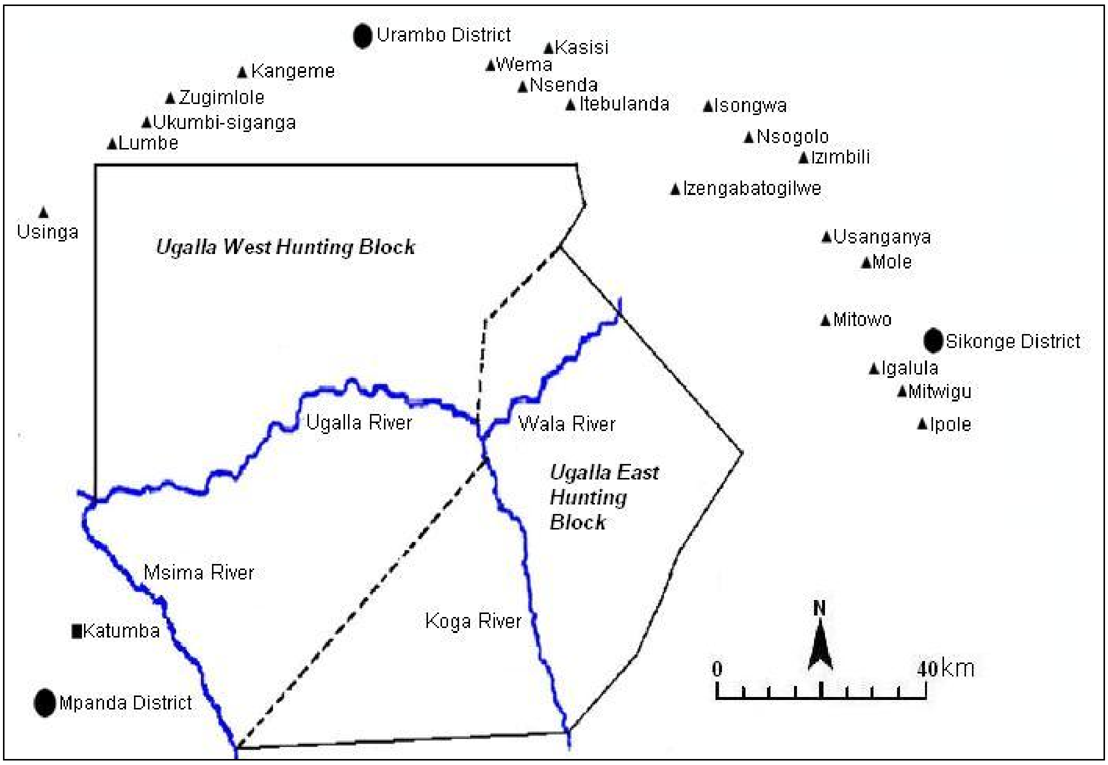

Figure 3.1 Ugalla Game Reserve. Thick line denotes the boundary. Dashed line demarcates the hunting blocks. Triangles represent the study villages. Mpanda, Sikonge and Urambo are districts surrounding the reserve. Katumba area in which the refugee camps are located is also shown. Meanders of lines show the main rivers.

Statistical analysis

All statistical analyses were carried out using the statistical package GenStat version 10 (Payne *et al.*, 2007). Descriptive statistics (mean and percentages) were used to describe sample characteristics. Generalised linear models, with appropriate error structures, were used to explore factors associated with wildlife poaching and bushmeat consumption. For some analyses the responses of individual households were used as the dependent variable while for others village average responses were used as appropriate for the level of the analysis. In all these models, the best predictors or determinants of a response variable were identified from a list of potential variables. This was done systematically by dropping predictors in order of their F-values, lowest first, until all remaining predictors contributed significantly to a model. Except in the case of monthly poaching frequency, the study village distance from Ugalla Game Reserve was controlled for in all the analyses. The level of significance for all tests was set as $\alpha = 0.05$.

Results

Respondents

I surveyed 573 households, 319 (56%) near Ugalla west hunting block and 254 (44%) near Ugalla east hunting block. More than half of the respondents (63%) admitted that there had been poaching incidents in their villages during the six months prior to the time they were interviewed. Thirty eight percent were in villages adjacent to Ugalla west, and 24% were in the villages adjacent to Ugalla east. Almost half of the respondents (46%) had consumed bushmeat in the six months prior to the survey. Of these, about 60% were from Ugalla west and 40% from Ugalla east. Other sources of protein were also available in the study area; for example, 54% of the respondents had consumed fish with mean frequency of 2.46 ± 0.17 month^{-1}. In addition, 76% of the respondents had consumed livestock meat in the previous six months with mean frequency of 2.78 ± 0.03 month^{-1}. The mean village distance from Ugalla Game Reserve was 16.89 ± 2.33 km (range 2.5 – 35 km). Study villages located east of Ugalla had a mean distance of 24.11 ± 2.0 km (range 16 – 35 km) whereas those adjacent to Ugalla west were located at a mean distance of 10.4 ± 2.3 km (range 2.5 – 21 km).

Wildlife species poached

Respondents mentioned a total of 40 wildlife species targeted by poachers. Species with very small frequencies were pooled under a single name; for example, different species of snakes and primates were named as "snakes" and "primates" respectively (Fig. 3.2). Generalised linear model (GLM) with binomial errors revealed that the frequency with which species were mentioned as being poached varied significantly across species (deviance $\chi^2_{39} = 22.15$, $p<0.001$), the most mentioned species being impala, followed by dik-dik and common duiker, whereas most of the small carnivores, primates and snakes were seldom mentioned (Fig. 3.2). I used a GLM with a normal error structure to test the influence of the village distance from the reserve and the village location with respect to

Ugalla east and Ugalla west hunting blocks (hunting block) on the number of species mentioned as being poached village^{-1}. The fixed terms were village location distance, hunting block, and their interaction. Only village location distance significantly predicted species mentioned as being poached ($F_{1, 18} = 28.33$, $p<0.001$), meaning that number of species mentioned declined with village location distance from the Ugalla Game Reserve boundary (Fig. 3.3). Hunting block ($F_{1, 17} = 0.98$, $p = 0.336$) and hunting block x village location distance ($F_{1, 16} = 1.06$, $p = 0.320$) were not statistically significant.

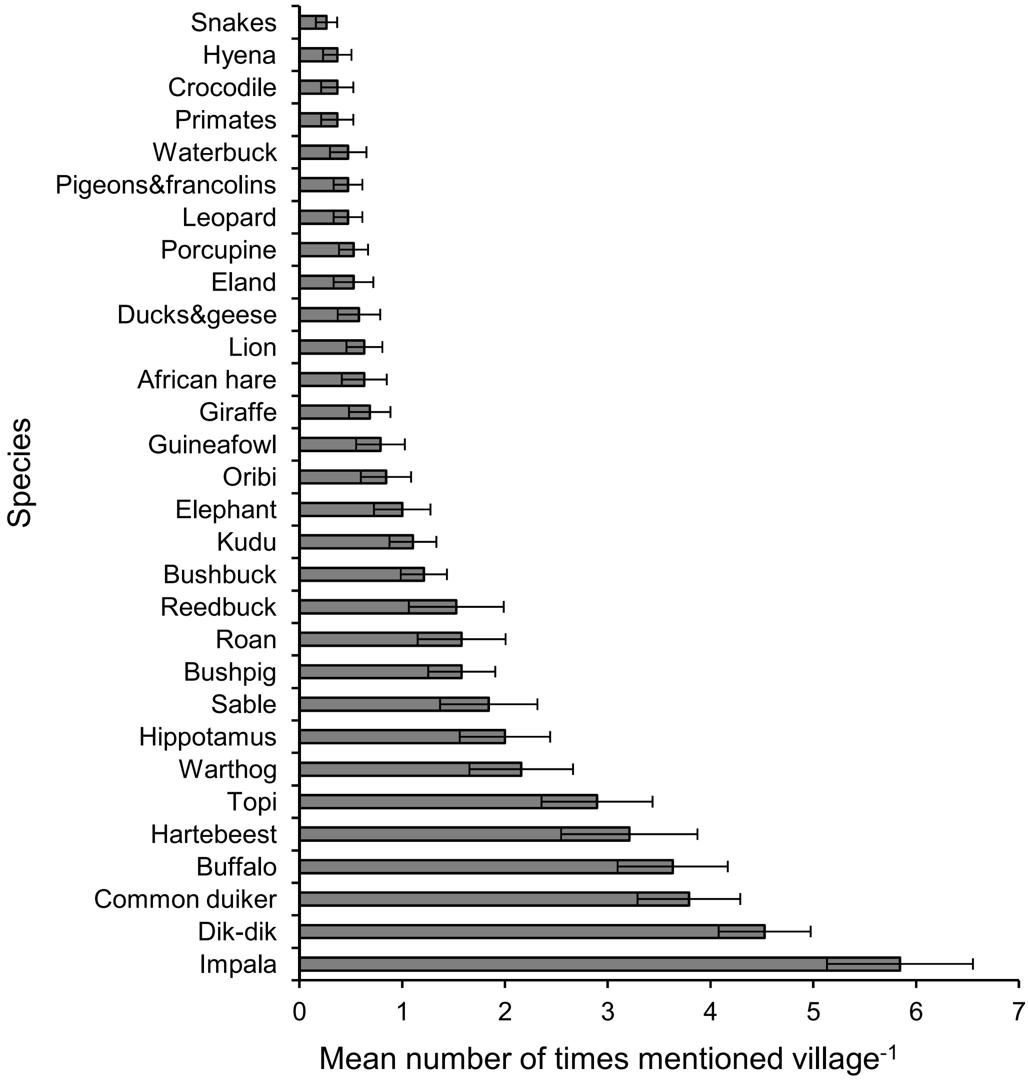

Figure 3.2 Frequency with which different species were mentioned by villagers as being poached. Error bars are the standard error of the mean.

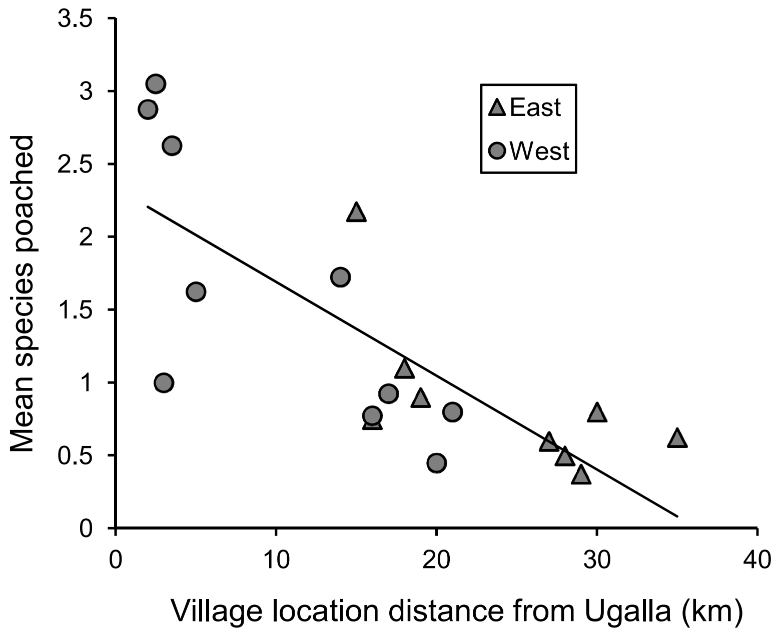

Figure 3.3 Relationship between location distance from the Ugalla Game Reserve boundary and mean number of species poached village^{-1} for villages neighbouring Ugalla east and west hunting blocks.

Poaching frequency

A binomial GLM was used to find the best predictors of mean village poaching frequency. The predictors (as shown in Table 3.2) were: village distance, hunting block, fish consumption – mean number of times respondents ate fish prior to the dates of the interview, village population – total number of residents in a study village (obtained from the village register in 2009), retained crop yield – amount of food crops in kgs. (maize, beans, groundnut, sunflower, cassava, potato, rice, sorghum, sesame etc.) kept during the survey dates, retained livestock – number of livestock (irrespective of species name) kept during the survey (mainly cattle, goat, sheep, chicken), livestock consumption – number of times respondents ate livestock meat prior to the dates of the interview. Poaching frequency had a significant negative relationship with village distance (Table 3.2). Most of the study villages adjacent to Ugalla west hunting block were closer to the reserve boundary than those adjacent to Ugalla east. Therefore, mean poaching frequency near Ugalla west exceeded that of Ugalla east. However, poaching frequency declined more rapidly with distance from Ugalla in the former than in the latter (Table 3.2 & Fig. 3.4).

Fish consumption also had a negative association with poaching frequency, meaning that villages with high fish consumption rates had low poaching frequency. Likewise, study villages with higher number of inhabitants and higher mean amount of retained food crops tended to have lower poaching frequency, but this was not statistically significant (Table 3.2).

Table 3.2 General linear model output showing the factors associated with wildlife poaching frequency among the communities around Ugalla Game Reserve.

	Estimate ± s.e.	d.f. (change, residual)	Deviance	Probability
Constant	2.00 ± 0.90			0.027
Village distance	-0.04 ± 0.03	1,14	32.10	<0.001
Hunting block (West)	1.70 ± 0.82	1,13	5.23	0.022
Fish consumption	-0.23 ± 0.10	1,13	4.88	0.027
Village distance x Hunting block (West)	-0.08 ± 0.03	1,13	6.53	0.011
Village population	-0.0001 ± 0.00007	1,13	3.19	0.074
Retained crop yield	-0.009 ± 0.008	1,13	3.03	0.082
Retained livestock	0.00854 ± 0.008	1,12	0.91	0.339
Livestock consumption	-0.072 ± 0.15	1,11	0.23	0.632
Hunting block reference level: East block				

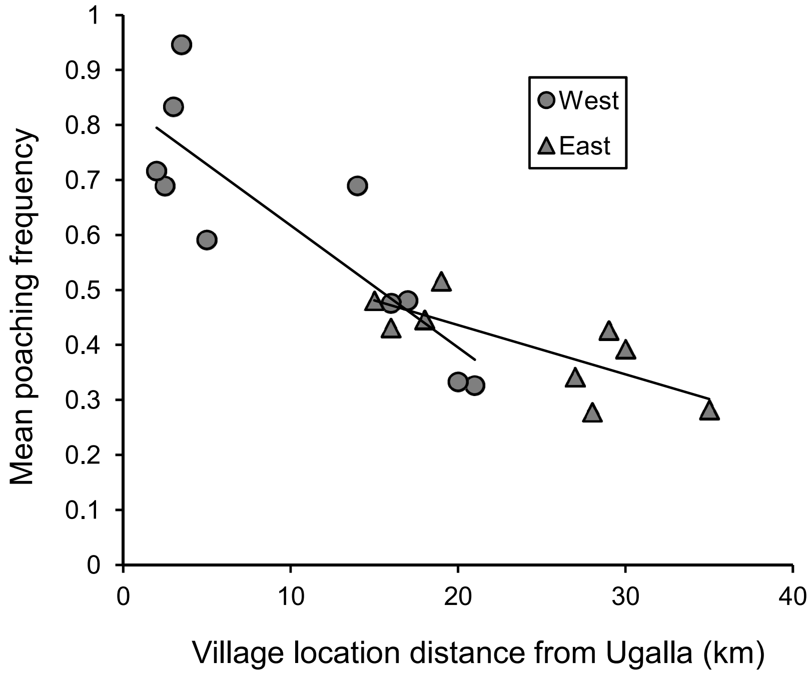

Figure 3.4 Relationship between mean poaching frequency and village distance from Ugalla for villages adjacent to Ugalla east and Ugalla west hunting blocks.

The influence of climate seasons and rainfall on wildlife poaching was explored using a separate model. Climate seasons are presented according to the distinction given by the Ugalla Game Reserve office in line with what is accepted by local people, so may not necessarily tally with the amount of rainfall. The amount of rainfall varied significantly between dry and wet seasons (GLM with normal errors, $F_{1,11} = 15.89$, $p = 0.003$), and between months ($F_{11,467} = 60.16$, $p<0.001$) (Fig. 3.5). Monthly poaching frequency was analysed using a GLM with a binomial error structure. The model included rainfall, season and month as fixed effects. Rainfall ($\chi^2_{11} = 19.88$, $p<0.001$) and season ($\chi^2_{1} = 10.71$, $p = 0.001$) were significant, but month ($\chi^2_{11} = 1.72$, $p = 0.071$) was not statistically significant. Monthly poaching frequency increased with amount of rainfall (Fig. 3.6). Overall, poaching frequency was also higher in the wet season than dry season (Fig. 3.7).

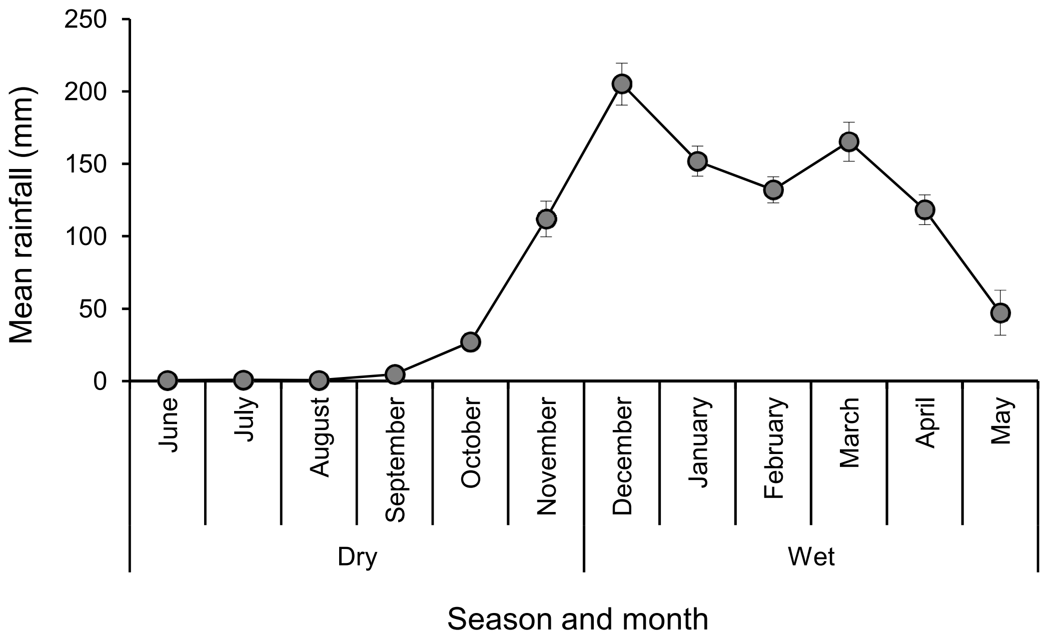

Figure 3.5 Mean annual rainfall (from 1970 – 2008) across different months and seasons in the Ugalla ecosystem. Error bars are the standard error of the mean.

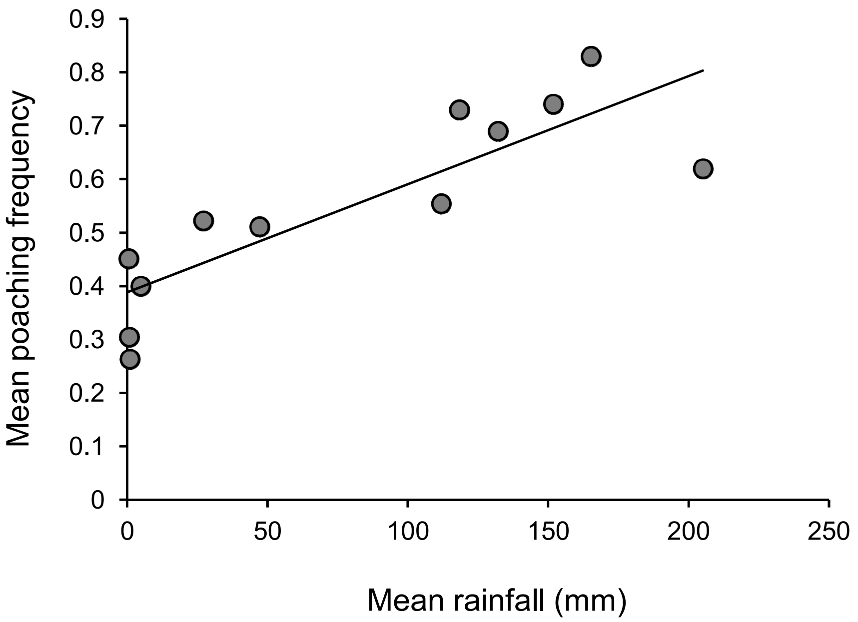

Figure 3.6 Relationship between mean poaching frequency and rainfall (1970 – 2008) across different months in the Ugalla ecosystem.

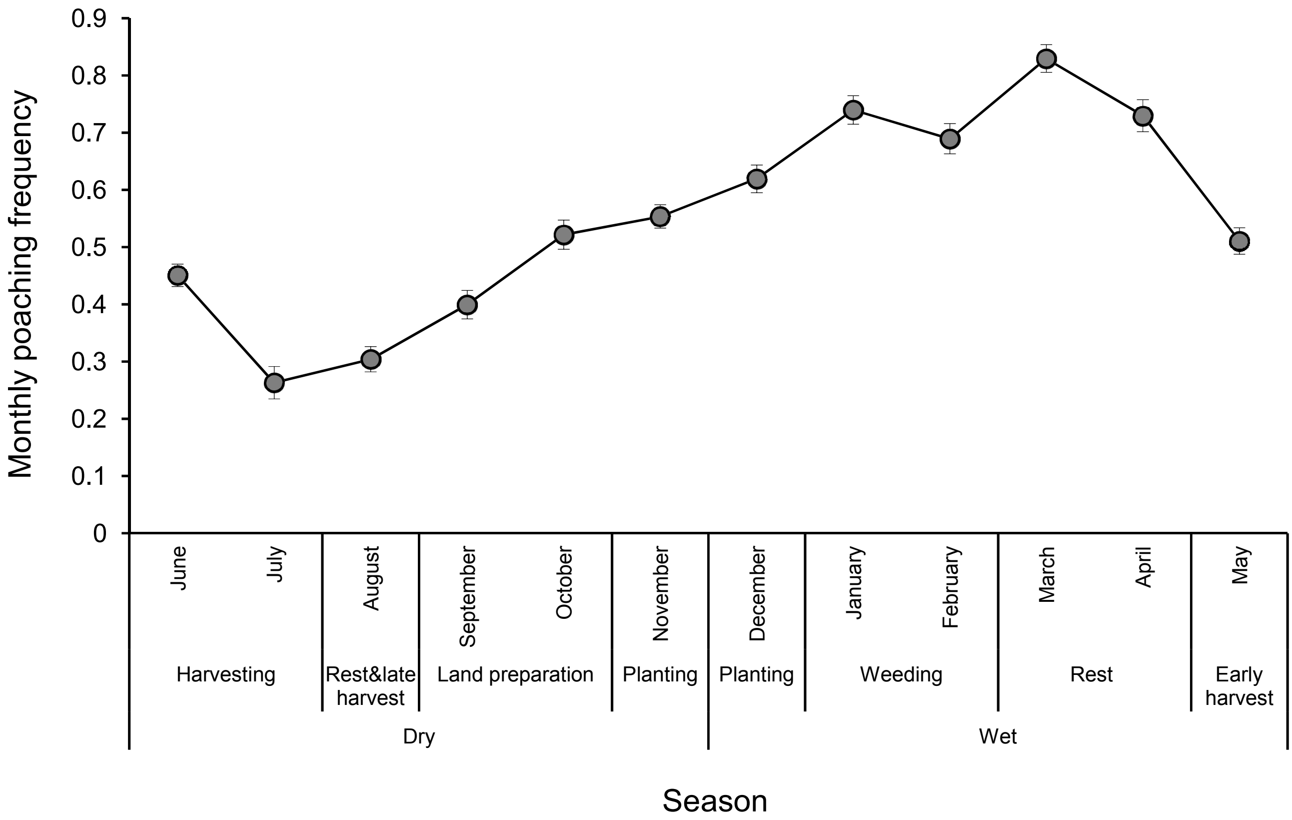

Figure 3.7 Poaching frequency between dry and wet seasons. Months and agricultural seasons are included for illustration purposes. Error bars are the standard error of the mean.

Bushmeat consumption frequency

Bushmeat consumption rate can be one of the best indicators of poaching intensity. A GLM with binomial errors was used to identify the best predictors of the bushmeat consumption frequency at both household and village levels. At a household level, the following parameters were tested: retained crop yield, retained livestock, fish consumption, livestock consumption, household income, household assets – the value of the productive assets in a household, household size – total members of a household, and family labour – members of a household aged 18 and above. Of the parameters tested, fish consumption frequency, crop yield, family labour, household income and household assets were significant predictors of bushmeat consumption (Table 3.3). Respondents who ate fish more often also ate bushmeat fewer times. Equally, villagers who kept quite large stocks of food crops claimed not to have consumed bushmeat in the period of 6 months prior to the interview. Consumption of bushmeat was more common among those families with more labour, high income and high value of household assets.

To find the best predictors of bushmeat consumption at a village level, all predictors tested under poaching frequency (see Table 3.2) were included in a GLM. Other predictors such as household size, family labour, household income and household assets (Table 3.4) were also included in the model. Generally, there were different bushmeat consumption trends between study villages adjacent to Ugalla east and west hunting blocks. Near Ugalla west, consumption decreased sharply with an increase in distance, the opposite was true for villages neighbouring Ugalla east (Fig. 3.8).

Table 3.3 General linear model output showing factors influencing household bushmeat consumption in the villages around Ugalla Game Reserve.

	(Estimate ± s.e.) x 10^{-3}	d.f. (change, residual)	Deviance	Probability
Constant	1076 ± 486			0.027
Fish consumption	-59.8 ± 10.1	1,568	47.62	<0.001
Retained crop yield	-0.19 ± 0.045	1,568	23.23	<0.001
Family labour	56.1 ± 61.1	1,568	5.66	0.017
Household income	0.17 ± 0.089	1,568	4.67	0.031
Household assets	48.5 ± 23.6	1,568	4.11	0.043
Retained livestock	-1.88 ± 1.58	1,567	1.45	0.229
Livestock consumption	-93 ± 141	1,566	0.48	0.486
Household size	20.6 ± 33	1,565	0.39	0.534

Table 3.4 General linear model output showing the influence of different predictors on the mean village bushmeat consumption.

	Estimate ± s.e.	d.f. (change, residual)	Deviance	Probability
Constant	-1.54 ± 0.49			0.002
Hunting block (West)	2.33 ± 0.53	1,18	14.50	<0.001
Village distance	0.05 ± 0.02	1,17	1.14	0.286
Village distance x Hunting block (West)	-0.11 ± 0.03	1,16	20.67	<0.001
Household size	-0.28 ± 0.27	1,13	2.54	0.111
Family labour	0.33 ± 0.29	1,11	1.49	0.222
Mean household income	-0.0002 ± 0.0004	1,12	1.21	0.272
Household assets	-0.02 ± 0.08	1,15	0.88	0.347
Fish consumption	-0.009 ± 0.11	1,15	0.49	0.485
Retained crop yield	-0.02 ± 0.009	1,15	0.12	0.730
Village population	-0.00003 ± 0.00008	1,15	0.09	0.761
Retained livestock	0.02 ± 0.009	1,15	0.08	0.772
Livestock consumption	-0.06 ± 0.18	1,10	0.06	0.810
Hunting block reference level: East block				

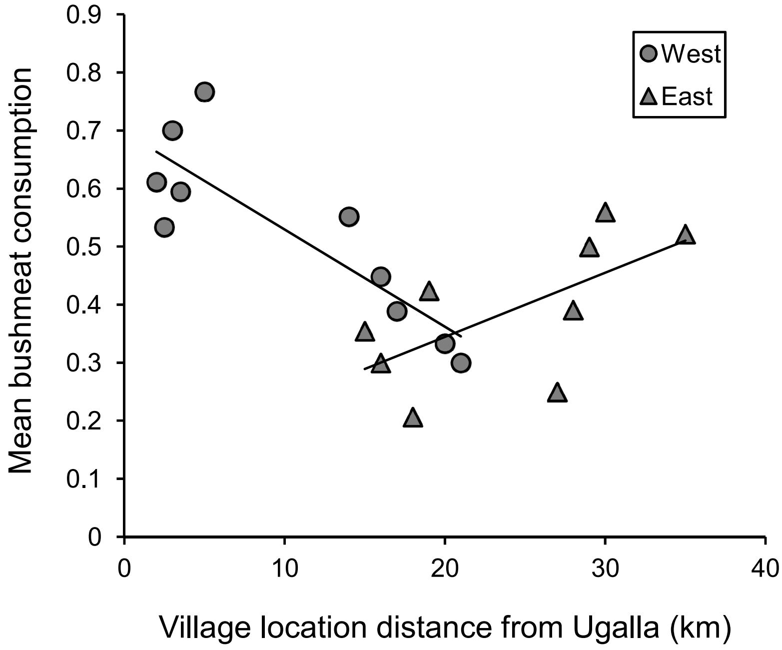

Figure 3.8 Relationship between location distance from Ugalla and mean bushmeat consumption in study villages neighbouring east and west Ugalla hunting blocks.

Discussion

This study has found that wildlife poaching and bushmeat consumption are strongly linked to the village distance from the Ugalla Game Reserve boundary in the Ugalla ecosystem. The relationship between wildlife exploitation and distance from protected areas has also been reported in other bushmeat studies elsewhere (Hofer *et al.*, 1996; Nielsen, 2006; Coad, 2007; Smith, 2008; Nyahongo *et al.*, 2009). The results of this study suggest that poaching frequency near Ugalla west hunting block is fairly high because villagers are closer to the reserve than in the Ugalla east. About half of the study villages adjacent to Ugalla west hunting block were within 5 km of the reserve boundary. These were the most problematic villages as far as wildlife poaching was concerned. Being closer to the reserve might be advantageous because of easy or cost effective access to wildlife resources. Elsewhere, the extent of poaching/bushmeat hunting is a trade-off between the cost involved in hunting (often time and financial resources) and proximity to a hunting area (for example, Nielsen, 2005; Coad, 2007).

Wildlife poaching

The first ten most poached wildlife species appearing in this study were the ones also commonly hunted for bushmeat elsewhere in Tanzania. For example, impala are commonly hunted in the Serengeti ecosystem (Setsaas, *et al*., 2007). Caro (2008) highlighted some species favoured by poachers in the Katavi-Rukwa ecosystem, among them were: buffalo, warthog, hippopotamus and bushpig. In the areas around Urumwa Forest Reserve in western Tanzania, common duiker and dik-dik are the primary species hunted for bushmeat (Carpaneto & Fusari, 2000). The large number of bushmeat species mentioned as being poached is an indication that wildlife is among the primary sources of animal protein in the Ugalla ecosystem.

Low poaching frequency among villages near Ugalla east, in Sikonge district, is also possibly due to active participatory conservation approaches. Currently there is an international non-governmental organisation called Africare, which has spearheaded the development of a successful wildlife management area (WMA), namely Ipole (2,500km^2) (Nelson, 2007). Ipole WMA encompasses a large part of Ugunda Game Controlled Area, and is broadly aimed at reducing poaching amongst local communities in Sikonge. According to Wilfred (2010), local communities are one of the main stakeholders in the utilisation of wildlife resources within WMAs. Therefore, WMAs have a potential to slow down poaching when the bottlenecks to their success are effectively dealt with. A caveat here is that WMAs may not deter villagers from buying and/or consuming bushmeat poached elsewhere.

Other factors associated with poaching frequency in the study area included rainfall and fish consumption. Rainfall determines the pattern of agricultural activities across different months, thus influencing wildlife poaching. A close relationship between rainfall, agriculture and poaching was also reported in Serengeti by Barrett & Arcese (1998). Poaching is least common at harvest time, when villagers are busier and food is most abundant, and most common immediately before harvest when villagers have little to do and food is scarce. During periods of rain, villagers depend predominantly on food stocks accumulated in preceding cropping seasons. Owing to the fact that a rainy season may last for 5-6 months, food stocks are always inadequate or quickly depleted resulting in villagers' increased dependence on wildlife resources (URT, 1998*a*). Additionally, rains influenced some other factors related to poaching; for example, animals dispersing outside the reserve, thereby encountering villagers who are desperately in need of animal protein (Carpaneto & Fusari, 2000). Rainfall seasons also pose major setbacks to anti-poaching efforts in the Ugalla ecosystem. The Ugalla Game Reserve headquarter is in Tabora town, which is more than 100 km from the reserve. Anti-poaching teams have to travel regularly from Tabora town to the reserve in specific periods throughout the year. The wet season is the most difficult time of the year for game rangers to access the reserve as many roads become muddy and impassable (FCF, 2008). It is this time of the year when poachers devastate wildlife populations in Ugalla, taking advantage of the patrol teams' infrequent visits and poor coverage of their operations within the reserve (see WD, 1998).

The importance of fish as an alternative source of animal protein cannot be overemphasised. Fish among local communities in the Ugalla ecosystem come from two main sources. Firstly, through licenced fishing within Ugalla Game Reserve. This is the only game reserve in Tanzania where the surrounding local communities are allowed (through permit) to carry out fishing and honey gathering activities each year, but in specific seasons (normally from July – December). These activities are aimed at furnishing local people with alternative sources of protein or income (WD, 1998; UGR, 2006). A second source is the fishing areas (rivers) outside the reserve. These are largely controlled by the districts' fishery offices, and may be subject to substantial illegal and unsustainable fishing. Fish consumption was very important in reducing poaching and bushmeat consumption at village and household levels respectively. This observation contrasts with the findings of a study in Serengeti (Nyahongo *et al.*, 2009). These authors found that fish were not a viable alternative to bushmeat when the latter was cheaply and abundantly available. However, unlike in the Lake Victoria basin, fishery in Ugalla is dominated by locally-based subsistence fishermen, and this leads to high availability of fish in the area. Elsewhere, Brashares *et al.* (2004) found that the increase in fish supply reduces bushmeat utilisation in west Africa.

Bushmeat consumption

While poaching frequency and village distance were positively correlated among the local communities adjacent to both hunting blocks, bushmeat consumption around Ugalla west hunting block was common in villages close to the reserve, but the opposite was true in the

Ugalla east. There are some reasons that might be causing this particular pattern. First is the distance from the closest large town (Tabora city centre) to the study villages. Most of the study villages near Ugalla east are found in the Sikonge district which is 68 km/42 miles from Tabora, and the ones near Ugalla west are found in the Urambo district which is 83 km/52 miles from Tabora. The presence of better road and shorter distance between Sikonge and the city centre might have perpetuated a market chain for wildlife products between the two areas, which encourages bushmeat consumption as it moves towards the city centre or away from the reserve. Such bushmeat supply chains are explained in Caro & Andimile (2009) who argued that where there is an active market, present poachers will normally sell bushmeat to middlemen and the latter trade it in towns. There are two main roads that connect Tabora to other regions such as Mbeya and Mpanda. The two roads meet in Ipole village to form one road extending to Tabora town centre. According to Mr. K. Twaha (Game Officer in Sikonge district), this road has facilitated the flourishing of the market for natural resources including bushmeat, timber and many other wildlife products, since it leads to reliable and easily accessible means of transport from Sikonge to the town centre. Although the local government in Sikonge has raised two natural resources inspection roadblocks (operating for 24 hours) in Ipole and Pangale villages, their effectiveness is questionable especially at night. Most of the bushmeat consumers come from Mole, Mitowo, Ipole, Chabutwa, Tutuo, Udongo and Sikonge villages (K. Twaha pers. comm.). The last four villages were not part of this study.

This study has shown that factors related to household wealth/purchasing power; for instance: household income, family labour and household assets have positive effects on bushmeat consumption. Caro & Scholte (2007) also found that bushmeat consumption increases with the rise in people's living standards/wealth. The presence of a large local hunting concession area (approximately 2,101,500ha) (K. Twaha pers. comm.; see also Chapter 6) coupled with a high purchasing power as a result of higher incomes (Wilfred & MacColl, 2010) has probably encouraged a high turnout in purchasing local hunting licences in Sikonge. This made it difficult not only to distinguish between poached and legally obtained bushmeat in the district, but also to ascertain whether bushmeat comes from wildlife management areas, game controlled areas, or Ugalla Game Reserve. Therefore, unlike Urambo district, villagers in Sikonge were more reluctant to admit the presence of poaching activities in their areas than they were to state that they had consumed bushmeat.

When surveying Mole, Mitowo and Usanganya villages, the research team spent a considerable amount of time at Tutuo and Sikonge town centres. These are the most popular business centres along the Ipole-Tabora road. Most timber and bushmeat deals are carried out at these centres. There is a road that connects the two centres to Urambo district at a place called Ussoke. It was learned that many middlemen in the timber and bushmeat trade at the aforementioned centres have established contacts with remote villages in Sikonge and some parts of Urambo as well. As a result (around Tutuo, Sikonge and also Ipole centres) villagers would not admit to the presence of poachers partly for fear of damaging what was perceived to be a well-paid bushmeat and timber business. Olupot *et al.* (2009) also reported such secretive networks of the bushmeat trade (which included poachers, middlemen/dealers and consumers) in Uganda.

Reasons for bushmeat consumption in villages close to Ugalla west differed from Ugalla east. Those within 5 km from the boundary consumed more bushmeat. As pointed out by the respondents, the main reasons for consumption included being given bushmeat by game scouts and other workers from some tourist hunting companies when they visited the villages for official or private reasons. Another source of meat for villagers was from their fellows who hunted in areas immediately outside the reserve (most of these were actually agricultural lands). Such hunters did consider that they were poachers, and some villagers thought that was a good way to scare off crop raiding animals. Hunting of problematic animals by villagers has also been reported in Serengeti (Kaltenborn *et al.*, 2005). Ugalla has a long history of ungulates going out of the reserve in search of palatable grazing during the rainy season (when the reserve often floods) and a short period afterwards (Thomas, 1961), but in most cases they come across village farms which are very near the reserve boundary, and hunters take advantage of the situation under the umbrella of protecting their crops. Caro (1999*a*) noted that this could be one of the reasons for very low densities of species outside protected areas in the Katavi ecosystem, in the Mpanda region, of western Tanzania. Wild ungulates from Ugalla used to move even longer distances towards human settlements; for example, they were reportedly killed by villagers in the areas around Urumwa near Tabora town (Carpaneto & Fusari, 2000). Currently, most of the miombo woodlands outside the reserve have been severely degraded as a result of unsustainable crop farming, extensive grazing and ad hoc resettlement of local people (URT, 1998*a*; UGR, 2006), it is unlikely for animals from Ugalla to move over such longer distances.

Consumption decreased in the villages within 10-20 km of the boundary on the northern side of the reserve (adjacent to Ugalla west hunting block). These villages were close to Urambo town centre, and there was no reliable transport from Urambo to villages much closer to the reserve. Unlike Ugalla east, transport infrastructure do not appear to contribute to the bushmeat trade and the consumption pattern in the villages adjacent to Ugalla west hunting block. Nonetheless, since poachers make use of bicycles as their primary means of transport (see Chapter 5), I cannot rule out the possibility that bushmeat consumption frequency might have been underreported by the villagers adjacent to Ugalla west.

At a household level, crop farming was among the factors that were associated with reduced consumption of bushmeat. Retained agricultural crop yield provided a safety net for villagers by ensuring food security (see also de Klerk *et al.*, 2004). Villagers with adequate stocks of food crops had significantly low dependency on wildlife resources. Elsewhere, Shrestha & Alavalapati (2006) also found that agriculture lessened farmer dependency on wildlife in Nepal. Fa *et al.* (2003) pointed out that improving subsistence agriculture would bring down demand for bushmeat in rural areas.

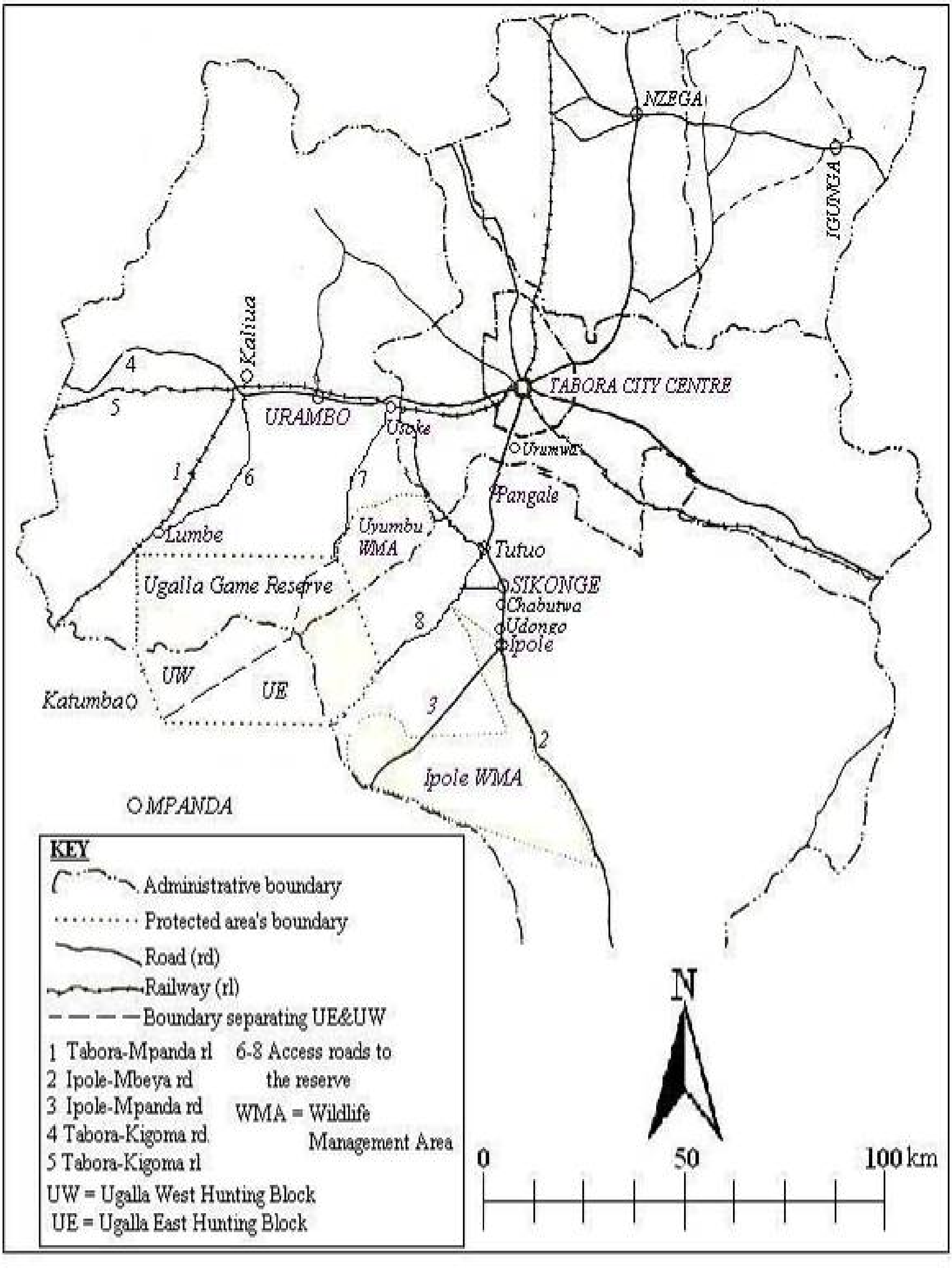

Figure 3.9 Tabora map showing different places and transport infrastructure (roads and railways) mentioned in the text.

Conclusions

This chapter has revealed some important factors when seeking ways to strengthen the conservation of Ugalla Game Reserve. Results indicate that wildlife poaching activities are largely carried out by villagers closer to the reserve. Currently, there are some forms of participatory conservation approaches through licenced local fishing and honey gathering activities, and wildlife management areas aimed at sensitizing people around Ugalla to the importance of conservation. Community conservation service (see Emerton & Mfunda, 1999) in conjunction with wildlife management areas would speed-up realisation of conservation benefits among local communities; for example, in northern Ugalla where wildlife management areas and other local hunting concession areas are few. Nevertheless, the introduction of community conservation services needs further research to understand its sociological, cultural and economic implications.

Rainfall seasons in conjunction with agricultural cycle influenced poaching activities. Ineffectiveness of anti-poaching operations in wet seasons is a serious matter in the conservation of Ugalla. Accessibility to the reserve should be enhanced through shortening of the distance to the reserve and improving access road conditions to cope with wet seasons. The reserve management team has done its best to construct and maintain three active game posts at Ipole, Ussoke and Lumbe. Strengthening these game posts and ensuring adequate means of transport would boost anti-poaching activities in the ecosystem. There is a Wildlife Division's anti-poaching unit in Tabora (popularly known as KDU-Tabora – a Swahili abbreviation for Kikosi Dhidi Ujangili-Tabora) which oversees all anti-poaching activities in western Tanzania. I think that KDU-Tabora is already weighed down as it currently covers a large area (for example, Tabora, Kigoma, Mpanda and Kahama) with few staff. Among the areas with notorious poachers is Katumba refugee camp in Mpanda (not covered by this study). It is enormously challenging to deal with illegal exploitation of wildlife in the "refugee hosting areas" (Jambiya *et al.*, 2007). In fact, extensive and thorough assessment of the effectiveness of anti-poaching activities in the Ugalla ecosystem needs to be carried out, and practical recommendations developed and implemented.

Fish provided the best alternative to bushmeat at a household level. In order to ensure sustainability of fisheries resources in the area, aquaculture production should be promoted. Mwangi (2008) defined aquaculture as "the growing (farming) of fish and other aquatic organisms in controlled environments". The author argued further that aquaculture is a great replacement of wild fish stocks during times of fish scarcity. Plans to ensure sustainable food security, in the periods of low food crop harvests, for villagers around Ugalla are also of supreme importance.

CHAPTER 4: INCOME SOURCES & THEIR RELATION TO WILDLIFE POACHING

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Abstract

In the Ugalla ecosystem, wildlife conservation is constantly and pervasively challenged by the local communities looking for ways to improve their livelihoods. The need to curb illegal hunting (or poaching) of wildlife continues to spark debate amongst conservation stakeholders in the area. Assessing contributions to livelihood of different sources of income in light of wildlife poaching is vital to inform conservation efforts in Ugalla. The heads of households in villages bordering Ugalla Game Reserve (an integral component of the Ugalla ecosystem) were interviewed to obtain data on poaching and income sources. Income from crops (tobacco, maize and groundnut), and livestock (cattle) was positively correlated with household income, but also with decreasing poaching frequency. Other economically important crops were rice, sesame and sunflower, although these were not significantly correlated with wildlife poaching. Household income from other sources (wildlife, forests, small businesses, formal employment and remittances) were not significantly associated with wildlife poaching. Villages with lower mean income had higher poaching frequency. Additionally, those closer to Ugalla Game Reserve tended to have higher poaching frequency than the ones further from it. Improving agricultural production would help to lessen pressure on wildlife resources in Ugalla.

Introduction

A large proportion of rural communities in the developing world depend on renewable natural resources such as forest products (Butler, 2006) and wildlife (TNRF, 2008) for their livelihoods. These resources are a basic safety net for the rural people. Forest as an alternative source of income offers a range of timber and non-timber products, such as fuel wood, honey, beeswax, building poles, fodder resources, fruits and medicinal plants (Sunderlin *et al.*, 2005; Giliba *et al.*, 2010). Rural communities also depend on wildlife-based products such as bushmeat, fur, skin, claws, horns and teeth as sources of income and/or protein (LWAG, 2002; Pattiselanno, 2004; Bennett *et al.*, 2006; Carpaneto *et al.*, 2007). Because of the rapidly accelerating human population, exploitation of natural resources has increased (Songorwa, 2004; Wilfred, 2010); thus, the need for effective conservation measures to balance people and wildlife needs.

The main approach to conservation until the end of the 20th century has been the establishment of protected areas (Johannesen, 2007). A protected area is "a clearly defined geographical space recognized, dedicated and managed through legal or other effective means to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (UNEP-WCMC, 2008). Certain types of protected areas provide alternative sources of income to local people mainly through tourism, but in some cases can deny local people access to these resources (Roe & Elliott, 2005). The latter can be a problem because it can lead to poor support, by local communities, for the principle of conservation (Shemwetta & Kideghesho, 2000; Arjunan *et al.*, 2006; Allendorf, 2007). Conservation has therefore found itself at crossroads between meeting the demands of local people for sustainable livelihoods, and ensuring the preservation of natural resources (Roe & Elliot, 2005).

From a wildlife conservation perspective, unbalanced relationships between protected areas and local communities perpetuate illegal hunting activities (Wilfred, 2010). Such hunting activities are referred to as wildlife poaching since they are carried out regardless of whether the wildlife laws permit them. Wildlife poaching is often unsustainable and is mainly done to harvest bushmeat although it may also involve small-scale trade of by-products such as skins, horns, teeth and claws (Taylor & Dunstone, 1996). A variety of different income-based factors behind bushmeat exploitation have been put forward, apparently mostly location-specific and thus operating at a local scale. For example, Coad (2007) found that rich households dominated commercial use of wildlife in Dibouka and Kouagna villages in Gabon, precisely because they had the resources necessary to invest in the bushmeat exploitation. Loibooki *et al.* (2002) found that keeping fewer livestock, in particular goats and sheep, coupled with a "lack of alternative income sources" were the main reasons for an increased dependency on wildlife in the Serengeti ecosystem. Shrestha & Alavalapati (2006) established that lower agricultural incomes were one of the main reasons for an increase in dependency on wildlife in the Koshi Tappu Wildlife Reserve of Nepal.

In the case of Tanzania, wildlife poaching is increasingly becoming a problem (Carpaneto & Fusari, 2000; Baldus, 2002; Holmern *et al.*, 2004; Rustagi, 2005; Caro, 2008). One of the predominant reasons for poaching is the need to improve living standards (see Caro & Scholte, 2007; Kideghesho, 2008). The government of Tanzania is determined to improve the livelihoods of its people by integrating different local sources of income sustainably; for example, agriculture, wildlife, forests, and small-scale businesses (URT, 1998*b*; 2005). Assessing the relative importance of such sources of income in the light of wildlife poaching would reveal priority options for both improving livelihoods and minimising human pressure on wildlife protected areas.

This chapter explores the relative contribution of different sources of income to the livelihoods of rural communities, and how this relates to wildlife poaching around Ugalla Game Reserve, western Tanzania. Ugalla Game Reserve was first occupied by Wagalla people in the 1950s, who were hunters, fishermen and honey gatherers. These people were allowed to carry out their livelihood activities in the reserve until 1965, when the area was officially gazetted as a game reserve. Owing to increased pressure on wildlife resources, all unauthorised use of resources were prohibited within the game reserve and local people were forced to move out occupying the neighbouring areas (UGR, 2006). Demand for wildlife resources in the area then continued to create conservation challenges despite the availability of alternative sources of livelihoods. Regrettably, the relationship between different livelihood opportunities (or sources of income) and wildlife poaching has received far less attention. The results presented here suggest opportunities for reducing poaching through improving living standards of people in Ugalla.

Methods

Study area

The study was conducted in the Sikonge and Urambo districts, western Tanzania. These districts contain a substantial part of the Ugalla ecosystem, in which Ugalla Game Reserve is a key component. The area falls between 4^0 – 7^0 South and 31^0 – 34^0 East, with an altitude ranging from 1100 – 1300 m above sea level. The land areas of Sikonge and Urambo districts are 21000 km^2 and 21299 km^2 respectively (URT, 1998*a*). According to the 2002 population census, Sikonge District had a population size of 132,733, and Urambo District 369,329. In general, the human population of the Tabora region is among the fastest growing in Tanzania with a growth rate of 3.6% (NBS, 2002).

The climate is defined by a distinct wet season from December – May, and a dry season from June – November. Rainfall varies between 700 – 1000 mm per year, and the mean maximum and minimum temperatures are between 28 – 30 ^{0}C and 15 – 21 ^{0}C respectively (Mbwambo, 2003; Hazelhurst & Milner, 2007). As in many other rural areas in Tanzania, the livelihoods of the local people around Ugalla Game Reserve rely fundamentally on a mixture of activities such as keeping livestock, crop farming, fishing, hunting, beekeeping, and the harvesting of forest products (UGR, 2006). Rain–fed agriculture plays a central role in the people's livelihoods, but soil fertility is relatively low (URT, 1998*a*; Hazelhurst & Milner, 2007). Popular crops grown in the area include maize, cassava, sweet potatoes, rice, groundnuts, tobacco, and sunflower (Kikoti, 2009).

Data collection

Following the theory for sampling techniques by De Vaus (2002), a sample of 19 study villages was drawn randomly from a total of 122 villages (15% sampling intensity) in the Sikonge and Urambo districts. The study villages were Isongwa, Kangeme, Lumbe, Nsenda, Ukumbi-Siganga, Usinga, Zugimlole, Mole, Izengabatogilwe, Igalula, Ipole, Mitowo, Mitwigu, Wema, Usanganya, Kasisi, Izimbili, Nsogolo and Itebulanda. Data were collected from these villages through structured questionnaires completed by interviewing the heads of 573 randomly selected households (out of about 11000 households in all the study villages: a sampling intensity of 5.2%) in the period from March – October, 2009. Random sampling was adopted in order to ensure that the estimated parameters (for example, income and poaching in this case) represent the population as adequately as possible (Levy & Lemenshow, 1999; De Vaus, 2002).

The survey gathered information on income generation through seven sources (see Table 4.1) in addition to wildlife poaching. Firstly, respondents were asked about production and sale of their crops and livestock in the preceding harvest season. They were then asked to estimate income from small business, formal employment, forest- and wildlife-based products, and remittances in the previous 6 months. Additionally, the second portion of the survey encompassed questions of direct relevance to wildlife poaching. Respondents were asked to state whether or not poaching incidents had occurred in their villages in the previous 6 months. Responses concerning poaching were used as an indication of poaching frequency or intensity. Elsewhere, wildlife researchers have also obtained information on wildlife exploitation from the views of local people (see Holmern *et al.*, 2004; Ndibalema & Songorwa, 2007; Caro, 2008; Rist *et al.*, 2008; Smith, 2008). The assessment of wildlife poaching is enormously challenging not only because of its illegal nature, but also because capturing reliable information depends chiefly on the ability of respondents to remember any of the details (Knapp *et al.*, 2010). But, due to good rapport established with the villagers, most were willing to report poaching activities in their villages anonymously. In addition, following DeWalt & DeWalt (2002), participant observation was carried out to verify various answers provided by the respondents. In the data analysis and interpretation of the results, I was also mindful of any potential bias associated with the survey. For each of the study villages, the distance from the centre of the village (as agreed with the village chairman) to the closest point on the Ugalla Game Reserve boundary was estimated using a hand-held GPS unit.

Statistical analysis

All statistical analyses were carried out using the statistical package GenStat version 10 (Payne *et al.*, 2007). Non-parametric Kruskal-Wallis one-way analysis of variance was used in the comparison of different sources of income in order to determine their relative importance. The rest of the analysis was done using generalised linear models (GLMs). The interest here centred on the comparisons of household income across crops, livestock species and study villages. The relationship between different sources of income and the frequency of wildlife poaching was also investigated. The best predictors or determinants of household income were identified from a list of potential variables: age of a respondent, household size, family labour (number of household members aged >18 years), productive assets (e.g. significant items of agricultural equipment), tribe and level of education (number of years of formal schooling). This was done systematically by dropping predictors in order of their F-values, lowest first, until all remaining predictors contributed significantly to the model. Thus the minimum sufficient model is presented in the tables below.

Results

Respondents' characteristics & *income sources*

The majority of the households surveyed had male heads (69.3%). Average household size (± s.e.) was 9.1 ± 0.24 people. The age distribution revealed that 28.8% of the respondents were young adults between 18 and 35 years old, 46.8% were aged between 36 and 55 years, and 24.4% were more than 56 years of age: the mean age was 45.3 ± 0.63 years. Of the respondents, almost 40% had no formal education, 60% had acquired primary education ranging from 1 to 7 years of schooling, and 0.03% had achieved secondary education. There were no respondents with college and/or university education. On average, the years of formal education were 3.5 ± 0.13. The largest proportion of the respondents (37.5%) belonged to the Sukuma tribe, whereas 29.7% and 15.2% were members of the Nyamwezi and Muha tribes respectively. There was a total of another 22 tribes (17.6%) in the area: Bemba, Bende, Bungu, Chaga, Fipa, Gogo, Haya, Hehe, Kimbu, Kanonko, Lungwa, Lwila, Wagalla, Gogo, Hyao, Wajita, Ngoni, Nyakyusa, Nyaturu, Nyiramba, Pimbwe and Tutsi.

All respondents (99%) practiced small-scale farming, 80% of whom sold some of their produce. About 90% of the respondents kept livestock, 33% of whom sold some of them. Other sources of income; for example, forests, wildlife, small business, remittances and formal employment, provided additional income to 38.4%, 15.7%, 8.6%, 7%, and 4.9% of the respondents, respectively. The mean self-assessed household annual income was U.S. $

967 ± 59, composed of income derived from various sources. Income sources differed significantly in their contribution to the total (Kruskal-Wallis test: χ^2 = 1473, d.f. = 6, p<0.001, Table 4.1); the most important one being crop sales, followed by livestock, whereas remittance income was the least one.

Table 4.1 Mean income (U.S. $) ± standard error (s.e.) from different sources in study villages around Ugalla Game Reserve, western Tanzania. Number of households that obtained income from different sources (n) and the total number of observations (N) are shown.

Income source	n	Income[¶] ± s.e.	Income[§] ± s.e.(N = 573)
Crop sales	460	1057.63 ± 67.64	849.05 ± 57.10
Livestock	173	249.45 ± 35.44	75.31 ± 11.70
Forests[1]	220	8.87 ± 0.87	21.19 ± 2.69
Small business[2]	49	147.31 ± 14.93	12.60 ± 2.14
Formal employment[3]	28	134.24 ± 24.45	6.56 ± 1.69
Wildlife[4]	90	134.89 ± 11.17	3.41 ± 0.38
Remittances[5]	40	33.62 ± 3.88	2.35 ± 0.45

Official exchange rate in 2009: 1 US dollar = 1300 Tanzanian Shilling (TZS). ¶Income divided by "n". §Income divided by "N". [1]Forest-based products: timber, charcoal, building poles, ropes, firewood, honey, beeswax, medicinal plants. [2]Self-employed activities: carpentry, local village midwifery and traditional healing practices; day labourers on farms; selling fruits, vegetables, fishes, soft drinks and local alcoholic drinks; kiosks; and maize mills. [3]Formal employment: primary school teaching, village healthcare practitioners, village agricultural extension officers, village executive officers, and working with non-governmental organisations. [4]Wildlife-based products: bushmeat, teeth, claws, skins, skulls, feathers, horns, jaws, and other bones and organs. [5]Money sent home by children and/or other relatives working in towns or other regions.

Crop & livestock sales

Owing to the central role played by crops and livestock in the household economy, I quantified the contribution of each element to household income. Table 4.2 presents the estimates of income for crops grown in the study area. Maize was the most commonly grown food and cash crop (98% of households), followed by groundnut (90%), whereas other crops such as cowpea, green gram and millet, were not common in the study area. Tobacco was grown exclusively for sale by 37% of the respondents. A GLM with a normal error structure showed that seven crops were important in determining household income (Table 4.4). The most profitable crop was tobacco, followed by groundnut, whereas others; for example, sorghum, beans and cassava had lower contribution to the household income.

In the case of livestock, common species were chicken, goat and cattle (Table 4.3). Only cattle had significant positive impact on the household income (Table 4.4). Other livestock; for example, goat, sheep and duck, had low contribution to the household income.

Table 4.2 Crops produced and sold annually in study villages around Ugalla Game Reserve, western Tanzania.

Crop	Respondents (%)	Mean amount produced (kg) ± s.e.	Mean amount sold (kg) ± s.e.	Mean income (U.S. $) ± s.e.
Tobacco	37	418.2 ± 35.5	418.2 ± 35.5	658.3 ± 54.0
Groundnut	90	1097.9 ± 65.0	568.7 ± 50.4	70.5 ± 7.3
Maize	98	1313.5 ± 77.2	285.3 ± 33.2	51.6 ± 6.4
Rice	28	467.3 ± 71.3	194.1 ± 36.6	43.9 ± 8.3
Sesame	10	19.3 ± 7.3	16.3 ± 3.4	17.8 ± 3.7
Sunflower	18	92.0 ± 20.8	38.3 ± 13.5	12.9 ± 1.3
Potato	32	152.9 ± 15.4	27.8 ± 6.3	7.6 ± 0.6
Cassava	40	238.4 ± 27.2	14.4 ± 4.6	1.1 ± 0.4
Beans	16	19.6 ± 5.3	1.3 ± 0.6	0.9 ± 0.4
Sorghum	10	40.2 ± 7.3	3.6 ± 1.9	0.7 ± 0.5
Other*	3	23.1 ± 7.9	5.5 ± 3.3	0.09 ± 0.04

Official exchange rate in 2009: 1 US dollar = 1300 Tanzanian Shilling (TZS). *includes cowpea, green gram and millet.

Table 4.3 Ownership of livestock in study villages around Ugalla Game Reserve, western Tanzania.

Livestock	Respondents (%)	Mean owned	Mean sold	Mean income (U.S. $)
Cattle	32	10.7 ± 1.4	0.4 ± 0.1	64.2 ± 11.2
Goat	42	5.1 ± 0.5	0.3 ± 0.1	5.9 ± 1.3
Chicken	81	15.6 ± 0.7	1.2 ± 0.2	3.9 ± 0.6
Sheep	13	0.9 ± 0.1	0.01 ± 0.004	0.2 ± 0.1
Duck	72	0.5 ± 0.1	0.02 ± 0.01	0.07 ± 0.04

Official exchange rate in 2009: 1 US dollar = 1300 Tanzanian Shilling (TZS).

Table 4.4 Results of a GLM examining the association between the overall household income and income from important crops and livestock. d.f. (change, residual) = 1,565.

	(Estimate ± s.e.) x 10^{-3}	F- value	Probability
Constant	1152 ± 358		<0.001
Tobacco	12.85 ± 0.23	3211.66	<0.001
Groundnut	18.72 ± 1.86	100.79	<0.001
Cattle	16.4 ± 1.12	78.80	0.001
Maize	13.64 ± 1.65	60.33	0.011
Rice	20.38 ± 3.03	42.37	0.021
Sesame	12.64 ± 2.35	28.95	0.028
Sunflower	28.68 ± 7	19.30	0.033
Potato	41.4 ± 20.5	11.09	0.049

Income determinants

Study villages varied significantly in their mean household income (GLM with normal errors, $F_{17,571}$ = 6.26, p<0.001, Fig. 4.1). Mean household income increased with distance from Ugalla Game Reserve ($F_{1,572}$ = 4.57, p<0.033). Villages located at least 10 km away had higher mean incomes. However, there were a few exceptions: Kangeme village had a somewhat high mean income despite being close to Ugalla Game Reserve. Similarly, Igalula village had a very low mean income even though it was far from Ugalla Game Reserve. A number of factors were found to be associated with household income (Table 4.5):

household assets (the value of productive assets in the household) had a significant positive relationship with the household income; households that were larger in total size tended to have lower income, although those with more members aged 18 and above had higher household income; educated individuals had higher income than non-educated ones. Age and tribe of the respondents had no significant influence on income.

Table 4.5 Determinants of the household income.

	Estimate ± s.e.	d.f. (change, residual)	F- value	Probability
Constant	7.85 ± 3.18			0.014
Household assets	1.21 ± 0.1	1,565	134.88	<0.001
Household size	-0.82 ± 0.26	1,565	9.80	0.002
Education	0.59 ± 0.24	1,565	6.24	0.013
Family labour	0.91 ± 0.45	1,565	4.15	0.042
Age	-0.07 ± 0.05	1,565	2.09	0.149
Tribe		3,567	1.55	0.200
Sukuma	-2.67 ± 2.2			
Nyamwezi	-2.36 ± 2.21			
Other*	1.23 ± 2.45			
Tribe reference level: Muha				

*Includes Bemba, Bende, Bungu, Chaga, Fipa, Gogo, Haya, Hehe, Kimbu, Kanonko, Lungwa, Lwila, Wagalla, Gogo, Hyao, Wajita, Ngoni, Nyakyusa,Nyaturu, Nyiramba, Pimbwe and Tutsi.

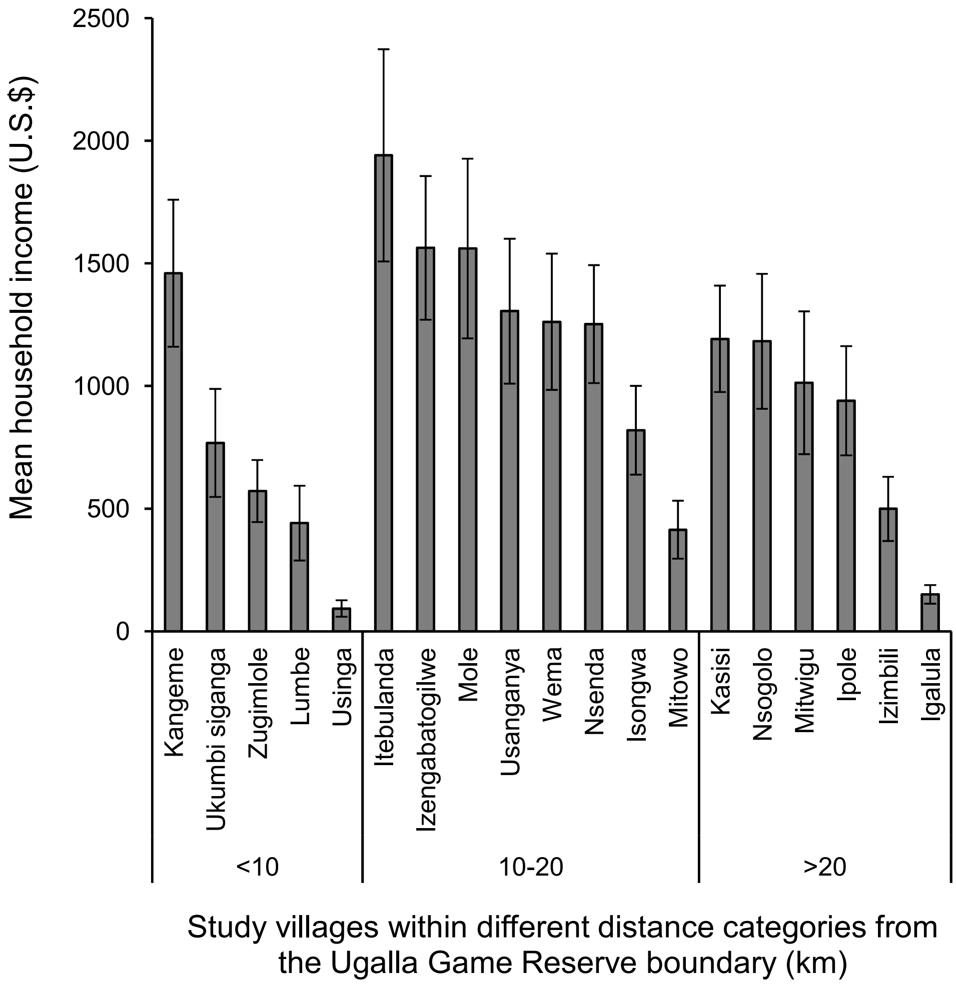

Figure 4.1 Comparison of mean household income across study villages. Error bars are the standard error of the mean.

Income & *wildlife poaching*

Wildlife poaching frequency differed significantly across the study villages (GLM with binomial errors, d.f. = 17, deviance χ^2 = 51.4, p<0.001, Fig. 4.2). Generally, mean poaching frequency decreased with increasing distance from the Ugalla Game Reserve boundary (GLM with binomial errors, d.f. = 1, deviance χ^2 = 41.3, estimate ± standard error [slope] = -0.058 ± 0.009, p = 0.001), especially for study villages with higher mean household income (p = 0.001, Fig. 4.3). Of all the income sources, livestock sales (GLM with binomial errors, d.f. = 1, deviance χ^2 = 12.34, [slope] x 10^{-3} = -3.82 ± 1.1, p<0.001) and crop sales (d.f. = 1, deviance χ^2 = 17.09, [slope] x 10^{-3} = -0.73 ± 0.179, p<0.001) were the best predictors of wildlife poaching, meaning that the increase in income from these sources was associated with a significant decrease in poaching.

An attempt was made to identify the effect of individual crops, livestock species and income determinants on wildlife poaching. Income from tobacco, groundnut and maize was negatively correlated with wildlife poaching (Table 4.6). Likewise, number of cattle, goat and chicken were the livestock species which significantly predicted poaching frequency. Only income from cattle and goat correlated with lower poaching frequency. Conversely, poaching frequency was high in villages where most respondents relied on chicken as their main source of income. Of the income determinants, study villages with higher mean value of productive assets had significantly lower mean poaching frequency (GLM with binomial errors, d.f. = 1, deviance χ^2 = 18.29, slope = -0.16 ± 0.023, $p<0.001$). Increase in manpower led to significant decrease in poaching frequency (d.f. = 1, deviance χ^2 = 5.00, slope = -0.29 ± 0.092, p = 0.002), whereas household size had a positive impact on poaching frequency (d.f. = 1, deviance χ^2 = 9.31, slope = 0.18 ± 0.057, p = 0.002).

Table 4.6 General linear model showing the influence of crops and livestock on wildlife poaching. d.f. (change, residual) = 1,13.

	(Estimate ± s.e.) x 10^{-3}	Deviance	Probability
Constant	1022 ± 222		<0.001
Cattle	-9.5 ± 2.19	19.17	<0.001
Chicken	194.3 ± 48.1	16.99	<0.001
Tobacco	-0.27 ± 0.07	13.59	0.007
Maize	-11.29 ± 3.26	12.11	0.013
Groundnut	-0.02 ± 0.01	7.85	0.024
Goat	-39.4 ± 14.6	4.42	0.057

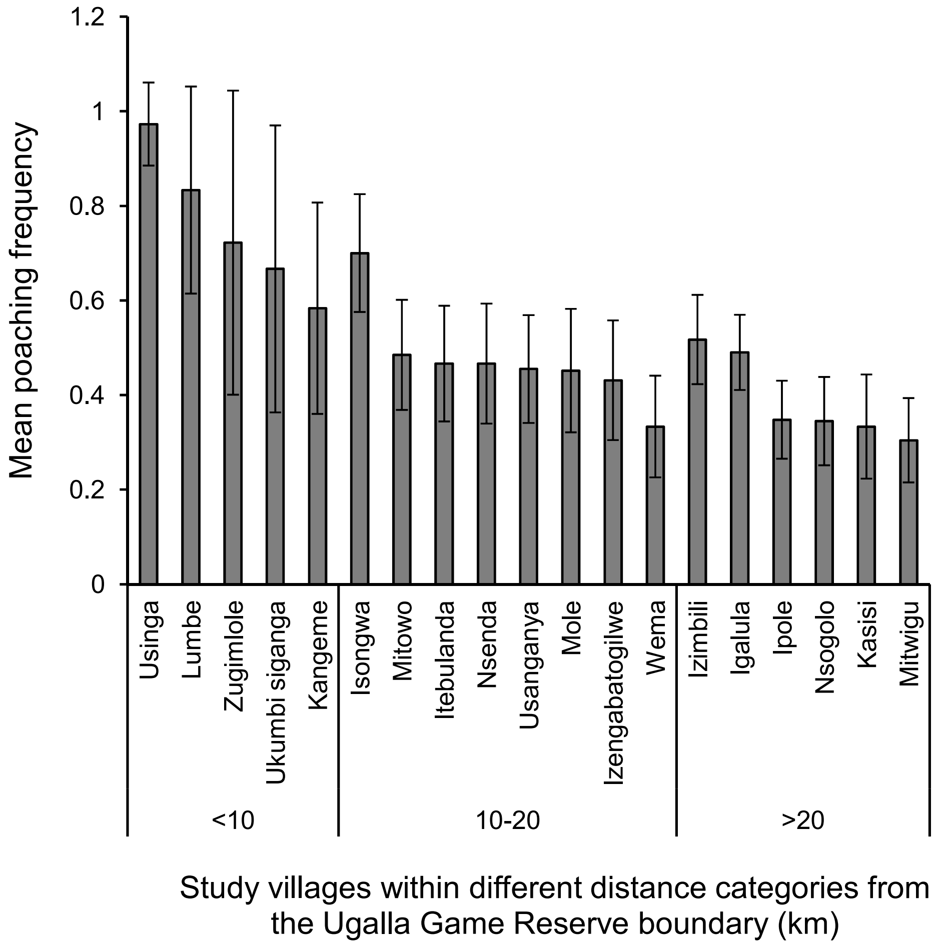

Figure 4.2 Comparison of mean poaching frequency across study villages. Error bars are the standard error of the mean.

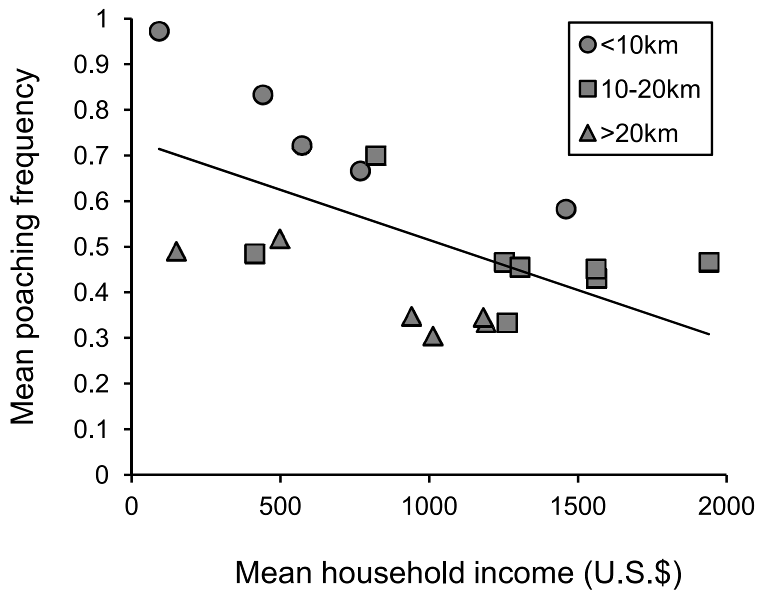

Figure 4.3 Relationship between household income and poaching frequency across study villages in different distance categories from Ugalla Game Reserve.

Discussion

Increases in income were associated with decreases in wildlife poaching, which suggests that hunting activities were carried out by low-income villagers. This has been confirmed in other wildlife conservation studies (Loibooki *et al.*, 2002; Robinson & Bennett, 2004; Bennett *et al.*, 2006). Greater income from different activities determined the extent to which villagers were involved in poaching. Families who earned considerable income from agriculture were less likely to engage in illegal hunting activities. Shrestha & Alavalapati (2006) reported similar observations in Nepal and found that farmers who earned high incomes from agriculture were not very dependent on wildlife resources.

Crops and livestock sales provided substantial income to households. Tobacco was the most profitable crop and the only non-food crop grown for commercial purposes. Income from tobacco far exceeded income from other crops due to its high market price. Respondents admitted that the price per kg of tobacco had increased significantly between 2008 and 2009 farming seasons compared to the 1990s. Nevertheless, this represents a conservation setback since it may attract many farmers into extensive tobacco cultivation at the expense of the miombo ecosystem (Yanda, 2010). Other important food and cash crops that were not predictors of wildlife poaching were rice, sunflower and sesame. In general, crop farming in the study area was confronted by a number of challenges. Some of them, as mentioned by one of the farmers in Usanganya village, were: the expensive agricultural inputs such as fertilizer, power tillers, pesticide, water pumps and ploughs; poor access to credit facilities together with tightened eligibility criteria; poor soil fertility; and fewer agricultural extension officers (Ellias R. Kaugilla, pers. comm.). Ensuring adequate and affordable inputs is required to increase profits from crop farming because of the huge loss of soil fertility (Hazelhurst & Milner, 2007).

Livestock was the second most important source of income, as Bucheyeki *et al.* (2010) also found. Only cattle had a significant effect on both household income and poaching frequency. But, some of the villagers were reluctant to sell their livestock and, in most cases, only sick or very weak animals were sent to seasonal local markets in the region. According to URT (1998*a*), for some communities in Tabora, keeping large numbers of livestock signals wealth and prestige. This has been a worrying scenario in terms of both the amount of land cleared to provide grazing space, and disputes between pastoralists and farmers over land resources, in particular due to mobile and largely uncontrolled keeping of livestock (URT, 1998*a*; Abdallah & Monela, 2007; Matata *et al.*, 2010).

Apart from agriculture (crop farming and livestock keeping), villagers were also dependent on other income sources. Despite the fact that non-agricultural sources of income could not significantly predict wildlife poaching, respondents who made use of some of such sources (small businesses, formal employment and wildlife) had important earnings from them.

Forest-based products were consumed by a large number of respondents, which is why the average income from "forests" seemed to be lower. From observations, it was established that charcoal, honey and timber were the most marketable and profitable forest-based products. Bucheyeki *et al.* (2010) reported substantial income earned from forest products by local communities in western Tanzania. Utilisation of forest products (mostly through commercial logging and charcoal burning) has had a noticeable impact on miombo woodlands in the area (Mkanta & Chimtembo, 2002), which has undesirable effects on wildlife habitats.

Wildlife made a fairly low contribution to household income. According to most respondents, income from wildlife was mainly generated through selling bushmeat, and much of the bushmeat was consumed for non-commercial purposes. Carpaneto & Fusari (2000) saw that bushmeat hunting was less important as a source of income for local people in western Tanzania. Owing to the fact that in Tanzania selling bushmeat is illegal and, in most cases, the people involved sell it on black markets for fear of being arrested for poaching (Baldus, 2002; Knapp *et al.*, 2010), the possibility that income from wildlife might be underestimated cannot be ruled out. Other wildlife by-products such as skin, claws and teeth were either sold or used for ritual and traditional purposes.

Income from other off-farm sources (formal employment, small business and remittances) accounted for only 2.2% of the total income in the study area. Formal employment was uncommon as would have been expected from the prevalent low literacy level, but this was not the case. Apart from a few government employees like primary school teachers and village executive officers, most of the unemployed respondents had at least a primary school education. In fact, in the rural areas, economic openings are few (URT, 2005); therefore, formal employment opportunities are limited. Small businesses were carried out by a handful of people in the study area. Due to their importance in the economy of rural areas, the government of Tanzania is committed to undertake policy changes necessary to improve rural livelihoods through small-scale business enterprises (URT, 2004; 2005).

Differences in the mean household income explain the observed variation in wildlife poaching frequency among the study villages. Wildlife poaching was more important to villages that were close to Ugalla Game Reserve than those farther away. In Panama, Smith (2008) found a similar relationship between higher wildlife exploitation and declining distance of human settlements from wildlife areas. Household production assets such as ploughs, water pumps, hand hoes, wheel barrows, traditional carts, tobacco barns and traditional grain storage baskets were vital for increasing household income, thereby reducing villagers' dependency on wildlife resources. Byarugaba (2003) pointed out productive assets as one of the missing ingredients in poverty-stricken rural communities whose lives are dependent on the natural ecosystem. The importance of family labour in crop farming and livestock keeping cannot be overstated. On the other hand, the increase in the household size heightened family demands, thus exacerbating the household economic situation for larger-sized households with insufficient family labour available for agricultural activities simply because most household members had migrated to urban areas in search for a better life. Some respondents claimed that they had to hire additional labour for agricultural production, thereby further worsening their economic situation. Results also showed a significant positive impact of formal education on household income. Formal education is a tool for making sound decisions that would improve income from both livestock keeping and crop farming (Inoni *et al.*, 2007; Serin *et al.*, 2009). Additionally, it facilitates adoption and successful implementation of new technologies that can improve agricultural productivity (Weir, 1999; Serin *et al.*, 2009).

Conclusions

Assessing different sources of income in the Ugalla ecosystem from a wildlife poaching standpoint offers a good understanding of the tradeoffs existing between local livelihoods and wildlife conservation. This study has revealed some factors that influence both people's livelihoods and wildlife poaching. I have shown that crops and livestock are not only important sources of income, but also reliable options for curbing wildlife poaching.

Other sources of income, namely; forests, wildlife, small business, formal employment and remittances, although subsidiary, signify the presence of additional livelihood options for local people in the study area. None of these were relevant in reducing poaching. While income is an important ingredient in lessening local people's dependence on wildlife resources, the results suggest that villagers carry out poaching activities not only because of their economic hardship, but also because of their close proximity to wildlife areas.

Attempts to improve people's livelihoods in Ugalla should pay particular attention to local communities neighbouring Ugalla Game Reserve. This would contribute to commendable conservation work carried out by the Ugalla Game Reserve Management Team. A considerable amount of emphasis should be put on the cultivation of important food and cash crops.

CHAPTER 5: POACHING ACTIVITIES IN UGALLA GAME RESERVE

Abstract

Although widely acknowledged as affecting wildlife populations across Africa, bushmeat poaching in east Africa is less often tackled from the viewpoint of its relationship with other types of illegal exploitation of wildlife in a protected area. Here, I tried to address this issue using a combination of fieldwork and existing records in the Ugalla ecosystem of western Tanzania. Logging was the main poaching type with fairly large group size of poachers apprehended, followed by bushmeat hunting, fishing and illegal entry into Ugalla Game Reserve. Exploration of poachers' belongings revealed little overlap between poaching types, which suggests that poachers are specialised in their respective poaching activities. This is also reflected in the spatial distribution of poaching signs across different areas within the reserve. Therefore, anti-poaching efforts should incorporate the nature (intensity, trade, distribution etc.) of each type of poaching to ensure effective conservation of wildlife and their habitats.

Introduction

Although poaching (illegal exploitation) of natural resources is a significant problem for most of Africa's protected areas (Milner-Gulland & Rowcliffe, 2007), it has often been considered as a general exploitation of whatever natural resources are perceived to be valuable by people living in poverty around the protected areas (Taylor & Dunstone, 1996; Davies & Brown, 2007). Yet, implementing an effective anti-poaching strategy requires an understanding of the extent to which the exploitation of particular resources is specialised, in order that the level of organisation and/or the need for a particular resource such as protein or wood may be ascertained and appropriate measures taken to halt its use.

Controlling the problem of bushmeat hunting has become central to the escalating debate over the illegal exploitation of natural resources (Hofer *et al.*, 1996; Fa *et al.*, 2005; Rist *et al.* 2008; Willcox & Nambu, 2007; Mfunda & Røskaft, 2010). Currently, bushmeat hunting is a major concern in central/west Africa (Blom *et al.*, 2005; Waite, 2007). Various approaches have been used to understand patterns of the problem in this part of Africa. For example, studies have assessed status, destination and actual quantification of bushmeat as well as conducting hunter follows (Fa *et al.*, 2006; Willcox & Nambu, 2007; Jachmann, 2008*b*; Rist *et al.*, 2008). It has also been possible to record some species-specific parameters (such as age, sex and weight) for hunted species (see Muchaal & Ngandjui, 1999; Fa & Garcia Yuste, 2001; Coad, 2007).

Bushmeat hunting also represents a conservation problem in Tanzania (Baldus, 2002; Caro & Andimile, 2009), but assessing its extent and impacts on wildlife species is challenging. This is because the illegal nature of poaching makes it difficult to get hunters to cooperate when much of the bushmeat trade is carried out in the black market (Kaltenborn *et al.*, 2005; Ndibalema & Songorwa, 2007; Nyahongo *et al.*, 2009; Knapp *et al.*, 2010). This is a serious problem for quantifying impacts of wildlife poaching (Caro & Andimile, 2009). Therefore, bushmeat studies in Tanzania have tended to employ different approaches from central/west Africa. Often these are surrogate approaches for quantification of bushmeat hunting data. They include working with villagers' responses about bushmeat issues using questionnaire surveys (Nyahongo *et al.*, 2009; Mfunda & Røskaft, 2010; Wilfred & MacColl, 2010). In some other cases, bushmeat hunting information has been anonymously captured through a few willing hunters or village-based informers (Nielsen, 2006; Caro, 2008). Although these efforts have been useful in highlighting the breadth of the problem, getting a grip on it using data on poachers caught and their activities collected by game rangers is also important in gaining a more realistic understanding of these issues (Milner-Gulland & Rowcliffe, 2007; Jachmann, 2008*b*).

A few recent bushmeat studies in Tanzania have incorporated poaching information gathered by game rangers/scouts in partially protected areas (protected areas other than national parks and game reserves; for example, wildlife management areas, game-controlled areas and open areas) (Holmern *et al*., 2007) and game reserves (Knapp *et al*., 2010). These areas are often subjected to high hunting pressure (Waltert, *et al*., 2009). Unfortunately, in central-western Tanzania, poacher characteristics and poaching information recorded by game rangers in their normal patrol duties have remained largely unexploited in anti-poaching and monitoring programs. The predominant reason for this could be either lack of awareness of the usefulness of the information gathered or inability to make a meaningful interpretation of it for the betterment of conservation.

The purpose of anti-poaching patrols is to deter poaching activities, and to make offenders (poachers) bear responsibility for their actions (Milner-Gulland & Rowcliffe, 2007; Fischer, 2008; Jachmann, 2008*a*). While punishment for poaching may differ widely depending on the severity of the infraction (Holmern *et al*., 2007) and poaching type (for example, timber harvesting, bushmeat hunting, fishing and honey gathering), patrols to catch poachers do not normally make a distinction between different types of poaching. This is because damage of one natural resource may lead to damage of another. For example, Kinnaird *et al.* (2003) reported negative impacts of forest loss on the large mammal populations in Bukit Barisan Selatan National Park on the Indonesian island of Sumatra. Unsustainable bushmeat hunting has also been reported to cause forest degradation as a consequence of removing effects of pollination, seed dispersal and seed predation (Fa & Brown, 2009). As a result, in assessing the impact of poaching activities on wildlife, all indicators of poaching like tree stumps, snares and poachers' camps, and poachers caught are given due attention (Campbell & Loibooki, 2000; Blom *et al*., 2004; Milner-Gulland & Rowcliffe, 2007).

Poachers may also conduct more than one type of poaching in a protected area. There are records of timber poachers also hunting for protein (Guariguata *et al.*, 2009). Corlett (2007) argued that when logging and bushmeat hunting co-exist in a protected area, wildlife species become increasingly threatened. Therefore, it is worthwhile considering the degree of interaction/specialisation between different types of poaching. This would improve our understanding of the intensity of bushmeat hunting with respect to other illegal activities. It can also help us discern the magnitude of the impact suffered by different natural resources (for instance, forest, fish and wildlife) and determine the extent to which poaching types influence each other, for effective anti-poaching measures. In light of this, the present study explores the degree of specialisation between different poaching types using data from arrested poachers and their confiscated belongings, as well as on the spatial distribution of different poaching signs. The study shows the place of illegal hunting in a poaching system, and emphasises the importance of paying attention to all types of illegal activities as a way forward towards reducing wildlife poaching in a protected area.

Methods

Study area

This study was carried out in Ugalla Game Reserve (5000 km^2), western Tanzania, between 5^o – 6^o South and 31^o – 32^o East. The reserve lies within the Rukwa (in Mpanda district) and Tabora (in Sikonge & Urambo districts) regions. It is characterised by miombo woodland vegetation containing highly valuable timber species of the genera *Brachystegia*, *Julbernardia* and *Isorberlinia*. A wide range of wildlife species including large mammals such as hippopotamus *(Hippopotamus amphibius)*, giraffe (*Giraffa camelopardalis*) and African elephant *(Loxodonta africana)* are also found in the area. There are four main rivers: Msima, Ugalla, Koga and Wala. These rivers support a diverse range of fish; for example: tilapia (*Tilapia* spp.), African butter catfish (*Schilbe mystus)*, African lungfish (*Protopterus aethiopicus*), eastern bottlenose mormyrid (*Mormyrus longirostris*) and Long-finned Tetra (*Brycinus longipinnis*). The main economic activity within the reserve is tourist hunting conducted in two hunting blocks: Ugalla west and Ugalla east. Local communities around Ugalla are also allowed (through permit) to carry out fishing and beekeeping activities during specific seasons of the year, normally from July – December. Ugalla is the only game reserve that allows multiple use of natural resource in a sustainable manner, the aim being to increase local participation in conservation while minimizing illegal offtakes. Nonetheless, as with all other game reserves, poaching is a problem in Ugalla, particularly during the rainy seasons when anti-poaching patrols are hampered by muddy and hardly passable roads/tracks. The rainy season spans from December – May, with annual rainfall ranging between 700 – 1,000 mm. The common types of poaching are: timber harvesting, bushmeat hunting, and illegal fishing. The Ugalla Game Reserve Management Team has stipulated in its management objectives that it aims "to conduct monitoring on the impact of resource utilisation activities in the reserve so as to control these activities more effectively" (WD, 1998). Although monitoring is a good conservation tool in protected areas, its execution might turn out to be financially impractical (see Milner-Gulland & Rowcliffe, 2007). Assessing patterns and spatial hotspots of illegal activities would contribute knowledge about strategic but effective monitoring and conservation of Ugalla.

Ugalla poaching record

Data on poachers arrested in Ugalla Game Reserve were obtained from the Wildlife Division's Ugalla Game Reserve office based in Tabora town. The information spanned from 2003 to 2009, and consisted of the following components: the date a poaching incident occurred, a record of the belongings confiscated from poachers, place within the reserve where poaching took place (often recorded at the level of hunting block), and number of species of plants and/or animals identified with the arrested poachers. The data set lacked some useful information; for example: poacher age and sex, and villages from where arrested poachers originated. Additionally, there was neither a measure of the effort/effective anti-poaching patrol days per game ranger, nor a record of the number of members of the patrol team in each anti-poaching trip. This made it difficult to control for anti-poaching effort when assessing factors associated with poachers caught. However, it was important to use the available data to understand basic patterns of poachers' activities in the reserve.

Since rainfall influences the effectiveness of the patrols in terms of accessibility to the reserve and the coverage of operations within the reserve, I compared how it affected the number of poachers caught. Rainfall data were obtained from Tabora metrological station.

The metrological office records monthly amount of rainfall (in mm) for various government and research uses. The data covered all districts in which the Ugalla ecosystem falls (i.e. Sikonge, Mpanda and Urambo districts); therefore, I am confident that they represent the amount of rainfall in the whole of the ecosystem. I used rainfall data from 2003 to 2009 to match the poaching data obtained from the Ugalla Game Reserve office.

Poaching signs

A survey of poaching signs was carried out monthly on randomly placed road transects during the dry season, between June – September, 2009. The sampling units were the eight anti-poaching units within Ugalla Game Reserve: Isimbira, Kakoma, Kamakala, and Ugalla (in the Ugalla east hunting block); and Muhuba, Luganzo, Siri and Msima (in the Ugalla west hunting block). Within each month, hunting blocks and anti-poaching units were sampled in a random order. Each anti-poaching unit was surveyed for 10 consecutive days in each month, and 5 days between surveys for other logistics. This ensured approximately equal effort invested in surveying the anti-poaching units. A total of 36 transects were randomly selected, 24 in Ugalla east and 12 Ugalla west depending on available roads and resources (financial, manpower and time resources). The roads were those used for patrolling purposes by the game rangers. Transects began at randomly selected points, and at least three transects were surveyed in each anti-poaching unit. All surveyed transects within the reserve covered a total of 782 km. Transects varied in length and there was no fixed distance on either side of the transect within which poaching signs were searched. This allowed optimal coverage of the surveyed area because poachers carried out their activities away from patrol roads to avoid being detected by patrolling rangers. Surveys were conducted during the afternoon hours (13 – 18 hrs) by three people in an open vehicle at a speed not exceeding 20 km h^{-1} to allow rigorous search of poaching signs. Owing to the low-lying/flat landscape characteristic of Ugalla Game Reserve, and the fact that the survey was conducted during the dry season (i.e. June – December) when much of the reserve was burnt for conservation purposes, visibility was generally relatively high. Therefore, with the aid of binoculars, observers searched for poaching signs on both sides of the transect. All poaching signs were recorded, whether structural; for example, fish and bushmeat smoking racks and poacher camps, abandoned poacher belongings, or animal remains. Positions were recorded for each poaching sign using a handheld global positioning system unit

(GPS) (Garmin GPSMAP® 60Cx). Poaching signs were first ascertained by game rangers before they were recorded, and where necessary the vehicle stopped and observers got off the vehicle to take a closer look at poaching signs. Due to their long-term experience in patrolling activities, game rangers were very knowledgeable about the reserve and the poaching activities therein.

Statistical analysis

All statistical analyses were carried out in GenStat®10 (Payne *et al.*, 2007). Generalised linear models (GLMs), with appropriate error structures, were used to test predictors of interest to total numbers and group sizes of poachers caught, and poaching sign encounter rate (SER). The "SER" was calculated as [number of encounters of different types of poaching signs along a transect]/[length (in km) of the same transect]. The fixed terms of the GLMs were dropped in the ascending order of their F-values until the minimum adequate models were obtained. The correlation between mean monthly amount of rainfall and mean number/group size of poachers was tested using a Pearson correlation and a two tailed significant test.

To examine how poachers were specialised in different poaching types using their belongings, a canonical variate analysis (CVA) (Shaw, 2003) was used. Only the first three axes or dimensions [Canonical variate (CV) 1, CV2 and CV3] were extracted, representing much of the variation in the poaching types. Then bi-plots were generated using resulting scores of the dimensions along with co-ordinates or loadings of some selected poacher belongings. The bi-plots were useful in showing the degree with which certain poacher belongings or poaching gear were related to their respective poaching types, and whether there was a distinct separation between them. A CVA was also used to show how different types of poaching activities were spatially distributed across different anti-poaching units in the reserve using poaching signs. Three axes (CV1, CV2 and CV3) were also presented in this case, but just two captured most of the variation in poaching types. Consequently, a bi-plot was generated using scores of the first two axes and co-ordinates of the poaching signs.

Results

Poachers caught

A total of 944 poachers was caught in Ugalla Game Reserve for different poaching types (Table 5.1). Poachers were caught in groups of varying sizes ranging from 1 – 14 poachers. A GLM with normal errors was used to test for the effect of different terms associated with numbers of poachers caught. The response variable "total number of poachers" was square root transformed in order to achieve normality. The tested predictors were: poaching type, year, hunting block, month, hunting block x poaching type, month x poaching type, and year x poaching type. Of these, poaching type, year, and hunting block x poaching type were the best predictors of total number of poachers (Table 5.2). Number of poachers caught differed according to poaching types. The majority of the poachers were arrested for illegally harvesting timber and bushmeat (Table 5.1 & Fig. 5.1). Poachers caught increased significantly over the years (Fig. 5.2). There was also a significant difference in number of poachers caught for different poaching types between the hunting blocks (Fig. 5.1). Although the difference in poachers caught across months was not statistically significant (Table 5.2), Fig. 5.3 suggests that the number of poachers caught between May and November was slightly higher than other months. Factors used in the model above were also tested in another GLM with a normal error structure to determine the best predictors of group sizes of poachers. Here, the response variable "poacher group size" was log-transformed. Only poaching type had a significant effect on poacher group size (Table 5.3), indicating that group sizes of poachers differed significantly between poaching types (Fig. 5.4). There was no significant correlation between rainfall and number of poachers caught ($r = -0.4093$, $n = 12$ months, $p = 0.1864$), nor between rainfall and group sizes of poachers ($r = -0.0612$, $p = 0.8500$).

Table 5.1 Number of poachers (poacher groups in parentheses), group-size range and mean number of poachers group^{-1} caught for different poaching types in Ugalla east and Ugalla west hunting blocks, in Ugalla Game Reserve.

Poaching type	Ugalla East	Ugalla West	Total Ugalla	Group size range	Mean ± s.e.
Timber harvesting	338(98)	210(74)	548(172)	1 – 14	3.20 ± 0.17
Bushmeat hunting	64(32)	171(60)	235(92)	1 – 11	2.55 ± 0.21
Fishing	97(28)	23(10)	120(38)	1 – 9	3.16 ± 0.36
Illegal entry	12(7)	29(14)	41(21)	1 – 5	1.95 ± 0.24

Table 5.2 General linear model output showing terms associated with total numbers of poachers caught in Ugalla Game Reserve from 2003 – 2009.

	d.f. (change, residual)	F-value	Probability
Poaching type	3,182	5.24	0.002
Year	1,180	6.46	0.012
Hunting block	1,179	1.44	0.232
Hunting block x Poaching type	3,178	4.21	0.007
Month	11,175	1.64	0.091
Year x Poaching type	3,164	0.47	0.702
Month x Poaching type	31,169	0.79	0.779

Table 5.3 General linear model output showing terms associated with group sizes of poachers caught in Ugalla Game Reserve from 2003 – 2009.

	d.f. (change, residual)	F-value	Probability
Poaching type	3,280	4.57	0.004
Year	1,277	2.12	0.147
Hunting block	1,276	1.79	0.182
Hunting block x Poaching type	3,275	1.46	0.227
Month	11,272	0.83	0.612
Month x Poaching type	31,258	1.00	0.471
Year x Poaching type	3,261	0.30	0.822

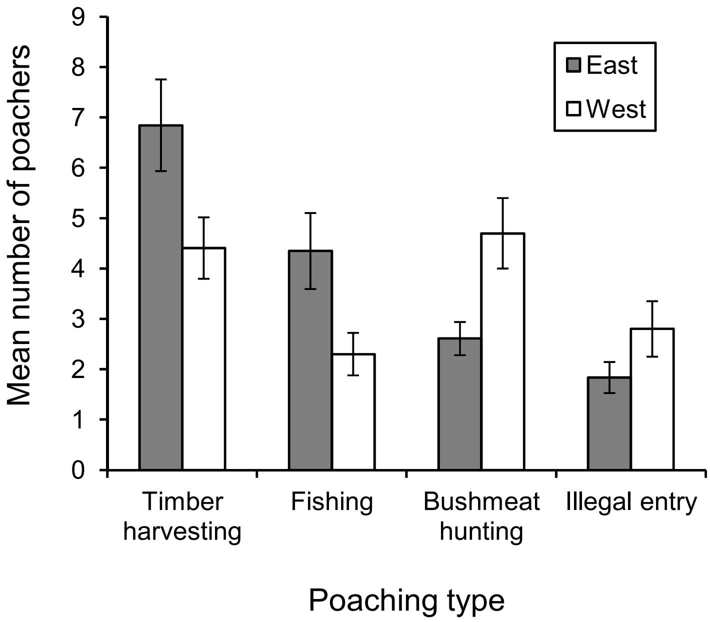

Figure 5.1 Number of poachers across different poaching types between Ugalla east and west hunting blocks in Ugalla Game Reserve. Error bars are the standard error of the mean.

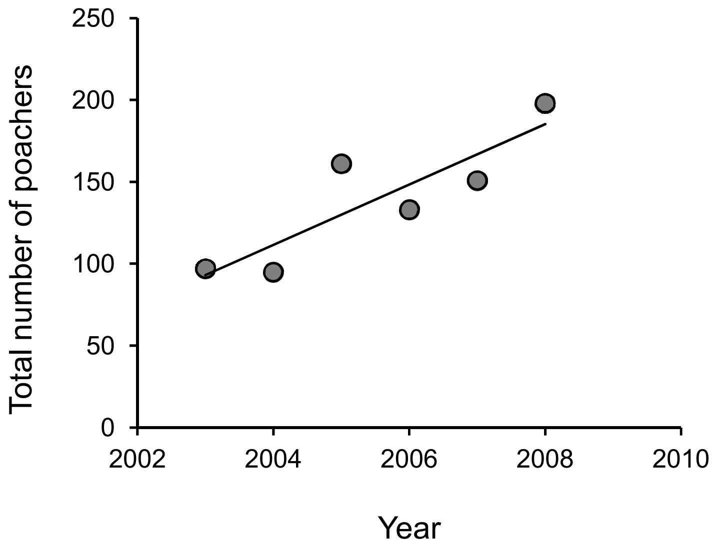

Figure 5.2 Total numbers of poachers caught in Ugalla Game Reserve across different years.

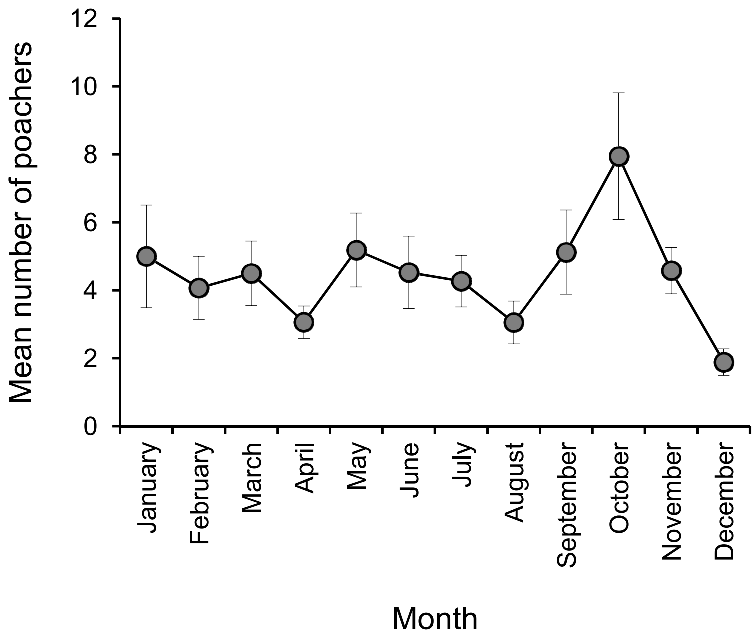

Figure 5.3 Mean monthly number of poachers caught in Ugalla Game Reserve from 2003 – 2009. Error bars are the standard error of the mean.

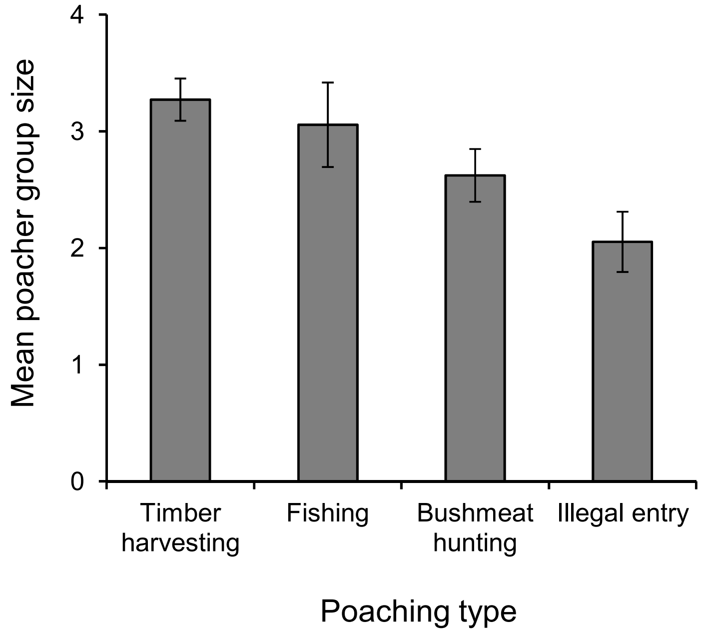

Figure 5.4 Mean group sizes of poachers caught in Ugalla Game Reserve from 2003 – 2009 plotted against different poaching types. Error bars are the standard error of the mean.

By comparing the composition of confiscated poacher belongings listed in Table 5.4, I investigated how different groups of poachers were specialised to different poaching types.

Table 5.4 Belongings identified with arrested poachers in Ugalla Game Reserve.

Item	Total	Number of Poachers
Bicycle	722	231
Saw	327	129
Axe	273	156
Knife and bush knife combined	269	163
Fishing net	133	46
Fishing hook	117	5
Cooking pot	83	39
Water container	68	23
Muzzle loader	46	30
Hoe and Spade combined	33	30
Sharpening steel, hammer and chisel combined	32	25
Tape measure, balance scale, and torch combined	26	20
Gun	24	21
Radio	20	17
Watch	10	9
Isuzu truck	1	1

Ordinations of poacher belongings from poaching types using a CVA were carried out. For interpretation purposes, groups of poachers were named according to different poaching types. There were four main types of poachers: bushmeat poachers (bushmeat), timber poachers (timber), fish poachers (fishing), and illegal entry into the reserve (entry). The common poacher belongings were collapsed into 12 main categories (Table 5.5). The CVA was carried out on transformed data. The loadings (coordinates) of the 12 categories of poacher belongings along the first three dimensions (axes) are shown in Table 5.5.

Table 5.5 Latent vectors (loadings) of different poachers' belongings for the first 3 axes.

	Axis		
Item	1	2	3
Gun	0.2186	-0.248	-0.3856
Fishing net & hook	0.0364	-0.1875	0.6933
Hoe & spade	0.9716	0.6355	-0.1513
Knives	0.3603	-0.373	0.3485
Muzzle loader	0.2478	-0.3853	-0.4525
Pots and buckets	-0.0297	0.0597	0.015
Saw	-0.5656	0.5887	-0.292
Sharpening equipments	-0.1061	0.0129	0.0173
Radio & watches	-0.0892	0.1346	0.0723
Torches	-0.0902	-0.2466	-0.3331
Bicycle	-0.1255	-0.0287	-0.0871
Axe	-0.0805	-0.1411	-0.2764
Eigenvectors	0.7816	0.473	0.1848
Percentage variation	54.30	32.86	12.84

Figs. 5.5 & 5.6 present two dimensional ordinations from the CVA. Only those poacher belongings with noticeably high and low loadings along each of the axes are shown on the plot to avoid congestion. In Fig. 5.5, the first axis (CV1) appears to separate timber poachers with saws from those with digging equipments and hunting gear. The second axis (CV2) represents the difference between those who poached for protein (illegal fishermen and bushmeat hunters) and other types of poachers, namely, timber poachers and those caught for illegal entry into the reserve (most of whom possessed hoes and spades). The third dimension (CV3) shows the difference between fish poachers with knives, fishing nets and hooks from timber and bushmeat poachers (Fig. 5.6).

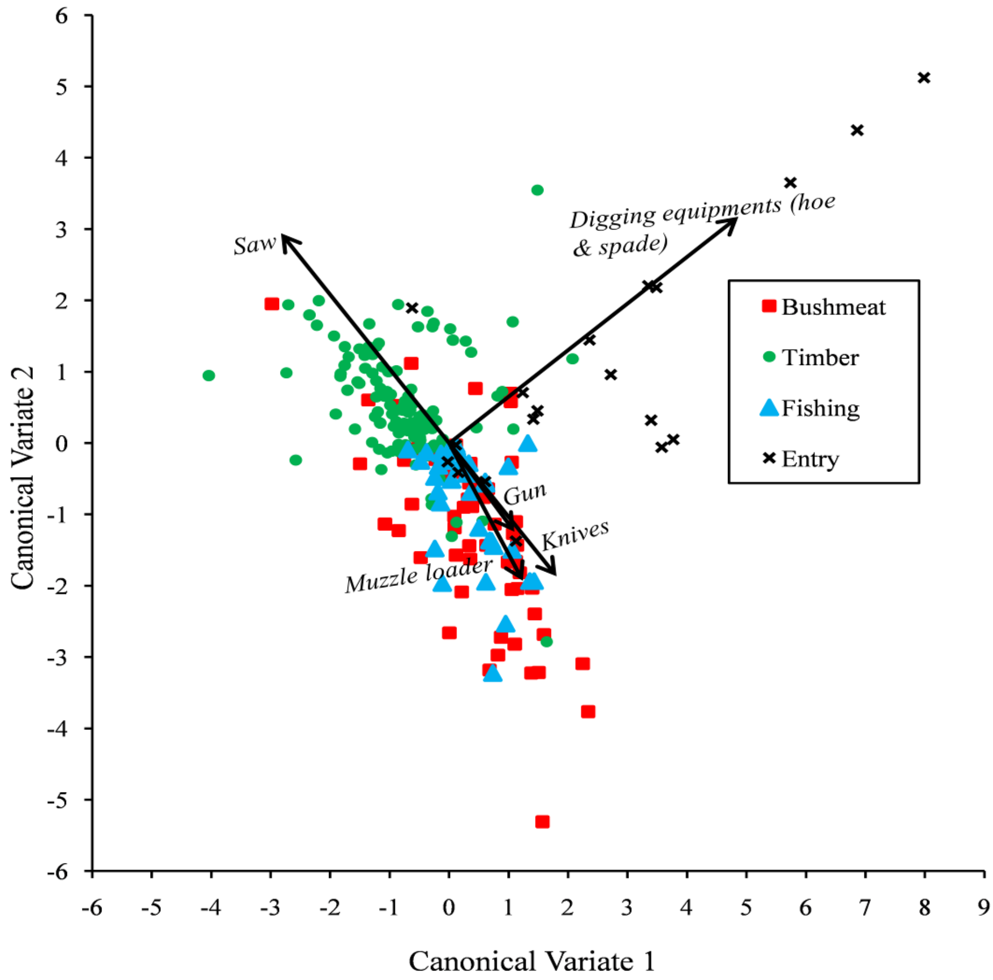

Figure 5.5 Scatter plot from a CVA showing poachers' belongings with fairly high loadings on axes 1 & 2. Coordinates of poachers' belongings were multiplied by 5.

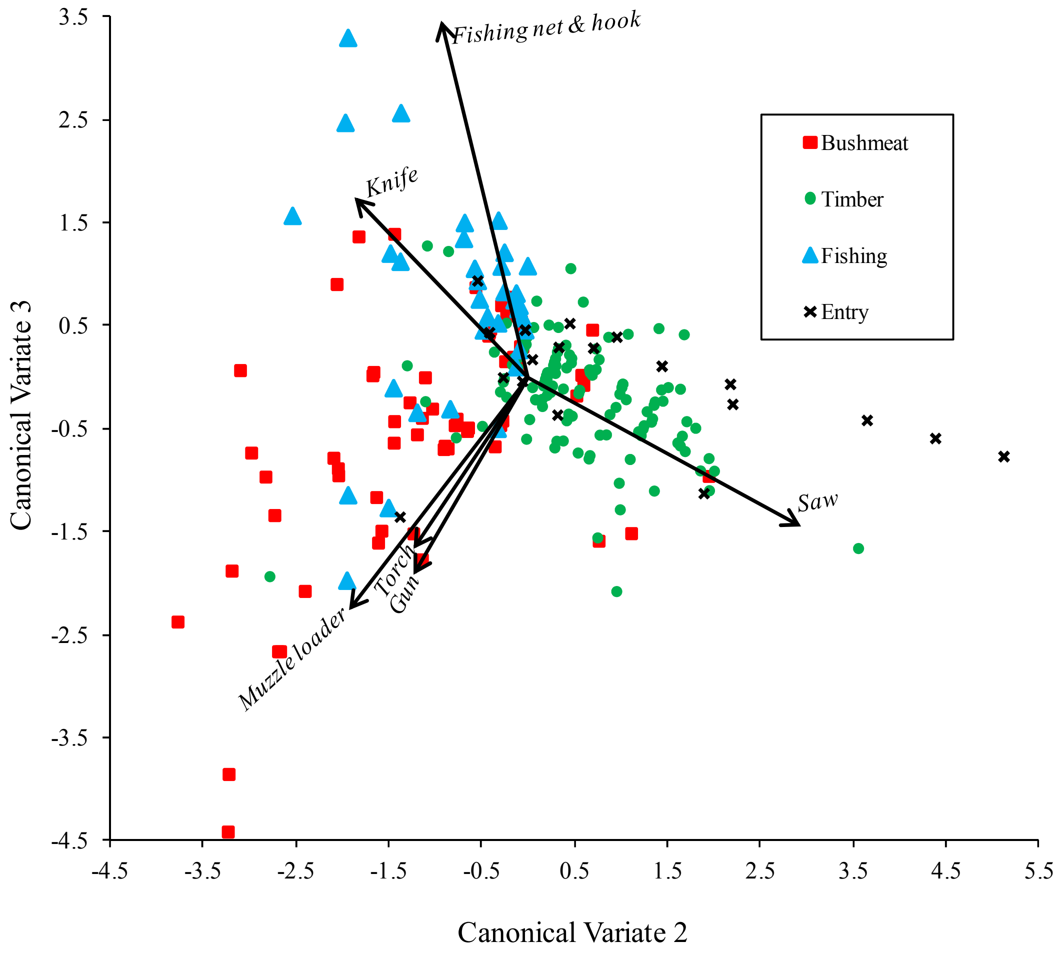

Figure 5.6 Scatter plot from a CVA showing poachers' belongings with high loadings on axes 2 & 3. Coordinates of poachers' belongings were multiplied by 5.

Poaching signs

Poaching signs were also grouped into 12 categories reflecting illegal fishing, timber and bushmeat harvesting (Table 5.6). A total of 764 encounters of different poaching signs was recorded across anti-poaching units in both east and west hunting blocks in the period of four months from June – September, 2009. Two separate GLMs with normal errors were used to test the effects of hunting block, anti-poaching unit and poaching sign type on SER. Hunting blocks and anti-poaching units had to be in separate models because they were aliased. Anti-poaching unit and poaching sign were tested in the first model. Here, SER varied significantly across different types of poaching signs ($F_{11, 88} = 3.93$, $p<0.001$, Fig. 5.7) and among anti-poaching units ($F_{7, 87} = 2.36$, $p = 0.031$). Poaching sign was also tested in the second model along with hunting block and their interaction. Again poaching sign type was a significant predictor of SER ($F_{11, 95} = 3.53$, $p<0.001$). Neither hunting block ($F_{1, 84} = 1.18$, $p = 0.280$) nor poaching sign x hunting block ($F_{11, 83} = 0.84$, $p = 0.597$) explained significant variation in SER.

Table 5.6 Categories of different poaching signs encountered in Ugalla Game Reserve.

Poaching sign	Description
Animal remains	Remnants of different animal species (other than elephant) killed by poachers. Most of these were found around bushmeat poachers' camps.
Bicycle	Any bicycles abandoned by poachers.
Boat	Traditional fishing boats made of tree barks or hollowed tree trunks also known as locally made canoes.
Elephant	Dead African elephant *Loxodonta Africana*. In most cases these were killed by ivory poachers. Elephant remnants were considered as a separate category of poaching signs because they were commonly encountered and easily identified during the survey.
Fish rack	Wooden racks used for drying fish.
Fishing net	A net used for fishing.
Honey camp	Camps where illegal honey gatherers sleep, cook their meals, and temporarily stock their honey while in the reserve.
Meat rack	Racks used for smoking bushmeat.
Sawpit	Dug-out pits over which wood logs are placed in order to facilitate timber sawing.
Sawn wood	Any cut wood as a result of timber poachers' activities; for example: sawn tree stumps, wood logs, poles, timber etc.
Snare	Wire snares located across animal paths.
Track	Clusters of poachers' foot prints and bicycle tracks.

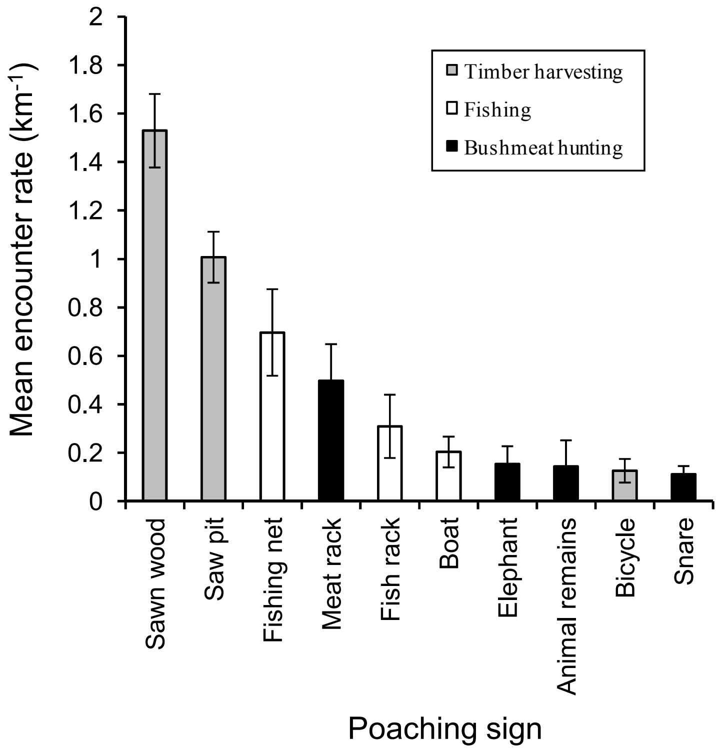

Figure 5.7 Mean poaching sign encounter rates across different types of poaching signs in Ugalla Game Reserve. Error bars are the standard error of the mean.

Fig. 5.8 presents a biplot from a CVA showing spatial variation of poaching signs across different anti-poaching units in Ugalla Game Reserve. The loadings of the 12 categories of poaching signs along the first three axes are shown in Table 5.7. The first axis, which represents 60% of the variation, separates poaching signs associated with timber and bushmeat harvesting from illegal fishing. Timber poaching signs (sawpit, bicycles, and sawn wood) have higher values along the second axis, which represents 30% of the variation, whereas illegal fishing and bushmeat harvesting signs have lower values.

Table 5.7 Latent vectors (loadings) of different poaching signs for the first 3 axes.

	Axis		
Item	1	2	3
Animal remains	0.0442	-0.2815	-0.119
Bicycle	0.2045	0.0874	-0.2804
Boat	-0.1384	-0.1822	0.2688
Elephant	0.1137	-0.5469	-0.0426
Fish rack	-0.3415	0.0361	0.4521
Fishing net	-0.2765	-0.0264	0.0169
Honey camp	-0.0718	0.0031	-0.802
Meat rack	0.0942	-0.7144	0.1351
Saw pit	0.6798	0.3368	-0.1242
Sawn wood	0.671	0.2142	0.0856
Snare	0.1131	-0.4391	0.0925
Track	0.4729	-0.0445	0.3838
Eigenvectors	1.6048	0.8024	0.1479
Percentage variation	60.28	30.14	5.56

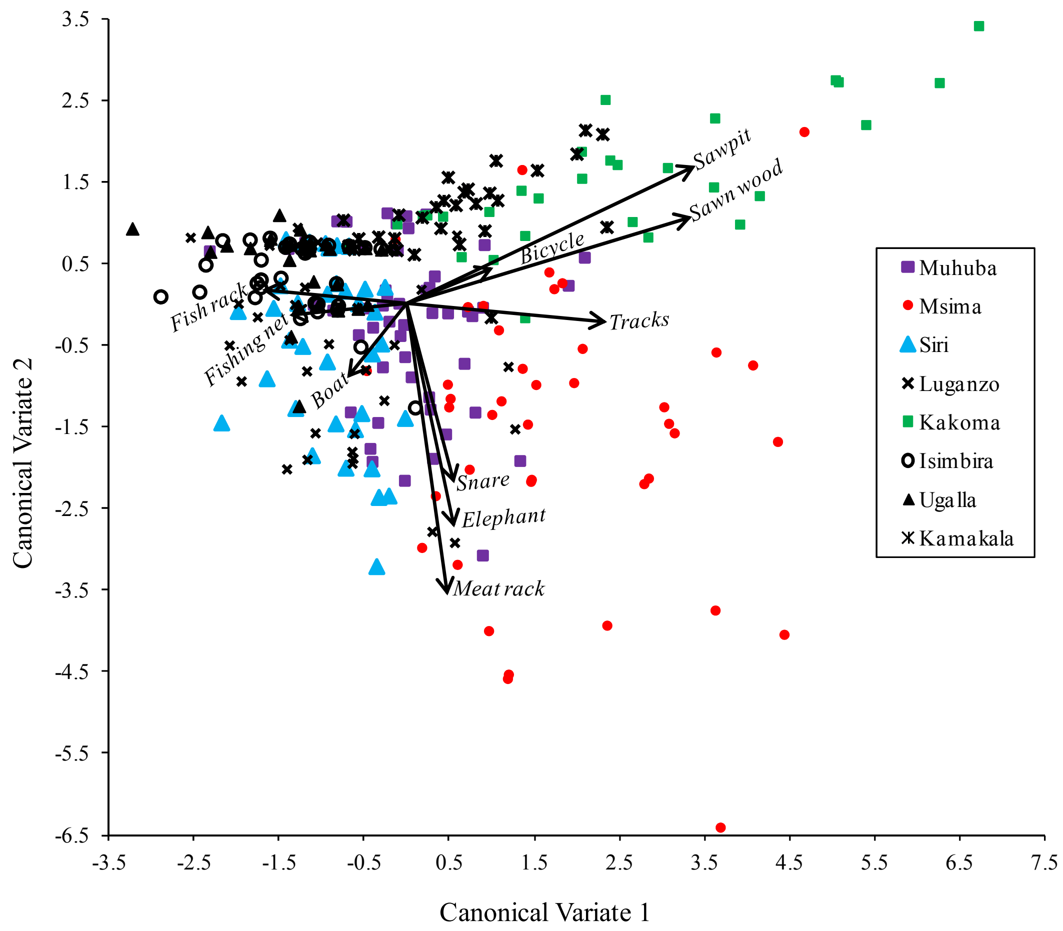

Figure 5.8 Scatter plot from a CVA showing the distribution of poaching signs in different anti-poaching units. Coordinates of poaching signs were multiplied by 5.

The GPS co-ordinates of the poaching signs showed that anti-poaching units in Ugalla east, especially Kakoma and Kamakala (Fig. 5.9), had more timber poaching activities than Ugalla west. Illegal fishing was common in Siri, Ugalla and Isimbira anti-poaching units. Bushmeat harvesting was more prevalent in Ugalla west, around Msima and Luganzo. Furthermore, the distribution of some poaching signs strongly suggests co-occurrence of poaching activities in a particular anti-poaching unit. This was common for most of the anti-poaching units in Ugalla west. For example, Muhuba had illegal fishing, timber and bushmeat harvesting signs. Likewise, Siri had signs of both illegal fishing and bushmeat harvesting, whereas Msima consisted of predominantly bushmeat and timber harvesting signs.

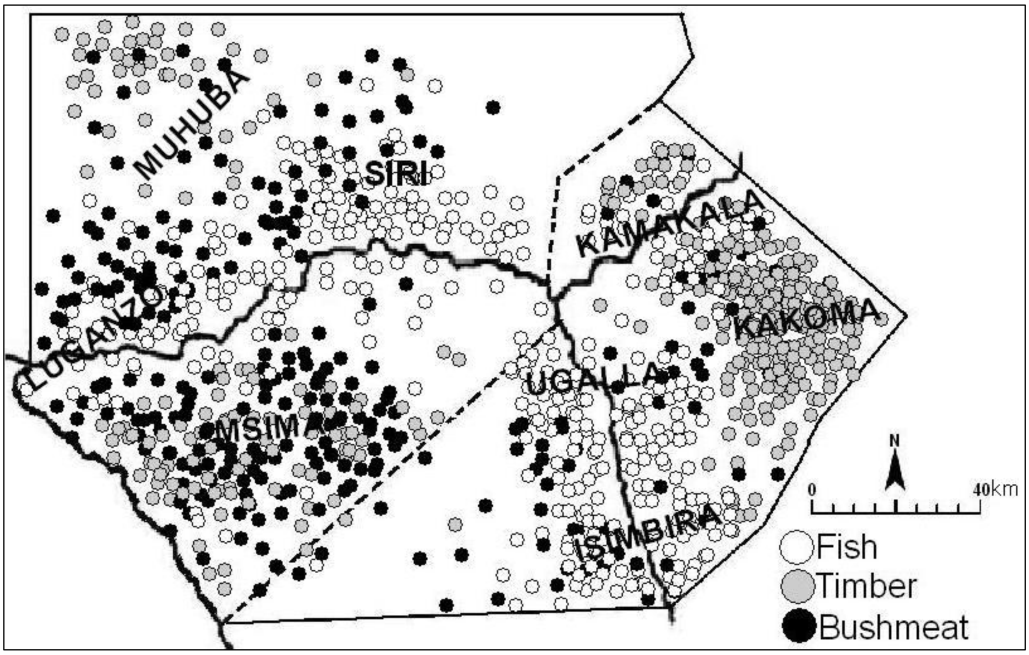

Figure 5.9 Locations of poaching signs encountered within Ugalla Game Reserve. Signs were classified into three main types of poaching (fish, timber and bushmeat) depending on the location and nature of a sign. Names of the anti-poaching units are at approximate centres for anti-poaching units.

Discussion

Differences in numbers, group sizes and poaching gear of the arrested poachers, and spatial distribution of poaching signs strongly suggest that the different forms of poaching in Ugalla are specialist activities that are largely independent of each other. The information about poachers and their activities helps in understanding who poachers are (Gavin *et al.*, 2009), their behaviour against anti-poaching efforts (Forsyth, 2008) and the intensity and distribution of their activities in a protected area (Blom *et al.*, 2005; Holmern *et al.*, 2007; Gavin *et al.*, 2009). This is vital for ensuring cost-effective anti-poaching efforts in Ugalla for the benefit of wildlife and their habitat (WD, 1998). Subsequent paragraphs give an account of different types of poaching and poacher activities and their implications for wildlife conservation in Ugalla.

Timber poaching

The most common poaching type in Ugalla Game Reserve was illegal timber harvesting. Timber poaching has also been considered to be problematic in other miombo ecosystems in Tanzania (Luoga *et al*., 2000). Timber harvesting can cause both forest fragmentation (Giliba *et al*., 2011) and wildlife disturbance (Kinnaird *et al*., 2003). Fragmentation creates habitat patches of different sizes, qualities and carrying capacities (Caro & Sherman, 2011). This, coupled with hunting pressure, leads to an increased movement of animals between habitat patches (Navaro *et al*., 2005) and thus biased distribution of animals in the reserve. Disturbance also involves increased animal wariness, and destruction of the vegetation upon which wildlife depend for food. These may consequently alter reproductive abilities of different wild ungulate species (Caro & Sherman, 2011). In addition, during the transect surveys, most of the wooded areas dominated by logging activities were fairly open and easily penetrable with our vehicle. Such openness is likely to encourage bushmeat hunting through enhancing quarry visibility and poachers' access to areas with higher concentrations of wildlife (Wilkie & Carpenter, 1999; Kinnaird *et al*., 2003).

The large group size of timber poachers is a result of the nature of timber exploitation. Much of the timber processing was done by pitsawing, as seen by Luoga *et al.* (2000) in Kitulanghalo Forest Reserve, Tanzania. The technique involves a range of activities from digging pits to setting up logs for sawing, which requires adequate manpower (Wall & Wells, 2000). Furthermore, unlike other types of poachers, timber poachers usually spend much time in their camps when they come across areas with abundant trees suitable for timber extraction. Taking all these factors into consideration, it was much easier for rangers to detect and apprehend timber poachers. Forsyth (2008) noted that it is easier to catch poachers who work in larger groups. I also suspect that a few rich people from Tabora town and some other major cities in Tanzania hire cheap labour from the villages around Ugalla Game Reserve and supply them with all the necessary gear for pitsawing within the reserve. Some of such casual labourers are given logging trucks by their employers to facilitate transportation of the sawn timber out of the reserve. Unless these illegal timber business owners are held responsible for extensive timber poaching, neither punitive sentences to their employees nor confiscation of poacher belongings are likely to discourage timber harvesting.

Employer-employee issues in the logging industry and their influence on bushmeat hunting have been highlighted elsewhere in tropical forests (Wilkie & Carpenter, 1999; Guariguata *et al.*, 2010). Employers normally increase the number of their employees to earn substantial income, and thus pressure on wildlife populations is increased (URT, 1998*a*: Wilkie & Carpenter, 1999). For example, a study in the Katavi-Rukwa ecosystem of western Tanzania by Caro (1999*a*) noted that illegal loggers occasionally use bushmeat as a supplementary diet. This study, though, suggests that this might not be very common in Ugalla. Firstly, because Ugalla east and west hunting blocks had higher number of timber poachers and lower number of wildlife poachers and vice versa respectively. Secondly, despite the fact that most of the areas such as Kamakala and Kakoma in Ugalla east are dominated by timber harvesting activities, in another study (Chapter 2), I showed that pressure on wildlife is somewhat lower than Ugalla west.

The analysis of poacher belongings further proved that timber poachers are very different from bushmeat poachers since a great majority of them were identified with saws; nevertheless, there are some cautions we should bear in mind when dealing with timber poacher belongings as explained to us by one of the most experienced game rangers in Ugalla, Mr. G. Mwanakusha: firstly, timber poachers do not keep all their belongings (for example, radios, bicycles, muzzle loaders and saws) at the camp all the time. Instead, they tend to hide some of them in the bushes a little distance away from their camps and remain with belongings which are really necessary for them to carry out certain activities at certain times. Thus, when they are arrested, the only belongings that get confiscated are the ones at the camp, and in the majority of cases these are saws. Secondly, a few poachers at the camp are usually assigned some other duties that involve staying out of the camp most of the time; for example, going for hunting or fishing expeditions and fetching water from the rivers. Therefore, quite often, when arresting timber poachers any belongings out of their camps with their fellow poachers would never be confiscated.

Bushmeat poaching

Bushmeat was the second most common illegal activity in Ugalla Game Reserve. Bushmeat signs were widespread in Ugalla west, mainly in areas around Msima. This is not surprising because of the hunting pressure exerted on this part of the reserve by refugees from Katumba refugee camp, and villagers (most of whom reside in close proximity to the reserve) (UGR, 2006). The presence of a refugee camp also ensured the availability of modern/automatic guns used in hunting. A study on the relationship between refugee livelihoods and bushmeat hunting by Jambiya *et al.* (2007) acknowledges the fact that availability of modern guns in wildlife areas near refugee camps perpetuates wildlife poaching and increasingly jeopardizes the survival of wildlife populations. Arresting poachers who hunt with modern guns poses a huge challenge to game rangers (Forsyth, 2008); thus, making it difficult to confiscate most such guns. The locally made guns (muzzle loaders) were commonly confiscated because they were used mainly by local hunters from villages around the reserve. Another bushmeat study in western Tanzania also observed extensive use of muzzle loaders for hunting (Carpaneto & Fusari, 2000). The use of guns as a dominant means of hunting makes poachers in the Ugalla ecosystem interestingly different from some other ecosystems. For example, the main bushmeat hunting gear in the Serengeti ecosystem is "wire snares" (Hofer *et al.*, 1996; Kaltenborn *et al.*, 2005; Holmern *et al.*, 2007). A study of discrepancies in wildlife poaching gear/techniques between Ugalla and other ecosystems, and resultant implications for conservation would contribute valuable knowledge towards lessening poaching activities in Tanzania.

According to the Ugalla Game Reserve Management Team, poachers with guns target large-bodied animals such as elephant, giraffe and hippopotamus. The majority of bushmeat hunting signs consisted of meat drying racks and elephant remains. This is also an indication of the commercial use of wildlife resources. When local hunting is progressively shifting from being predominantly subsistence to being one of economic activities (Corlett, 2007), species such as elephant are profitable (Barrett & Arcese, 1995) and are progressively subjected to severe hunting pressure (Jambiya, *et al.*, 2007; Caro, 2008). Elephants are hunted for their ivory (Blake *et al.*, 2007) and sporadically for bushmeat (Barnes, 1996). The killing of elephants in Ugalla dates back to the 1800s during the famous "caravan" trade when people resorted to the lucrative ivory trade at the expense of elephant survival

(Roberts, 1968). Had it not been for the decision of the government of Tanzania to upgrade Ugalla to Game Reserve status in 1965 (Fisher, 2002) and manage wildlife in this area (UGR, 2006), elephants would probably not have survived.

Like timber poachers, the large number and group size of bushmeat hunters in Ugalla reflect the behaviour of most bushmeat poachers. They normally set up camps (bases) where they smoke the meat before taking it out of the reserve. Then there is a division of labour where at most three poachers who are experienced or skilled at shooting animals (known as "Fundi" in Swahili) go hunting while the majority remain at the base camp to continue smoking the already hunted meat, build meat-smoking racks or collect firewood. When game rangers stealthily surround the camp to catch them, none or at most a few are able to escape.

Fish poaching

The group size of fish poachers exceeded bushmeat poachers, but the latter were arrested at a frequency higher than the first (see Table 5.1). Since it is acknowledged that number of poachers is the best indicator of exploitation intensity in a protected area (Holmern *et al.*, 2007; Forsyth, 2008), from the number of poachers caught, illegal fishing seems less common than bushmeat hunting in Ugalla. This is probably because fish were also obtained through legal means and the conditions for obtaining a fishing licence from the Ugalla Game Reserve office in Tabora were achievable (WD, 1998). The aim of introducing such a scheme along with beekeeping was to discourage wildlife poaching (UGR, 2006) because fish is one of the viable alternatives to bushmeat (see Chapter 3). Further research is required to evaluate the relevance of legal fishing and beekeeping activities in subsistence hunting.

I acknowledge the possibility that legal fishing might have confounded the analysis of fish poaching activities. Some fish poaching signs (fishing nets and boats) were mostly seen along the rivers and may not be associated with illegal fishing because it was difficult to tell whether such fishing gear were used for licenced fishing. "Fish drying rack" was the only sign which could be straightforwardly linked with poaching. Poachers dry their fish far from rivers to make it difficult for rangers to detect their presence either physically or through fires and smoke (G. Mwanakusha, pers. comm.). On the contrary, the drying racks for legal fishers are located at the official sites (normally close to the main rivers) to ensure proper monitoring of the legal fishing activities according to the 1998 Ugalla Game Reserve management plan (WD, 1998). Such official drying racks could not be confused with poacher racks. Again on a few occasions it was difficult to make a distinction between bushmeat and fish smoking or drying racks, particularly very old ones. This necessitates the consideration of time-scale when assessing poaching signs (see Milner-Gulland & Rowcliffe, 2007), since it may influence not only identification but also the usefulness of poaching signs as a measure of the resource exploitation intensity.

Like in other ecosystems in Africa (for example, Ajayi *et al.*, 1981; Caro, 2008), most rivers in Ugalla stop flowing in the dry seasons and form chains of disconnected pools, which act as dry-season watering points for animals. Since fish poaching is characterised by widespread illegal fishing activities along the rivers (in areas other than those officially earmarked for licenced fishing), they are likely to be preventing animals from congregating around the pools of water (or water sources) during the dry seasons. Such human disturbance of wildlife represents an "incidental form of poaching", which may compromise animal population performance (see Dobson & Lynes, 2008).

Illegal entry

The majority of poachers arrested for illegal entry into Ugalla Game Reserve were caught with digging equipment. It could not be established whether the intentions of such poachers were to hunt for bushmeat, harvest timber, do some fishing or engage in other activities. Unfortunately, the poaching data obtained from the reserve office do not show clearly what their intentions were, but they were just punished for illegally entering the reserve. The punishment involved paying a comparatively small fine and/or going to jail for a period not exceeding 12 months. Human presence in a protected area regardless of whether they have committed a particular poaching offence has often been considered by many wildlife researchers as one of the primary indices of resource exploitation pressure (Wright *et al*, 2000; Blom *et al.*, 2004; Blom *et al.*, 2005; Lwanga, 2006; Holmern *et al.*, 2007; Waltert *et al.*, 2009). Therefore, the punishment given to poachers arrested for illegal entry could be increased to match deterrence measures applied on other types of poachers.

The observed small number of poachers who committed illegal entry in Ugalla Game Reserve (about 4%), as opposed to timber (58%) bushmeat (25%) and fish (13%), suggests that virtually all poachers commit a poaching offence before falling in the hands of game rangers. Therefore, one would cast some doubt on the effectiveness of anti-poaching efforts in apprehending poachers well before they carry out activities that could lead to devastation or destruction of natural resources. But, anti-poaching in Tanzania is generally hampered by the lack of adequate resources such as vehicles, personnel and financial resources for other logistics (Carpaneto & Fusari, 2000; Holmern *et al.*, 2007). This leaves open the possibility that poachers would only be effectively apprehended after engaging themselves in poaching activities which make them less likely to escape game rangers.

Conclusions

This study has shown the importance of considering a wide range of illegal activities in order to minimize bushmeat exploitation. There are three main issues that emerge from this analysis of poaching.

Firstly, poaching pressures on different types of natural resources differ considerably. Among other things, this reflects differences in wealth among illegal consumers which in turn determine the extent to which a natural resource is exploited (see Coad, 2007). Thus, deterrence of some of the poaching activities such as commercial logging may not be effective simply by confiscating poaching gear or apprehending poor villagers hired to work as loggers in the reserve. There must be a means of identifying and dealing with "power sources" or owners of such businesses. Also, preference for a particular natural resource may determine differences in exploitation intensity across natural resources. This needs to be explored in the future.

Secondly, different types of poachers rarely have overlapping interests in a range of poaching types. This signifies that there are different motives behind different poaching types. Since (as highlighted in the discussion) all illegal human activities in Ugalla Game Reserve are liable to have undesirable influence on wildlife populations, we need to explore empirically the nature and contributing factors of each poaching type. Understanding poachers' motives would enable us to devise more effective ways of dealing with poaching.

Thirdly, poacher activities are spatially unevenly distributed across different habitats within Ugalla Game Reserve. This was evident from the distribution of poaching signs. Wildlife researchers do acknowledge the significance of poaching signs as indicators of the distribution of human activities in protected areas (Wright *et al.*, 2000; Blom *et al.*, 2004; Gavin *et al.*, 2009; Hayward, 2009). Therefore, poachers seem to be endowed with inherent knowledge (Forsyth, 2008) of where in the reserve are the best places to find the natural resources they want to extract. It is important that the reserve management team establish official anti-poaching zones within the reserve, from their own perspective, and enforce anti-poaching laws accordingly. The law enforcement in such zones could also incorporate knowledge about the abundance and distribution of animals, and how they respond to exploitation intensities in general.

On balance, although we should commend the commitment so far of the Ugalla Game Reserve management to anti-poaching, there is still work to do. Most of the reserve is still experiencing unacceptable levels of exploitation (WD, 1998). Apart from law enforcement, a number of other proposals to reduce human pressure have been put forward. One suggestion has been to involve local communities in conservation matters, so wildlife management areas; for example, Uyumbu and Ipole have been established in this regard in the anticipation that their sustainability would halt poaching (see Chapter 7). Anti-poaching efforts in the areas around Msima and Kakoma need to be tightened up. I strongly suggest that all the necessary anti-poaching resources be made adequately available to the Ugalla Game Reserve Management Team.

CHAPTER 6: SUBSISTENCE HUNTING IN THE UGALLA ECOSYSTEM

Abstract

Although most bushmeat hunting is illegal and can cause wildlife population decline to unrecoverable levels, sustainable legal hunting has a place in conservation and often occurs in buffer zones around protected areas. The long term success of such schemes depends on them being well managed. However, major challenge lies in reconciling the various aspects of harvesting management; for example: allowable hunting quotas, the number of licenced animals, and the biological or ecological attributes of the quarry. Here, I assess the effectiveness of legal subsistence hunting around Ugalla using data from the local licencing scheme. The present hunting scheme in the buffer zones around Ugalla Game Reserve is not well managed and wildlife populations are contracting. I put forward recommendations for its improvement, the main one being to integrate it into the Wildlife Management Areas – the current conservation approach outside state-owned protected areas.

Introduction

Concern over direct impacts of consumptive utilisation on wildlife populations has gained wide attention amongst conservationists. For example, hunting can result in reduced population sizes as well as high female-to-male ratio in exploited populations (Setsaas *et al.*, 2007). Alterations in behaviour (Caro, 1999*c*) and social structure (Marshall *et al.*, 2005) can affect animal reproduction in hunted areas (Milner *et al.*, 2007). Consumptive use of wildlife can involve commercial, sport and subsistence hunting (Baldus & Cauldwell, 2004). The last is carried out illegally (wildlife poaching) (Holmern *et al.*, 2007; Chapters 3-5) or legally (subsistence hunting through permit or licence) (Gibson & Marks, 1995), either of which can remove substantial numbers of animals, especially when they occur together (Caro, 2008; Caro & Andimile, 2009). Illegal bushmeat hunting has been widely acknowledged as one of the main challenges confronting conservation efforts, and there is a desperate need to contain the problem (Fa *et al.*, 1995; Milner-Gulland & Akçakaya, 2001; Bowen-Jones *et al.*, 2002; Wilkie *et al.*, 2005; Willcox & Nambu, 2007; Mfunda & Røskaft, 2010). However, the sustainability and place of legal subsistence hunting in conservation are less often explored, even though well-managed legal bushmeat hunting is an important conservation tool in two ways: first, it acts as a sustainable means of meeting protein demands of people (Milner-Gulland & Rowcliffe, 2007); and second, it occurs in areas outside or adjacent to core wildlife protected areas (buffer zones) (Msoffe *et al.*, 2007).

Buffer zones may be considered as sinks from a demographic perspective (Martino, 2001; Newmark, 2008). They help to lessen resource use pressure on core protected areas with relatively less consumptive use (Naranjo & Bodmer, 2007) and those with no consumptive use (Robinson & Albers, 2006). In this scenario, the hunted sink population is maintained by immigrants from the source population (Peres & Nascimento, 2006; Ohl *et al.*, 2007). Buffer zones often experience severe wildlife poaching (Shauri & Hitchcock, 1999) because of little enforcement of conservation laws (Stoner *et al.*, 2007). This can adversely affect wildlife in core areas (Novaro *et al.*, 2005; Newmark, 2008). In addition, buffer zones can contain other human livelihood activities (for example, crop farming and livestock keeping) (Hackel, 1999). Although some livelihood activities can co-exist with wild ungulates (Naughton-Treves *et al.*, 2003), it is generally accepted that most human land-use activities adversely affect wildlife populations and their habitats (Du Toit, 2002; Kideghesho *et al.*,

2006) precisely because the human population size is escalating (Joppa *et al.*, 2009; Wilfred, 2010). Human land-use activities fragment the landscape, forming isolated pockets of natural habitat (Kideghesho *et al.*, 2006) which can hardly maintain wildlife populations amid intensified off-takes (animals killed) (Laurance *et al.*, 2008).

Owing to the conservation significance of buffer zones, ensuring their sustainability is vital (Shauri & Hitchcock, 1999). One approach is the frequent monitoring of population changes of the hunted species (Stoner *et al.*, 2007); for example, by surveying from the air (Reading *et al.*, 2001; Ancrenaz *et al.*, 2005) or ground-based from walked and/or driven transects (Caro, 1999*a*; Waltert *et al.*, 2008; Chapter 2). These approaches are important, but costly (Milner-Gulland & Rowcliffe, 2007; Msoffe *et al.*, 2007); therefore, are carried out infrequently (Witmer, 2005). Conversely, data from wildlife harvesting can be a very useful cost-effective monitoring component, which can act as an indicator of the status of exploited wildlife (Milner-Gulland & Rowcliffe, 2007) on a medium- to long-term basis. Using information from legal (licenced) subsistence hunting, I explore levels of exploitation suffered by different wildlife species in the areas (buffer zone) immediately adjacent to a state or core protected area in Tanzania.

Licenced resident hunting in Tanzania takes place in game-controlled areas and open areas (hereinafter collectively referred to as partially protected areas) (Mabugu & Mugoya, 2001). Most of these adjoin core protected areas such as game reserves and national parks, so they act as buffer areas (Shauri & Hitchcock, 1999). Ecological destruction has been a central issue in their conservation because of unsustainable land use activities (Holmern *et al.*, 2004; Jones *et al.,* 2009; Wilfred, 2010). Wildlife hunting in the partially protected areas is licenced for subsistence purposes (Msoffe *et al.*, 2007) to meet local people's protein needs legally but in a sustainable manner. Some tourist hunting activities also take place (Baldus & Cauldwell, 2004). Subsistence hunting is administered by the Wildlife Division of Tanzania and district game offices (Mabugu & Mugoya, 2001). District game officers suggest or apply for hunting quotas to the Wildlife Division, along with submitting quota utilisation reports (hunting reports) for the previous hunting seasons. The Wildlife Division authorises quotas for the subsequent seasons based on the hunting reports (see Baldus & Cauldwell, 2004). After receiving quotas allocated to different species, district game officers then issue hunting licences to local people (Mabugu & Mugoya, 2001) under a strict condition that Wildlife Division quotas must not be exceeded.

Owing to poor supervision and abuse of the hunting quotas, the sustainability of licenced resident hunting is uncertain (Baldus & Cauldwell, 2004; Holmern *et al.*, 2004; Caro & Andimile, 2009). Its administration does not take into account the ecology of the hunted species (Msoffe *et al.*, 2007), and because of this the current system of granting licences and setting quotas may not be sustainable. While 100% of the licenced resident hunting revenue goes to respective districts (Mabugu & Mugoya, 2001), fees payable for hunting different species are low (Baldus & Cauldwell, 2004); therefore, the revenue can neither pay for conservation (Msoffe *et al.*,2007), nor provide adequate economic gains (Baldus & Cauldwell, 2004).

Tanzania has been promoting the establishment of wildlife management areas (URT, 1998*b*) to encourage effective conservation of wildlife outside core protected areas, address livelihood needs of local communities through wildlife, and minimize other human-wildlife related problems. Most of the wildlife management areas encompass partially protected areas (Wilfred, 2010). However, creation of wildlife management areas is a complex and a time-consuming process (Baldus & Cauldwell, 2004; Nelson, 2007), which means that partially protected areas will continue to exist for an extended period of time. The 1998 Wildlife Policy of Tanzania emphasises that "it will continue to manage" partially protected areas while promoting participatory conservation through wildlife management areas. Therefore, it emphasises the importance of ensuring the sustainability of the partially protected areas (URT, 1998*b*). The present study is aimed at contributing towards realising this ambition by assessing the licenced resident hunting in the partially protected areas adjacent to Ugalla Game Reserve. Specific objectives of the study were four-fold: first, to explore the variation in the number of hunting licences issued and animals removed across different study areas; second, to examine the variation in the number of different species licenced by the district game officers versus their respective quotas set by the Wildlife Division; third, to examine how hunting success rate (offtake per licence) differ across species and districts along with assessing trends over time in the hunting success for districts and species; and fourth, to explore the relationship between hunting success and wildlife poaching in the Ugalla ecosystem.

Methods

Study area

Ugalla Game Reserve lies between longitude 31°26′ to 32°23′ E and latitude 5°31′ to 6°03′ S, covering an area of approximately 5000 km² in the western part of Tanzania. Four administrative districts (Urambo, Tabora, Sikonge and Mpanda) are located close to the reserve. The reserve constitutes a critical component of the Ugalla ecosystem (UGR, 2006). It borders seven forest reserves (Fig. 6.1), which are also linked to other forest reserves; for example, Swangala, Mpembapazi and Itulu. The forest reserves form a buffer zone around Ugalla Game Reserve, and contain partially protected areas in which licenced resident hunting takes place. Ugalla Game Reserve is a source of animals for the adjacent partially protected areas and forest reserves (Hazelhurst & Milner, 2007). With the exception of tourist hunting, and licenced fishing and honey gathering, no human activities are allowed in the reserve (UGR, 2006). Therefore, all other livelihood activities are concentrated in the adjacent areas. Such activities include: livestock grazing, crop farming, licenced hunting, timber harvesting, fishing, beekeeping, human settlements, and illegal utilisation (poaching) of forest and wildlife resources (WD, 1998; Hazelhurst & Milner, 2007). Poaching is a serious problem because of poverty and a massive increase in demand for animal protein (Wilfred & MacColl, 2010). Refugees from the nearby Katumba camps also intensify poaching activities (UGR, 2006). The rapid loss of wildlife habitats in the partially protected areas (as a result of extensive tobacco cultivation and timber harvesting for commercial purposes; overgrazing; expanding human settlements; and burning) cannot be overstated (URT, 1998*a*; Hazelhurst & Milner, 2007).

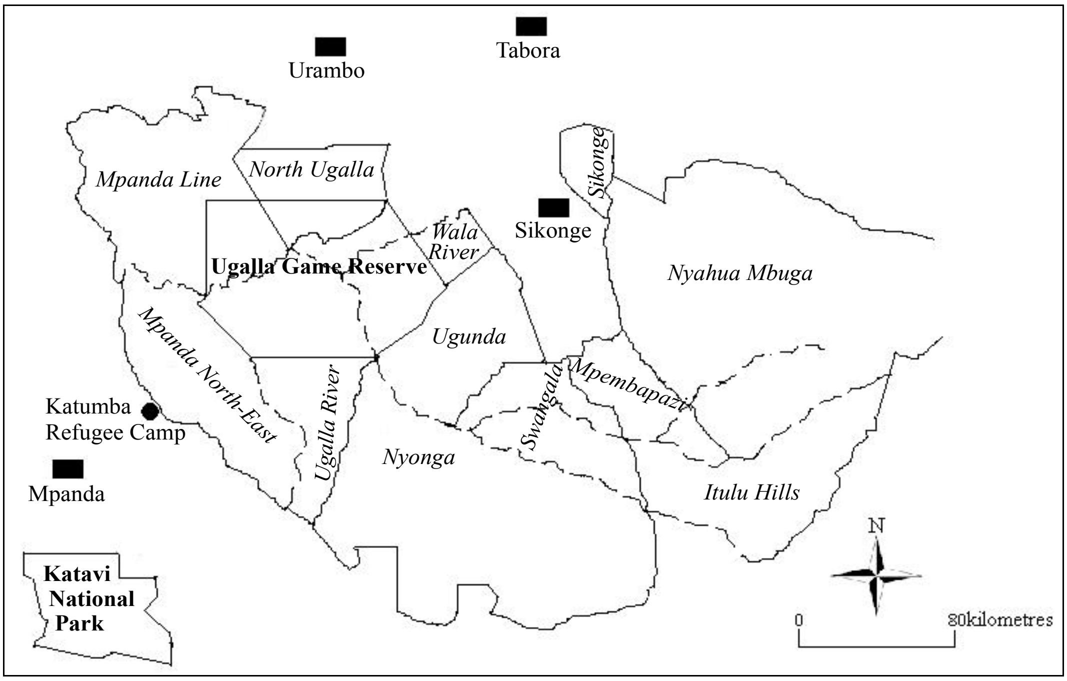

Figure 6.1 A map showing forest reserves around Ugalla Game Reserve. Licenced resident hunting takes place in these forest reserves (partially protected areas). Filled rectangles show approximate centres of the administrative districts around the game reserve. Katavi National Park and Katumba refugee camp (mentioned in the document) are also shown. Broken lines represent different rivers traversing the ecosystem.

Subsistence hunting

The local management of licenced resident hunting in the Ugalla ecosystem is conducted by the district game offices in Sikonge, Urambo, Tabora and Mpanda districts. The first three districts are found in Tabora Region, whereas Mpanda is in Rukwa Region. This study was based in Tabora Region. Much of the information on licenced resident hunting comes from 1997 – 2004, years outside this period had insufficient data. Hunting in Ugalla is legally allowed between 1st July and 31st December each year (dry season). During this time, a hunting licence allows a hunter to hunt a specified number of each animal species for 14 consecutive days during the hunting season. This period (the 14 days) may be renewed or extended when there are special hunting requests from some local institutions (such as academic and religious institutions) or during celebrations of the "Mbio za Mwenge wa Uhuru" (a Swahili phrase meaning "the Freedom Torch Race"). Hunters who cannot use all of their 14 days due to unforeseen circumstances are given a short extension. There is also the possibility of re-applying for another hunting licence in the same hunting season.

Hunters are allocated to hunting sites (Fig. 6.1) in a haphazard manner. In most cases, they are allocated to hunting sites near the district where they applied. For example, Sikonge and Urambo hunters concentrate on areas adjacent to eastern and western Ugalla Game Reserve respectively, whereas residents from Tabora can hunt on either side. Residents are only licenced to kill adult male individuals of a number of ungulate species, but gamebirds of either sex can be taken. The hunting of giraffe, hippopotamus and elephant is prohibited. Upon completion of the hunting expedition, the numbers of animals killed and/or injured are reported to the district game officers on the back of the hunting licence together with any other required information for future reference. The district game officers then send copies of the hunting licences along with hunting reports to the Wildlife Division for further cross-checking. Hunting information available for Tabora District included hunting quotas set by the Wildlife Division, and the numbers of licenced and killed animals species^{-1} year^{-1}. Data from Sikonge and Urambo were much more detailed, with animals licenced and killed species^{-1} year^{-1} licence^{-1}.

The information on wildlife poaching used in this study was obtained from poaching records available in the Ugalla Game Reserve office, and villagers around Ugalla (Chapters 3 & 5). The poaching record spanned from 2003 – 2009, and included names of different species identified with apprehended poachers. Species were identified for each arrest; the number of times a species was recorded represents its poaching frequency during that period. Poaching frequency around Ugalla Game Reserve was also assessed with formal interviews held with 573 residents around the reserve from March – October, 2009. Respondents were asked which wildlife species was mostly hunted in their areas in the preceding year (villager poaching perspective). Their responses were regarded as indices of poaching frequency for each species. I also made use of density estimates of some species from Chapter 2.

Statistical analysis

All analyses were conducted in GenStat (release 10, VSN International Ltd., Hemel Hempstead, U.K.). Generalised linear models (GLMs), with appropriate error distributions, were used to test potential predictors of the number of hunting licences, hunter days, and animals removed in the study area. A GLM was also used to assess the coherence of the annual hunting quotas granted (the Wildlife Division quota), and the adherence to these quotas by licenced individuals for the various species in Tabora district. Both the quota and licenced animals were related to density (for species with density estimates) using a Pearson correlation and a one-tailed significance test. Significance of fixed effects in each GLM was assessed by noting the change in deviance, compared to a chi-squared distribution with the appropriate degrees of freedom. Pearson's correlation analysis was used to examine the relationship, at the species level, between biomass, payable hunting fees and off-takes. Trends in the number of individuals of different species removed hunter^{-1} year^{-1} were analysed with generalised linear mixed models (GLMMs). Statistical significance of fixed effects was assessed by Wald F tests. The relationship between hunting success and over-licencing index (considered as licenced animals per Wildlife Division quota) as well as between over-licencing and the decline trend (the slopes generated by GLMM models) were tested using Pearson correlations. The relationship between legal subsistence hunting (as the mean annual hunting success rates) and wildlife poaching (mean poaching frequencies from Ugalla Game Reserve records and villager perspectives) across species was investigated using a GLM. The significance level for all statistical tests was set at 5%.

Results

Hunting licences & *licenced species*

There was a total of 1,944 hunting licences (with an average of 243 ± 11.9 licences yr^{-1}) issued in Sikonge, Urambo and Tabora districts between 1997 and 2004. A GLM with normal errors was used to investigate differences in the number of licences issued among districts. The response variable "number of licences" was square-root transformed to ensure normality. The predictors were: year (a continuous variable) and district (a fixed factor). The number of licences did not change linearly through time ($F_{1,21} = 0.73$, $p = 0.402$), but varied considerably across study districts ($F_{2,23} = 10.72$, $p<0.001$, Sikonge [mean licences ± s.e.] = 110.13 ± 4.86, Urambo = 69.38 ± 8.23, Tabora = 63.50 ± 8.28). Sikonge (881 licences) had more hunting licences issued than Urambo (555) and Tabora (508). Information on the number of hunting days $licence^{-1}$ was only available for Sikonge and Urambo. The average number of days was 13.87 ± 0.05, ranging from 5 – 29 days. The number of days per hunting licence did not differ significantly between the two districts (GLM with Poisson error structure, $\chi^2 = 0.06$, $p = 0.779$).

A total of 17 species was removed through licenced resident hunting in the Ugalla ecosystem (Table 6.1), with a pooled total of 5,991 animals hunted in all districts from 1997 – 2004. Differences in the number of animals hunted among districts were analysed using a GLM with a normal error structure. The response variable (number of animals hunted) was log-transformed to attain normality. Predictors were: year (covariate) and district (fixed factor). The number of animals hunted varied significantly across districts ($F_{2,23} = 16.28$, $p<0.001$, Sikonge [mean animals ± s.e.] = 344.5 ± 39.6, Urambo = 118.75 ± 24.35, Tabora = 277.6 ± 28.3). Sikonge had the highest number of animals hunted (2,882 animals), followed by Tabora (2,160) and Urambo (949). There was no significant trend in the number of animals killed with time ($F_{1,21} = 0.30$, $p = 0.589$). Fees payable for hunting larger species were higher than smaller species (Pearson correlation, n = 17 species, r = 0.90, $p<0.001$, Table 6.2), but off-take was uncorrelated with either the fee ($r = -0.35$, $p = 0.17$) or the biomass ($r = -0.30$, $p = 0.24$).

Table 6.1 Wildlife hunted in the Ugalla ecosystem through licenced resident hunting. Species are listed in descending body mass.

Species	Local name	Biomass (kg)	Fees (U.S.$)	Off-take
Buffalo (*Syncerus caffer*) Sparrman, 1779	Nyati	450	4.62	201
Eland (*Taurotragus oryx*) Pallas, 1766	Pofu	340	7.69	24
Kongoni (*Alcelaphus buselaphus cokii*) Günther, 1884	Kongoni	125	2.31	618
Topi (*Damaliscus korrigum*) Ogilby, 1837	Nyamera	100	2.31	438
Bushpig (*Potamochoerus porcus*) Linnaeus, 1758	Nguruwe	54	1.54	24
Warthog (*Phacochoerus aethiopicus*) Pallas, 1766	Ngiri	45	1.54	290
Impala (*Aepyceros melampus*) Lichtenstein, 1812	Swalapala	40	1.54	588
Reedbuck (*Redunca redunca*) Pallas, 1767	Tohe	40	1.15	557
Bushbuck (*Tragelaphus scriptus*) Pallas, 1766	Pongo	30	0.92	134
Common Duiker (*Sylvicapra grimmia*) Linnaeus, 1758	Nsya	15	0.46	780
Oribi (*Ourebia ourebi*) Zimmermann, 1782	Taya	14	0.38	352
Dik-dik (*Madoqua kirkii*) Ogilby, 1837	Digidigi	5	0.35	549
Suni (*Nesotragus moschatus*) Von Dueben, 1846	Paa	4.5	0.38	149
African hare (*Lepus capensis*) Linnaeus, 1758	Sungura	2	0.38	71
Ducks & gees (*Anatidae*) Vigors, 1825	Mabata	1	0.23	385
Helmeted guineafowl (*Numida meleagris*) Linnaeus, 1758	Kanga	1	0.23	541
Francolins (*Francolinus*) Stephens, 1819	Kwale	0.5	0.23	290

Official exchange rate in 2009: 1 US dollar = 1300 Tanzanian Shillings.

Wildlife Division quota vs. licenced animals

Investigation of the association between annual resident hunting quotas for different species set by the Wildlife Division and the number of individuals of different species licenced year^{-1} by district game officers was carried out using a GLM with a normal error structure. The response variable "licenced animals" represented number of animals purchased by hunters in Tabora district from 2000 – 2004. The predictors were year (covariate), species (fixed factor), Wildlife Division quota (covariate), and the interactions Wildlife Division quota x species and Wildlife Division quota x year. Of these; year, species and Wildlife Division quota x species were significant predictors in the model (Table 6.2). The number of animals purchased by hunters decreased with time. There were significant differences in the number of animals licenced among species (Fig. 6.2). The slope of the relationship between hunting quota and licenced individuals varied significantly among species (Fig. 6.2). Most species (76.5%) shown in Fig. 6.2 had more individuals licenced to be hunted than the hunting quotas authorised by the Wildlife Division. The difference between hunting quota and licenced individuals was notably large for common duiker, dik-dik, guineafowl and reedbuck.

The correlation between density (individuals km^{-2}) and licenced animals was positive but not significant ($r = 0.57$, $n = 7$ species, $p = 0.18$). Equally, the correlation between density and quota was negative but not statistically significant ($r = -0.50$, $n = 7$, $p = 0.25$).

Table 6.2 Results from a general linear model showing terms associated with licenced animals in Tabora district.

	Estimate ± s.e.	d.f. (change, residual)	F-value	Probability
Constant	99.5 ± 32.3			0.003
Year	-0.05 ± 0.016	1,73	9.34	0.003
Species		18,90	5.96	<0.001
Wildlife Division quota		1,72	1.23	0.271
Wildlife Division quota x Species		14,71	2.47	0.008
Year x Wildlife Division quota		1,57	0.56	0.458
Species' reference level: African buffalo				

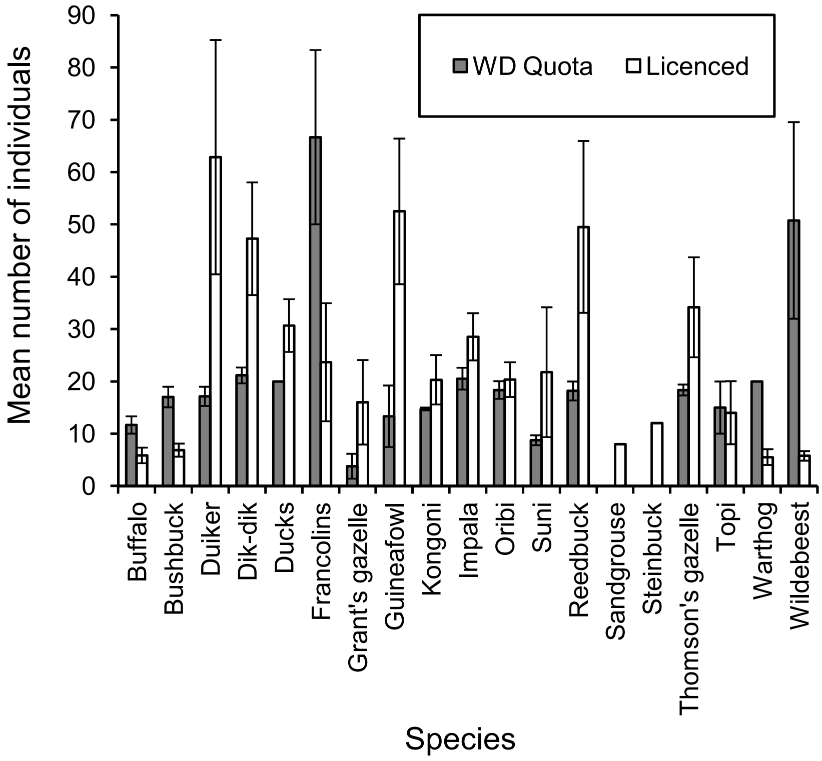

Figure 6.2 Comparison between licenced individuals and hunting quota granted by the Wildlife Division (WD Quota) across different species in Tabora. Error bars are the standard error of the mean.

Offtakes & hunting success per individual hunters

Hunting success, in this case, refers to the number shot as a proportion of individual animals that were licenced per hunter (quota hunter^{-1}). But, first, I examined variations in quota and offtake trends species^{-1} hunter^{-1} using two generalised linear mixed models (GLMMs) with Poisson error structure and a logarithm link function. The fixed effects in both models were species, year, and their interaction. The hunter's licence number (normally shown on the top of the hunting licence) was included in each of the models as a random effect. In the first model (the variance component for the random effect – licence – [±s.e.] = 0.0505 ± 0.0028) the response variable was quota hunter^{-1}. The predictors: species, year and species x year were found to be the best predictors of hunting quota hunter^{-1}. Mean hunting quota hunter^{-1} increased significantly with time (slope ± s.e. = 0.02 ± 0.01,

$F_{1,\ 1512.8}$ = 18.42, p<0.001, Fig. 6.3*a*). Quota hunter^{-1} also varied among species ($F_{16,\ 4364.1}$ = 236.23, p<0.001, Fig. 6.3*c*). The trend, over time, of hunting quota hunter^{-1} varied significantly across species ($F_{16,\ 4427.1}$ = 3.43, p<0.001, see Fig. 6.4). The second model (variance component = 0.0449 ± 0.0087) included animals shot hunter^{-1} as a response variable. All the predictors above were also statistically significant in this model. Animals shot hunter^{-1} decreased significantly with time (slope ± s.e. = -0.03 ± 0.02, $F_{1,\ 1355.3}$ = 63.92, p<0.001, Fig. 6.3*b*). In addition, animals shot hunter^{-1} differed significantly between species ($F_{16,\ 4795.6}$ = 30.84, p<0.001, Fig. 6.3*d*). The trend of animals shot also differed significantly between species ($F_{16,\ 4808.2}$ = 2.42, p = 0.001, Fig. 6.4). All ungulate species, except common duiker, had mean animals removed hunter^{-1} less than 1 across the years of subsistence hunting data (Fig. 6.4).

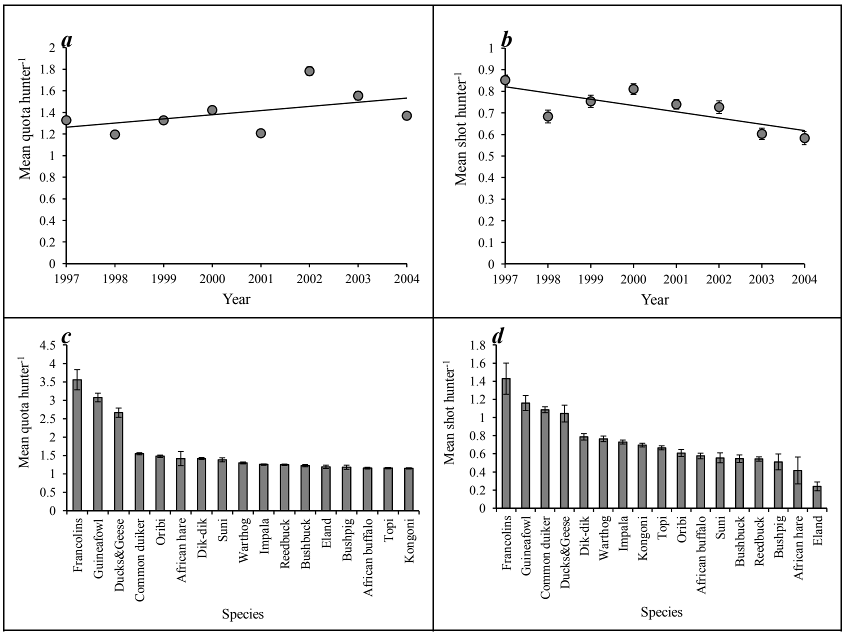

Figure 6.3 Mean animals per hunter, licenced (or quota) and shot, plotted according to time and species. Error bars are the standard error of the mean.

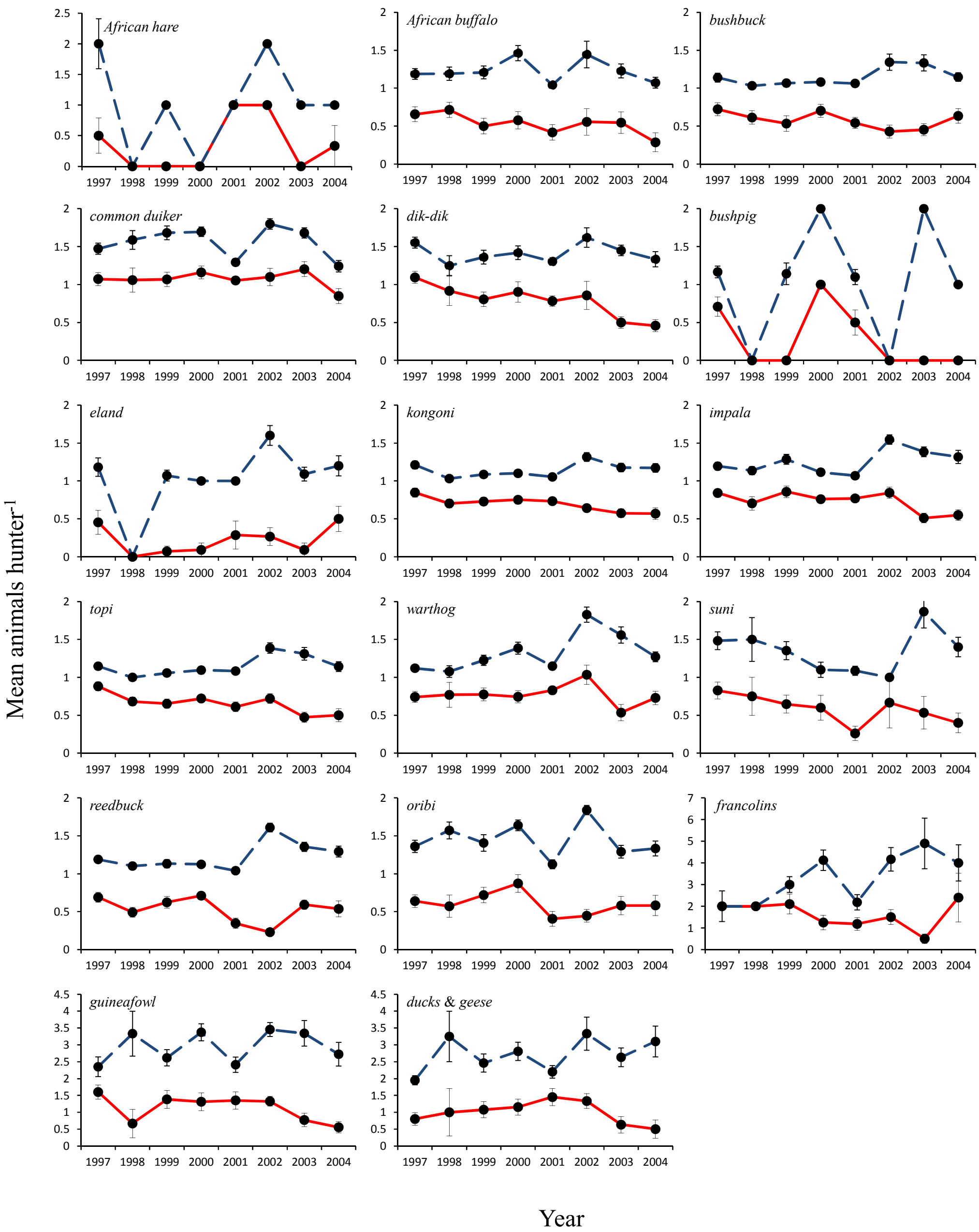

Figure 6.4 Time (years) plotted against mean of animals hunter^{-1} among species removed through legal subsistence hunting in the Ugalla ecosystem. Broken lines represent quota species^{-1} hunter^{-1}, and continuous lines represent animals shot species^{-1} hunter^{-1}. Error bars are the standard error of the mean.

A GLMM was then used in the analysis of hunting success, where the response variable "hunting success" was modelled with a binomial error structure and a logit link function.

The random effect was the hunter's licence number. The predictors were: district (fixed factor), species (fixed factor), year (covariate), and relevant interactions. All predictors except "district x species x year" significantly influenced hunting success (Table 6.3). Individual hunters in Urambo were more likely to shoot animals they had paid for than those in Sikonge (Fig. 6.5). Overall, hunting success tended to decrease with time (slope = -0.099 ± 0.05), more so in Sikonge (-0.13 ± 0.051) than Urambo (-0.021 ± 0.032) (Fig. 6.5). Hunting success varied significantly across species. With the exception of African hare and bushpig, success rates for all species were higher in Urambo than Sikonge (Fig. 6.6). Trends in hunting success also differed significantly between species (Fig. 6.7). For most of the ungulates, success rate decreased with time. Of the birds, francolins and guineafowl had similarly decreasing trends in hunting success rates.

There was no significant correlation between the over-licencing index and hunting success ($r = 0.03$, $n = 12$ species, $p = 0.92$) or the decline trend ($r = -0.18$, $n = 12$, $p = 0.57$).

Table 6.3 GLMM output showing terms associated with hunting success for licenced hunting scheme in the Ugalla ecosystem. Variance component for licence = 0.195 ± 0.038.

	n.d.f., d.d.f	F-statistic	Probability
District	1,2249	56.78	<0.001
Species	16,4903	17.25	<0.001
Year	1,1339	106.71	<0.001
District x Species	15,4931	2.08	0.008
District x Year	1,2569	11.62	<0.001
Species x Year	16,4910	1.93	0.014
District x Species x Year	15,4916	1.23	0.238

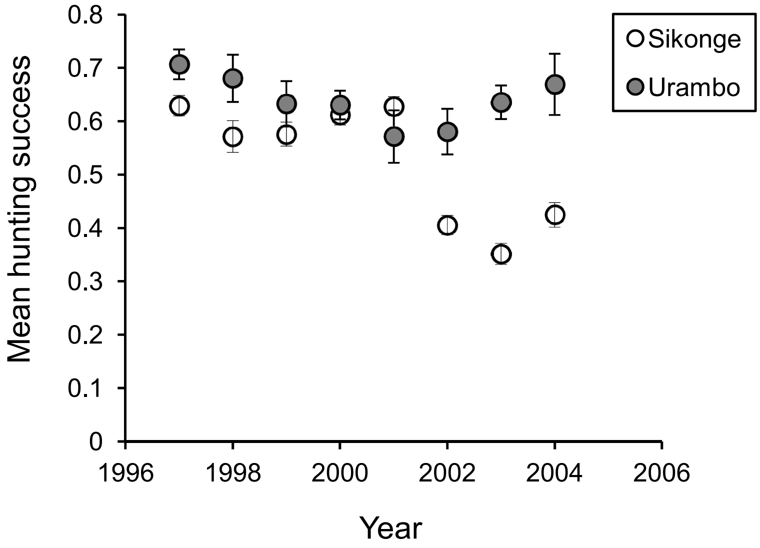

Figure 6.5 Trend in hunting success rate between Sikonge and Urambo districts. Error bars are the standard error of the mean.

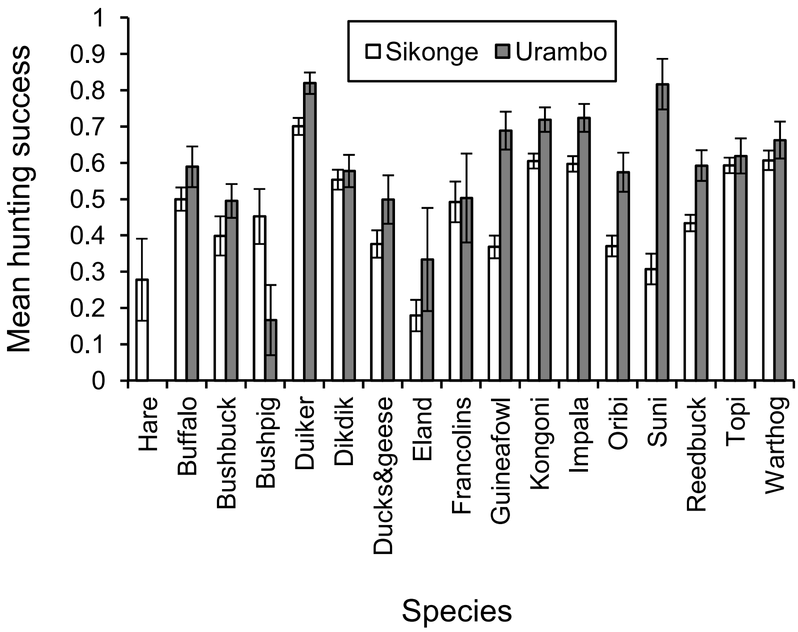

Figure 6.6 Hunting success across different species in Sikonge and Urambo districts. Error bars are the standard error of the mean.

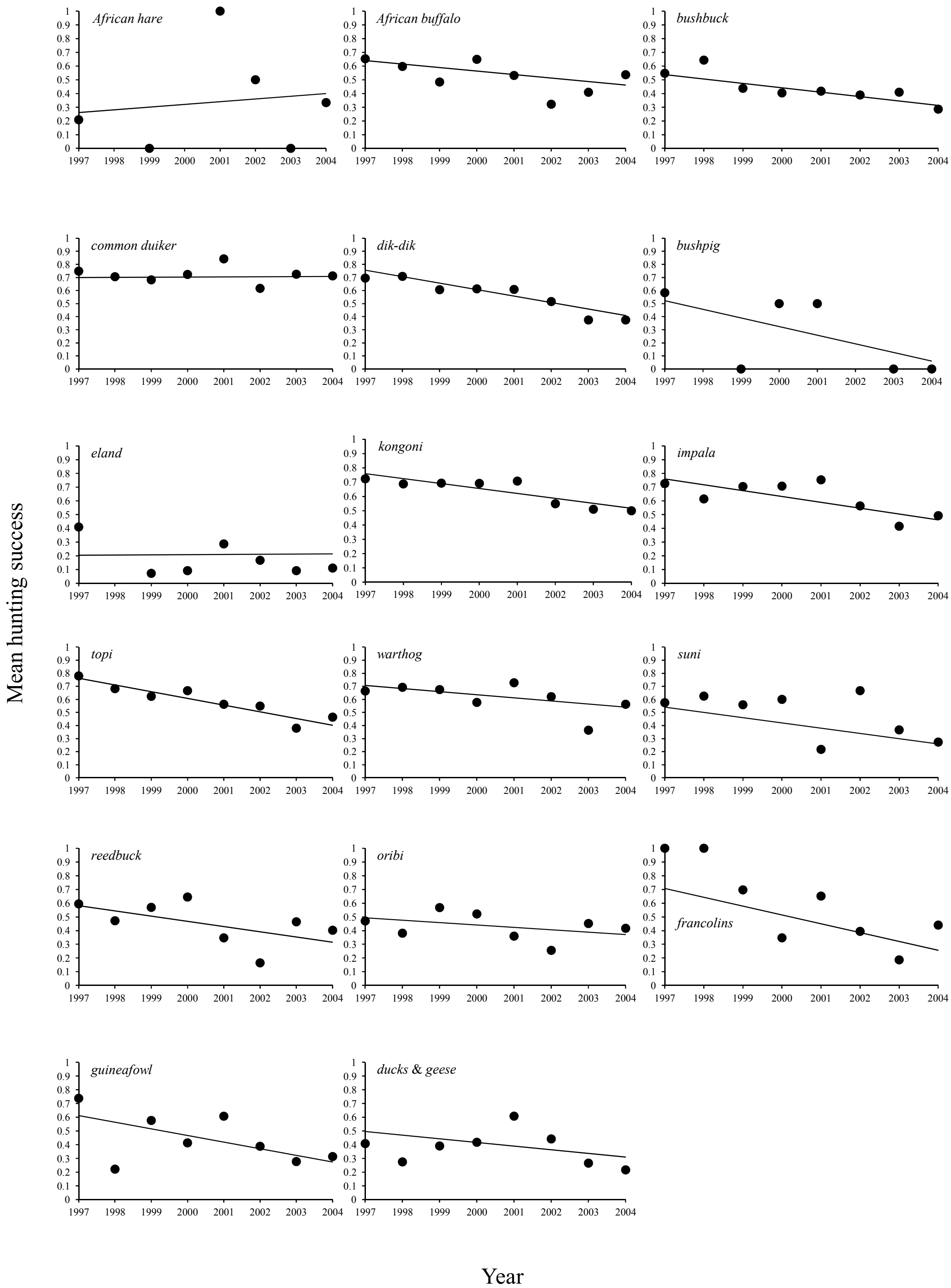

Figure 6.7 Time (years) plotted against hunting success rate for different species removed through legal resident hunting in the Ugalla ecosystem. Trend lines were fitted using estimates (effects) generated by a GLMM model.

Poaching & licenced hunting success rate

Exploration of the relationship between wildlife poaching and hunting success was carried out using a GLM with a normal error structure. The response variable in this case was "mean hunting success rate", whereas actual poaching in the Ugalla Game Reserve (mean number of times a species was illegally hunted), and villager poaching perspective (frequency with which a species was poached outside Ugalla Game Reserve – as estimated by villagers) were fitted as covariates. Species body mass was also included in the model as a covariate. The best predictor of mean hunting success in the model was villager poaching perspective ($F_{1, 16} = 19.14$, $p<0.001$, estimate ± s.e. = 0.06 ± 0.01). This means that hunting success rate was considerably higher for species mentioned by villagers as being frequently poached. Actual poaching in Ugalla Game Reserve and biomass were not significantly related to mean hunting success (actual poaching in Ugalla Game Reserve: $F_{1, 14} = 3.61$, $p = 0.080$, 0.020 ± 0.011; biomass: $F_{1, 15} = 3.20$, $p = 0.095$, -0.0003 ± 0.0002).

Discussion

These results suggest that licenced resident hunting in areas outside Ugalla Game Reserve may not be effective, as widely acknowledged across wildlife studies elsewhere (for example, Baldus, 2001; Newmark, 2008; Wittemyer *et al.*, 2008; Abensperg-Traun, 2009). About 76% of the ungulate species in Ugalla are hunted under the licenced resident hunting scheme. This prompts the need for sustainability owing to the presence of other impacts on ungulate populations highlighted in the bushmeat literature. For example: natural predation (van Schalkwyk *et al.*, 2010), tourist hunting (Caro, 2008), diseases (Lloyd-Smith *et al.*, 2005), and poaching (Wilfred & MacColl, 2010). Areas that demand special attention include the allocation of hunting quotas and the issuing of hunting licences (Balsus & Cauldwell, 2004) as well as trends in off-take rates (Taylor & Dunstone, 1996).

Hunting licences

The mean number of hunting licences in the Ugalla ecosystem was lower than other ecosystems; for instance, the 1998 local hunting licences issued in the Serengeti ecosystem (Holmern *et al.*, 2004). Apart from other reasons specific to Ugalla, this discrepancy is apparently due to the high abundance of ungulates in Serengeti (Dobson *et al.*, 2010). The number of hunting licences differed between study districts, with Sikonge having more licences issued than the other districts. One reason behind this variation could be the availability of hunting concession areas. Loss of local hunting areas and its impacts on hunting activities in Tanzania has been highlighted by Baldus & Cauldwell (2004). Sikonge contains most of the local hunting areas in Ugalla (see Fig. 6.1), whereas other districts (for example, Urambo) had only a few hunting areas around Mpanda Line and North Ugalla forests. The high number of local hunting concessions also attracts outsiders or foreign clients, who look for ways to escape the much higher fees levied for hunting in the game reserve. The 1998 Wildlife Policy of Tanzania acknowledges this by stating that "while there is a thriving resident hunting industry in open areas, it is now recognized that this serves the richer urban-dwelling Tanzanians and non-citizen residents… on the other hand, richer urban-dwelling Tanzanians apply to shoot a number of animals at well below market prices and at considerable opportunity cost to those rural communities on whose land they hunt" (URT, 1998*b*, page 17). The concern shown by the Wildlife Policy of Tanzania follows the realisation that district game officers do licence tourist agents companies contrary to the provisions of the Tourist Agents (Licencing) Act No. 2 of 1969 (URT, 1969).

Ability to pay hunting fees could be one of the factors influencing number of licences issued (URT, 1998*b*). Hunting requirements such as the possession of an appropriate firearm and one's own transport (see Holmern *et al.*, 2004), abundance of animals outside protected areas (Newmark, 2008), and awareness among the local residents of the possibility of hunting legally (Shauri & Hitchcock, 1999) are additional factors determining the number of local people involved in the licenced resident hunting, and hence the number of animals removed.

The mismatch between hunting quotas & *licenced animals*

The comparison between hunting quotas set by the Wildlife Division and licenced animals by the district game officers is a good indicator of whether licenced resident hunting is a conservation oriented enterprise. It also reflects conflicting interests between Wildlife Division, district game officers and hunters. For example, while authorised quotas for francolins, warthog and wildebeest exceeded others, they were far less licenced than common duiker, dik-dik, guineafowl and reedbuck. Since my analysis in this section was based on the information from only one district (Tabora district), I cannot rule out the possibility of it being unrepresentative of other districts. Nevertheless, it throws some light on issues worthy of consideration in the planning and implementation of the licenced resident hunting.

The ungulates which are usually licenced are likely to be among the most common ones in the hunting areas outside and around Ugalla Game Reserve. Milon & Clemmons (1991) argued that hunters normally go for certain species depending on their past experience of their availability and distribution. For example, common duiker and dik-dik are the most frequently hunted species outside Ugalla Game Reserve (Carpaneto & Fusari 2000). Yet, for some reason, the authorities responsible for administering licenced resident hunting in Ugalla have been setting hunting quotas and issuing licences for species such as wildebeest, Thomson's and Grant's gazelles, which are either very rare or absent altogether (see also Carpaneto & Fusari, 2000). These species are found in some other ecosystems where licenced resident hunting also takes place; for example, Tarangire-Manyara (Msoffe *et al.*, 2007) and Serengeti (Holmern *et al.*, 2004). Here, Wildlife Division does not seem to care much about the availability of the various species when deciding hunting quotas, simply because district game officers propose or apply for quotas while paying too little attention to knowledge about species (species density and distribution) and the frequency with which they are licenced. Baldus & Cauldwell (2004) and Msoffe *et al.* (2007) argued that both hunting quotas and licences for resident hunting should be granted according to the reality on the ground. In an effort to address this challenge and ensure effective administration of the licenced resident hunting, Wildlife Division usually specifies some crucial issues that district game officers are supposed to observe when preparing hunting reports soon after the hunting season. These include: species abundance and distribution; habitat availability and distribution; rarity and endemism of the hunted species; and unique behaviour such as migration, feeding and breeding habits. However, such parameters are much harder for district game officers to survey without expertise; as adequate human and financial resources are also a limiting factor (Baldus & Cauldwell, 2004).

Another strategy employed by the Wildlife Division to try to counteract overharvesting, because of the paucity of knowledge concerning status of species utilised under licenced resident hunting, is to set conservative quotas for species (Haule *et al.*, 2002; Holmern *et al.*, 2004; Caro, 2008). But, the majority of species are licenced over and above their quotas. Although over-licencing is not obviously caused by the pressure from hunters to get licences for the common species, it might be caused by district game officers issuing hunting licences just before receiving authorised quotas from the Wildlife Division. On top of that some local community members are haphazardly licenced to hunt through memos and letters from the district game officers instead of the official hunting licences prepared according to the Wildlife Conservation Act No. 12 of 1974 (URT, 1974). This study has no evidence that over-licencing causes species declines. Nevertheless, it might have undesirable consequences on the harvested populations in the long run, specifically because monitoring to assess population performance is less often carried out. Elsewhere, Olson *et al.* (2005) noted that the lack of routine monitoring of the population status coupled with the presence of incentives to hunt would lead to increased off-takes, thereby obviating the value or importance of having quotas.

Hunting success patterns

To the best of my knowledge, this is the first study to carry out the vitally informative analysis of resident hunting success in western Tanzania. The two districts involved in this analysis, Urambo and Sikonge, are important in the management of the Ugalla ecosystem because they contain resident's hunting areas most of which are directly linked to Ugalla Game Reserve. Over-exploitation of wildlife within these areas may have direct impacts on the reserve (UGR, 2006). In this study, hunting success actually represents the degree to which the individual hunter's hunting quota is realised, and not prey abundance as one might have expected. As pointed out by Milner-Gulland & Rowcliffe (2007), before we can attribute harvesting data to population abundance, there are some caveats of which we should be aware. First of all, we should control effectively for hunting effort. Second, assumptions such as "equal catchability", absence of migration and precise measurement of hunting effort must be fulfilled. Apart from the number of hunter days, which barely varied among hunters, my analysis could not take into account any other form of effort. Moreover, the management system of Ugalla allows for the possibility of animals migrating between Ugalla Game Reserve and adjoining habitats (WD, 1998).

All things considered, hunting success has decreased tremendously with time. The most probable reason could be loss of animals resulting from poorly managed licenced resident hunting, poaching (Chapters 3-5) and habitat destruction (Hazelhurst & Milner, 2007). The last is problematic because it goes hand in glove with the maintenance of the livelihoods of local people that is based on natural resources (Naughton-Treves *et al.*, 2003). The predominant environmentally destructive livelihood activity is agriculture (Ramadhani *et al.*, 2002; Kikoti, 2009; Wilfred & MacColl, 2010), particularly the extensive cultivation of tobacco, which offers a great deal of financial income (URT, 1998*a*; Wilfred & MacColl, 2010). Tobacco production in rural areas involves slash-and-burn to ensure the availability of enough land for an increased profit (Mangora, 2005), and the removal of substantial amounts of wood for curing tobacco leaves (Geist *et al.*, 2009) at the expense of wildlife habitats. Extensive livestock grazing (URT, 1998*a*) alters vegetation (Brockington & Homewood, 2001; Krausman *et al.*, 2009), and causes severe competition for water resources between wildlife and pastoralists (Brockington & Homewood, 2001).

The establishment of human settlements in the partially protected areas also claims a substantial amount of wildlife habitat since it expands concomitantly with the need for agricultural land, and building materials (for example, construction poles) (Shauri & Hitchcock, 1999; UGR, 2006). The government in Tabora has been working hard to discourage resettlement in the forests and forcing the invaders out of the partially protected areas in order to save the ecosystem (Hazelhurst & Milner, 2007). Regrettably, the success of such conservation movements depends on political will (Kajembe *et al.*, 2004; Hausser *et al.*, 2009). Local political leaders seeking to renew their terms normally persuade potential voters with mouth-watering promises related to natural resources. There have been few cases where local communities demanded amendment of the Ugalla Game Reserve boundary to give them some additional land for livelihood activities. One of these occurred in 1991/1992 when about 150 km² of the reserve was lost as a result of "improper boundary demarcation" (WD, 1998). The affected areas were northern, and north-eastern (around Wala River Forest) parts of the reserve where there is a high concentration of agro-pastoralists (WD, 1998). Political influence is widespread in conservation areas elsewhere in Tanzania; for example, in the Serengeti ecosystem where the aborted plan to build "a two-lane road through 50 km of the Serengeti National Park" was propelled by a strong political motivation (Dobson *et al.*, 2010).

The difference in the hunting success between Urambo and Sikonge can be explained by the location of the hunting concession areas and the number of the licenced animals. Though Urambo consists of fewer local hunting areas, most of them are probably more directly influenced by Ugalla Game Reserve than Sikonge, since they are closer to the reserve (Fig. 6.1). Hazelhurst & Milner (2007) argued that the reserve does replenish the immediately adjoining hunted areas with ungulate species. The conservation literature suggests that the closer the hunted areas (or sinks) are to the "source" areas, the more effectively the wildlife species they contain are maintained through "source-sink population dynamics" (Begazo & Bodmer, 1998; da Silva, *et al.*, 2005; Novaro *et al.*, 2005). Furthermore, apart from limiting the number of licences issued year^{-1}, the district game officer in Urambo is compelled to limit the number of licenced animals to match the size of the (fewer) available areas (Mr. Katondo, DGO-Urambo, pers. comm.). The mean number of animals ($\pm$ s.e.) licence^{-1} in Urambo was 2.7 $\pm$ 0.08, while that of Sikonge was 6.4 $\pm$ 0.16; Therefore, it was easier for hunters in Urambo to realise the number of animals they were licenced to kill.

Hunters in Sikonge were mostly licenced to hunt in the areas within Swangala, Mpembapazi, Nyahua and Itulu forests, which are comparatively far from Ugalla. Although Carpaneto & Fusari (2000) noted that villagers around Urumwa Forest Reserve (not shown on the map) in Tabora district, which is >100 km from Ugalla Game Reserve, could occasionally hunt animals dispersed from the reserve, it is currently unlikely for animals from Ugalla to migrate so far because of environmental destruction (UGR, 2006). With the exception of a few hunters lucky enough to carry out their hunting activities in the areas around Ugunda and northern Inyonga, the majority of hunters elsewhere in Sikonge are hardly able to realise their individual hunting quotas (Mr. K. Twaha, DGO-Sikonge, pers. comm.).

Species-wise differences in hunting success between Urambo and Sikonge were also found, with the former exceeding the latter. When the two districts were pooled, the rate of decline of hunting success over time was different for each species. Similar results were reported in Arabuko Sokoke Forest in Kenya by FitzGibbon *et al.* (1995), who found that off-take success patterns varied among species for both hunted and trapped mammals. Despite the general decline in hunting success, the vast majority of species had success rates above 40% in the 1999 and 2000 hunting seasons. One possible explanation for this is the El Nino rains in 1997/1998, which probably caused most animals to move and spend much of their time outside Ugalla Game Reserve to avoid massive floods, exposing them to local hunting. Much of the Ugalla Game Reserve is located on the low-lying flood plain alongside Ugalla River, which is extensively flooded in prolonged heavy rains (WD, 1998) like those of El Nino. One main factor for the observed variation in the hunting success across species is the prey "vulnerability". Hunted larger mammals are more vulnerable (FitzGibbon *et al.*, 1995), more "profitable", have "lower intrinsic productivity" (Milner-Gulland & Rowcliffe, 2007), and are thus more affected than smaller mammals. In Gabon, Coad (2007) found that gun-hunted larger species were more preferred and pursued in the areas far from the village because nearby areas were void of such species.

The two largest mammals under licenced resident hunting were eland and buffalo. Eland had a consistently low success rate, which is possibly due to overexploitation outside Ugalla Game Reserve. A study in the Katavi-Rukwa ecosystem of western Tanzania (with habitats similar to Ugalla) found overexploitation of eland in hunted areas (Waltert, *et al.*, 2009). Hunters licenced to kill buffalo were more successful in 1997 and 2000, but hunting success lowered in other years. On the contrary, elsewhere in Africa, buffalo are common in hunted areas (Bouché *et al.*, 2009). Of course I admit the reality that habitat type and quality (Waltert *et al.*, 2009), and illegal hunting (Hilborn *et al.* 2006; Waltert *et al.*, 2009) influence the availability of buffalo in hunted areas. Annual buffalo quotas allocated in Ugalla suggest that the rate of buffalo licences issued may be decreasing as a response to hunting success. Basing on Tabora district, the mean Wildlife Division quota allocated for buffalo from 2000 – 2004 was 11.7. This is lower than that of the Katavi-Rukwa ecosystem in 1993 and 1996 (32 buffalo) (Caro, 2008), Kilombero partially protected area in 1995 (15) (Haule *et al.*, 2002), and Serengeti in 1998 (18) (Holmern *et al.*, 2004). In addition, the ratio of the number of licenced buffalo to the quota in Ugalla (57%) was lower than Serengeti (90%) (Holmern *et al.*, 2004).

A decreasing trend in hunting success was also a characteristic of other mammal species in Ugalla except common duiker and African hare. For some of the species, such as bushpig and African hare, hunting success patterns were influenced by the availability of data. These species had no hunting information for 1999 and 2003, and their off-takes seem to be strongly influenced by cultural and traditional backgrounds of consumers. For example, in Serengeti Muslims would not consume bushpig (Kideghesho, 2006). Common duiker had the most stable success rate among all the species. Its success rate was at least 60% throughout. Notwithstanding the fact that it is commonly preferred as a bushmeat species (Lwanga, 2006), common duiker is one of the species which can endure anthropogenic habitat disturbance; therefore, they are unsurprisingly abundant outside protected areas (Averbeck, 2009). Carpaneto & Fusari (2000) found that common duiker had the largest number of individuals removed by local hunters in the Ugalla ecosystem. Dik-dik, bushbuck, kongoni, topi and warthog showed a consistently declining pattern; which is very informative when considering their exploitation pressures. Like common duiker, dik-dik is a common species in western Tanzania (Carpaneto & Fusari, 2000); thus, its continuous downward trend should be closely monitored. Other species, as warthog (Waltert *et al.*, 2009) and impala (Setsaas *et al.*, 2007), are known to be under continuous pressure from both legal and illegal hunters in the partially protected areas (Stoner *et al.*, 2007).

Hunting success rates for large birds declined sharply from 2001 – 2004. This cannot straightforwardly be related to overexploitation as in the case of mammal species. Since subsistence hunting is becoming economically more significant (Damania *et al.*, 2005), the trend in hunting success for large birds could be a trade off between the cost involved in successfully killing the prey and the ensuing anticipated profit. Therefore, mammals are preferred over large birds (Redford, 1992). In Chapter 3, I showed that birds such as duck and francolin were much less frequently mentioned by villagers as being important to hunters. In other hunted areas in the Serengeti ecosystem, exploitation does not affect large birds (Magige *et al.*, 2009) in the same way it affects mammals (see Makacha *et al.*, 1982; Setsaas *et al.*, 2007). Nevertheless, the presence of large birds on hunting quotas shows their potential conservation value in the partially protected areas. The Ugalla ecosystem is rich in gamebird species (UGR, 2006); and if bird hunting could be sustainably promoted, it could be of both economic and conservation importance like duck hunting in southern Australia (Bennett & Whitten, 2003).

The hunting success patterns for gamebirds explain further the quality of the habitats in the partially protected areas. Birds are good indicators of ecological integrity (Canterbury *et al.*, 2000). Thiollay (2006) found that the decline in species of large birds outside protected areas in Burkina Faso was a result of habitat destruction and fragmentation induced by anthropogenic activities. An inverse relationship between the degree of human forest disturbances and the abundance of large birds is also detailed in an avifauna study by Martin & Blackburn (2010) conducted on Buton Island, southeast Sulawesi. This is because birds are more sensitive to habitat changes caused by human beings (Fjeldså, 1999). For instance, francolin and guineafowl are listed among six bird species whose populations suffer from habitat loss and fragmentations (Fuller *et al.*, 2000). Owing to the ecological importance of birds, further study would be required to determine whether the abundance and distribution of different bird species would reflect habitat integrity, and the applicability of this in conservation and sustainable use of other wildlife species in the partially protected areas.

Hunting success & *wildlife poaching*

The positive relationship between wildlife poaching and hunting success suggests that species killed by legal hunters are the ones sought after by poachers. These species are probably either commonly poached in Ugalla (Chapter 3) or they are easy to catch subject to factors influencing legal hunting success discussed earlier on. This scenario complicates wildlife conservation in general and bushmeat trade in particular since it is very challenging to discriminate between legal and illegal bushmeat, especially when consumers offer much less cooperation in bushmeat surveys (Nyahongo *et al.*, 2009; Chapter 3). Legal resident hunters are strictly prohibited to conduct any kind of bushmeat trade, but since the meat is normally sold on the black market (Ndibalema & Songorwa, 2007; Knapp*, et al.*, 2010) it is difficult to determine whether legally hunted bushmeat is actually intended for sale. The Wildlife Division game rangers and village game scouts occasionally carry out inspections. When they come across someone in possession of bushmeat without a valid hunting licence, the meat is deemed illegal and measures may be taken against offenders (K. Twaha, pers. comm.). Again difficulties may arise here since legal hunters are supposed to return their hunting licences to the district game officers for future records, and in some cases licences are either misplaced or outdated and the stock of legally hunted bushmeat from the previous season is not yet finished. Under such circumstances, ensuring justice to people caught with bushmeat remains very questionable and poachers are likely to be taking advantage of the licenced resident hunting to intensify their activities. Suffice it to say that contemporary bushmeat exploitation is extraordinarily dependent upon income generation (Loibooki *et al.*, 2002; Coad, 2007). This runs counter to the popular claim that legal subsistence hunters are trustworthy enough not to engage in the bushmeat trade.

Conclusions

This study has shown that wildlife in areas around Ugalla Game Reserve (partially protected areas) is under an unacceptable intensity of exploitation. The sustainability of Ugalla Game Reserve depends greatly on wildlife utilisation in the partially protected areas. These provide a crucial buffer zone to the reserve (UGR, 2006). There are two main ways through which wildlife is utilised in the partially protected areas, namely: wildlife poaching and licenced resident hunting. The former is more detrimental and normally takes place against wildlife conservation laws, but the latter is legally allowed and carried out under the control of responsible local authorities. In Tanzania, poaching is hard to quantify because it is not regulated and illegal hunters are very reluctant to share information about their activities (Caro & Andimile, 2009; Knapp *et al.*, 2010) for fear of punishment. Owing to the significance of buffer zones, monitoring to ensure sustainable use of wildlife resources must be more frequently carried out. Information on licenced resident hunting can provide a cost-effective assessment of the exploitation intensity suffered by wildlife in partially protected areas. This has been my intention throughout the present study, which culminates in the following key conclusions:

- Residents in Sikonge bought more local hunting licences than any other study district. This shows widespread awareness of the importance of legal bushmeat hunting among Sikonge dwellers together with availability of hunting concession areas.

- The number of animals licenced declined with time. Considering the case of Tabora district, three ungulate species seem to be over-licenced. These are: common duiker, dik-dik and reedbuck, licenced over and above their Wildlife Division quotas. Of the gamebirds, guineafowl were licenced more than their authorised quota. Some other species such as sandgrouse and steinbuck are licenced without any quota allocation from the Wildlife Division, while others (for example, wildebeest and Grant's gazelle) are licenced in spite of the fact that they are not common in Ugalla. Generally, I suspect that the exercise of deciding hunting quotas and granting licences is not carried out according to wildlife population performance. If this situation is not responsibly dealt with, hunted areas around Ugalla may end up being "empty forests", as argued elsewhere by Redford (1992). This would severely affect the Ugalla Game Reserve buffer zone, and more so the reserve itself.

- With the exception of common duiker, the rate at which all ungulates were successfully removed decreased with time. Therefore, hunting pressure is increasing for certain species which seem to be common and easy to catch. This is a worrying situation because both legal and illegal hunters appear to share interests in terms of what species to hunt.

I further strongly recommend the following:

- As it stands, resident hunting is not compatible with conservation aspirations, and may need to be revamped. In my opinion, I think that the conservation approach "wildlife management areas" should encompass licenced resident hunting as soon as possible. Currently, there are two wildlife management areas in Ugalla: Ipole and Uyumbu. Effective administration of the wildlife management areas will probably benefit local communities, buffer zones and core protected areas. In Chapter 7 I give a detailed account of how wildlife management areas can be sustainably utilised to ensure that wildlife in Tanzania receives the protection it deserves.

- Since studies do not recommend prohibiting all bushmeat hunting outside protected areas (see Holmern *et al.*, 2007), introduction of ecologically viable and socially acceptable species would help to alleviate pressure on the local wildlife. Such an approach was used in the Pantanal wetland portion of Brazil, where the introduction of feral pigs is said to have benefitted conservation (Desbiez *et al.*, 2011). In Cameroon, Njiforti (1996) recommended domestication of the acceptable species; citing the example of successful guineafowl domestication project in Nigeria. However, Alho *et al.* (2011) highlighted some caveats with regard to implications such approaches might have to conservation.

- Active supervision coupled with effective law enforcement, raising of awareness and outreach programs are necessary to minimize hunting impacts.

- District game officers should be educated on the sustainable use of wildlife, and be made aware that animals protected in Ugalla Game Reserve are the ones removed through legal resident hunting in the partially protected areas, and that overexploitation in this buffer zone would be destructive to the reserve.

- Hunting licences should not be extended or issued irresponsibly. Any licence extensions issued based on private or personal reasons may create dangerous loopholes, which can be used to fulfil interests of the district game officers and hunters alike. Though requests for extensions are made within the same hunting season (i.e. calendar year), there is a possibility for hunters to apply for extensions more than once or apply to a different district even if this would mean changing their identity.

CHAPTER 7: TOWARDS SUSTAINABLE WILDLIFE MANAGEMENT AREAS IN TANZANIA

A slightly modified version of this chapter appears as: Wilfred, P. (2010). Towards sustainable Wildlife Management Areas in Tanzania. *Tropical Conservation Science* **3**: 103-116.

Abstract

Within the last few years, Tanzania has witnessed mushrooming growth of "wildlife management areas" (WMAs). These are broadly meant to halt (or reduce) loss of wildlife populations, and ensure that local people benefit from their conservation. However, human pressure is rapidly increasing and creating management problems in the WMAs. Some human land-use activities also limit wildlife dispersal, potentially destabilizing wildlife population dynamics. In addition, poor resource use diversification and lack of creativity constrain sustainable use of natural resources in the WMAs; consequently, their contribution to sustainable livelihoods is seriously undermined. A key question is how WMAs can be a sustainable and competitive land-use option that meets their predetermined objectives? Without doubt, a road map to sustainable WMAs should responsibly engage the government, non-governmental organisations, and community-based organisations in a joint effort towards realisation of simple and flexible WMAs establishment process, quality wildlife habitat, and reduced human pressure on the wildlife resources, as well as successful and sustainable wildlife-based enterprises.

Introduction

Sustainable conservation of wildlife resources has been one of the core objectives of wildlife managers and biologists in many countries in Africa. For centuries, wildlife has been utilised not only for subsistence but also for commercial purposes. As human population expands, wildlife resources are increasingly subjected to severe pressure, which threatens their existence and sustainability (Milner, *et al.*, 2006; Caro & Scholte, 2007; Smith, 2008). Apart from consumptive utilisation, other anthropogenic activities such as agriculture have indirectly influenced the survival of wildlife species through manipulation of their habitats (Kideghesho *et al.*, 2006). Since most local communities have a historical link with wildlife in rural areas, efforts to ensure sustainability have been focusing on involving local people in conservation. Many governments have adopted a participatory approach to conservation as a result of pervasive loss of wildlife species and the challenges of a "fences and fines" approach (Adams & Hulme, 2001; Nishizaki, 2004; Büscher & Whande, 2007). Countries in the southern part of Africa such as Namibia, Botswana, Zambia, and South Africa have had a good experience in community-based conservation (Munthali & Mughogho, 1992; Lewis & Alpert, 1997; TNRF, 2008). In the rest of Africa, for example in east Africa, participatory conservation has been confronting some challenges. This has led to a considerable concern over community-based conservation initiatives in this wildlife-rich part of Africa (Baldus *et al.*, 2001; Rutten, 2004; Bett, 2005; Saito, 2007).

In attempting to promote sustainability in the use of wildlife resources, the government of Tanzania has introduced the concept of Wildlife Management Areas (WMAs). These are areas of community land in which local people have usage rights over the wildlife resources. Conservation of natural resources in WMAs is therefore a shared responsibility and local communities must significantly benefit from it (URT, 1998*b*; Stolla, 2005). WMAs started as one of the tools in a new approach to managing wildlife resources in the early 1990s. According to URT (1998*b*), wildlife ownership will be decentralized to local government, and to the rural communities that are recognized as important stakeholders in the wildlife conservation. The logic behind WMAs is that when local communities develop a sense of resource ownership and realise the tangible benefits that can accrue from wildlife conservation, they will develop a positive attitude towards conservation issues (Severre, 2000; Felix, 2004). This chapter seeks to present some factors that need to be given attention in the current escalating interest in WMAs in Tanzania in particular and indeed in other parts of Africa.

Background on wildlife management areas in Tanzania

The history of wildlife conservation in Tanzania goes back to 1891 when colonial laws controlled the use and management of wildlife resources (URT, 1998*b*). Due to this top-down approach to conservation, integration of wildlife conservation into rural development was not a priority. As a result, in the 1970s and 1980s, Tanzania saw a pervasive decline of its wildlife. Factors involved in this decline included poverty, flourishing markets for wildlife products, increased human population and demand for bushmeat, and lack of trained personnel and financial resources to do conservation work, as well as local people's negative attitude towards conservation (WSRTF, 1995). Therefore, much of the wildlife (especially outside protected areas) became increasingly scarce (Shemwetta & Kideghesho, 2000). In response to this rapid loss of wildlife, the government, through the National Parks Authority and Wildlife Division, began to emphasise collaboration with local communities as part of a protected areas management strategy. By 1995, the Wildlife Sector Review Task Force [WSRTF] had suggested the creation of village-based WMAs in order to lay the basis for sustainable management and utilisation of wildlife resources at the grass-roots level (WSRTF, 1995).

Three years after the WSRTF report came out, the wildlife policy of Tanzania was put in place. The 1998 wildlife policy reflects the willingness of Tanzania to decentralize wildlife management issues, and accommodates much of the people's needs and interests in its conservation plans. Implementation of the policy is evidenced in the current mushrooming of WMA projects in the country. So far, there are about 16 pilot projects in 16 districts encompassing more than 135 villages. It is envisaged that through WMAs, local communities will attach a considerable value to wildlife as they do in other forms of land use, for instance, agriculture. This would in turn lead to a reversal of wildlife declines and enhanced movements or dispersal of wildlife species (URT, 1998*b*; TNRF, 2008).

The process of establishing a WMA involves the following steps: creation of awareness among villagers of the merits and disadvantages of having a WMA; a village assembly's approval of an application for WMA formation taking into consideration village council recommendations; formation of a community-based organisation; preparation of a strategic plan; preparation of a land-use plan; carrying out of an Environmental Impact Assessment prior to approval of a land-use plan; preparation of village by-laws that support the land-use plan; and preparation of a resource management zone plan. The community-based organisation then makes application to the director of wildlife for designating part of village land as a WMA; the director considers the community-based organisation's application and sends his recommendation to the Minister of Natural Resources and Tourism; and finally the minister declares a designated WMA by order in the gazette. After this, the community-based organisation applies to become an authorised association, and the authorised association applies for a user right and hunting block to the director of the Wildlife Division. The authorised association may also enter into investment agreements with potential investors (IRA, 2007; Miniwary, 2009). From local people's perspective, this seems to be a complicated process, which may delay the formation of WMAs and the realisation of tangible benefits accrued from them.

WMAs are one of the categories of wildlife conservation areas in Tanzania. Other categories include Game Reserves and National Parks and National Conservation Areas of varying sceneries (see Chapter 1, Fig. 1.1 & Fig. 7.1). At a local scale, however, there are some challenges to realisation of sustainable wildlife management areas. Some of these challenges can be immediately linked to human influences (Callicott *et al.*, 1999; Miniwary, 2009), for example, loss of wildlife habitat and widespread wildlife poaching. In the sections that follow, I explain most of WMAs' challenges and the possible solutions (opportunities), giving examples of wildlife projects elsewhere in Africa.

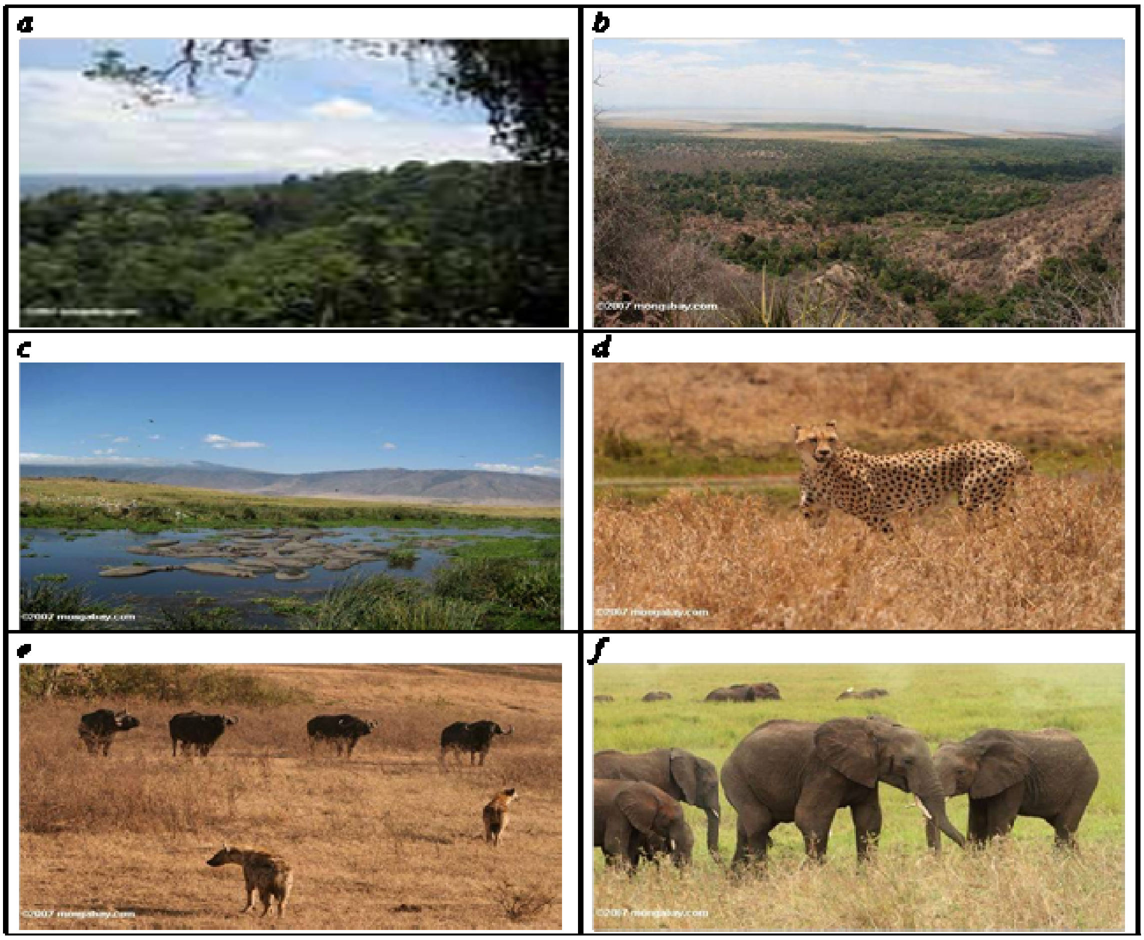

Figure 7.1 Photos of some wildlife areas' landscapes and wildlife in Tanzania. *a*-African montane forest, *b*-Lake Manyara National Park as seen from the rift valley wall, *c*-Hippo pool in the Ngorongoro Crater, *d*-Cheetah (Acinonyx *jubatus*), *e*-Group of buffalo eyeing two spotted hyena, *f*-African elephants. Photos by Rhett Butler – see: http://travel.mongabay.com/tanzania/topics/ngorongoro%20crater10.html.

Wildlife management areas in fragmented landscapes: what can be done?

The importance of land as a fundamental resource in conservation of wildlife cannot be overemphasised. In rural areas where most wildlife is found, a significant proportion of the landscape is used for agriculture, grazing, and settlement. As human population density near wildlife rich areas increases (Wittemyer *et al.*, 2008), even more land is needed for livelihood maintenance. This has increasingly brought human land-use zones into contact with conservation areas (Crooks & Sanjayan, 2006). Therefore, there has been a negative trade-off between rural communities' interest in land use and conservationists' interest in healthy wildlife populations. Some examples in Africa include the Zambezi valley of Zambia at Livingstone, where expansion of farmlands into forested areas has led to ecological devastation and widespread human-wildlife conflicts (Karidozo, 2007). In some wildlife areas of Kwale District in Kenya, local people have been forced to leave their productive land because of crop raiding animals such as elephants, baboons, and monkeys (KWS, 1996). The conflict between the Bénoué Wildlife Conservation Area and the adjacent communities in northern Cameroon (Weladji & Tchamba, 2003) is another example of the tension arising from co-existence between human land-use activities and wildlife conservation. The root cause of the negative attitudes towards conservation among Khwai communities around Moremi Game Reserve in the Okavango Delta in Botswana is the displacement of these communities in order to provide land for gazettement of the game reserve (Mbaiwa, 2005). In some areas of western Serengeti National Park in Tanzania, wild animals have found themselves on the frontline of land-use conflict with pastoralists (Holmern *et al.*, 2006). Displacement of Wagalla people from the Ugalla Game Reserve in western Tanzania in the 1960s (UGR, 2006) has contributed to the current poor support of local communities for conservation efforts. All these are only a few examples showing how land and its resources have become a source of friction between wildlife and human beings.

Land-use conflicts between wildlife and humans have resulted in wildlife habitat fragmentation and biodiversity loss. In regions with high population growth rates (Table 7.1) along with unsustainable land-use activities, wild animals often find themselves in a hostile environment. An important question is, "What is the most effective pattern of habitat fragments/patches to ensure sustainable co-existence between wildlife species and local communities?" (Bennett, 1999). There has been substantial discussion about habitat patches and the movement of wildlife species between different patches amid human pressure, with special attention to the decline of wildlife populations (Merriam, 1984; URT, 1998*b*; Shauri & Hitchcock, 1999; Hassan, 2007). Some wildlife biologists have argued that "species connectivity" may help to stabilize wildlife populations. Connectivity is defined as the extent to which individuals of different species can move from one habitat patch to another in a fragmented landscape. Spatial arrangement and the quality of different habitat elements influence species connectivity (Crooks & Sanjayan, 2006). Species-specific connectivity entails having knowledge of different species and their different habitat requirements (Merriam, 1991; Taylor *et al.*, 1993; Forman, 1995).

Owing to the importance of species' ability to move between suitable habitat patches, WMA stakeholders, namely central government, local governments, and non-governmental organisations, should put emphasis on the regular assessments of the land-use systems and how they influence quality of the actual and potential wildlife habitats in the WMAs. Since tourist hunting and hunting for subsistence by local communities (through permits) are among the land uses in the WMAs (Baldus & Cauldwell, 2004; Nelson, 2007, Chapter, 6), and poaching is often a problem (Chapters 3, 4 & 5), intensive utilisation of wildlife resources is most likely. In such a scenario, enhanced species' dispersal can effectively stabilize wildlife populations, especially in areas of habitat isolates created as a result of fragmentation and destruction of natural vegetation (Pulliam, 1988; Pulliam & Danielson, 1991; Begon *et al.*, 2006).

Population size & pressure on wildlife management areas

Human population density will affect sustainability of WMA projects. The intercensal (1988 - 2002) population growth rates show that, of the regions where WMAs have been initiated, population in Arusha, Manyara, and Tabora regions are rapidly increasing (Table 7.1). The WMA projects in these regions: Enduiment, Loliondo, Burunge, Makame, Ipole, and Uyumbu, have been facing conservation challenges related to resource use and local participation (Sosovele, 2005; Nelson, 2006). The Lindi region, where the Liwale WMA is found, has the lowest growth rate of 1.4%, far below the total Tanzania growth rate of 2.9% (NBS, 2002). Comparison of the population size by districts reveals that Tarime is the highest, followed by the Kilosa, Urambo, and Babati districts (Fig. 7.2). The WMAs in these districts are; Tarime (Tarime district), Twatwatwa (Kilosa district), Uyumbu (Urambo district), and Burunge (Babati district), respectively. These WMAs have also been confronting varied resource use conflicts, which are partly due to high population density (IRA, 2007).

Adequate knowledge about human population size and growth rates is helpful in setting conservation priorities, because population density may be used to determine resource use intensity and act as a surrogate measure of the degree to which wildlife resources in WMAs are under threat. For example, in the game-controlled areas and open areas (where most of the WMAs are established), density of human habitation is high and bushmeat hunting is also a serious problem. Consequently, densities of most wild ungulate species are relatively low (Caro, *et al*., 1998). In districts with high population size such as Tarime, bushmeat exploitation is a critical problem (Ndibalema & Songorwa, 2007). Mung'ong'o and Mwamfupe (2003) reported that high population density in Kilosa district has significantly contributed to devastation of natural resources. The high population growth rate of Urambo district and its rapid development, as well as demand for a better quality life, have encouraged illegal hunting of wildlife for both commercial and subsistence purposes. In an interview with Rolf Baldus on May 21, 2006, Tim Caro (a scientist researching wildlife issues in Tanzania) argued that demand for bushmeat in Tanzania is partly caused by increasing living standards of people. Elsewhere in Africa, for example, in west Africa, hunting for bushmeat has hugely contributed to decline in wildlife populations. This has always been attributed to rapid human population growth along with the demand for higher standards of living (Milner-Gulland & Bennett, 2003; Fa, *et al*., 2006; Caro & Scholte, 2007).

Table 7.1 Intercensal population growth rates in regions with WMAs in Tanzania.

Region	1988-2002 growth rates (%)	Wildlife Management Areas
Arusha	4.0	Enduimet and Loliondo
Manyara	3.8	Burunge and Makame
Tabora	3.6	Ipole and Uyumbu
Morogoro	2.6	Twatwatwa, Ukutu and Wamimbiki
Ruvuma	2.5	Songea and Tunduru
Mara	2.5	Ikona and Tarime
Pwani	2.4	Ngarambe-Tapika
Iringa	1.5	Idodi-Pawaga
Lindi	1.4	Liwale

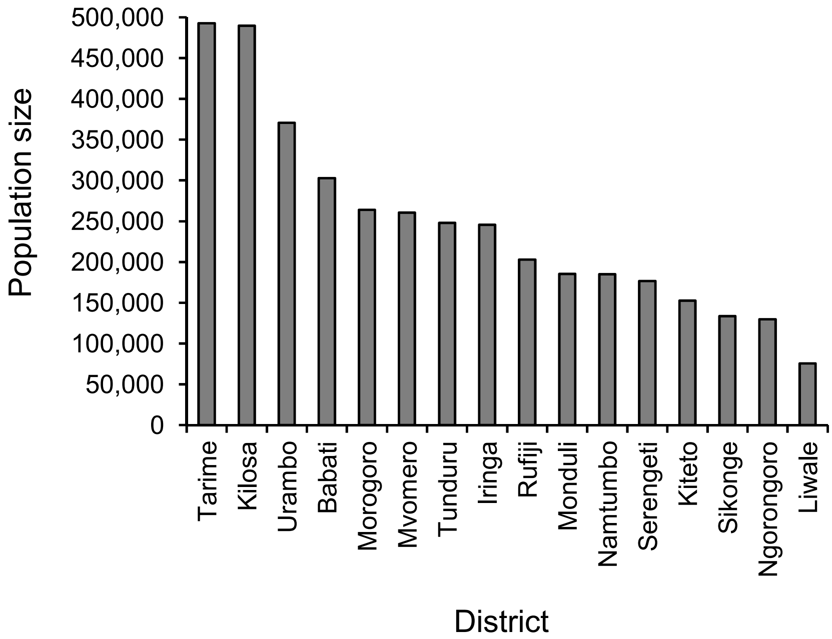

Figure 7.2 Population size in districts with wildlife management area projects, Tanzania.

Factors influencing resource access & utilisation

The primary natural resources in WMAs include forest, wildlife, and fish. Although highest priority is currently given to wildlife utilisation as the main activity, all other natural resources should also be considered in the utilisation schemes of the WMAs (Arntzen, 2003). A study on economic opportunities in WMAs identified, among others, four main economic openings through which rural communities can optimise the use of WMAs. These are: subsistence hunting, non-consumptive tourism, beekeeping, and utilisation of forest resources (Christophersen *et al.*, 2000). Making effective use of these opportunities calls on the local communities to be equipped with resource utilisation technologies and entrepreneurial skills. Such skills can unleash creativity and innovation for improved ways of resource exploitation. For example, construction and use of fuel-efficient stoves may reduce wood consumption and thereby contribute to a reduced deforestation rate (FAO, 2006). Initiation of successful small-scale income-generating activities in WMAs, which can improve people's livelihoods and take care of the environment, demands proper marketing strategy (Barstow, 2002). Beekeeping in Uyumbu and Ipole WMAs, for instance, has been one of the important economic activities among the villages involved in the WMA projects. Yet in order to enable local communities to expand their beekeeping enterprises, training and firm market structures are needed (Caroll, 2002).

Sustainable natural resources accessibility plans should also be developed and clearly documented in the terms of reference of any WMA project. To enhance resource accessibility and reduce conflicts, all the stakeholders (see Fig. 7.3) in WMA projects are obliged to observe important roles played by all the institutions involved. Institutions provide "rules of the game" (Norfolk, 2004); proper institutional arrangements will provide a good link between WMAs and local communities. It is, however, regrettable that most of the WMA projects are lacking stable institutional structures. Pragmatically, there are no clear boundaries between the roles played by the Wildlife Division; regional, district and village governments; non-governmental organisations; tourist hunting companies; and local communities (Nelson, 2006). In order to make a real change in resource accessibility and ownership at local levels, institutional structures need to be flexible enough to allow access for local institutions' voices to be heard at all levels of the WMA project plan formulation and decision making, and a genuine bottom-up approach should be employed.

The extent of resource ownership is defined in different ways by different official documents governing the use and management of natural resources in the country. For instance, while the wildlife policy of 1998 maintains that natural resources in WMAs will be under the control of local communities, the Forest Act of 2002 declares on page 98, section 69 (1), that "all biological resources and their intangible products whether naturally occurring or naturalized within forests including genetic resources belong to the government..." (URT, 2002). Furthermore, when forests are found in WMAs, the Forest

Act stipulates that forest management plans may contain forests other than village land forest reserves, and the plans will control the use and management of resources in such forests. This may bring some confusion on utilisation of forest resources within WMAs, and limit the span of communities' resource ownership.

From the government's perspective, the central economic role of WMAs is commercial hunting (e.g., in the Mbomipa, Okutu, Ikona, Ipole, and Uyumbu WMAs) (Walsh, 2002). This may perpetuate conflicts because people will have higher expectations and depend too much on revenues accrued through tourist hunting instead of initiating alternative ways of benefiting from WMAs. Despite the revenues believed to accrue to hunting activities, in the situation where the government is taking a lion's share of such revenues, successful participatory conservation will be a dream, which will never come true. For example, in Burunge WMA conflicts between the district government and the local communities exist because the government does not want to respect communities as important stakeholders and their wildlife-based needs are disregarded (Nelson, 2006).

There have been arguments about the capacity of community based organisations to properly administer sustainable use and management of WMAs. The College of African Wildlife Management in the Kilimanjaro region has been training government officials and local people from areas with WMA projects. Among other things, the college is conducting short entrepreneurial courses for district and village government officials as well as selected community members (Cooksey *et al.*, 2007). The effectiveness of this exercise may be constrained by uncertain market structures and lack of adequate experience as well as capital on the side of the local communities.

Participatory and sustainable resource accessibility and utilisation plans have been a matter of greater concern in many other countries of Africa. Tanzania has a lot to learn from the experience of other countries. In the southern part of Africa, in Malawi for example, local communities had negative attitudes towards establishment of Kasungu National Park. But, the development of simple wildlife-based enterprises increased local participation and made people realise the tangible values of wildlife management (Munthali & Mughogho, 1992). The Administrative Management Design (ADMADE) program in Zambia has been funding different development projects for communities in and around Game Management

Areas (GMAs). The ADMADE-funded projects include classrooms, houses for teachers, clinics, shelters for hammer mills used to grind maize, village shops, and capital for cottage industries. The program also trains village game scouts in order to reduce poaching and expand the scope of local communities' involvement in wildlife conservation (Lewis & Alpert, 1997). The Living in a Finite Environment (LIFE) project in Namibia has emphasised wildlife conservancies where local communities have legal rights to consumptively utilise wildlife and enter into contracts with investors in tourist hunting and photographic tourism (Jones, 2006). The Community-Based Natural Resources Management (CBNRM) program in Botswana attempted to reduce conservation costs tolerated by local communities. This was done by letting the government own wildlife resources, but the user rights were delegated to communities (Arntzen, 2003). Korup National Park has been one of the few protected areas in Cameroon, areas which are successful in integrating people's needs into conservation plans. Through developing proper use programs, the park has reduced land-related conflicts with the surrounding communities (Schmidt-Soltau, 2003).

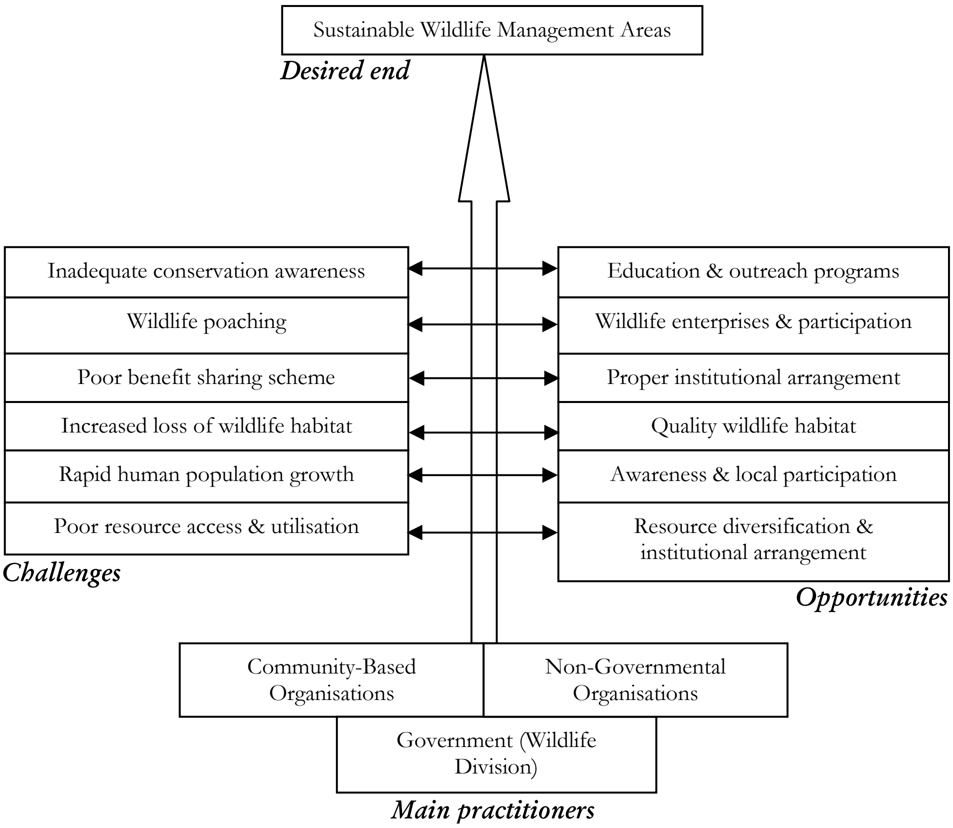

Figure 7.3 A theoretical framework for sustainable wildlife management areas in Tanzania.

Implications for conservation

There has been an overwhelming need for ensuring that WMAs are sustainable and can better meet their intended objectives. This section presents some recommendations as an attempt to stimulate further discussion on sustainable conservation of WMAs in Tanzania.

- There should be effective interventions for dealing with procedural complexity that involves distinct and time-consuming steps in the establishment of a WMA. Miniwary (2009) pointed out the example of Enduimet WMA, in which local people had to wait for about 10 years (from 1997 to 2007) before they were issued a certificate of authorisation. Such scenarios belittle the importance of WMAs and degrade their empirical credibility as one of the valuable land-use options.

- Effective monitoring of WMA projects will enhance their performance (Monique *et al.*, 2007) and help participants understand habitat dynamics of the wildlife species at the landscape scale as well as determinants of species' dispersal, particularly in the fragmented landscapes.

- Conservation awareness and extension programs toward advocating sustainable utilisation of wildlife resources should be emphasised. Such outreach programs are an important vehicle for the dissemination of conservation awareness and education in the rural areas. Caro & Scholte (2007) consider "outreach programs" to be among the most crucial activities in reducing escalating pressure on wildlife resources.

- A diverse range of natural resources in the areas with WMA projects prompts two significant conservation activities. The first is the widening of sustainable wildlife-based economic opportunities for local communities (Christophersen *et al.*, 2000) in order to promote a sense of belonging to WMA projects among the local people. Nonetheless, extensive conservation training programs that are blended with entrepreneurship skills are necessary in building the capacity of local people, and unlocking their creativity and innovation to initiate natural resources-based income-generating ventures in areas with WMAs. The second activity is promoting harmonization of the resource utilisation schemes. In the guidelines for the designation and management of WMAs, the Wildlife Division of Tanzania pointed out that the authorised associations may allow resource utilisation in the WMAs based on the regulations of the respective resource management authorities. For example, utilisation plans for fish resources should adhere to the Fisheries Act of 1970, while the utilisation of forest and bee resources should follow the regulations in the Forest Act of 2002 and Beekeeping Act of 2002, respectively (URT, 2003). This may create a jumble of differing conservation obligations that authorised associations must understand and meet unless such utilisation regulations are harmonized and simplified at a grassroots level.

- A theoretical framework (Fig. 7.3) depicts in general the factors influencing realisation of sustainable WMAs. Community-based organisations initiate WMA projects in collaboration with the government, and non-governmental organisations also play a role in conjunction with both the government and community-based organisations. In order to achieve a desired outcome (sustainable wildlife management), pragmatic collaboration among the practitioners is key to addressing conservation challenges through the available opportunities.

CHAPTER 8: GENERAL DISCUSSION

Since it is the first extensive and rigorously carried out wildlife exploitation study in the Ugalla ecosystem of western Tanzania, this thesis serves as a baseline study for wildlife conservation in the area. Its goal has been to shed some light on different aspects indicating pressures faced by wildlife and their habitats inside and outside Ugalla Game Reserve. In this discussion, I recapitulate the key results of the chapters while considering their conservation and ecological implications. The discussion will also encompass comparisons and contrasts between the main findings in this study and conservation studies elsewhere, underscoring the limitations as well as pin-pointing areas for further research.

The fact that wildlife resources across different ecosystems in Africa are under pressure is nothing new (Taylor & Dunstone, 1996; Davies & Brown, 2007). What the contemporary conservation community is striving to achieve is sustainability of the remaining wildlife populations (Baldus & Caudwell, 2004; Ling & Milner-Gulland, 2006; Milner-Gulland & Rowcliffe, 2007; Allebone-Webb *et al.*, 2011). Research plays a critical role towards realising this goal. Investigations have focussed on consumptive utilisation (overexploitation), wildlife population dynamics and demographics, and habitat manipulation (Kideghesho *et al.*, 2006; Mills, 2007; Sinclair *et al.*, 2008). These are in fact influenced by an array of factors predicated on human behaviour and the maintenance of livelihoods (Coad, 2007; Sinclair *et al.*, 2008). Thus, Hurt & Ravn (2000, page 304) stressed that "the future of wildlife in Africa rests in the hands of its indigenous people. Good ecology and economics should never ignore the human being". Owing to different backgrounds of resource users across different ecosystems, wildlife studies have tended to be ecosystem specific. This is absolutely acceptable as solutions to tackle conservation problems in one area might not necessarily be valid in another. Unfortunately, most wildlife researchers conduct their studies in areas that would allow for species connectivity. For example, wildlife areas that are geographically linked where one of them is strictly protected and the other one severely exploited. While there is clear scientific evidence about "source-sink" scenarios in such areas (Pulliam, 1988; Novaro *et al.*, 2000), ensuing research output might bear very little relevance in exploited areas which are isolated or not directly linked to unexploited areas. Therefore, research should also focus on ecologically isolated wildlife areas in Africa if it is to help resolve the current bushmeat crisis.

This study aimed to find out what would bushmeat exploitation mean (from a conservation view point) in a largely isolated and data poor, yet important African game reserve. As a tourist hunting site, Ugalla Game Reserve is divided into two tourist hunting blocks (each approx. 2500 km^2). Ugalla east and Ugalla west hunting blocks offered a good opportunity to compare the distribution of the exploitation intensity between adjacent hunting blocks. This was one of the concerns of the Ugalla Game Reserve Management Team. Because if wildlife in one of the hunting blocks is overexploited, also considering the potential for source-sink dynamics, there could be ecological degradation in the reserve and the Ugalla ecosystem as a whole. Chapter 2 showed that pressure on wildlife in Ugalla west hunting block exceeded that of Ugalla east hunting block. Some indicators used to arrive to this conclusion included species density, sex ratios and group sizes. These parameters have also been used elsewhere to assess wildlife utilisation intensities (for example, Milner *et al.*, 2007; Waltert *et al.*, 2008; Topp-Jørgensen *et al.*, 2009).

Generally, the wildlife density in Ugalla east hunting block was higher than Ugalla west hunting block. At least half of the individual species with density estimates had higher densities in Ugalla east hunting block (Table 2.1). When comparing densities of the ungulates in Ugalla Game Reserve with their counter-parts in a more protected Katavi National Park as reported by Caro (1999*b*), the latter far exceeded the former suggesting higher utilisation impact in the game reserve. Sex ratios did not statistically differ between the hunting blocks, but most of the species had sex ratios significantly skewed towards female. While this is common in nature (FitzGibbon & Lazarus, 1995; Kiøboe, 2006), studies suggest that the pattern is more pronounced in exploited populations (Ginsberg & Milner-Gulland, 1994; Milner-Gulland *et al.*, 2003; Milner *et al.*, 2007; Setsaas *et al.*, 2007; Marealle *et al.*, 2010). Animals in Ugalla east hunting block were observed in larger groups than Ugalla west hunting block. Caro (1999*c*) argued that exploited species do tend to congregate in either larger or smaller groups in response to hunting pressure. This study suggests that exploited ungulate species are more likely to disperse in smaller groups. Manor & Saltz (2003) reported the impact of human disturbance on reducing group sizes of the mountain gazelle *Gazella gazella* in Israel. This often minimizes resource competition, and natural predation through increased vigilance (Roberts, 1996; Lima *et al.*, 1999). Such "group-size effect" might be one of the antipredator strategies among wildlife populations in Ugalla west hunting block. Nonetheless, future studies should empirically take this, and differences in habitat into account. Caro (1999*c*, 2008) described group-size effect in Katavi National Park. Indeed, work is still needed (in conservation areas) to put in proper context the trade-offs between anthropogenic intervention, vigilance, natural predation, resource competition and group size (Roberts, 1996; 2003).

The observed difference in species parameters between the hunting blocks served as suggestive evidence of the existence of trends and driving factors in the exploitation of wildlife in Ugalla. To comprehend these factors, and more importantly to identify the entry points for dealing with over-exploitation of wildlife in the area, I carried out further studies presented in other chapters of this thesis from the following viewpoints: wildlife poaching and bushmeat consumption; legal subsistence hunting; conservation measures.

Wildlife poaching & bushmeat consumption

In chapter 3, I showed that proximity to Ugalla Game Reserve was correlated with wildlife poaching. Human settlements near Ugalla west hunting block were much closer to the reserve than Ugalla east hunting block; this promoted accessibility for poachers. Bushmeat hunters tend to consider travel distance when choosing a hunting destination (Coad, 2007; Nyahongo *et al.*, 2009). The distance from the Ugalla Game Reserve boundary also determined the number of species poached. Species mentioned as poached around Ugalla west hunting block exceeded Ugalla east hunting block. This further indicates that the reserve harbours a diverse range of species. Considering all species targeted by poachers, at least 30 species were removed. Following the order of their frequency, the first ten were: impala, dik-dik, common duiker, buffalo, hartebeest, topi, warthog, hippopotamus, sable and bushpig. Some of these species are also frequently hunted in other ecosystems in Africa. Impala, warthog and buffalo are among the most hunted species in Zimbabwe (Lindsey, *et al.*, 2011). Extensive killing of dik-dik alongside other antelopes for food and wildlife trade in Somalia has been reported by Amir (2006). Hippopotamus, bushpig, buffalo and warthog are common species on bushmeat menus in Uganda (Bean, 2009; Olupot *et al.*, 2009).

Interestingly, bushmeat consumption had a different pattern. It increased with the distance from the reserve boundary. Therefore, communities near Ugalla east hunting block appeared to have higher consumption rate than Ugalla west hunting block, suggesting the existence of a bushmeat trade network where consumers (most of whom are far from Ugalla) determine the frequency of poaching. Caro & Andimile (2009) explained such bushmeat chains in Tanzania. They involve poachers, middlemen and consumers. In open bushmeat markets, studies have shown that hunters spend much effort not only on increasing their catches but also on catching profitable species (Fa *et al.*, 2000; Cowlishaw *et al.*, 2004; Willcox & Nambu, 2006; Coad, 2007). The disadvantage of the secretive bushmeat black markets in Tanzania is the difficulty of quantifying the meat trade, so parameters like meat price, amount, source and profit margins are harder to assess (Caro, 2008; Caro & Andimile, 2009; Chapter 3). Currently, selling any kind of bushmeat whether legal or illegal is strictly prohibited (URT, 1974). This raises some important questions about where in the Ugalla ecosystem much of the bushmeat on sale originates from. Does it actually come from Ugalla Game Reserve? Or is it hunted in the buffer zones? We also don't know how much of the meat is illegally or legally extracted. Although genetic structuring of the population may be difficult to do, where possible we should probably think of maximising the use of genetic approaches in the conservation and bushmeat systems for countries like Tanzania (Bitanyi *et al.*, 2011). In Ugalla, such approaches should be integrated into the approaches used in this thesis in order to address the bushmeat hunting problem in a broader scale in the future.

Special consideration in reducing bushmeat poaching and consumption should also be given to promoting the development of animal protein alternatives, poverty eradication, agriculture and law enforcement as explained below:

Protein alternatives to illegal bushmeat: The most important bushmeat substitute was fish (Chapter 3). Other studies have made similar observations (Wilkie & Carpenter, 1999; Brashares *et al.*, 2004; Wilkie *et al.*, 2005; Nyahongo *et al.*, 2009). Although some precautions should be taken with regard to this in a larger scale bushmeat and fish system (Rowcliffe *et al.*, 2005), for a subsistence fishing like Ugalla, fish can be a reliable bushmeat alternative. Fish were obtained through illegal and legal means. Respondents were not specifically asked whether the fish they ate were legally obtained, but from the law enforcement record it was evident that fish poaching was common (Chapter 5). The frequency of fish consumption at a household level influenced the rate of bushmeat consumption and the overall extent of wildlife poaching at a village level. By looking at the factors influencing the exploitation of natural resources at different levels within a community, this thesis crucially identifies the importance of each factor to its respective level.

The ownership and consumption of livestock species are said to have a profound positive impact on the reduction of bushmeat hunting (Rushton *et al.*, 2005). In chapter 4, I assessed the influence of livestock on wildlife poaching in Ugalla. Surprisingly, of all the livestock species, chicken ownership correlated positively with wildlife poaching. Chicken keepers in Ugalla were mostly operating on a small scale; and the more the number of chicken a household had, the less the number of other larger livestock it possessed. General poultry disease management and control could help to promote the positive impact of chicken on the bushmeat hunting as observed by Knueppel *et al.* (2009) in Ruaha Landscape. The comparison of the rate of livestock consumption across individual livestock species with the rate of bushmeat consumption would help to find out how each of the livestock contributes to wildlife exploitation in Ugalla. This is particularly important because as a component of agriculture, livestock keeping must be sustainably promoted as one of the solutions to bushmeat hunting and consumption.

Agricultural output: Reserved crop yields were helpful in reducing the household's bushmeat consumption frequency (Chapter 3). Farmers who kept larger amounts of their harvests (those who sold a small amount or none of their yields) ensured food security and their dependence on the wildlife resources reduced significantly. A study in the Congo Basin advised that "increasing smallholder agricultural productivity is essential to reduce demand for bushmeat from urban as well as rural areas" (Fa *et al.*, 2003). Okello *et al.* (2011) highlighted the willingness of farmers to support wildlife conservation in the Amboseli ecosystem of southern Kenya provided their agricultural activities are not compromised. The main challenge in Ugalla remains how to promote sustainable agriculture. Intensive agriculture instead of the current extensive agriculture (URT, 1998*a*) is important to strike a balance between farming activities and natural resource conservation. The miombo woodlands of Ugalla are characterised by poor soil fertility (URT, 1998*a*; Hazelhurst & Milner, 2007). Farm encroachments in the forests outside Ugalla Game Reserve are widespread (Shishira & Yanda, 1998) to cope with the loss of soil fertility. Wherever farmers settle (temporarily) they clear a substantial piece of land for farming and settlements and other livelihood activities; and these increase concomitantly with the expansion of their families. Such conservation problems are well explained in the Maasai Mara ecosystem by Mundia & Murayama (2009).

Due to the importance of food crops and the need to conserve wildlife habitats, we must address the problem of extensive cultivation in Ugalla; and do it quickly. The majority of the farmers admitted that they lacked expertise to boost their agricultural yield. Extension officers were also very few, and in some places unavailable altogether. Graaff *et al.* (2011) highlighted the importance of extension services in the rural agricultural production. Agriculture extension workers play the role of advising farmers on various technical aspects of yield maximization (Sanchez *et al.*, 2009; Graaff *et al.*, 2011). Tobacco farmers in Ugalla benefit a lot from knowledge and fertilizers offered by the foreign tobacco companies. Other non-tobacco farming villagers were forced to cultivate tobacco so that they use some of the agricultural inputs aimed at raising tobacco for their other food crops (Ellias R. Kaugilla, pers. comm.).

Rainfall: Efforts to curb poaching activities in Ugalla should never ignore the influence of rainfall. This study showed that rainfall had a positive correlation with wildlife poaching (Chapter 3). Rainfall was important for agricultural production, but on the other hand it flooded the Ugalla Game Reserve access and patrol roads; and its extended periods led to short term food shortages in the area. Due to the poor coverage of the area by the wildlife law enforcement teams, during the rainy season of the year the wildlife populations are probably substantially damaged. Rainfall influences poaching activities in various ways in other ecosystems as well. In India, it is said to help poachers to avoid detection by rangers and the species they target by walking slowly and quietly on the wetter surfaces (Velho *et al.*, 2011). In Mara-Serengeti region, most of the patrol roads become hardly passable during the rain seasons hampering anti-poaching patrols (Ogutu *et al.*, 2009).

During the field work, it was learned that in the wet seasons Ugalla west hunting block is poorly patrolled. The access roads to Ugalla west hunting block; for example, from Kaliua via Lumbe (Fig. 5.9) become more muddy and flooded (see Fig. 8.1) than the ones to Ugalla east hunting block (from Usoke, Tutuo and Ipole). Ugalla Game Reserve rangers quite often access the reserve through Ugalla east hunting block during heavy rains, but again reaching some remote areas in Ugalla west hunting block such as Msima area by crossing the flooded Ugalla and Koga rivers (Fig. 2.1) is almost an impossible task. Thus, Msima becomes dangerously exploited especially by poachers from nearby Katumba refugee camps who also take the advantage of floods to get logs, game meats (often hippopotamus meat) and other natural resource products from Ugalla Game Reserve by using traditional wooden boats (UGR, 2006; Chapter 5).

Further studies specifically aimed at assessing the influence of rainfall on livelihood activities, wildlife population dynamics and natural resource exploitation pressures would contribute substantially to the conservation of Ugalla. Such studies have been very useful in the Serengeti (Sinclair *et al.*, 2008) and Maasai Mara (Ogutu *et al.*, 2008) ecosystems.

Figure 8.1 A photo showing our survey vehicle stuck in a muddy flooded Kaliua-Ugalla road in Lumbe village, in one of our visits to Ugalla Game Reserve during the 2009 rainy season.

Income sources: Involvement in poaching was negatively correlated with mean household income (Chapter 4). This result parallels Wilkie & Godoy (2001), who observed that increase in household income halted the demand for bushmeat among the Amerindian societies in south and central America. However, this is contrary to Coad (2007) in Gabon, who saw that higher income households participated actively in hunting activities. Higher income households in Ugalla had instead higher rate of bushmeat consumption, which also indicates the existence of an active game meat trade operating certainly on black markets as described beforehand. I showed that income is a trade-off between wildlife poaching and bushmeat consumption. Income lessened poverty and increased purchasing power for smaller and larger families with adequate family labour and productive assets, who appear to constitute a significant proportion of the bushmeat customers. While income determinants enhanced agricultural production, it is worth noting that some families with increased agricultural yield were not selling their produce. Such families guaranteed food availability, and depended upon selling natural medicinal products, forests, formal employment and other small scale businesses; which earned them monetary income.

Households who were involved in tobacco farming maximised their income in a relatively short period of time. This observation disagrees with Kibwage *et al.* (2009) who found that tobacco was not a profitable crop among the communities in the Nyanza region of Kenya. Perhaps we need to take into account tobacco farmers' profit margins and compare them with those of non-tobacco farmers (Van Minh *et al.*, 2009). Owing to its environmental consequences, I am suggesting that other sustainable and environmentally friendly income generating enterprises be developed as explained in Chapter 7; and the central attention of the agricultural production be directed to the cultivation of the food crops and food availability.

The nature of poaching within Ugalla Game Reserve: To have a clear picture of the intensity of wildlife exploitation in Ugalla, I assessed wildlife poaching in the context of other poaching activities using poaching signs and law enforcement records (Chapter 5). This was achieved by looking at the size and extent of the interdependence of different types of illegal natural resource utilisation. The analysis encompassed fishing, timber harvesting and bushmeat hunting. With the exception of fishing, bushmeat studies have often paid little attention to the influence of the illegal exploitation of other types of natural resources (for example, Brashares *et al.*, 2004; East *et al.*, 2005; Rowcliffe *et al.*, 2005; Holmern *et al.*, 2007; Jachmann, 2008*a*,*b*). This study has shown that motives and patterns differ among poaching types in Ugalla. For example, timber poaching might be involving richer businessmen far from Ugalla who hire the local cheap labour and give them all the logistics necessary to harvest timber from the woodlands. The increasing price and demand for timber in the face of escalating human population in Tabora (URT, 1998*a*) and other neighbouring regions as well as major cities (Shayo, 2006), might have encouraged timber poachers to intensify their activities. This has far-reaching consequences on the wildlife status. Poulsen *et al.* (2009) argued that timber harvesting industry should be taken into consideration when dealing with challenges facing wildlife conservation. The employment of timber trucks in Ugalla, for example, has led to a loss of plant species (own field observation). Trees and other plant species are cleared by poachers in order to cut roads for their trucks and create camping sites. The digging of sawing pits may jeopardize the survival of the small mammals and other creeping and crawling forest creatures. While Beale (2007) warned about the "effects and impacts" of human activities in conservation areas, we are not sure about how much impact the timber poachers and their timber cutting machines have had on the behaviour and productivity of the animals in Ugalla. The law enforcement teams in the area should seriously extend their operations and deal with timber business owners if at all we are to counteract forest loss and conserve wildlife habitats within the reserve effectively.

Spatial analysis of the poaching signs revealed that much of the timber harvesting activities were carried out in the Ugalla east hunting block, whereas Ugalla west hunting block was dominated by illegal hunting activities. Msima area in Ugalla west hunting block (Fig. 5.9) contained many timber and bushmeat poaching signs, reflecting a massive pressure of wildlife in this area. The distribution pattern of poaching signs provides useful information not only about where most wildlife off-takes take place within the reserve, but also about the fact that wildlife poachers have good knowledge of the spatial concentration of animals. Therefore, we need to focus our anti-poaching efforts and match them with the knowledge of the distribution of poaching signs. This would ensure the long term survival of game species since animals disperse following the availability of breeding sites (Kideghesho *et al.*,2006), food resources and as a response to natural predation (Hopcraft *et al.*, 2011).

Anti-poaching patrols or wildlife law enforcement measures may have contributed to the distribution pattern of the poaching signs since poachers would prefer not to get apprehended. Nonetheless, since bushmeat hunters normally strive to maximise their off-takes (Wilkie & Carpenter, 1999), in some cases law enforcement may as well have little influence on poaching activities (de Merode *et al.*, 2007). There is also a possibility of poachers to process their catches or kills in places far from where they hunted to avoid detection, this may have potentially altered the comparison of the poaching signs across anti-poaching units. But, it largely depends on whether a poaching activity in a particular anti-poaching unit was conducted adjacent to another anti-poaching unit. Additionally, for some species such as elephant moving the whole carcass is not possible because all the elephant poachers are interested in is tusks unless they wanted to chop off a chunk of meat for consumption. In another study (Chapter 2), I showed that the density of animals in Ugalla east hunting block exceeded Ugalla west hunting block. Considering the distribution of poaching signs in the reserve, one might think that the latter is richer in wildlife than the first. Nevertheless, chapters 3 & 4 suggest other factors, apart from animal abundance, which may well have accounted for the observed wildlife poaching signs in Ugalla west hunting block.

Conservation efforts

The preceding sections have highlighted different illegal wildlife exploitation contexts and their conservation impacts in Ugalla. Here, I describe some conservation measures being taken to bring down the level of poaching.

Conservation laws are prohibitive in nature, and they interfere with the human and natural resource interaction (Pimbert & Pretty, 1995; Morgera & Wingard, 2008) in search of sustainability (Morgera & Wingard, 2008). This has created tensions between humans and conservation principles (Kideghesho, 2006). As a result, we should not expect a voluntary willingness of people to stop illegal utilisation of natural resources unless deliberate measures are taken to motivate them to do so (Sinclair *et al.*, 2008). Despite the fact that law enforcement has always been the main alternative (Keane *et al.*, 2008), it is often constrained by the lack of resources in the developing countries (Rowcliffe *et al.*, 2004). Therefore, conservationists have been seeking complementary and alternative options to better protect natural resources (Songorwa, 1999; Kideghesho, 2006). The most promoted and debated option is the involvement of people who bear the costs of conservation (Campbell *et al.*, 2003). This includes consenting to legal but regulated subsistence utilisation (Kaimowitz & Sheil, 2007; Mfunda & Røskaft, 2010). All these measures are often exercised in the buffer zones, the majority of which are immediately adjacent to core protected areas (Neumann, 1997; Naughton-Treves *et al.*, 2005). This thesis presents a first study to assess the effectiveness of the legal subsistence utilisation which has, for decades, been employed as a conservation tool in the Ugalla buffer zones. The idea is that by allowing subsistence hunting around the reserve, poaching would be reduced and the reserve itself would be properly protected. Chapter 6 suggests that the responsible authorities have been overly optimistic about this. Problems associated with the issuing of hunting licences and species hunted are common. Elsewhere in Tanzania studies have also indicated that legal subsistence hunting is not properly administered (Baldus & Caudwell, 2004; Holmern *et al.*, 2004; Rija, 2009). Poor knowledge of important species parameters such as abundance, richness and distribution, and habitat destruction were among the probable root causative factors of the observed shrinking of the species successfully removed in Ugalla. Hunting success rates can be used as indicators of contracting populations (Coad, 2007). The decreasing trends of the hunting success rates across species are a sign of not only overwhelming exploitation pressure in the buffer zones, but also inefficacy of the legal subsistence hunting as a conservation measure.

At the moment, Tanzania is piloting wildlife management areas that would take over most of the subsistence hunting sites as a new category of protected areas (Chapter 7). There are at least 16 wildlife management areas across the country. Ipole and Uyumbu wildlife management areas in the Ugalla ecosystem are among the advanced projects in the country (IRA, 2007; Nelson, 2007). Nevertheless, their coverage of the Ugalla buffer zone represents only the tip of the iceberg (Figs. 3.9 & 6.1), so they might not have a big contribution to the protection of Ugalla as some reports suggest (UGR, 2006). One of the Africare officers in the area (the ones spearheading wildlife management areas around Ugalla) admitted that there would be more wildlife management areas in the future that adequately surround the reserve in Mpanda, Sikonge and Urambo districts (C. Metta, pers. comm.). If this is realised, then Ugalla Game Reserve will apparently benefit from their conservation. In Chapter 7, I highlighted issues that need serious attention in order to warranty sustainable wildlife management areas, drawing examples from similar projects in and out of Tanzania. The presented recommendations would inform the management and research priorities for these projects for the betterment of Ugalla and other game reserves in the country. Sinclair (2008) advised that core protected areas are no longer pristine and safe from human disturbances. The same author noted that to ensure successful protection of the core areas in Tanzania, community involvement through wildlife management areas must be emphasised. Thirgood *et al.* (2008) insisted that the wildlife management area projects are "the most exciting and challenging new conservation initiative in Tanzania" and their success would protect biodiversity in the country.

The future

This study has presented several salient findings related to the conservation of the Ugalla ecosystem. In particular, it has exposed, for the first time, wildlife management challenges of the ecosystem from both buffer zone and core protected area contexts. The perspectives assessed in this study included: factors influencing wildlife poaching, wildlife status, poaching signs, law enforcement records and legal subsistence hunting. It is, nonetheless, important to be explicit here that Ugalla still needs work. Throughout the thesis I have pinpointed some specific areas that need further research. This section presents a few additional, but rather important research areas.

Unlike many others in Tanzania, the Ugalla ecosystem lacks adequate amounts of research in the various aspects of bushmeat hunting at both large and small scales. There are always merits and demerits of conducting bushmeat studies at each of these levels (Coad, 2007). For example, when assessing factors influencing wildlife poaching the study incorporated the majority of the communities around the reserve. In such a large scale, it possibly ran the risk of missing out on some fine issues which could have been addressed had the survey been carried out for a relatively small sample. Dealing with a small area allows the researcher to spend a substantial period of time with respondents. But, on the other hand such studies bear a narrow focus of the key issues at stake, and resultant findings might not be applicable to other parts of the same ecosystem when the ecosystem in question is wide-ranging.

One of the hotly debated topics in the bushmeat circles that also need looking into is species preference. In chapter 3, I estimated the poaching frequency across different species. This is the outcome of preference which is in turn dictated by a number of other variables including: market structure, taste of the meat, demand, price and customer's background (Coad, 2007; Ndibalema & Songorwa, 2007). Unfortunately, the "invisible" bushmeat markets in Tanzania make it extremely challenging to realistically employ market approaches in the bushmeat systems as it is done elsewhere in Africa; for instance: in Gabon (Coad, 2007), Equatorial Guinea (Morra *et al.*, 2009) and Cameroon (Willcox & Nambu, 2007). In many cases we are forced to rely on non-market variables in assessing sustainability or extent of bushmeat exploitation. Alternatively, we integrate livestock consumption patterns to try to fathom the breadth of the problem. Further studies on the operation of the informal bushmeat markets in Tanzania are desperately needed. Allebone-Webb *et al.* (2011) confirmed that market surveys are an efficient approach to study bushmeat hunting and its sustainability provided "market filters" existing between suppliers and consumers are properly addressed.

Alternative sources of protein can positively influence the bushmeat markets, preferences and ultimately hunting frequency (Wilkie & Carpenter, 1999; Ndibalema & Songorwa, 2007; Willcox & Nambu, 2007). This study looked at the impacts of fish, livestock and bushmeat consumption on the poaching frequency in Ugalla. The bushmeat and fish consumed could not be identified as coming from legal or illegal sources. This challenge was also noted by Nyahongo *et al.* (2009) in the Serengeti ecosystem. Knapp *et al.* (2010) used dietary recalls and law enforcement records to assess poaching activities in Serengeti. The integration of legal sources in the consumption frequencies to establish the relative contribution of all protein sources to poaching reduction would also be very helpful. In Ugalla, legal fishing is a source of fish protein among the communities. Within the reserve legal subsistence fishing is administered by the Ugalla Game Reserve Management Team whereas outside the reserve district governments oversee the fishing exercise. Ugalla Game Reserve is the only game reserve in Tanzania where (apart from tourist hunting) local people are also allowed (by permit) to utilise natural resources, namely: fish, honey and wax (UGR, 2006). The main objective of this is to reduce levels of illegal hunting. Future studies should evaluate the performance of this multiple resource utilisation scheme from a standpoint of wildlife poaching intensity.

Tourist hunting is said to benefit conservation through the generated revenues which are partly used by game projects to reinforce anti-poaching activities (Caro, 1998; Hurt & Ravn, 2000). Before we can ensure the sustainability of tourist hunting, regular monitoring should be carried out on an individual game reserve basis. This would keep us informed of the biology, behaviour and ecology of the species subjected to trophy hunting across different reserves in Tanzania; as well as the impact tourist hunting has had on the local communities neighbouring the game reserves. Regrettably, for some reason, it has been overwhelmly difficult to get access to tourist hunting data. Much of this information is kept at the headquarters of the Wildlife Division in Dar es Salaam. I recommend that this information be made more readily available, for research purposes, to wildlife researchers in the country. In this way, the whole exercise of tourist hunting would be based on the scientifically informed decisions and not anecdotal reports as it currently is (LEAT, 2011).

Tourist hunting in Ugalla is taking place even outside the game reserve (URT, 1998*a*). We do not know its extent and effect on the species in these buffer zones. In wildlife management areas, tourist hunting activities are purportedly controlled by local communities who also realise tangible benefits (URT, 1998*b*). The current low pace of expanding wildlife management areas means that any possible ecological and biological impacts of tourist hunting in the buffer zones would continue unabated. This, and other undesirable outcomes of: habitat manipulation (Hazelhurst & Milner, 2007), legal subsistence hunting (Chapter 6) and poaching (Chapters 3, 4 & 5) are likely to jeopardize chances for the buffer zones to considerably support the Ugalla ecosystem. Future research should consider assessing the buffer zones from the following vantage points: biological and ecological parameters of the wildlife species, livelihood activities with respect to habitat and wildlife status, the effectiveness of law enforcement following Holmern *et al.* (2007) in Serengeti, and poaching signs and habitat quality. Failure to address these issues properly and put forth handy recommendations in the context of the conservation and ecology of Ugalla Game Reserve, the Ugalla ecosystem itself would be ecologically discontinuous. Just like the Sherwood Forest of England (Nikolakaki, 2004).

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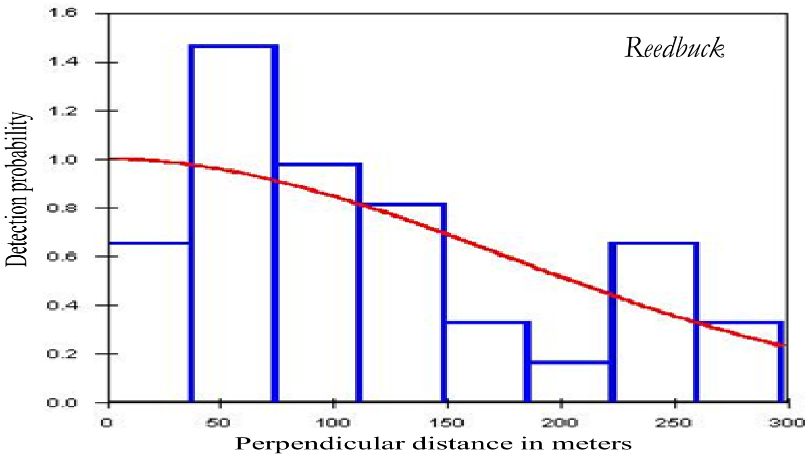
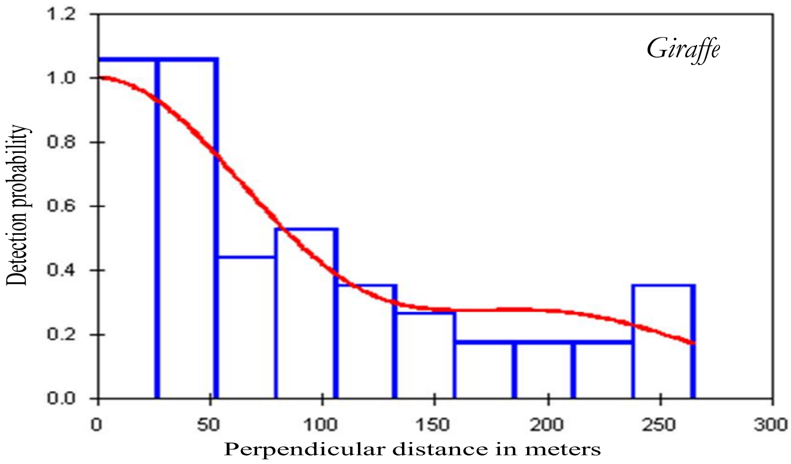
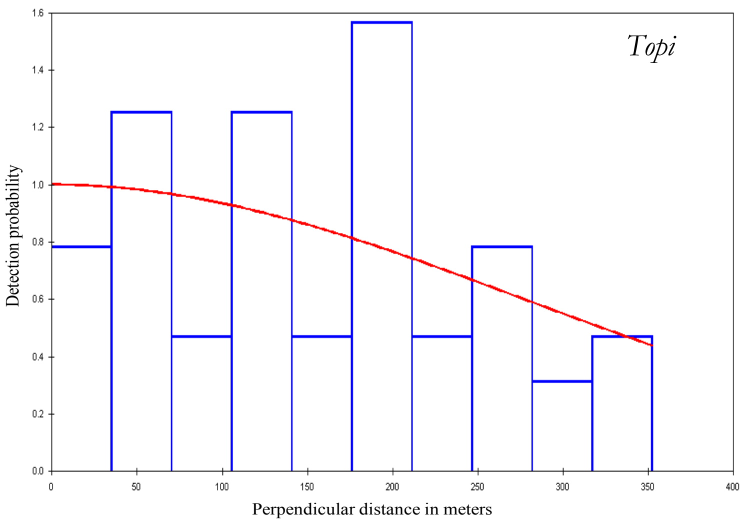
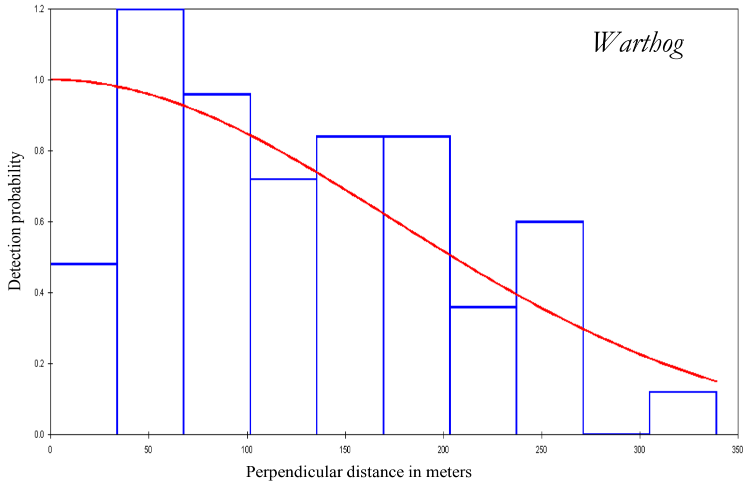
APPENDIX

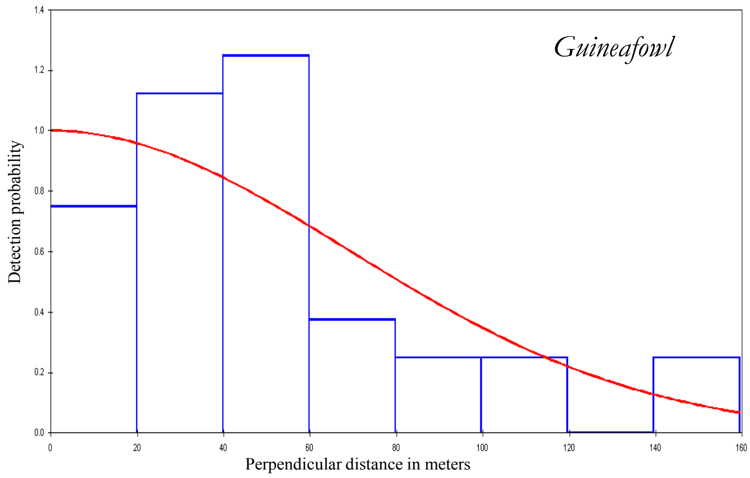
Detection probability curves for selected species in Ugalla Game Reserve.

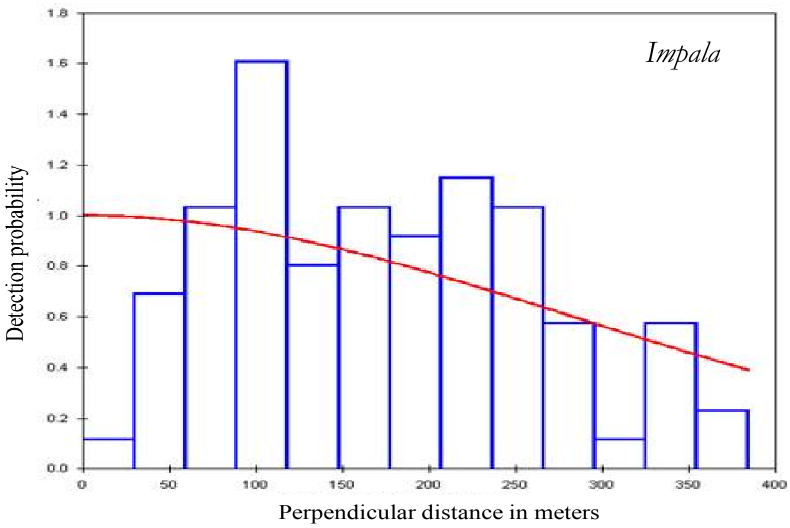







Warthog
Detection probability
0.0
0.2
0.4
0.6
0.8
1.0
1.2
0
50
100
150
200
250
300
350
Perpendicular distance in meters


Guineafowl
Detection probability
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
0
20
40
60
80
100
120
140
160
Perpendicular distance in meters


Impala
Detection probability
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
1.6
1.8
0
50
100
150
200
250
300
350
400
Perpendicular distance in meters