

A photograph of two giraffes in a savanna landscape. The giraffes are standing in a grassy field with green bushes and trees in the background. The sky is a clear, pale blue. The image is framed with rounded corners.

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The impact of crop raiding by wild animals in communities surrounding the Serengeti National Park, Tanzania

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Crop damage is a serious source of conflict in communities adjacent to protected areas. Data on crop raiding were collected through questionnaires in villages at different distances from the Serengeti National Park in northern Tanzania. Multiple response analyses were used to calculate the frequencies of the types of crops damaged, the levels of crops growth when damaged and the estimated economic losses caused per household. The results indicate that crop raiding differed significantly on farms along a distance gradient from the protected area. Baboons were reported to be the most destructive wild animals, followed by elephants, which were destructive especially in the villages located near the protected area. Wild animals caused significant economic losses for households. We recommend further studies on the crop yield gap caused by crop-raiding wild animals and human-primate conflict in communities around protected areas. For effective protection of crops from wild animals, we recommend that local communities adopt a combination of methods to reduce the levels of crop raiding.

Key words: Crop raiding, Serengeti ecosystem, human-wildlife conflict, wild animals.

INTRODUCTION

Crop damage is defined as feeding on cultigens by wild animals that causes substantial financial losses to farmers (Saj et al., 2001; Gunn, 2009; Fungo, 2011) and is a serious source of conflict between local communities and the management of adjacent protected areas (Newmark et al., 1994; Ntalwila et al., 2003; Ogra, 2008; Malugu, 2010). It is becoming a worldwide and complex problem (Saj et al., 2001; Sillero-Zubiri and Switzer,

2001; Nyahongo, 2007). It affects subsistence farmers directly through the loss of their primary food and cash resources and indirectly through a variety of social costs (Osborn and Parker, 2003; Marchal and Hill, 2009). For example, crop raiding was reported to affect 40% of all crops planted in a farm in the Ruaha Ecosystem (Kabepelle, 2011) and cause an average annual crop loss per household of 11% of the household income in the

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Serengeti District (Kaswamila, 2007). In general, crop raiding is reported to be limited to a distance of a few hundred meters outside the protected areas, with fields further away receiving little or no damage (Hill, 2000; Gillingham and Lee, 2003; Ole Meing'ataki, 2005; Ntalwila, 2007; Sarker and Røskaft, 2011).

Crop raiding by wild animals is caused by several species ranging from large mammals to smaller animals, such as birds, rodents and insects (Saj et al., 2001; Rao et al., 2002; Osborn and Parker, 2003; Sitati et al., 2003; Gunn, 2009). The animal species most often cited as responsible for crop damage in regions surrounding protected areas in Tanzania include African elephants (*Loxodonta africana*) (Ntalwila et al., 2003; Kaswamila, 2007; Malugu and Hoare, 2007; Kikoti et al., 2010; Malugu, 2010), primates (Ntalwila et al., 2003; Kaswamila 2007), bushpigs (*Potamochoerus porcus*) (Newmark et al., 1994; Ntalwila et al., (2003), dik diks (*Madoqua kirkii*) (Ntalwila et al., 2003), bushbucks (*Tragelaphus scriptus*) (Ntalwila et al., 2003), rodents (Newmark et al., 1994), African porcupines (*Hystrix cristata*) (Ntalwila et al., 2003), vervet monkeys (*Chlorocebus pygerythrus*) (Ntalwila et al., 2003) and Cape buffalos (*Syncerus caffer*) (Ntalwila et al., 2003).

Crop raiding may peak during the harvest season when the crop is mature (Parker and Osborn, 2001; Malima et al., 2005; Jackson et al., 2008; Gunn, 2009; Malugu, 2010), but it commonly occurs throughout the year (Walpole et al., 2004; Malugu and Hoare, 2007; Gunn, 2009; Ntalwila et al., 2011). Mature crops may offer a high nutritional benefit to the raiding wild animals (Malugu and Hoare, 2007) and are also the most palatable and contain most calories, reducing herbivore feeding time (Ntalwila et al., 2011). Crops that are at the highest risk include cassava (*Manihot utilissima*) (Naughton-Treves, 1998), maize (*Zea mays*) (Naughton-Treves, 1997; Ntalwila et al., 2003; Walpole et al., 2004; Kikoti et al., 2010), sweet potatoes (*Ipomea batatas*) (Ntalwila et al., 2003; Walpole et al., 2004), groundnuts (*Arachis hypogaea*) (Naughton-Treves, 1997), sorghum (*Sorghum vulgare*) (Walpole et al., 2004) and finger-millet (*Eleusine coracana*) (Ntalwila, et al., 2003; Walpole et al., 2004). Other crops subject to damage are beans (*Phaseolus vulgaris*), bananas (*Musa acuminata*), tomatoes (*Lycopersicon esculentum*) and fruits (Ntalwila et al., 2003). According to Malugu and Hoare (2007), the crops most often raided in the western Serengeti in northern Tanzania are sorghum and maize.

Local communities have adopted several measures to deter crop raiding by wild animals. Some of the most common measures include guarding (Hill, 2000; Walpole and Linkie, 2007; Malugu 2010), making noises (Malugu, 2010), lighting fires at night, digging trenches against bush pigs (Kagoro - Rugunda, 2004; Fungo, 2011), and constructing fences (Paterson, 2007; Malugu and Hoare, 2009; Malugu, 2010; Fungo, 2011). These methods are used either individually or in combination. Selection of a

particular method depends on the size of the field, the crop being grown; guard labour availability and vulnerability of the crop to raiding (Fungo, 2011).

Expanding scientific knowledge regarding the nature and extent of human-wildlife conflicts is important because such conflicts tend to affect the livelihoods of people (Newmark et al., 1993). The monitoring of crop damage is largely limited to incident reports by local people to their local governments and later to the wildlife authorities in order for an appropriate action to be taken (Hoare, 1999). Most studies on human-wildlife conflicts are based on surveys examining perceptions of the problem held by local people. However, it is recognized that perceived and actual extent of such conflicts do not match. An exaggeration of the extent of damage is therefore the consequence (Naughton-Treves, 1998). In addition, careful documentation of economic losses is essential in order to assess the extent of damage. The present study intends to determine the extent of crop raiding by elephants, birds, primates, bushpigs and rodents in households along a gradient of distance from the protected areas in the Serengeti Ecosystem in northern Tanzania and to analyse the economic effects crop raiding has on different area at household level. We hypothesise that the amount of crop damage by wild animals per household per year is similar in villages adjacent to Serengeti National Park and in villages farther away from the park.

METHODOLOGY

Study area

The study area is located in the north-eastern corner of Tanzania, northwest of the Serengeti National Park (SNP) (Figure 1). The western Serengeti corridor extends westward to Lake Victoria (1°30' - 2°30' and 33°50'S - 34°45'E). The SNP is the central part of the greater Serengeti Ecosystem in the northern Tanzanian highlands. Rainfall in the Serengeti is seasonal and is determined by large-scale weather patterns, modified by local topography (Pennycuick and Norton-Griffiths, 1976). The rain normally falls in two seasons; the short rains from November to December and the long rains from late February until mid May (Hopcraft, 2010). There is a rainfall gradient from the dry southwest plains to the wetter northwest. Rainfall ranges from 514-688 mm per year in the southeast plains to 857-976 mm per year in the central woodlands and western corridor and 972-1100 mm per year in the north (Campbell and Hofer, 1995). It has been reported that, in many areas of Tanzania, crop raiding occurs mostly during rain seasons when crops are matured (Ntalwila et al., 2011). The monthly average maximum temperature in the western Serengeti fluctuates between 25 and 32°C (Campbell and Hofer, 1995). The minimum daily temperature ranges between 13 and 19°C.

People inhabiting this region are either agro-pastoralists or pastoralists. The human population in the area is estimated to be over two million (URT, 2012). The main ethnic groups in the study areas are Ikoma, Sukuma, Nata, Ikizu, Jita and Kurya. Local communities in western Serengeti pursue differences economic activities that include fishing, livestock rearing, game meat hunting and trading (Loibooki, 1997; Loibooki et al., 2002). Approximately 96% of the respondents in the Serengeti ecosystem depend on

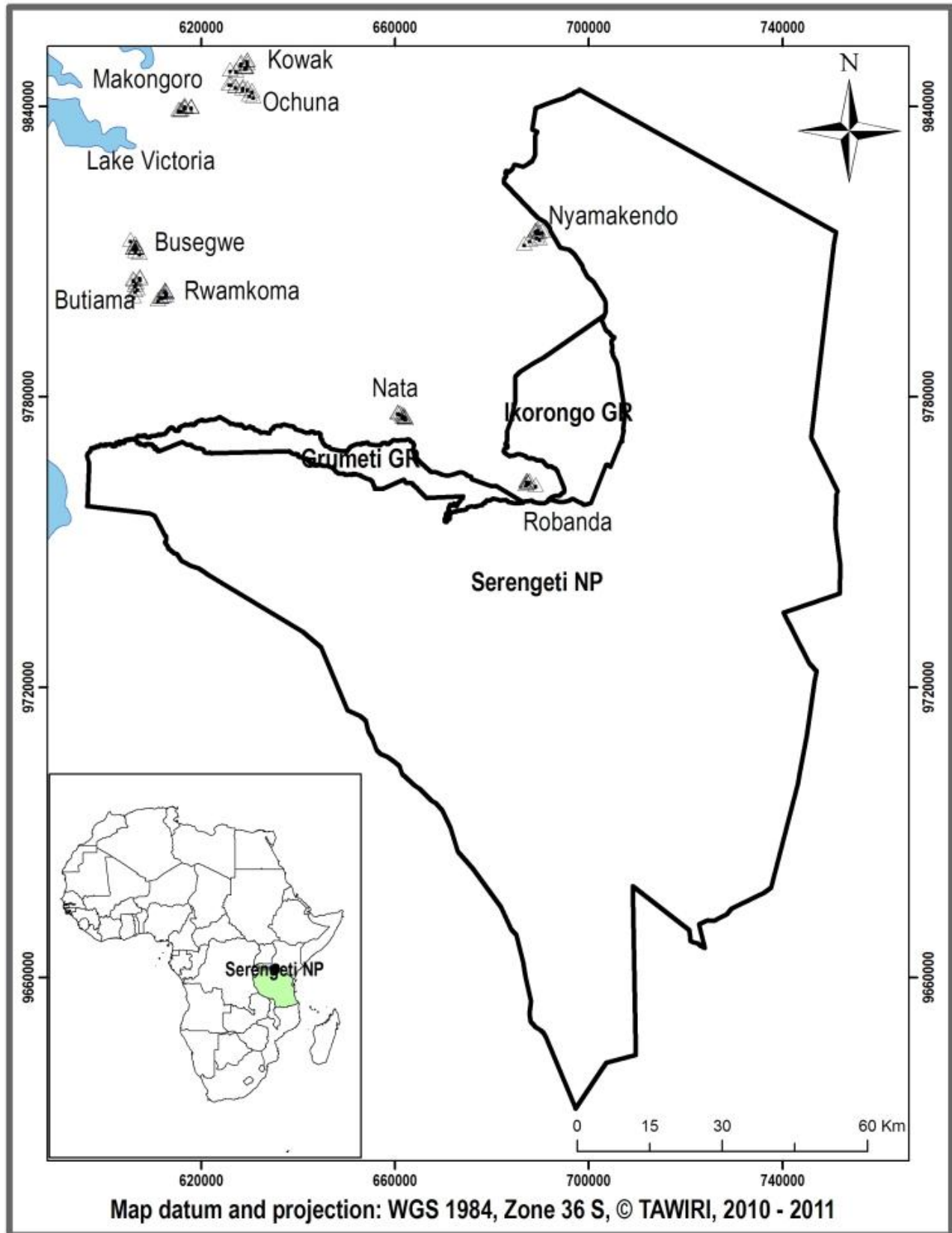


Figure 1. Map of the study area showing Serengeti National Park, Grumeti and Ikorongo Game Reserves, Lake Victoria and the surveyed villages (Robanda, Nyamakendo, Nattambisso, Butiama, Busegwe, Rwamkoma, Ochuna, Makongos and Kowak) indicated with triangles.

Box 1. Research questions on crop raiding.

-
1. Do you grow crops in your farm?
 2. How many cropping season do you have in the area?
 3. Have you experienced any crop raiding incidences in your farm for the last cropping season?
 4. How many incidences of crop raiding occurred in your farm for the last cropping season?
 5. What coping strategies do you apply to prevent your crops from being destroyed by wild animals?
 6. Mention type of crops damaged and type of wild animals responsible for the damage
 7. What was the growth stage of crop damaged by wild animals (seedling, flowering or fruiting)?
 8. What is the estimate of economic losses resulting from crop damaged caused by wildlife in percentages (%) and Tanzania Shillings (TShs) in the last cropping season?
-

crop-based agriculture for their socio-economic development (Kaswamila, 2007). The crop cycle follows rain patterns. Maize, sorghum and millet are planted twice a year, in February-March and August-October, and the harvesting periods for the plantings are June-July and January-February, respectively. Livestock includes cattle, goats, sheep and poultry, although a few households keep pigs and donkeys. The western Serengeti District is one of the areas that are highly affected by massive attacks on field crops (Kaswamila, 2007).

Data collection

Data were collected throughout the year from January 2010 to December 2010 in nine selected villages along a gradient of distance from SNP. The selection of the villages in gradient was done in such a way that three villages were located within 10 km from the protected area (Robanda, Nyamakendo and Nattambisso), approximately 40 km (Butiama, Busegwe and Rwamkoma) and approximately 80 km from the protected area (Ochuna, Makongos and Kowak) (Figure 1). Crop raiding data were collected using the following techniques: key informant survey, group discussions, and questionnaires (Box 1). The questionnaire interviews covered a total of 459 households that were randomly selected from the village and sub-village registers. We interviewed household heads, their wives or resident adults (≥ 18 years old). The villages and sub-villages were picked based on a random selection. In terms of gender, 36.2% of the interviewed respondents were females and 63.8% were males. The questions were both close-ended and open-ended and were aimed at extracting the respondent's opinion in an open minded atmosphere. The questionnaire was written to elucidate patterns of crop raiding incidences, type of crops damaged, type of wild animals responsible for the damage, estimates of crop losses and coping strategies against crop raiding. For each village, we selected ten households whose farms were monitored for crop raiding. We hired an enumerator in each village who recorded any crop damage within the project villages and other villages that were not selected. After any incidence involving crop damage, enumerators were instructed to record and report the events to the Village Executive Office where similar data were recorded and compiled.

Data analysis

Statistical analyses were conducted using the Statistical Package for Social Sciences (SPSS, version 17). A multiple response analysis was used to calculate the frequency of damage to individual crop types, the developmental stage of plants when the damages occurred and the estimated economic loss due to crop

damage per household. Pearson Chi-square tests were applied to test the different independent variables in relation to distance from the protected area.

RESULTS**Crop raiding by wild animals based on the distance from the SNP boundary**

Fifty four percent (54.3%) of respondents ($n = 451$) reported crop damage caused by wild animals. The frequency of crop damage differed significantly among the surveyed villages along the distance gradient from the park (Pearson Chi-square: $\chi^2 = 6.0$, $df = 2$, $n = 451$, $P < 0.05$, Figure 2). The percentage of farms that experienced crop damage was highest in villages that were closest to the protected area (Figure 2).

Types of wild animals involved in crop raiding

Wild animals involved in crop raiding differed significantly among the surveyed villages (Pearson Chi-square: $\chi^2 = 446.1$ $df = 10$, $n = 644$, $P < 0.001$, Figure 3). Baboons were reported to be the most destructive wild animals in the surveyed villages at middle and far distances from the park (36.8%, $n = 644$). Elephants (35.1%, $n = 644$) were especially destructive in the villages nearest to the protected area (Figure 3). Other wild animals reported to damage crops were birds, rodents, squirrels, bushpigs, warthogs and porcupines.

The types of crops damaged by wild animals differed significantly among the surveyed villages (Pearson Chi-square: $\chi^2 = 41.7$ $df = 10$, $n = 703$, $P < 0.001$, Figure 4). Maize was the crop with the most reported damage (38.8%, $n = 703$), while sorghum was the second most affected crop (25.7%, $n = 703$). Other crops that experienced damage by wild animals were cassava, potatoes, finger millet, groundnuts and beans. The developmental stage of the damaged crops differed significantly (Pearson Chi-square: $\chi^2 = 27.8$ $df = 6$, $n = 213$, $P < 0.001$, Table 1). Crop-raiding was most intense during harvest time when crops were mature (47.9 %, $n =$

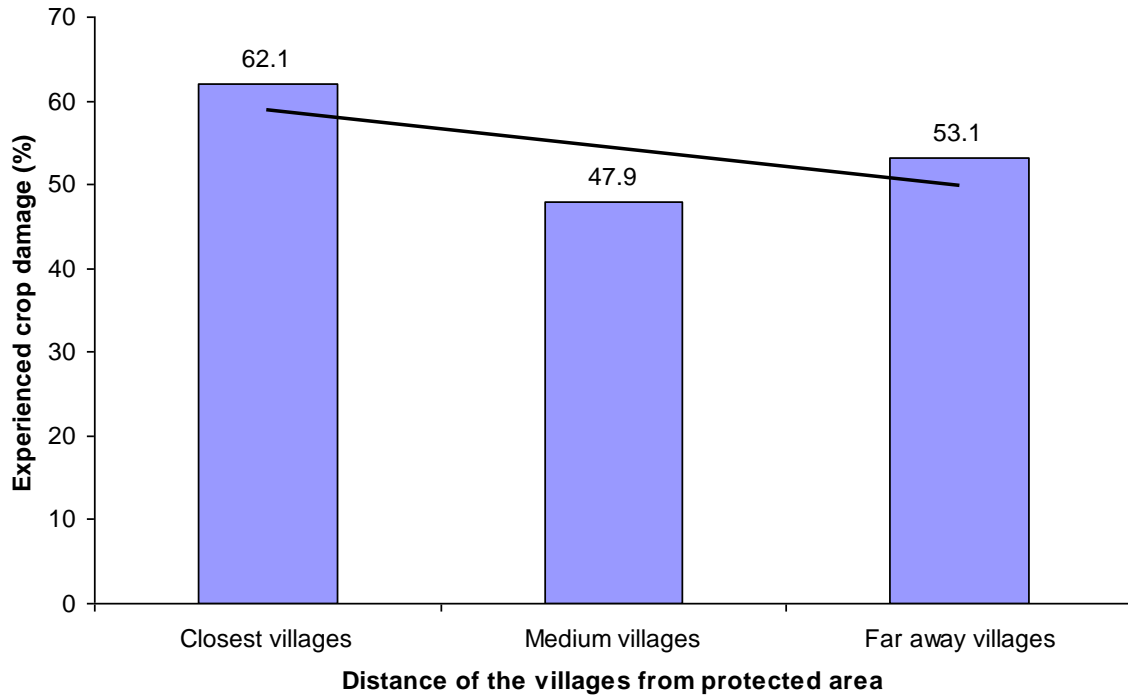


Figure 2. Percentage of households experiencing crop damage in the surveyed villages.

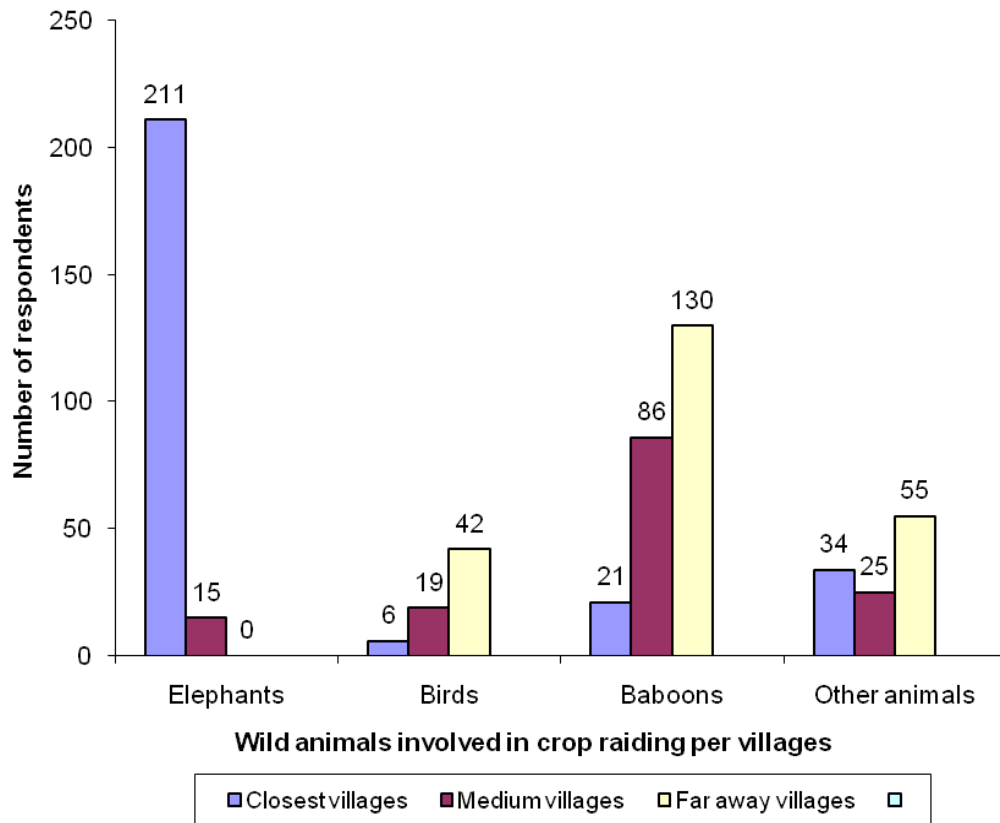


Figure 3. Wild animals involved in crop raiding at different distances from the Serengeti National Park (other wild animals include rodents, squirrels, bushpigs, warthogs and porcupines).

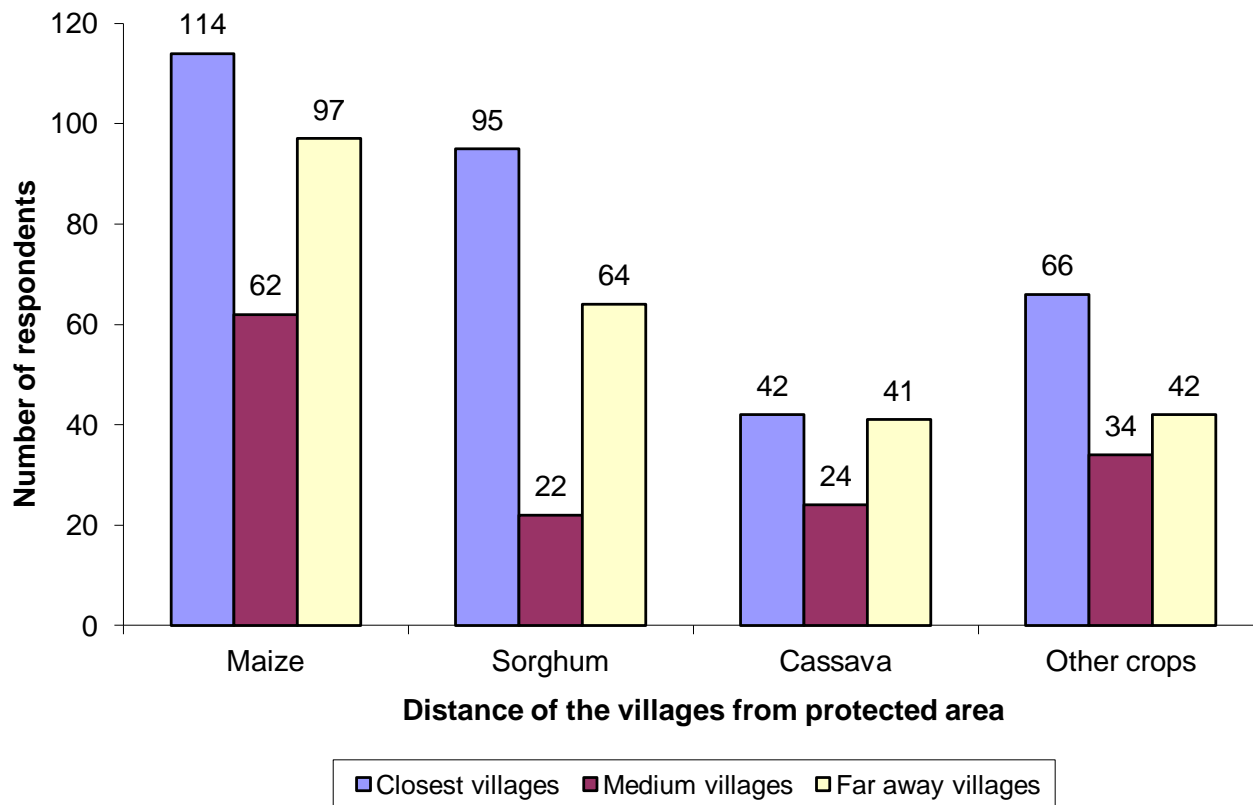


Figure 4. Number of respondents who reported different types of crops damaged by wild animals at different distances from the park (other crops include potatoes, finger millet, groundnuts and beans).

Table 1. Stage of crop growth when damaged.

Village	Stages of crop growth					Total
		Matured	Flowering	Growth	Seedling	
Closest villages	N	33	33	16	16	98
	%	33.7	33.7	16.3	16.3	100
Medium villages	N	43	6	5	4	58
	%	74.1	10.3	8.6	6.9	100
Far away villages	N	26	21	6	4	57
	%	45.5	36.8	10.5	7.1	100

213), followed by the flowering season (28.2%, n = 213), the growth season (12.7%, n = 213) and the seedling stage season (11.3%, n = 213).

Measures taken by local communities for deterring crop raiders

Methods used by local communities to protect their farms from destruction by wild animals differed significantly by

the distance of the village from the protected area (Pearson Chi-square: $\chi^2 = 13.4$ df = 6, n = 255, P = 0.037, Figure 5). The most commonly used crop protection strategy was constant guarding of the farm throughout the cropping season. Farmers reported guarding their farm throughout the day and night by patrolling their fields, actively chasing the wild animals away from the farms using dogs and building watch-out huts on the farms (85.1%, n = 255). Additionally, farmers

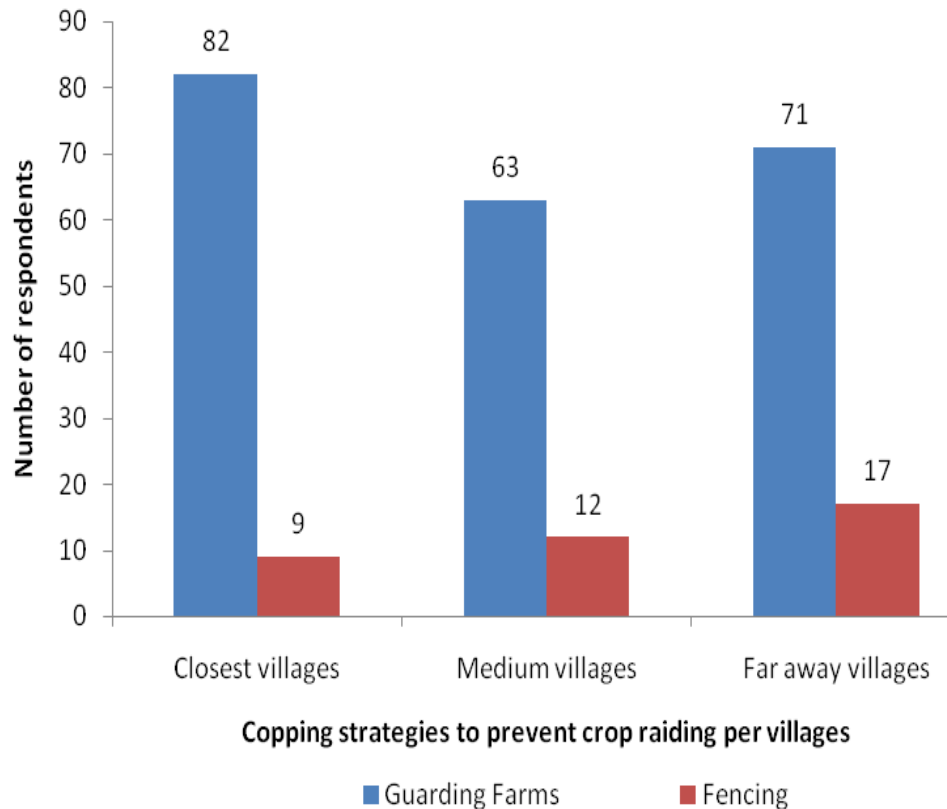


Figure 5. Number of respondents who reported different coping strategies to prevent crop raiding.

reported guarding their farms by making noise through shouting, beating drums, hunting and using firing-flashes to scare the wild animals. Moreover, respondents reported fencing their farms using thorn twigs, oily rugs and brightly coloured rugs (14.9%, $n = 255$).

Economic loss due to crop damage

Respondents were asked to produce an estimate of the total economic losses resulting from crop damage caused by wild animals on a per household basis. The reported estimated costs differed significantly between the villages, being highest in villages close to protected areas (Pearson Chi-square: $\chi^2 = 46.9$ $df = 10$, $n = 240$, $P < 0.001$, Figure 6). The mean loss estimated in the surveyed villages was Tanzanian Shillings TSh. 34,093 ($\pm SD = 1,224.10$) per household per season (equivalent to US\$ 22). This loss equated to an average loss of 16.7% of the total household income resulting from crop harvest per annum. The villages located closest to the protected areas reported the highest mean percentage loss in the total household annual crop harvest income (24.9%), which equates to a cost of TSh. 54,466 ($\pm SD = 2,017.26$) (equivalent to US\$ 35) per household per season. The villages located far away from the protected area reported the lowest mean percentage loss in the

total household annual crop harvest income (11.0%), while villages in between reported a mean estimated loss of TSh 20,938 ($\pm SD = 565.90$) (equivalent to US\$ 13.5) per household per season.

DISCUSSION

Crop raiding by wild animals with respect to distance from the park boundary

The results of this study suggest that the distance of a farm from the protected area is an important factor in determining the extent of crop raiding by wild animals. In addition, our results show that the villages closest to the protected area experience much higher frequencies of crop damage as well as higher costs of crop damage relative to villages located farther away. This result is contrary to our hypothesis that the amount of crop damage by wild animals per household per year is similar in villages adjacent to Serengeti National Park and in villages farther away from the park. The most serious damage was caused by wildlife from the park, especially elephants. However, baboons were reported to cause more crop damage in the villages most distant from the park. This is because elephants cannot move far away

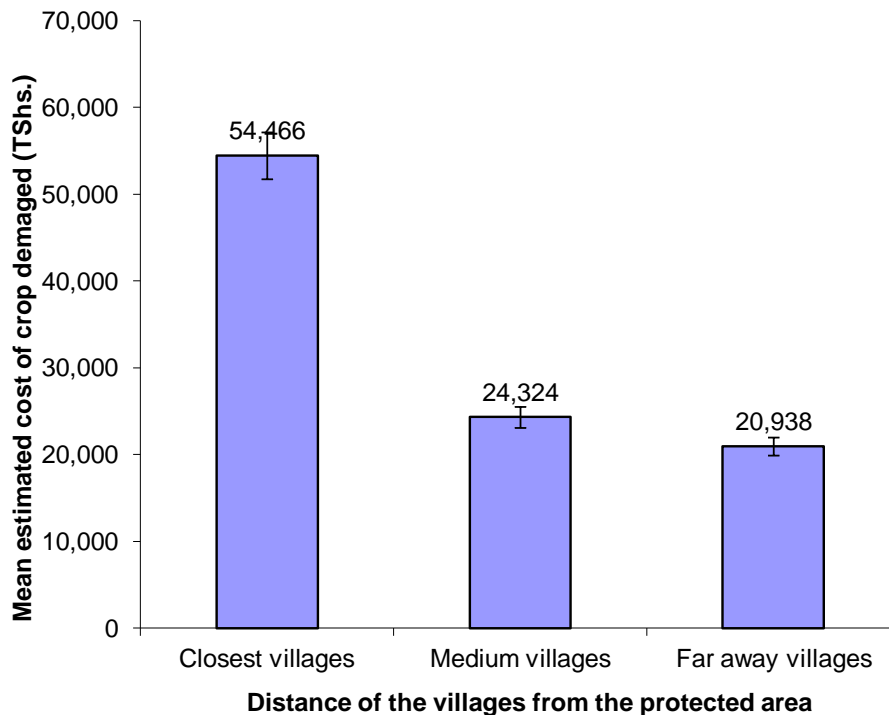


Figure 6. Estimated average cost of damaged crops per household (1 US\$ \approx 1550 TShs).

from the protected areas while other raider species in habitats near these far villages cannot move back to the park because the distance they have to move back to SNP is too long. Baboons are able to thrive in human-dominated habitats, provided there are some kopjes, hills and bushes in the village lands to serve as hiding places. Some of the farthest villages surveyed were near Lake Victoria, where some forest fragments still serve as primate habitat (Hill, 2000; Gillingham and Lee, 2003; Ole Meing'ataki, 2005; Ntalwila, 2007).

Wild animals involved in crop raiding

Different wild animals were involved in crop raiding. Primates were the most destructive wild animal in the surveyed villages at middle and far distances from the park, followed by elephants, which were destructive in the villages located adjacent to protected areas. Elephants have been reported to be the most damaging animals elsewhere in Tanzania (Ntalwila et al., 2003; Kaswamila, 2007). Moreover, African elephants, whose populations expanded rapidly following the ban of ivory trade, are presently becoming a large problem in the crop production economy, especially in villages adjacent to SNP (Ntalwila et al., 2003; Kaswamila, 2007; Malugu and Hoare, 2007; Malugu, 2010). In agreement with previous studies, the results of this study indicate that elephants

and primates are the animals most destructive to crop production. However, the ranking of the problem differs from many previous studies, which rank elephants as the most destructive (Kaswamila, 2007; Malugu and Hoare, 2007; Malugu, 2010). This may be because previous studies only concentrated on villages that were close to the park boundaries; the current study was conducted along a distance gradient up to 80 km away from the park. Sampling villages along such a gradient of distance is important in order to analyse the level of conflict and the animals involved. It is important for conservationists to focus on all rural regions, not only on areas close to protected areas, if they intend to better understand and solve the crop raiding problem. Also, this observation might be attributed to the fact that elephants cannot move far away from the protected areas, while baboons thrive in human dominated habitats as long as there are kopjes, hills and bushes in the village lands. Similar findings have been reported elsewhere (Ntalwila, 2007; Sarker and Røskaft, 2011).

Maize was the most preferred crop and thus the most damaged, followed by sorghum. This observation agrees with a report by Malugu and Hoare (2007) from similar locations. In our study, 38.8% of respondents indicated that maize was the most important crop, while 25.7% indicated sorghum. In contrast, Malugu and Hoare (2007) ranked sorghum first (20.8%) and maize second (18.9%). The fact that the two types of crops were mentioned and

respondents were able to rank them suggests that maize and sorghum are the major staple food for local communities in the area and are thus widely cultivated. Respondents in this study reported that crop-raiding peaked during harvest time in the wet season when crops were mature, followed by during flowering, as previously reported (Parker and Osborn, 2001; Malima et al., 2005; Jackson et al., 2008; Gunn, 2009; Malugu, 2010). This may be due to increased nutritional benefit from mature crops.

Measures taken by local communities for deterring crop raiders

To protect crop fields from wild animals, local communities around protected areas use different strategies. These strategies included guarding the farm during the day and night, making noises to scare wild animals, and fencing farms (Hill, 2000; Kagoro-Rugunda, 2004; Fungo, 2011; Malugu, 2010). Local communities in the study area developed and adopted different strategies to protect their farms because they realized that no single approach was efficient enough to deter the crop raiders. Karidozo and Osborn (2007) also found that a combination of crop protective methods would better protect the crop from wild animals and aid in conserving the animals.

However, the selection of available methods of farm protection depends on the species of animal that must be dealt with. For example, for deterring large animals such as elephants, guns may be fired. According to Fungo (2011), selection of the optimal method depends on the crop being grown; guard labour availability and vulnerability of the crop to raiders.

Economic losses caused by crop damage

Crop raiding by wild animals caused a significant economic loss in the local communities surrounding protected areas. The villages located closest to the protected areas reported the highest mean percentage loss per household per season, followed by medium villages and was reported lowest in villages located far away from the protected area. This observation might be contributed to the fact that in the closest villages, large animals such as elephants were also involved in crop damages which causes big losses as compared to baboons that were mostly reported to cause crop damages in further away villages. We found that the total estimated mean losses in the villages surveyed were highly variable; however, the mean was 16.7%. The mean percentage crop loss was higher as compared to that reported in Serengeti District (11%) (Kaswamila, 2007). This indicates the magnitude of the economic losses due to crop raiding in the local communities of the Serengeti ecosystem.

Recommendations

We recommend further studies on the crop yield gap resulting from crop raiding by wild animals. Second, because primates were reported to be the most destructive wild animal in the surveyed villages, we recommend further studies on human-primate conflict in communities around SNP and other protected areas.

Third, in order to prevent the movement of more people closer to the park, awareness creation on the losses that the community may incur for being closer to the park boundary is highly recommended. Lastly, for effective protection of crop farms from wild animals, we recommend that local communities adopt a combination of methods to deter crop-raiding wild animals from their farms for examples, Chill and bee fences (Fungo, 2011; Kabepole, 2011; Malugu et al., 2011), guarding and making noises (Hill, 2000; Malugu, 2010) as suggested by Karidozo and Osborn (2007).

Conflict of Interests

The author(s) have not declared any conflict of interests.

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Full Length Research Paper

Sustaining *Prunus africana* (Hook.f.) Kalkman (Rosaceae), a CITES listing tree species in the Teza Forest, Kibira National Park, Burundi

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Prunus africana (Hook.f.) Kalkman (Rosaceae)/Pygeum or African cherry is a mountain tree species of the tropical Africa. *Prunus* bark is exported dried, chipped or powdered to USA and Europe to produce an extract used to treat benign prostrate hyperplasia. The plant is considered as vulnerable and is listed in appendix II of the Convention on International Trade in Endangered Species of Fauna and Flora (CITES). This paper aims to estimate the average mass of mature stems of that tree species as a tool for establishing its non-detriment findings in Burundi. The study was carried out in July 2013 in a production forest of 2 800 ha found in the Teza sector, in the south of the Kibira National Park. Stem barks were gathered from a total of 36 mature trees of *Prunus africana* with diameter at breast height of more than 30 cm, known as mature trees. The harvesting of the barks was done according to the recommended sustainable technic which is, “for all trees above 30 cm DBH, only two quarters of the bark are taken from the main stem up to the first branch”. The average mass of the fresh bark of a *Prunus* mature tree is 47.0 kg. The quantity of the dried bark estimated to be harvested in sustainable manner each year in the Teza sector is 24.8 tons/year. This is known as the annual quota. If Burundi authorities agree to allow the exploitation of *Prunus* barks in the Kibira Park, the National Institute for the Environment and the Conservation of Nature (INECN), which is in charge of the management of protected areas in Burundi, can yield about 426 155.7 – 497 181.7 USD/year. And these funds can be used to enhance the strategy of conservation settled within the “Parks for Peace Project” in the Kibira national park.

Key words: *Prunus Africana*, Kibira National Park, mass, sustainable technic, annual quota, Burundi, CITES.

INTRODUCTION

Burundi is a small densely populated country with a topography dominated by hills and mountains interspersed with seasonal and permanent wetlands in the low-lying areas. Overall responsibility for environmental management and protection in Burundi is housed in the Ministry

of Water, Environment, Land Use and Urbanism (Ministère de l'eau, Environnement, Aménagement du Territoire et Urbanisme, MEEATU). Two autonomous institutions with environmental mandates exist under the Directorate General of Environment and Forests: the National Institute

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for the Environment and the Conservation of Nature (INECN) and the Geographic Institute of Burundi (IGEBU). INECN has direct management responsibility for the protected areas, including national parks and natural reserves (Hobbs and Knausenberger, 2003).

Prunus africana (Hook.f.) Kalkman is a mountain tree species of the tropical Africa including the Côte d'Ivoire, Bioko, Sao Tome, Ethiopia, Kenya, Uganda, South Africa, Madagascar, Congo, the Democratic Republic of Congo, Mozambique, Tanzania, Burundi and Cameroon. *P. africana* grows well in the sub-mountain and mountain forests at an altitude of 800 - 3000 m. *P. africana* is an evergreen canopy tree, 30 m tall with thick, fissured bark and straight bole that can reach a diameter of 1.5 m (Vivien and Faure, 1985; Fraser et al., 1996; Tchouto, 1996). The bark is the major source of an extract used to treat benign prostatic hyperplasia, an increasingly common health problem in older men in the western world. Prostate-related diseases increase in prevalence as men age. And as the average age of the world's population increases, the incidences of prostate diseases will increase as well, triggering a corresponding rise in demand for therapies. According to the World Cancer Research Fund International, prostate cancer is the second most common cancer in men worldwide. Around 910,000 cases of prostate cancer were recorded in 2008, accounting for approximately 14% of all new cancer cases in men (World Agroforestry Centre, 2012). The United Nations Food and Agriculture Organization (FAO) reported as far back as 1996 that the demand for the species' bark, which is used to produce treatments for prostate gland disorders, could lead to its over-exploitation (FAO, 1996). In 1997, the global need was about 4 000 tons of dried barks per year for a value of 220 millions of USD. Two hundred kilogram of dried bark yield 5 kg of extract (Cunningham et al., 1997). The trade in dried pygeum bark and bark extract is in the order of 3 000 to 5 000 tonnes a year (Page, 2003) and the main sources are in Cameroon, Madagascar, Equatorial Guinea, Kenya, Uganda and Tanzania. In Burundi, *P. africana* is found in five protected areas including two national parks (Kibira in the North and Ruvubu in the East), and three natural reserves (Bururi, Maragarazi and Monge). The species is also found in the sacred forest of Mpotsa. The species is well known by Burundi peoples who designate it as "Muremera" in Kirundi, the first national language. Here, the species is not only used as a medicinal plant, but also as a of high-value timber together with *Entandrophragma excelsum* (Meliaceae). *P. africana* is one of the plant species that can be used to enhance African Traditional Medicine as discussed by Kamsu-Foguem et al. (2003).

A key requirement of CITES is the non-detriment findings made by the Scientific Authority of the range state prior to export, certifying that export is not detrimental to the survival of the species. This requires information on the location, stocking, growth and condition of the species and on its ecology, regeneration and subse-

quent protection. Such information is often lacking, incomplete or imprecise making a proper evaluation of the sustainable levels of utilisation, establishment of quota and conditions attached to be difficult. The Scientific Authorities also face obstacles due to inadequately trained and resourced staff. The National Institute for the Environment and the Conservation of Nature (INECN) is the CITES Scientific Authority for Burundi. INECN has recently conducted *Prunus* inventories in the Kibira national park which led to the estimation of the stock of mature trees (Betti et al., 2013). This paper aims to estimate the mass of stem barks of mature *P. africana* in the Teza forest in the South of the Kibira national park as a contribution to making non-detriment findings on *P. africana* in Burundi.

MATERIALS AND METHODS

Study site

The Kibira National Park is a national park in north-western Burundi, overlapping with four provinces and covering 40 000 ha. Kibira National Park lies atop the mountains of the Congo-Nil divide. It extends north from the provincial town of Muramvya to the border of Rwanda where it is contiguous with the Nyungwe National Park. Kibira mountains are up to 2 600 m. The average annual rainfall is up to 1 600 mm while the average temperature is 17°C.

Main vegetation strata include: *Entandrophragma excelsum* (Meliaceae), *Parinari excelsa* (Chrysobalanaceae), *Albizia gummifera* (Mimosaceae) and *Prunus africana* (Rosaceae) stands, *Parinari excelsa* var. *holstii* and *Polyscias fulva* (Araliaceae) stands, *Polyscias fulva* (Araliaceae), *Macaranga neomildredian* (Euphorbiaceae) and *Syzygium parvifolium* (Myrtaceae) stands, *Hagenia abyssinica* (Rosaceae) and *Faurea saligna* (Proteaceae) secondary forest stands, *Philippia benguelensis* (Ericaceae) and *Protea madiensis* (Proteaceae) high-altitude stands on ridges, *Arundinaria alpina* (Poaceae) or pure bamboo stands, stands along thalwegs and stands formed by recolonization of previously cultivated and grazed land. The park is dominated by *afromontane* forests, including species such as *Parinari excelsa*, *Entandrophragma excelsum*, *Albizia gummifera* and *Prunus africana*. It is a zone rich in both animal and plant biodiversity: 644 plant species have been found in the park, as well as about 98 species of mammal. Bird life is also rich and varied, with 43 families and more than 200 are species identified (Hobbs and Knausenberger, 2003; Nzigidahera, 2000). More than three-quarters of the water in the country's largest dam – providing more than 50% of the hydroelectric energy consumed comes from this forest. Thus the park, situated as it is on the Congo-Nil ridge, plays a fundamental role in regulating the hydrological system and protecting against soil erosion. The Forest Code promulgated in 1985 (Law 1/02 of 25 March 1985) provides the basis for the various Burundian laws on forests, notably Law 1/010 of 30 June 2000 containing the environmental code for the country. The Kibira National Park was granted legal status in 2000 with Decree 100/007 of 25 January, establishing the park and four nature reserves. The management plan of the Kibira national park drafted in 2008 – 2009 with the help of the Wildlife Conservation Society (WCS) has led to the subdivision of the park into four small management units or sectors (one sector for each province) distributed as follows from south to the north: Teza, Rwegura, Mabay and Musigati. Each sector is divided in subsectors. The KNP has a park supervisor and four sector heads, while each subsector has a forest warden and an assistant.

The INECN and the Parks for Peace Project have established watchdog committees elected by the people at various levels – hills (the smallest administrative unit), sectors, zones and communes. The park supervisor supervises all activities in the park, the sector heads implement them in their respective zones, and the watchdog committees closely monitor the way the park is managed. Thus, the INECN, the local administration, the local population and the watchdog committees are the main actors in park management. Nearly 50 000 people from communes adjacent to the park are involved directly or indirectly in management of the park (Hobbs and Knausenberger, 2003; Nzigidahera et al., 2002; Nzojibwami, 2012). Areas bordering the KNP have always been considered as dense in terms of population, with Musigati and Muramvya being the two most dense councils of the park (Nzojibwami, 2012).

Method

Prunus stem barks were harvested inside the Kibira National Park, in a limit area of 2 800 ha of production forests, in the Teza sector, on July 2013. The method used is the sustainable harvesting, is often recommended for the harvesting of *P. africana* barks by trade companies.

Selection of trees

This study was conducted in July 2013 in the Teza, the south of the Kibira National Park. The study was conducted in the frame of the project “Evaluation du stock d’arbres sur pied de *Prunus africana* dans le pays en vue d’une exploitation durable” supported by the Secretary General of the Convention on International Trade in Endangered Species of Fauna and Flora (CITES) and executed by the National Institute for the Environment and the Conservation of Nature (INECN). The study started one month later after the teams in charge of assessing the abundance of *P. africana* trees in Teza forest (Betti et al., 2013) has finalised their work. The sheets of the teams in charge of counting were used to select the best trees that can allow for better appreciation of the mass of the stem bark of *P. africana* trees. Practically, trees were selected according to their diameter, their accessibility, their healthy and their conformity (only straight stems were selected). Once a given stem was identified in a counting sheet, we noted its geographical data (latitude and longitude) and we went with the global position system (GPS) in the field to take appropriated measures of that stem. A total of 36 *P. africana* mature trees were sampled.

Estimation of the mass of the stem-barks of the mature trees of *P. Africana*

To estimate the mass of the stem barks of *Prunus africana*, we used a direct method, the same method recommended to harvesters in the field to ensure the healthy and conservation of the stand tree in the forest. This method consists of, “for a given tree above 30 cm DBH, only two opposite quarters of the bark taken from the main stem up to the first branch”. We proceeded in four steps: (1) measuring the circumference of the tree, (2) dividing the circumference by 4, (3) harvesting the stem bark of the two opposite quarters and (4) measuring the mass of the stem bark harvested. The circumference of the tree was measured at DBH, which is at 1.30 m above the soil. The obtained value was then divided by 4 to obtain the width of each quarter. The stem bark of the two opposite quarters was then harvested starting from the breast high (1.30 m) to the insertion point of the first big branch. The bark harvested was then weighed, and its thickness measured.

The scale was used to climb on trees. To easily untie the bark from the sapwood, we used a bludgeon. We used a calliper rule to measure the thickness of the bark. Data obtained were recorded in an appropriate sheet precisely with the order number of the line (Figure 1), and the altitude of the tree.

For data analysis, we calculated the average mass of an mature tree, and we estimated the annual quota for the all sector of Teza.

The annual quota (Q_a) of the stem barks of *Prunus* trees is obtained by the formula:

$$Q_a = T_r \times \frac{M_s}{T}$$

T_r : Total number of mature trees estimated in the Teza sector, M_s : average mass of the dried barks of *Prunus* trees harvested on a sustainable basis (kg), T : In this paper, we distinguished the rotation from a small rotation.

The small rotation is the time that separates two consecutive harvesting campaigns on the same tree (6 years), while the rotation is the time that separates two consecutive harvesting campaigns on the same side for the same tree (6 x 2 = 12 years). This time is said to be enough to allow the regeneration of the bark.

RESULTS AND DISCUSSION

Mass of mature trees

Table 1 shows each of the 36 trees, their position (line), the diameter at breast high and the mass obtained. The average fresh mass of an mature (mature) *Prunus* tree in the Teza sector in the Kibira national park is 47.0 ± 21.9 kg. This value is different from the 55 kg found in mount Oku in the north west of Cameroon with the same method (Ondigui, 2001) but it is less as compared to the 69.3 kg found in the Adamaoua region in Cameroon using the indirect method (Betti and Ambara, 2013). Acworth (1999) suggested that an average mature tree of *P. africana* yields 75 kg of fresh bark.

Figure 2 illustrates the relation linking the diameter and the fresh mass of a mature *Prunus* tree in the Teza sector. The curve can be best approximated by the equation $M = 0.0004D^2 + 0.6865D - 2.2243$, M = mass of the fresh bark in kg, and D = diameter at breast high for mature trees (trees with diameter above 30 cm).

Annual quota of stem barks of *Prunus africana* (Rosaceae) in the Teza sector in the Kibira national park

The total surface area of the productive forest assessed in the Teza sector is 2 800 ha. The density of mature *Prunus* stems in that sector was estimated at 2.04 stem/ha while the total number of mature living trees or mature stock estimated was 6 347 (Betti et al., 2013). We used this value (6 347 mature trees) to estimate the total productivity of the stem barks in the Teza sector. The small rotation was estimated at 6 years while the ratio mass of dried bark/mass of fresh bark is 0.5. Table 2

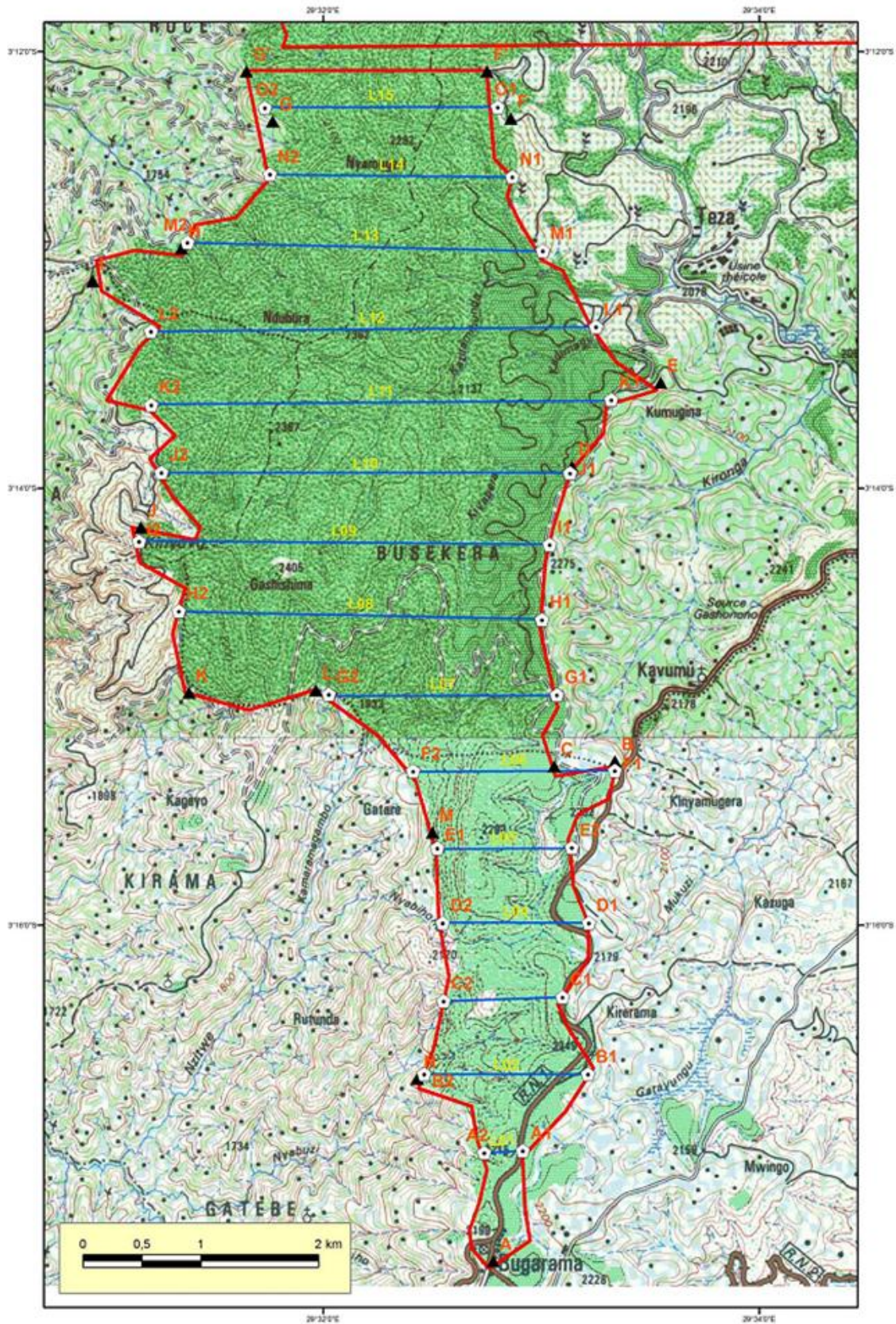


Figure 1. Map showing the disposal of the lines assessed for trees study in the Teza sector, KNP, Burundi. The coordinate system used is WGS84.

Table 1. Mass of fresh bark sampled in the Teza sector, in the KNP, Burundi.

No. of tree	Line	DBH (cm)	Mass (kg)
1	L02	89.2	58
2	L03	69.7	51
3	L03	54.8	31
4	L03	73.2	41
5	L03	34.7	35
6	L03	91.4	50
7	L03	40.4	17
8	L03	49.7	21
9	L03	83.8	54
10	L03	41.4	36
11	L04	63.7	38
12	L04	46.2	50
13	L04	51.9	48
14	L04	95.5	53.5
15	L05	62.7	75
16	L05	45.9	30
17	L05	30.6	14.5
18	L05	133.8	99
19	L05	46.5	10
20	L05	69.7	67
21	L05	66.6	26.5
22	L14	108.3	70
23	L14	68.8	39.5
24	L14	95.5	96
25	L07	56.1	28.5
26	L07	65.3	38
27	L07	77.1	39
28	L07	76.4	40.5
29	L07	66.2	53.5
30	L07	47.1	23
31	L07	53.8	29.5
32	L08	81.8	65.5
33	L08	68.2	68
34	L15	89.2	87
35	L15	79.6	68
36	L15	86.0	40
Mean		68.4	47.0
Standard deviation		22.0	21.9

presents the results obtained. The quantity of *Prunus* dried barks that can be harvested in a sustainable manner in the Teza sector is 24 859.1 kg/year. This is considered as the annual quota. Sustainable manner in this paper shows, as explained above, for all mature trees (DBH above 30 cm), harvesting of only two first opposite quarters of the bark from the main stem up to the first big branch, waiting till the bark regenerates and come back 6 years later to harvest the two second opposite quarters left on the tree.

The stock expresses the total number of mature stems found in a given area. A total number of 6347 trees of *P. africana* can be harvested since they have reached the MED which is 30 cm. The Kibira national park is vast (40 000 ha), and the production forest can cover more than 30 000 ha. If the density of 2.04 stems/ha of mature trees was normally distributed in the park, then the total number of the mature *Prunus* trees of the park will be estimated at 61 200, which is almost twice as compared to the 32 498 trees obtained in the mount Cameroon National Park (Betti et al., 2011).

Prunus bark is exported dried, chipped or powdered to USA and Europe to produce an extract used to treat benign prostrate hyperplasia. The extract is also a raw material for the burgeoning health, bio-product, diet supplement and pharmaceutical industry. If the bark is partially stripped according to norms (Cunningham, 1993; Hall et al., 2000) of two quarter panels from a tree of more than 30 cm diameter at breast height (approximate age of 12 to 15 years), it will regenerate and may be exploited at between 5 to 15 year intervals without killing the tree. About 2000 kg of fresh bark are equal to 1000 kg of dried bark, which is needed to make 5 kg of extract.

If we suppose that the relation linking the diameter and the mass of a mature *P. africana* tree obtained in the Teza sector does not change in the whole park, the total quota of the barks that can be found in the whole Kibira national park is 266 347.32 to 310 738.54 kg of dried bark/year or 532 694.3 to 621 477.0 kg of wet bark/year. In Cameroon, the price of fresh bark of *P. africana* at the village level by trade companies around the mount Cameroon national park is about 0.8 USD. If we consider this price for the Kibira National Park, the National Institute for the Environment and the Conservation of Nature (INECN), which is in charged of the management of protected areas in Burundi, can yield about 426 155.7 to 497 181.7 USD/year. This can (1) enhance the cooperation between the park board and villages settled around the park through development projects and (2) contribute to recruiting and equipping forest guards to enforce the control.

Conclusion

The trees study conducted in the Teza sector in the north of the Kibira National Park showed that a mature *P. africana* trees can yield 47.0 kg of wet bark. The quantity of dried bark that can be harvested in sustainable manner without killing the trees is 24.8 tons/year in the Teza sector or 266 347.32 to 310 738.54 kg of dried bark/year for the whole Kibira National Park. This is essential for addressing a non-detrimental report on *P. africana* for that part of the country. If the Government of Burundi authorizes the harvesting of *Prunus* barks in the Kibira National Park, the National Institute for the Environment and the Conservation of Nature, which is in charged of

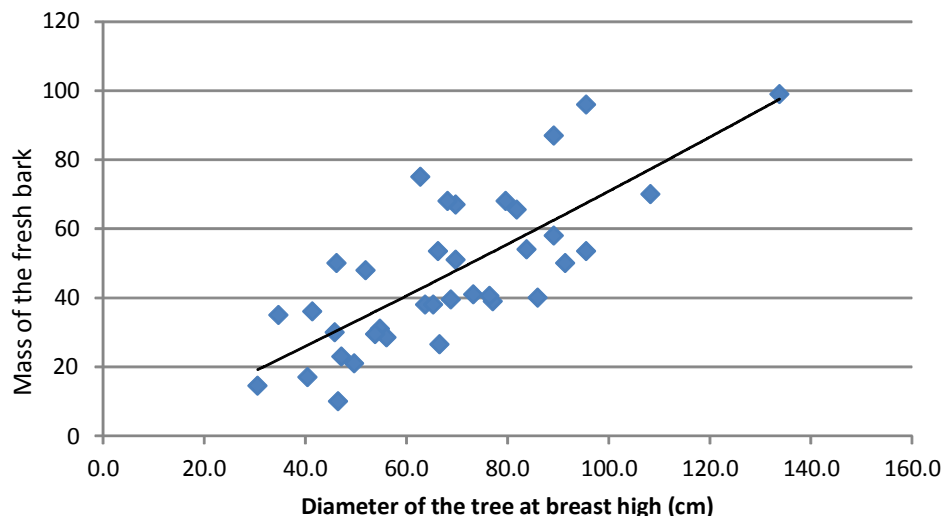


Figure 2. Relation (regression) linking the diameter at breast high and the mass of the wet bark of a mature *Prunus* tree in the Teza sector in the Kibira national Park, Burundi.

Table 2. Annual quota of dried bark of *P. africana* (Rosaceae) estimated for the Teza sector in the Kibira National Park, Burundi.

Total production area assessed in the Teza sector (ha)	Average density of mature trees (number of stems/ha)	Stock of mature stems	Average mass of dried bark (mass of fresh bark/2)	Small rotation (ans)	Annual quota (kg of dried matter)
2 800	2.04	6 347	23.5	6	24 859.1

the management of protected areas in Burundi, can yield about 426 155.7 to 497 181.7 USD/year. These funds can be used to enhance the strategy of conservation of the Kibira National Park settlements within the “Parks for Peace Project”.

Conflict of Interests

The author(s) have not declared any conflict of interests.

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Full Length Research Paper

Genetic diversity of qualitative traits of barley (*Hordeum Vulgare* L.) landrace populations collected from Gamo Highlands of Ethiopia

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Barley (*Hordeum vulgare* L.) has great adaptability to a wide range of environments. To determine genetic diversity in barley landraces, a total of 43 landrace populations were randomly sampled from the farmers' field on plant basis and characterized for eight qualitative traits; namely, kernel row number, spike density, lemma awn barb, glume color, lemma type, length of rachilla hair, kernel covering and lemma/kernel color. Morphological diversity was determined by the Shannon-Weaver index (H'). Overall barley landrace populations showed an average diversity index of 0.59, implying large diversity for the populations. Selection for adaptation to different altitude classes appears to be the main factor that has determined the observed variation, along with population-size effects. The result showed that barley landraces from Gamo highlands, Ethiopia are constituted by highly variable landraces that have large within-population diversity. These landraces are also shown to be locally adapted, with the major driving force that has shaped their population structure being consistent with selection for adaptation along an altitudinal gradient. Overall, this study highlights the potential of such landraces as a source of useful genes that can be exploited in crop improvement programmes.

Key words: Barley, characters, landraces/farmer varieties, morphological diversity, altitudinal gradient.

INTRODUCTION

Barley (*Hordeum vulgare* L.) is one of the world's most ancient food crops. It has been an important cereal crop since 8,000 to 10,000 years ago in the area of the Middle East known as "the Fertile Crescent" (Giles and von Bothmer, 1985; von Bothmer and Jacobsen, 1985). It is

ranked third among the major cereal crops on the basis of production tonnage after wheat and rice (FAOSTAT, 2013). In order of importance, barley is used for animal feed, brewing malts and human consumption (Hayes et al., 2002).

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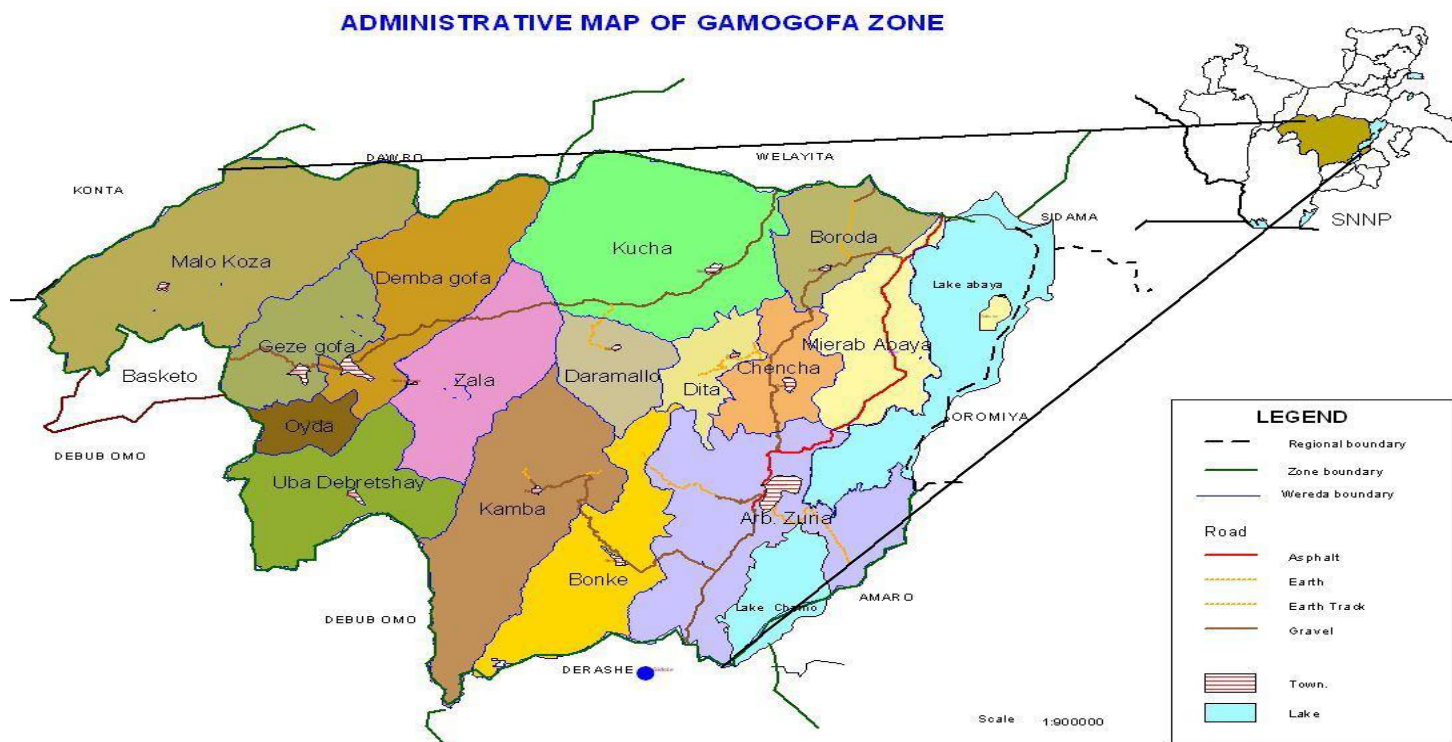


Figure 1. Map of Southern Nations Nationalities and Peoples Region showing the site of Gamo highlands.

Ethiopia is recognized as a major Vavilovian gene center. Earlier introduction from Mediterranean countries and centuries of natural and artificial selections on native crops has resulted in tremendous genetic diversity. Among the major cereal crops in which valuable genetic diversity observed were tetraploid wheat and barley are the most prominent crops for which the country is recognized as the secondary center of diversity (Demissie and Giorgis, 1991).

The characterization of genetic variability between and within populations is important for determining the rate of adaptive evolution and response to traditional crop improvement (Hunter, 1996). Genetic diversity is a raw material for evolution, thus enabling populations of species to survive, evolve and adapt to resist long-term changes in the environment. Differences within and between population can be of a strategic value to conservation as they provide a clear justification for protecting the valuable genes across their entire geographic range and all the subspecies of major populations.

The barley landraces exhibit variation both between and within populations. This within population diversity of these barley landraces might allow them to cope with environmental stresses, which is very important for achieving yield stability (Zhu et al., 2000).

The knowledge of the population structure of Ethiopian barley landraces, together with a deeper understanding of the nature and extent of their variation, is an important prerequisite for the efficient conservation and use of the existing plant materials. Several studies have reported a high level of genetic diversity in barley populations from Ethiopia, such as those based on morphological traits (Engels, 1994; Demissie and Bjornstad, 1996) and on biochemical data (Bekele, 1983a).

Thus, the study was conducted to assess the extent of morphological diversity in Gamo highlands of Ethiopia in relation to districts and altitudes. This characterization was carried out based on spike morphological traits, to classify landrace genotypes into morphologically similar clusters, and to identify the major traits responsible for most of genetic variation among the collected landrace populations.

MATERIALS AND METHODS

Plant collection

The barley landrace populations were collected in 2012 in the Gamo highlands of Ethiopia. A total of 43 populations/farmer varieties were collected during October and November, within three districts (Chencha, Dita and Bonke) in *Meher* growing season (Figure 1)

considering three altitude classes randomly chosen based on collection data: 2000-2395, 2395-2683 and 2683-3000 m.a.s.l.

The three districts were selected based on their barley crop production potential, long rainy season which covers from March to September. Agro-ecological condition varies from district to district; Dita district, has an altitude ranging from 1800-3500 m a.s.l. and the annual rainfall ranges between 801 to 1600 mm. The rainfall is of a bi-modal pattern, giving rise to two distinct seasons, the short rainy season from March to May (peak in April). It has three types of soil based on their colors: red (85%), black (13%) and brown (2%) and their texture is clay loam. The minimum and maximum temperature is 10.1 and 27.5°C, respectively. Chenchha district has an altitude range from 1800 to 3500 m.a.s.l. The area has bimodal rainfall. The annual rainfall ranges between 1201 and 1600 mm. Bonke district is highly degraded due to extensive tree cutting for fuel wood. This has resulted to continuous top soil loss due to erosion. Rainfall in the area is erratic and unreliable and ranges between 801 and 1600 mm. It has an altitude range of 801-3500 m.a.s.l. (Daniel, 1977).

The geographical position (latitude, longitude and altitude) of each sampling site was determined using a Geographical Positioning System (GPS). Each of the surveyed farmers' field size of barley was measured and found to have average range approximately 0.12 -0.25 hectare. The name of each landrace population was recorded during sampling (as informed by the farmers).

Twenty spikes per population were randomly sampled from each site. Accordingly, a total of 860 spikes/ears were used to measure the phenotypic diversity using qualitative morphological traits that are easily seen by the eye, expressed in all environments and governed by few genes. Data on the following eight morphological traits from the matured spikes were recorded. These included kernel row number, spike density, lemma awn barbs, glume color, lemma type, length of rachilla hair, kernel covering and lemma/kernel color (Table 1). The phenotypic character states of each qualitative trait were recorded using Bioversity International Barley Descriptor (1994). Color traits were recorded using the Eagle Spirit Ministry Color Chart which was developed by Kohe't (1996).

Statistical analysis

Data were analyzed for eight qualitative traits (Table 1) that have shown their usefulness in earlier studies (Engels, 1994; Demissie and Bjornstad, 1996; Tanto et al., 2009) and proved to be consistent over the years in characterization work and their scoring turned out to be reliable.

Polymorphism is measured by the percentage of polymorphic loci (PPL), Nei's genetic diversity (H), for single population or groups of populations (by landrace name, altitude class or district), within population diversity (H_S), coefficient of gene differentiation (G_{ST}) (Nei, 1973) and Shannon information index of diversity (H') (Lewontin, 1972), were all calculated by using Pop Gene ver. 1.32 (Yeh et al., 2000) software. The haploid option of the software was used for analysis in accordance with the assumption of Ferguson et al. (1998).

The actual sample sizes per altitudinal class and districts are presented in Table 2. For the comparative analyses of the diversity indices, the altitude classes were arbitrarily grouped into three altitudinal classes.

The Shannon diversity index (H') was also calculated as group of populations for districts, altitude classes and populations/farmer varieties by using the Shannon-Weaver equation as presented by Poole (1974) and it was in line with Pop Gene ver. 1.32 results.

$$H' = \sum_{i=1}^n p_i \log_e p_i \quad (1)$$

Where n represents the number of phenotypic classes of a given character, and P_i the proportion of the total number of accessions consisting of the ith class. Each value of H' was divided by ln n (where ln n is a natural logarithm of (n) in order to keep the values of H' between zero and one (Goodwin et al., 1992).

A one-way analysis of variance (ANOVA) for a normalized diversity index (H') was carried out for each trait using districts and altitude classes as classifying variables. These were treated as fixed effects and populations (farm fields) as random using SPSS Statistics 17 software.

SPSS statistics 17 software, was further used to analyze the total morphological data variation, according to two nested models of analysis of variance (ANOVA): (1) among populations, among individuals within populations (where the 'within populations' component is accounted for by the within individual barley fields); and (2) among 'groups', among populations within groups, and among individuals within populations (where the 'groups' were the landrace names/farmer varieties, the altitude classes, the districts and the 'populations' were represented by individual barley fields).

To study the patterns of diversity among the sampled populations, a dendrogram was obtained on the basis of Nei's unbiased pair wise distance matrix between populations (1973). This showed the relationships among all the 43 landrace populations. For this analysis, the Pop Gene ver. 1.32 (Yeh et al., 2000) software was used.

RESULTS

Morphological diversity of the whole population

A total of 90 different combinations of traits (morphotypes) were detected, of which 25 accounted for 75% frequency of the total collection (647 out of 860). Figure 2 shows the percentages of the morphotypes belonging to the 25 most frequent morphotypes ranging from 1.8-5.7%.

Diversity index analysis

Morphological diversity indices derived from qualitative traits

Estimates of Shannon diversity index (H') analyzed for individual traits are presented in Table 3. These estimates ranged from 0.03 (low polymorphic) for lemma awn barb to 0.91 (highly polymorphic) for spike density.

Phenotypic diversity index (H') across the three districts

Unlike the individual population estimates, the district wise comparison displayed a higher magnitude of differences. It ranged from average H' 0.52 ± 0.11 for Dita to 0.59 ± 0.11 for Chenchha. All the three districts showed medium to high (H' > 0.52 ± 0.11) level of polymorphism. Samples from Chenchha were the most diverse (H' = 0.59 ± 0.11) followed by Bonke (H' = 0.56 ± 0.11) and Dita (H' = 0.52 ± 0.11) (Table 4). The analysis of variance of H' for individual traits was performed and much of the variation was due to variations within populations rather than districts (Table 5).

Table 1. States (character) used for each of the eight traits considered.

Trait	State
Kernel row number (KRN)	1. Two rowed, large or small sterile lateral florets
	2. Two rowed, deficient
	3. Irregular, variable lateral florets development
	4. Six rowed, awnless or awnleted lateral florets
	5. Six rowed, long awns on all lateral florets
Spike density (SD)	1. Lax
	7. Dense
Lemma awn barb (LAB)	3. Smooth
	5. Intermediate (small barbs on upper half)
	7. Rough
Glume color (GC)	1. White
	2. Yellow
	3. Brown
	4. Black
Lemma type (LT)	1. No lemma teeth
	2. Lemma teeth (barbs on lateral nerves)
	3. Lemma hair
Length of rachilla hair (LRH)	1. Short
	2. Long
Kernel covering (KC)	1. Naked grain
	2. Semi-covered grain
	3. Covered grain
Lemma/Kernel color (LC)	1. Amber (=normal)
	2. Tan/red
	3. Purple
	4. Black/grey
	5. Yellow

Phenotypic diversity indices across altitude classes

Like district wise estimates, Shannon diversity index (H')

Table 2. Distribution of the landraces/farmer varieties collected across districts and altitude classes.

Altitude (m.a.s.l)	Districts			Total
	Chencha	Dita	Bonke	
2,000-2,395	3	0	10	13
2,395-2,683	7	1	7	15
2,683-3,000	6	9	0	15
Total	16	10	17	43

values pooled over all traits for altitudinal classes showed less variation (Table 6). For characters like glume color, kernel covering and lemma color, an increase of the diversity index was observed with increasing altitude from 2,000 to 2,683 m.a.s.l and after this altitude class, a decrease occurs but not statistically significant. When the diversity for single traits was considered, the diversity only in kernel row number increased with an increasing altitude. No comprehensible associations with altitude classes were seen for the traits: length of rachilla hair, lemma type and spike density. The above results indicated that the mean diversity indices indeed vary with altitude and that the indices are relatively highest for medium altitudes between 2,395 and 2,683 m.a.s.l.

The analysis of variance of diversity (H') by considering

Table 3. Phenotypic diversity index (H') for 8 qualitative traits for 43 populations collected from Gamo highlands, Ethiopia.

Trait	Diversity index (H')	
	43 populations	Mean H' ± SE
Kernel row number (KRN)	0.86	0.27 ± 0.03
Spike density (SD)	0.91	0.41 ± 0.06
Lemma awn barbs (LAB)	0.03	0.01 ± 0.01
Glume color (GC)	0.28	0.08 ± 0.02
Lemma type (LT)	0.57	0.20 ± 0.04
Length of rachila hairs (LRH)	0.89	0.40 ± 0.06
Kernel covering (KC)	0.47	0.16 ± 0.04
Lemma color (LC)	0.69	0.30 ± 0.04
Average diversity index	0.59 ± 0.11	0.23 ± 0.05

Table 4. Estimates of the diversity indices (H') for the three districts and mean diversity (H') and the standard error for the overall characters.

District/zone	KRN	SD	LAB	GC	LT	LRH	KC	LC	H' Mean ± S.E
Chencha	0.88	1.00	0.06	0.27	0.58	0.76	0.40	0.75	0.59 ± 0.11
Dita	0.82	0.76	0.00	0.22	0.57	0.87	0.39	0.50	0.52 ± 0.11
Bonke	0.72	0.82	0.00	0.22	0.54	0.97	0.56	0.65	0.56 ± 0.11
Mean of H'	0.81	0.86	0.02	0.24	0.56	0.86	0.45	0.64	0.56 ± 0.02
Gamo Goffa zone	0.86	0.91	0.03	0.28	0.57	0.89	0.47	0.69	0.59 ± 0.11

Table 5. Analysis of variance for districts of origin using 8 qualitative traits.

Traits	Among districts			Within districts			F	Sig.
	SSa	DF	MSa	SSw	DF	MSw		
KRN	0.522	2	0.261	2.223	40	0.056	4.699	0.150
SD	0.665	2	0.333	5.655	40	0.141	2.353	0.108
LAB	0.008	2	0.004	0.198	40	0.005	0.837	0.440
GC	0.053	2	0.026	0.923	40	0.023	1.139	0.330
LT	0.074	2	0.037	2.237	40	0.056	0.660	0.522
LRH	0.808	2	0.404	5.943	40	0.149	2.718	0.078
KC	0.080	2	0.040	2.293	40	0.057	0.701	0.502
LC	0.298	2	0.149	2.425	40	0.061	2.458	0.098

SSa: Sum of squares among, DF: degree of freedom, MSa: mean squares among, SSw: sum squares within, MSw: mean squares within districts.

individual traits were also performed and much of the variation was due to variations within altitude rather than among altitudes. However, length of rachila hair showed significant variation among altitude classes ($P < 0.05$) (Table 7).

Differentiation comparison between districts and altitudes

The divergence estimates between the districts, altitude

classes and populations are given in Table 8. When considering the genetic diversity differences between the three districts, an overall medium differentiation level (13.0%) was found; the differentiation was significantly different from zero ($p < 0.001$). The differentiation between districts was significant for six of the eight traits considered (kernel row number, spike density, lemma awn barb, length of rachilla hair, kernel covering and lemma color; 75%) with values ranging from 3.4% (lemma

Table 6. Estimates of the diversity indices (H') for the three altitude classes, eight characters and their standard errors.

Altitude (m.a.s.l.)	KRN	SD	LAB	GC	LT	LRH	KC	LC	H' mean \pm SE
2000-2395	0.64	1.00	0.00	0.08	0.67	0.91	0.37	0.53	0.52 \pm 0.13
2395-2683	0.85	0.75	0.00	0.41	0.48	0.83	0.58	0.76	0.58 \pm 0.10
2683-3000	0.88	0.88	0.06	0.23	0.53	0.92	0.40	0.61	0.56 \pm 0.11
Average H'	0.79	0.88	0.02	0.24	0.56	0.88	0.45	0.63	0.56 \pm 0.11

Table 7. Analysis of variance for altitudinal origin using eight loci.

Trait	Among altitudes			Within altitudes			F	Significance
	SSa	DF	MSa	SSw	DF	MSw		
KRN	0.313	2	0.157	2.432	40	0.061	2.576	0.089
SD	0.637	2	0.319	5.683	40	0.142	2.242	0.119
LAB	0.009	2	0.005	0.197	40	0.005	0.930	0.403
GC	0.035	2	0.018	0.940	40	0.024	0.749	0.479
LT	0.082	2	0.041	2.228	40	0.056	0.740	0.483
LRH	0.970	2	0.485	5.780	40	0.144	3.358*	0.045
KC	0.084	2	0.042	2.290	40	0.057	0.730	0.488
LC	0.288	2	0.144	2.435	40	0.061	2.365	0.107

* $p < 0.05$; SSa: sum of squares among, DF: degree of freedom, MSa: mean squares among, SSw: sum squares within, MSw: mean squares within altitude classes.

Table 8. Percentage (%) of the total morphological variation and significance levels for each spike trait considered.

Level	Percentage (%) of morphological variation for each trait								Mean
	KRN	SD	LAB	GC	LT	LRH	KC	LC	
Districts	39.4***	23.7***	3.4*	3.0	2.9	12***	12.3***	7.5**	13.0***
Altitude classes	19.4***	24.6***	3.8*	27***	16***	2.1	20***	11.5***	15.6***
Populations	49.7***	26.6***	4.8***	47.9***	34.1***	25.9***	37.3***	23.1***	31.2***

*, **, *** significant at $p < 5\%$, 1% , and 0.1% , respectively.

awn barb) to a maximum of 39.36% (kernel row number). Kernel row number and length of rachilla hair were the only traits for which the differentiation value was higher between the districts than among the altitude classes. The level of differentiation between the altitude classes (15.6%; $p < 0.001$) surmount that seen among the three districts. The differentiation among the landrace fields was relatively high (31.2%; $p < 0.001$) as compared to that of among the altitude classes and districts. Thus, it is remarkable that the largest amount of total morphological variance was seen among individuals within populations (68.8%).

Cluster analysis

Cluster analysis was used to examine the aggregation patterns for all 43 barley landrace populations (Figure 3).

However, four major groups of populations accounted for 38 of the 43 landrace populations (88%). These four clusters have different compositions in terms of districts and altitude classes (Table 9).

The number of landrace populations per cluster varied from 15 landraces in cluster II to 6 landraces in cluster I and IV. About 23% of landraces from Bonke grouped in cluster I and all of them have predominance of lemma teeth and covered grains since the majority of the landraces were from altitude class I. Furthermore, cluster I was the only cluster without landraces from altitude class II. Cluster II included landraces from all districts with the highest percentage from Chenchu (53%) which is a district contributing the highest percentage of landraces collected from altitude classes I, II and III and most of them have brown glume colors, lax and dense spike types, and

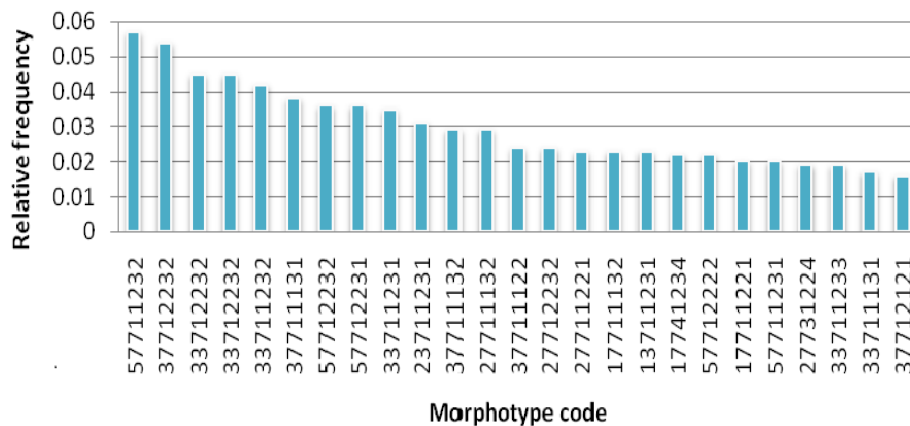


Figure 2. Frequencies of the 25 most frequent morphotypes.

Table 9. Distribution of 38 barley landraces over four clusters by three districts and altitude classes.

Districts/altitude classes	Number of landraces by clusters				Total
	I	II	III	IV	
Chencha	2	8	4	1	15
Dita	1	3	5	1	10
Bonke	3	4	2	4	13
Total	6	15	11	6	38
Altitude classes					
I	4	4	2	2	12
II	—	5	2	3	10
III	2	6	7	1	16
Total	6	15	11	6	38

covered grains. About 50 and 26% of landraces from Dita and Chencha, respectively grouped in cluster III, and all of them have tan lemma colors; white, black and yellow, glume colors; six rowed, long awns on lateral florets; two rowed types, since the majority of the landraces were from Dita district. Landraces with black glume colors clustered under cluster IV in which Bonke contributed the highest percentage (31%). Furthermore, most of the landraces included in cluster IV were two rowed, large or small sterile lateral florets; long rachilla; amber lemma colors and lemma teeth types, since the majority of them are from Bonke district

DISCUSSION

Population diversity

Shannon diversity index estimated from all data pulled

together varied from 0.03 to 0.91 with an overall mean of 0.59 (Table 3). This implied the existence of considerable variability in the studied barley population. There was also variability in barley population from studies of 51 Ethiopian barley accessions and it was found that the Shannon diversity varied from 0.00 to 0.62 for barley populations, and the diversity index estimated for an overall population ranged from 0.29 to 0.92 with an overall mean of 0.71 (Demissie and Bjornstad, 1996). This data implies that over all, Ethiopia has similar barley genetic diversity taking only one zone Gamo goffa as compared to the whole country.

Diversity based on districts

This study showed morphological variation for districts and altitude classes based on qualitative characters, which indicated that, the structure of morphological variation in Ethiopian barley landraces was influenced mostly by natural selection factors. So the degree of variation for characters differed with districts and altitudes from where the landrace populations originated. The presence of high level of phenotypic diversity in Ethiopian barleys was also reported by different authors (Negassa, 1985b; Vavilov, 1926; Asfaw, 1989; Engels, 1994; Tolbert et al., 1979). However, the estimate presented by Tolbert et al. (1979) is fairly low (mean $H' = 0.51$) when compared with Negassa's (mean $H' = 0.68$), Engels' (mean $H' = 0.70$), and the present study, the average diversity index of 0.59 was recorded for Gamo gofa zone alone (Table 4). This inconsistency might be due to inadequate sample size and/or dissimilar and different numbers of characters analyzed by others (Negassa, 1985b; Engels, 1994). The diversity index value in the present study is significantly different

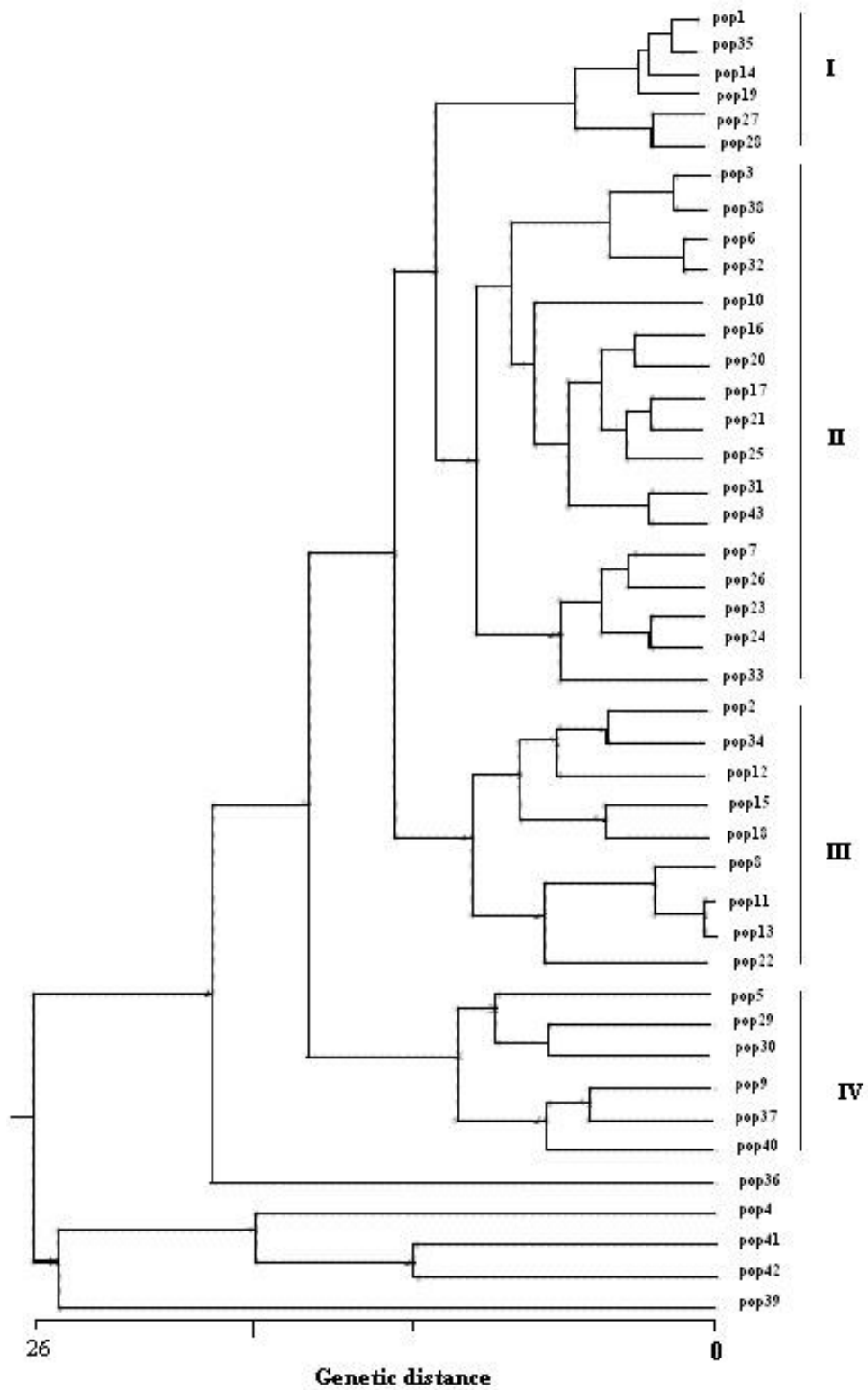


Figure 3. Unweighted pair group method of arithmetic mean (UPGMA) dendrogram based on Nei (1973) genetic distance, showing the relationships among all barley landrace populations.

from the highest diversity value, 0.70, for Ethiopia as reported by Engels (1994). This difference in average H' estimates observed by Engels (1994) and the present study could be due to: 1) the different number of populations studied; 2) the different sampling procedures and strategies; or 3) the different types of characters used for analysis.

The pooled indices over characters within districts are relatively different. Dita had the lowest values which might be explained by the possibly greater degree of selection pressure exerted in the extremely marginal production conditions in Gamo gofa zone as compared to the other two districts, for instance the lowest rainfall and high temperature availability in Dita district. Another possible factor could be the relatively less number of landrace populations collected from this district based on the availability of landraces. However, Chench district showed highest average diversity index value as compared to the other two districts. This is probably due to the presence of high rainfall and/or low temperature which is important for barley in such types of climatic conditions.

Phenotypic diversity indices over altitudes

The observed variations were mainly due to traits within populations and altitude classes. The estimates of H' for each of the characters and the altitude classes as well as the mean H' per altitude class are presented in Table 6. Despite different levels of contributions of the traits to variation, the average H' tends to decrease for samples above 2683 m.a.s.l. (Table 6). The highest polymorphism is concentrated in areas between 2395 to 2683 m.a.s.l. This range includes the major barley-growing areas in the Gamo gofa zone. These phenomena, taken together, are indicative of high genetic diversity and abundance of barley in the medium altitude class. Our results are in agreement with those reported by Engels (1994) that a maximum diversity occurred in a medium altitudinal class. In the present study, "glume color", "kernel covering" and "lemma color" showed such pattern of differences. For these characters, the phenotypic expression does depend on altitude class. The result indicated that for these characters genetic diversity increased with an increasing altitude and then decreased with decreasing altitudes, as a result of natural selection at the lower and upper extreme conditions. Or in other words, the diversity indices decreased towards lower as well as higher altitudes.

Differentiation comparison between altitudes and districts

Analysis of variance was conducted for districts, altitude classes and populations with the assumption that there

existed high variation between populations, followed by altitude classes and the districts. The result showed high variation between populations. Percentage (%) of morphological variation, which described the level of heterozygosity was low between districts (13%) and altitude classes (15%) and there was high genetic differentiation among populations (31%). Generally, high differentiation within population resulted in less gene flow among population but high gene flow within population due to some percent of cross pollination in the species coupled with planting of mixed barley populations by farmers.

A comparison of the present work with an earlier study (Tanto et al., 2009) showed a similar pattern of differentiation value among the districts, altitude classes and populations, but differentiation among populations (31%) is relatively higher as compared to differentiation among populations (25%) as studied by Tanto et al. (2009). This was probably due to sample and population size differences in the two cases.

Hierarchical clustering

The differentiation among the populations can be clustered into different coherent groups depending upon the origin/district or altitudinal classes. In the present study, the differentiation among population appeared to be weak on the basis of district/origin. Similarly, others like Demissie et al. (1998), Ould Med Mahmouda and Hamza (2009) and Tiegist et al. (2010) reported lack of geographical differentiation which failed to indicate clear pattern of division among barley accessions based on geographic origin. These results may reflect the impact of the seed exchange between farmers in small geographic areas which is likely to limit highlighting favorable genes due to local adaptation.

Grouping accessions into morphologically similar cluster of different groups is useful for selecting parents for crossing. However, clustering of collections based on the morphological traits under study revealed no distinct/origin grouping patterns because same or adjacent districts appeared in different clusters.

Conclusion

This study, presented the results of a morphological characterization of a collection of barley from Gamo highlands of Ethiopia.

The overall diversity index of Gamo highlands of Ethiopia barley collections used in this study and similar results from other studies made earlier, support the conclusion that Ethiopia is an important centre of genetic diversity for barley. This diversity is not evenly distributed over the

barley producing districts in the study area. However, there was decrease in diversity from the medium altitude towards the lower and higher altitudes. Therefore, morphological traits which are under direct influence of both human and natural selections are strongly associated with altitude. This implies that to capture the most diverse genotypes one should concentrate on the medium altitudes between 2,395 and 2,683 m.a.s.l., which correspond with the best growing conditions for barley. The concentration of some morphological traits at high or low altitudes and in different sites could result from farmers selection activity based on their selection criteria to the prevailing climatic and edaphic conditions, and because of presume stronger natural selection pressure towards the extremes of the altitudinal distribution range; for instance one might expect certain desirable genotypes for abiotic stress tolerance, such as frost or drought resistance to be found at higher frequencies in specific areas.

Although genetic differentiation was less both on districts and altitudinal classes, altitudinal differentiation was relatively greater than districts. Therefore, altitude difference was relatively more discriminative than origin / districts. This is probably due to the presence of low seed flow among altitude classes as compared to among districts and due to natural selection and adaptation to specific altitudinal classes.

Recommendation

The above conclusions were derived from results of studies conducted by using morphological traits. Therefore, the following recommendations are forwarded for future work. First to make this study more comprehensive, molecular marker techniques should be applied to confirm the morphological results obtained; second to study quantitative traits under field evaluation is required after seed increase by planting the genotypes on ear-to-row. Third using morphological, molecular and field evaluation, one can be able to reach the level of variety development. The conservation of such locally common variation is important, since it may represent genotypes adapted to specific environments. Thus, the patterns of variation described in this study may be useful for researchers in designing studies on barley germplasm elsewhere.

Conflict of Interests

The author(s) have not declared any conflict of interests.

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Full Length Research Paper

The state of biodiversity in Ghana: Knowledge gaps and prioritization

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Biodiversity conservation in tropical countries is of great importance due to high levels of endemism. Over the past century, Ghana has reserved over 300 ecologically important areas for biodiversity conservation, and a national strategy for this purpose has been developed under the framework of the Convention on Biological Diversity. However, inadequate relevant information has been a drawback to implementation. This paper synthesizes relevant occurrence records of species, together with recently published data, and provides a current overview of the country's biodiversity. A map of the distribution of biodiversity study sites in recent years, the taxa studied, total richness of each taxonomic group (except microbes) and list of the species, are presented. The results indicate that in addition to insufficient off-reserve biodiversity knowledge, its acquisition over the past two decades covers only less than 40% of the protected reserves. It is argued that, with the current low-level of protection in protected areas, it is necessary to prioritize agro-ecological studies in order to obtain the baseline data needed for developing effective land-use strategies, as well as the right information to attract the voluntary participation of the public in biodiversity conservation.

Key words: Ghana, biodiversity, conservation, species richness, off-reserve, protected areas, land-use.

INTRODUCTION

Biodiversity loss and conservation have been a global focus for at least two decades, mainly addressing issues of prioritization for efficient fund allocation (Myers et al., 2000; Mace et al., 2000; O'Connor et al., 2003; Brooks et al., 2006). At the national and local levels where all biodiversity driving forces converge, and where conservation needs to be implemented, prioritization is often biased by parochialism due to poor data availability (Hunter and Hutchinson, 1994; da Fonseca et al., 2000). Over the past century, different human activities especially in agriculture have degraded Ghana's biological resources

significantly. Exactly 80 years ago, 63% of the country's forests were in pristine or near-pristine condition in the forest zone (Dickson, 1969). Today, the landscape is mostly human-dominated with forest patches covering ~15% of the country's land area. This is mainly due to land conversion to agriculture, a phenomenon which may continue to biodiversity losses until the economy grows and becomes less dependent on agriculture (Beier et al., 2002). This trend can however be reversed, or at least stabilized, under land-use management regimes in which crop production is maximized with no significant losses to

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biodiversity (Tscharnkte et al., 2012). Essentially, every efficient management plan is driven by quality agro-ecological data, and should gravitate towards increasing ecological complexity through the cultivation of multiple resource-efficient crop varieties under enhanced fertilization while allowing some less-competitive native flora (Kumaraswamy and Kunte, 2013).

Like other African countries (da Fonseca et al., 2000), Ghana lacks location and landscape-scale ecological data, a setback to biodiversity conservation. The currently available biodiversity information has been described as scanty with inaccuracies and gaps, and lacking up-to-date knowledge (Ministry of Environment and Science, 2002; Ghana Report, 2009). Fortunately, a large collection of species occurrence data exists in different museums and online databases (see sources of information). This can be synthesized, together with data and results from recent studies, to obtain the baseline information needed to promote the development of new, effective strategies necessary for the conservation and sustainable use of the country's biological resources (da Fonseca et al., 2000). The objectives of this study were to: (i) compile relevant occurrence records of species, and together with recently published data, synthesize them to provide a current view of biodiversity, (ii) map the distribution of protected and non-protected areas previously studied, along with their corresponding studied taxa, for directing future studies, and (iii) identifying and discussing existing knowledge gaps.

MATERIALS AND METHODS

Study area

Ghana is located in West Africa (4.7° - 11.2°N, 3.3°W - 1.2°E), with its southern part lying on the coast of the northeast Atlantic Ocean (Gulf of Guinea) (Figure 1). To the north is Burkina Faso, to the west is Côte d'Ivoire and to the east is Togo. The country extends over two unique ecosystem zones; Eastern Guinean Forests and West Sudanian Savanna, with a transitional Guinean Forest-Savanna Mosaic ecosystem zone. It encompasses a total land area of 238,761 km², and has an annual bimodal rainfall distribution between 900 and 2,100 mm (Ntiemoa-Baidu et al., 2001). The current population is 24,658,823 (2010 estimate) with an annual population growth rate of 2.5% (Ghana Statistical Service, 2013). Divided into 10 regions, the country has ~15% of land under protection, the largest of which is in the Western Region where 32.3% of land is protected (Table 2 in Supplementary Information – Tables 1-3).

Sources of information

Species data totaling 296 (Table 1) was obtained from 73 peer-reviewed literature and several databases. The SCI-Web of Science was consulted for data on Ghana's biodiversity in November 2013. A search for Ghana species occurrence data was also conducted in large species databases like the Global Biodiversity Information Facility (GBIF) on Ghana, comprising 90 different organizations, AVIBASE (Lepage 2014), African Bird Club (Dowsett et al., 2014), Amphibia Web (University of California,

Berkeley), The Reptile Database (Uetz and Hošek 2014), African Butterfly Database (Sáfián et al., 2009), Afromoth (De Prins and De Prins, 2014), Orthoptera Species File (Eades et al., 2014) and the IUCN's Red List of Threatened Species. Finally, web pages of researchers working on different aspects of the country's biodiversity were visited for published literature containing additional information. Diversity data (species richness) and study locations were recorded for six taxonomic groups (birds, insects, mammals, amphibians, reptiles and plants) in three habitat systems: protected areas (forest and wildlife reserves, etc.), non-protected areas and sacred groves (Table 1; Table 1 in Supplementary Information – Tables 1-3) (Aalangdong, 2009; Ryan and Attuquayefio, 2000). Each article included in this review reported species richness of particular taxa for at least one site. In cases where different articles on the same location reported different species richness values for the same taxa, the maximum was used. Species lists were compiled from the records obtained from the databases and literature.

Data analysis

Box and whisker plotting were used to indicate differences in species richness across different habitat types for each taxonomic group. In addition, a combined map of the spatial distribution of sites where recent (1994 – 2013) taxonomic studies have been undertaken was generated for visual assessment of the present knowledge and possible gaps.

RESULTS

Birds

A total of 794 bird species belonging to 101 families have been recorded, of which 494 (65%) are resident, and 16 species (14 of which are resident) are globally endangered (Table 2; Supplementary Information - Birds). The Mole National Park, probably due to its large size, has the highest bird species richness of 314 (Ntiemoa-Baidu et al., 2001), while the Afrensu Brohuma Forest Reserve has the lowest richness of 44 species (Manu 2011). Table 2 shows the regional distribution of bird species in Ghana. A summary of the distribution of bird species richness for protected and non-protected areas of the country is presented graphically in Figure 2.

Studies on birds, like other species, have concentrated more on the protected areas (87.5%), with little attention to non-protected areas (11.25%) and sacred groves (1.25%) (Table 1). The Brong-Ahafo Region, with only one Important Bird Area (IBA) of ~1,830 km² has the highest bird species richness (533), while the Western Region with 18 of the 36 IBAs (total area ~3,850 km²) in the country, has a richness of 495 (Ntiemoa-Baidu et al., 2001; Table 2). This is surprising considering the fact that the Western Region occupies the largest portion of the richest biodiversity area of the country (Myers et al., 2000; Ministry of Environment and Science, 2002). Beier et al. (2002) gives strong evidence of a positive correlation between habitat size and species richness of birds, which offers some explanation on the basis of island biogeography. Moreover, the low diversity of birds in the

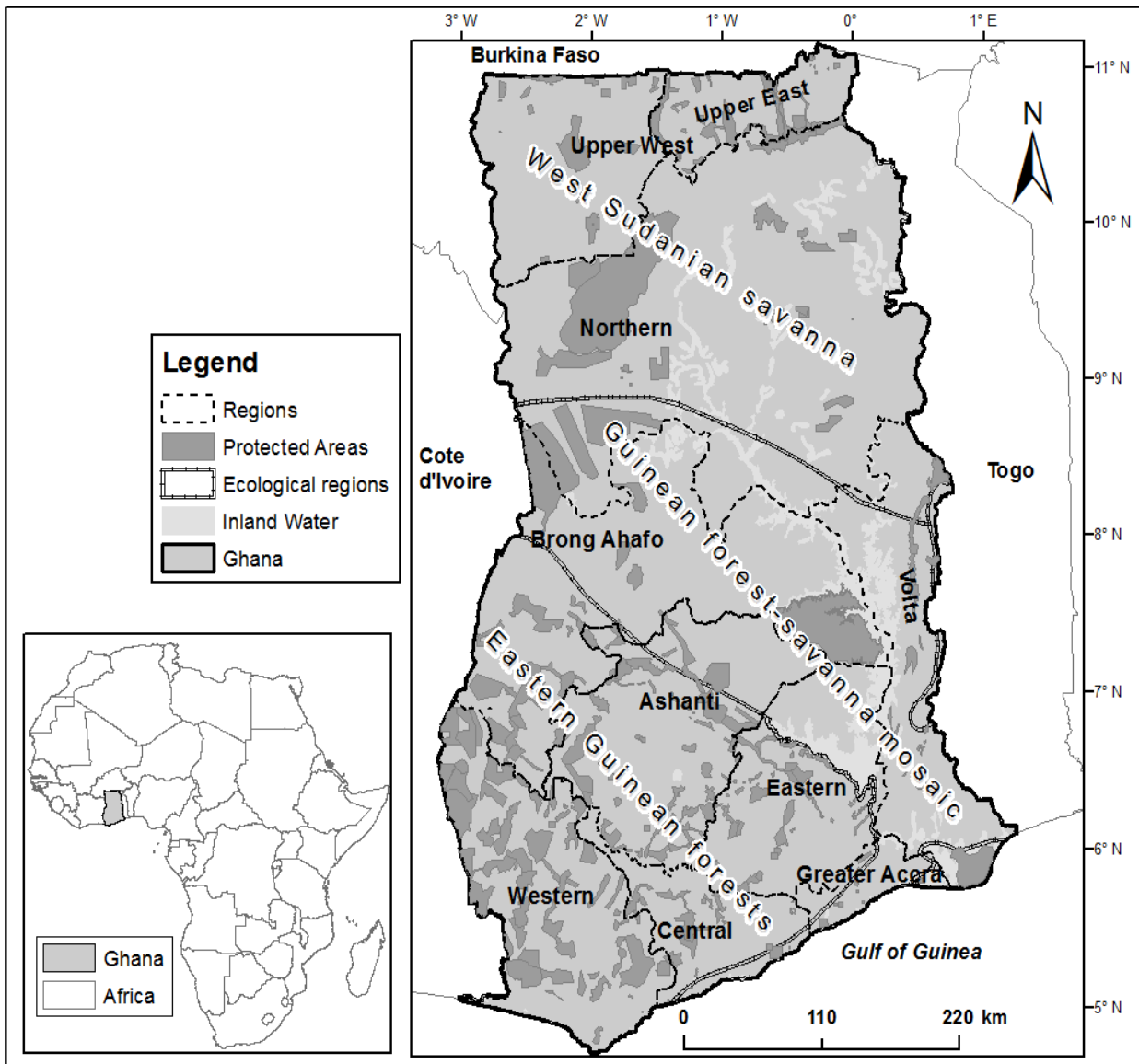


Figure 1. The map of Ghana.

Table 1 Summary of dataset distribution from literature for different taxa groups and habitat types in Ghana.

Taxa	Habitat type			Total	Percentage occurrence
	Protected	Non-Protected	Sacred groves		
Birds	70	9	1	80	27.0
Butterflies	42	0	7	49	16.6
Small mammals	27	12	4	43	14.5
Amphibians	25	12	0	37	12.5
Plants	22	11	0	33	11.1
Large mammals	25	0	0	25	8.4
Other insects	11	6	0	17	5.7
Reptiles	9	3	0	12	4.1
Total	231	53	12	296	
Proportions	78.0%	17.9%	4.1%		

Table 2. Summary of bird species per region.

Region	Richness	Proportion (%)	Rare Species	IBAs	GE	IN
Brong-Ahafo	533	70.4	11	1	10	2
Central	528	69.7	13	3	10	2
Eastern	528	69.7	10	2	7	2
Volta	526	69.5	16	2	5	3
Greater Accra	518	68.4	19	4	6	4
Ashanti	515	68.4	10	2	8	2
Western	495	65.4	12	18	9	2
Northern	472	62.4	15	2	8	0
Upper East	390	51.5	18	2	6	1
Upper West	375	49.5	13	0	5	2
Total	794*	100	-	36	16	-

Data source – Lepage (2014). GE- Globally endangered species, IN- Introduced species, IBAs - Important bird areas. *additional information used.

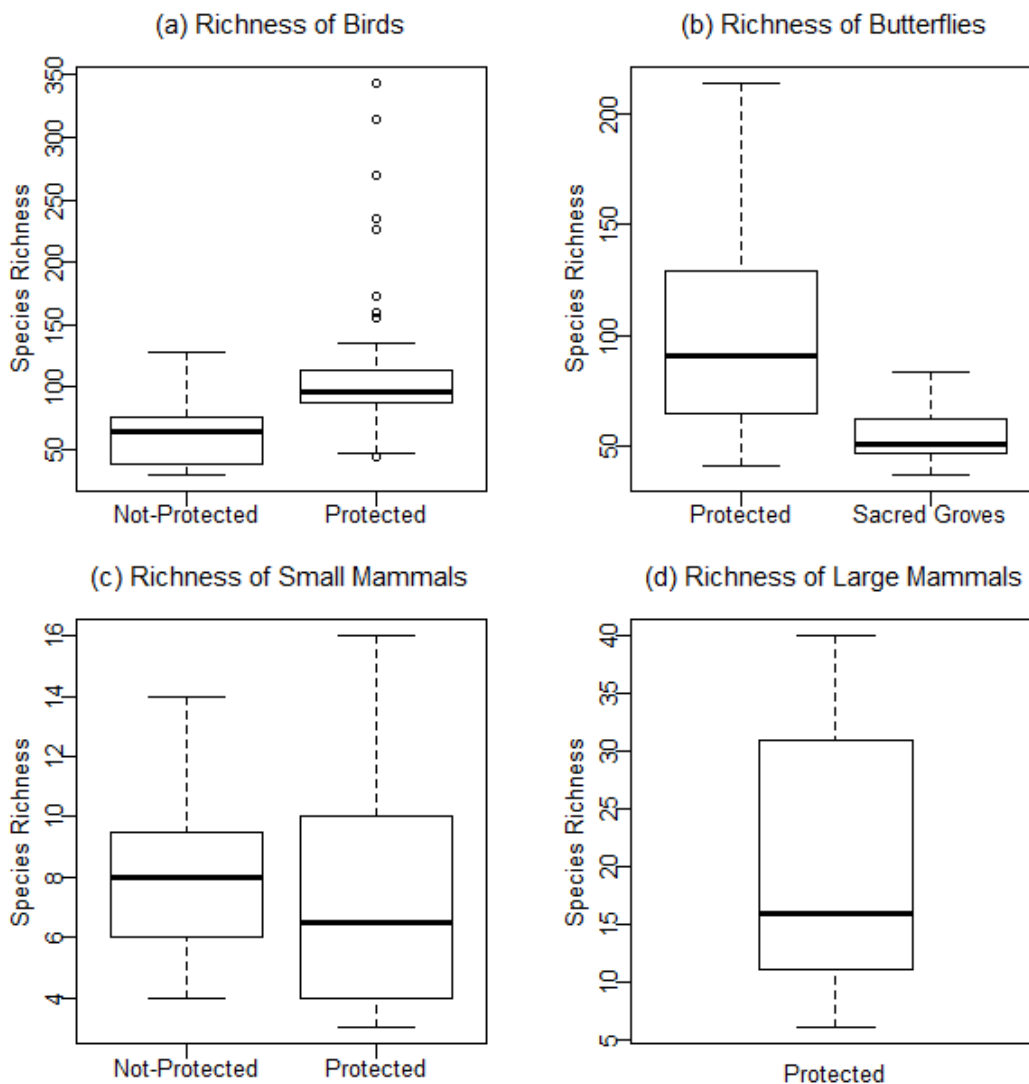


Figure 2. Richness of birds, butterflies, small and large mammals.

Table 3. Butterfly species richness and distribution according to rainfall and vegetation cover, produced based on estimated proportions in Larsen (2006). The proportions were calculated using the total butterfly richness (983) of the country.

Vegetation/Land-use	Annual rainfall (mm)	Richness (estimated)	Proportion
Forest			
Wet evergreen	1 900 – 2 100	718	0.73
Upland evergreen	1 700 – 1 800	688	0.70
Moist evergreen	1 600 – 1 700	629	0.64
Dry	1 300 – 1 600	511	0.52
Lightly logged wet evergreen	1 900 – 2 100	688	0.70
Secondary growth evergreen	1 600 – 2 100	462	0.47
Wholly-cleared evergreen	1 600 – 2 100	88	0.09
Savanna			
Guinea savanna	1 000 – 1 300	197	0.20
Sudan savanna	800 – 1 000	118	0.12
Wholly-cleared savanna	800 – 1 300	79	0.08

Western Region could suggest that, despite birds being the best studied taxa in terms of spatial coverage, the present knowledge on their diversity, distribution and abundance is still insufficient (Weckstein et al., 2009).

Despite the small sizes of the sacred groves, they may conserve small populations of forest-specialist without local extinction threats because, unlike butterflies (Larsen et al., 2009), birds from nearby source habitats can recolonize them. Attuquayefio (2008) found no species common to all five forest reserves in the Brong-Ahafo Region. This may suggest that (i) each forest-specialist bird species has a strict set of habitat requirements, and (ii) those forest reserves have different ecological conditions probably related to the different degrees of degradation within them. Details of the distribution of forest and savanna restricted species in the IBAs is given elsewhere (Ntiama-Baidu et al., 2001).

Butterflies and other insects

Out of a total of 983 butterfly species belonging to six families – Lycaenidae (349), Nymphalidae (336), Hesperidae (216), Pieridae (53), Papilionidae (26) and Riodinidae (3) (Supplementary Information – Insects), 33 are endemic to the Mount Afadjato area of the Volta Region, Atewa Range Forest Reserve and the wet forests of the area extending from western Ghana to eastern Côte d'Ivoire (Larsen, 1994, 2006; Larsen et al., 2009). There are 1,328 moth species belonging to 35 families with one (*Zekelita shamssia*) being endemic.

The Atewa Range Forest Reserve has recorded the highest cumulative butterfly richness of 575 (Aduse-Poku and Doku-Marfo, 2007), followed in decreasing order by Kakum Conservation Area (496 species) (Larsen et al., 2009), Bobiri Wildlife Sanctuary (454 species) (Larsen et

al., 2009), Bia Biosphere Reserve (453 species) (Aduse-Poku et al., 2012), Kyabobo National Park (401 species) (Larsen, 2006), Ankasa, Wli Falls area, Boabeng-Fiema and Aburi Botanical Gardens (390, 328, 288 and 267 species, respectively) (Larsen et al., 2009). A summary of the rest of the data, mostly from single, short-period assessments of different sites, is presented in Figure 2.

Using the habitat distribution ratios obtained from Larsen (2006), the number of forest-centered, savanna-centered and ubiquitous butterfly species are estimated at 798, 177 and 39, respectively. Table 3 shows the distribution of butterflies in different vegetation types and rainfall bands. About 20% of the butterfly species, including 16 endemics, are located within the human dominated areas and have not yet been recorded in the protected areas (Larsen, 2006). These species, unless they are ubiquitous, need conservation attention regardless of their endemic status. This is because land-use activities such as bush burning and insecticide application can potentially cause local extinctions due to their low mobility, especially if they are restricted to small geographic areas. For example, even though rare, all the three species (*Abisara intermedia*, *Abisara tantalus* and *Abisara gerontes*) of the Riodinidae, were recorded only in non-protected areas (Larsen, 2006).

More than three-quarters of Ghana's butterfly fauna is located within the forest zone, which stretches from the southwest to the mid-regions (Larsen, 2006). Concentration is unusually high in the southwest wet forests in the Western Region, upland moist forests within the Atewa-Range Forest Reserve and the forest areas of the Volta region (Emmel and Larsen, 1997). For example, even though the Atewa Range Forest Reserve covers only an area of ~232 km², it is home to ~580 species of butterflies with a projected estimate of over 700 species (Aduse-Poku and Doku-Marfo, 2007), which is more than

Table 4. Summary of species richness of invertebrates (excluding Lepidoptera (2,311 species)).

Order	Total richness
Hymenoptera	404*
Coleoptera	382 [#]
Diptera	128
Orthoptera	73
Mantodea	45
Hemiptera	34
Dermaptera	16
Neuroptera	12
Blattodea	8
Total	1,102

[#]Source: Browne (1963); *Source: Kolo and Hormenyo (2009).

the butterfly species richness of Western Europe, and more than the butterfly richness of any state in the USA (Larsen, 2005). This suggests that land-use activities that promote the encroachment of savanna vegetation into the forest zone can potentially cause the loss of many savanna-hostile butterfly species. Both vegetation and rainfall are known to affect butterfly richness with an approximately linear relationship (Larsen, 1994). Thus, any long-term perturbations, distributional shifts, and declines in rainfall may lead to loss of biodiversity and rapid turn-over of species (Dornelas et al. 2014).

Butterflies are the best studied fauna in the country's sacred groves, with at least 7 sacred groves studied. However, due to Larsen's (2006) estimated habitat area requirement of 200 km² for butterfly conservation, the continued local survival of some forest-specialists is doubtful in sacred groves (and even in some forest reserves) because of their small areas. This follows directly from the observation of Larsen et al. (2009) that local extinction of butterfly species in Boabeng-Fiema, an isolated reserve of ~ 4.4 km² area, cannot be reversed because there is no opportunity for continued gene-flow through re-colonization from nearby subpopulations, a process which is important for the continued local survival of butterflies than the surface areas of the habitats they occupy.

The total species richness of non-lepidopteran arthropods recorded under nine taxonomic orders is 1,102 (Table 4). The species richness of arthropods belonging to the Coleoptera (382 species from 130 genera) was based on Browne's (1963) estimate. However, 118 species belonging to the Scolytidae and Platypodidae which Browne omitted from his paper are not presented in the species list (Supplementary Information – Insects). Similarly, for Hymenoptera, only 221 of the species richness of 404 reported by Kolo and Hormenyo (2009) are listed. Recent studies (Davis and Philips, 2005;

Lachat et al., 2006; Norris et al., 2010; Kolo et al., 2011) have indicated that human land-use has negatively affected non-lepidopteran arthropods in West Africa. Larsen (2005), however, acknowledges that the population of savanna and adventive species has increased significantly within the forest zone, and Belshaw and Bolton (1993) believe that the change may not be significant.

Mammals

Currently, 327 species of terrestrial mammal have been recorded, of which four are endangered, eight are vulnerable and 15 are near threatened (IUCN, 2013; Table 5). Studies on mammals have concentrated primarily on small and medium-sized (or large) mammals with a ratio of about 3:1 (Table 1). Small mammals are the best studied in terms of species distribution and abundance on different land-use systems in the country. Small mammal diversity is high in areas that have stable habitats with dense vegetation, sufficient food availability and increased ground cover (Attuquayefio and Wuver, 2003; Ofori et al., 2013). Jeffrey (1977) also reported that the abundance of small mammals is determined by food availability, while richness is determined by availability of dense ground cover.

Diversity of small mammals has been found to be low in most forest reserves but high in built-up areas and other human dominated land-use systems (Jeffrey, 1977; Vordzogbe et al., 2005). Small mammals confronted by unfavorable food conditions in primary forests with sparse ground cover migrate to adjacent human-disturbed habitats with denser ground covers and sufficient food availability (Vordzogbe et al., 2005). Rainfall, being an agent for ground cover densification and food availability, also causes increases in small mammal diversity (Attuquayefio and Wuver, 2003). This explains why, sometimes, the diversity of small mammals is higher in human dominated land-use systems than in protected areas. The above observations suggest that the abundance and diversity of small mammals are less dependent on level of pristineness of habitats, even though other studies have found the opposite (Ofori et al., 2012). The species richness data available from recent single assessments of both small and large mammals in protected and non-protected areas are shown in Figure 2.

At habitat level, small mammal diversity depends on habitat size, distance from a colonization source, and the presence of suitable migration corridors (Decher, 1997). Hunting is the main cause of biodiversity decline in large and medium-sized mammals (Curry-Lindahl, 1969; Asibey, 1974; Vordogbe et al., 2005), while bush burning is the worst threat to small mammals (Decher and Bahian, 1999; Attuquayefio and Wuver, 2003; Vordzogbe et al., 2005).

Table 5. Summary of mammal species in Ghana. The terms near threatened, vulnerable and endangered are as defined by the IUCN.

Order	Richness	Near Threatened	Vulnerable	Endangered
Chiroptera	124	6		
Rodentia	86			1
Carnivora	30	2	3	1
Artiodactyla	29	1	2	
Primates	26	1	3	2
Soricomorpha	18	2		
Pholidota	5	2		
Hyracoidea	4			
Lagomorpha	2			
Erinaceomorpha	1			
Proboscidea	1		1	
Tubulidentata	1			
Total	327	14	9	4

Table 6. Summary of amphibian species in Ghana grouped according to families. NT- Near Threatened, CR- Critically Endangered (*Conraua derooi*), EN- Endangered, VU - Vulnerable (IUCN 2013).

Family	Richness	NT	CR	EN	VU
Hyperoliidae	34	4		2	3
Phrynobatrachidae	21	2		3	1
Arthroleptidae	17	2			
Ptychadenidae	14	1			
Bufonidae	13	1			
Ranidae	11	1	1		
Pipidae	4				
Dicoglossidae	2				
Hemisotidae	2				
Pyxicephalidae	2				
Caeciliidae	1				
Microhylidae	1				
Rhacophoridae	1				
Total	123	11	1	5	4

Herpetofauna

The terrestrial herpetofaunal richness of the country presently stands at 377 species (Supplementary Information – Amphibians; Supplementary Information – Reptiles) comprising a total of 119 species of amphibians belonging to 13 families (including six endemics), and 259 species of reptiles belonging to 18 families (Tables 6 and 7).

Available data on recent, single herpetological assessments of some protected and non-protected areas are presented graphically in Figure 4. Ghana's amphibian

diversity is highest in the Western Region of the Country (Rödel et al., 2005). Before the beginning of the 21st century, the highest amphibian richness recorded for a single site was 20 species (Rödel and Adjei, 2003). In recent years, however, higher values have been recorded, the highest being 40 species for the Ankasa Conservation Area (Rödel et al., 2005). The diversity of amphibians correlates negatively with elevation and density of ground litter (Wiafe and Adjei, 2013). Reptiles have also been studied in recent years, and like amphibians, the studies are few and not comprehensive. So far, protected areas studied include Kyabobo National

Table 7. Summary of reptile families in Ghana.

Family	Richness
Lizards	
Scincidae	42
Gekkonidae	21
Agamidae	10
Chamaeleonidae	7
Lacertidae	7
Gerrhosauridae	2
Varanidae	2
Polychrotidae	1
Snakes	
Colubridae	80
Atractaspididae*	18
Elapidae	14
Lamprophiidae	14
Viperidae	11
Leptotyphlopidae	10
Typhlopidae	6
Boidae	5
Chelonians	
Testudinidae	5
Amphibians	
Amphisbaenidae	4
Total	259

Source: The Reptile Database (2014).

**Aparallactus lineatus*- near threatened (IUCN, 2013).

Park (Leaché et al., 2006), Gyemira and Gyeni River Forest Reserves (Yahaya et al., 2013), and Draw River and Krokosua Hills Forest Reserves (Ernst et al., 2005).

Plants

The current total plant species richness of the country is 5,429, comprising 5,217 angiosperms (1,257 monocots, 3,959 dicots and a single species of magnoliids), 147 pteridophytes (ferns), 46 bryophytes (35 mosses and 11 liverworts), 12 lycophytes (club and spike mosses), and seven gymnosperms (Supplementary Information – Plants). Of these species, 121 are threatened, including three (*Talbotiella gentii*, *Salacia fimbriseipala* and *Aubreggrinia taiensis*) critically endangered, 19 endangered, four near threatened and 95 vulnerable species (IUCN, 2013). Historically, apart from *S. fimbriseipala* which was recorded at only two sites, *T. gentii* and *A. taiensis* appear to have had much wider geographic ranges in the forest zone (Figure 3).

Occurrence records between 1908 and 1974 indicate that *T. gentii*, despite being concentrated near Somanya (Eastern Region) and spread along the north-west direction into the Ashanti Region, was recorded in a few other parts of the country. Specifically, in the years 1952, 1960 and 1970, the species was sparsely recorded at different locations within the Guinea Savanna vegetative zone of the country. Also in 1952, *T. gentii* was recorded near Amisano in the Central Region. By extracting and examining the elevation and slope of each of the 102 sites where *T. gentii* was observed between 1908 and 1999, it has become clear that the species occurred more at lower and medium elevations (<300 m AMSL) than at higher elevations. The species occurred on flat and gentle-sloped terrains because 50% of the records were obtained from slopes from 0 to 7°. Recently, it has been reported that *T. gentii* occurs on high elevations and on rocky slopes (Anyomi et al., 2008), which is possibly due to overexploitation of the species that occurred at lower elevations. Despite earlier reports that the species is geographically restricted to Bandai Hills, Sapawusu and Yongwa Forest Reserves (Anyomi et al., 2008; Boshier et al., 2011), there is a strong indication of its occurrence in the other forest reserves within the area defined by the following coordinates: i. 0.273°E, 6.375°N; ii. 0.022°W, 5.744°N; iii. 1.362°W, 6.434°N; iv. 1.047°W, 7.084°N, especially Dome River, Southern Scarp, Afram Headwaters, Volta River and Bomfoum (Figure 3).

Similarly *A. taiensis*, known to occur only in Ghana and Côte d'Ivoire, appears to occur within the Eastern Guinean Forest block extending from the Sassandra River (Côte d'Ivoire) to the Volta River. In Ghana, its concentration is within the Atewa Range Forest Reserve area, and spreads towards the wet evergreen forest zone in the Western Region. The historical distribution range of *A. taiensis* is slightly different from that of *T. gentii* because it recorded only in the Eastern Guinean Forests ecoregion. The last record of the species was made in the Aiyala Forest Reserve (0.95°W, 6.15°N) in March 1970 which should be a starting point towards establishing its conservation status. The last critically endangered species, *S. fimbriseipala* is believed to be endemic to Ghana and Cameroun. In Ghana, only two records of the species have been filed in the University of Ghana Herbarium; one of which was recorded ~1.25 km, 026° from Anhwem in the Western Region in 1935 while the other (date unknown) was recorded near Ahinsan, ~2.5 km, 031° from Dompouse in the Ashanti Region. Unfortunately, both sites occur in human dominated land-use areas; however, search for the existence and conservation status of the species in the nearby Subuma and Bosomtwi Range Forest Reserves respectively could yield some positive results.

For the protected areas, the average species richness of 22 reserves whose data were recorded in recent studies is 306. Single study records on the richness of groups of plants species are presented in Figure 4. Plant

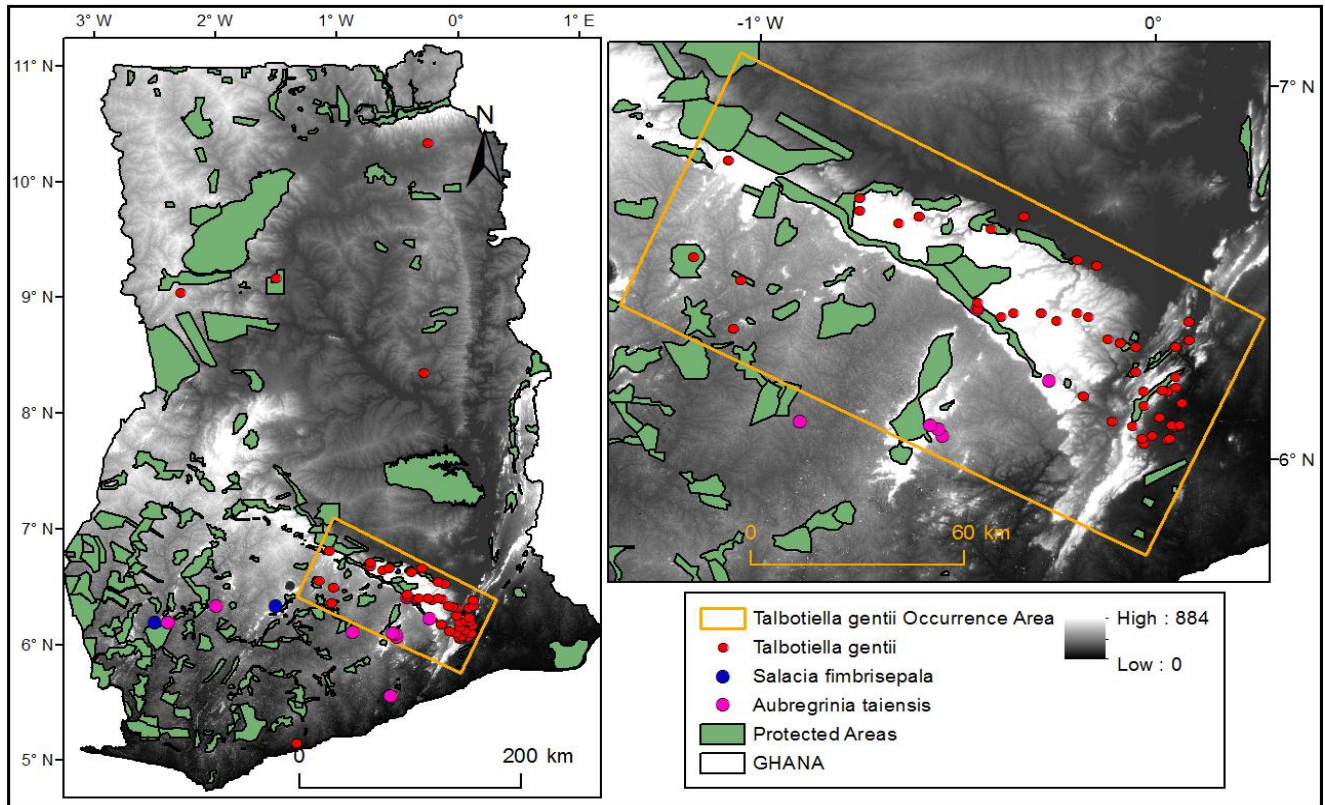


Figure 3. Historical distribution of the critically endangered species *T. gentii*, *S. fimbrisepala* and *A. taiensis*. Spatial distribution of *T. gentii* (an endemic species to Ghana) is highlighted. Data sources: Aster GDEM data (<http://gdem.ersdac.jpacesystems.or.jp/>), protected areas of Ghana (IUCN and UNEP-WCMC 2013) and species occurrence data (GBIF).

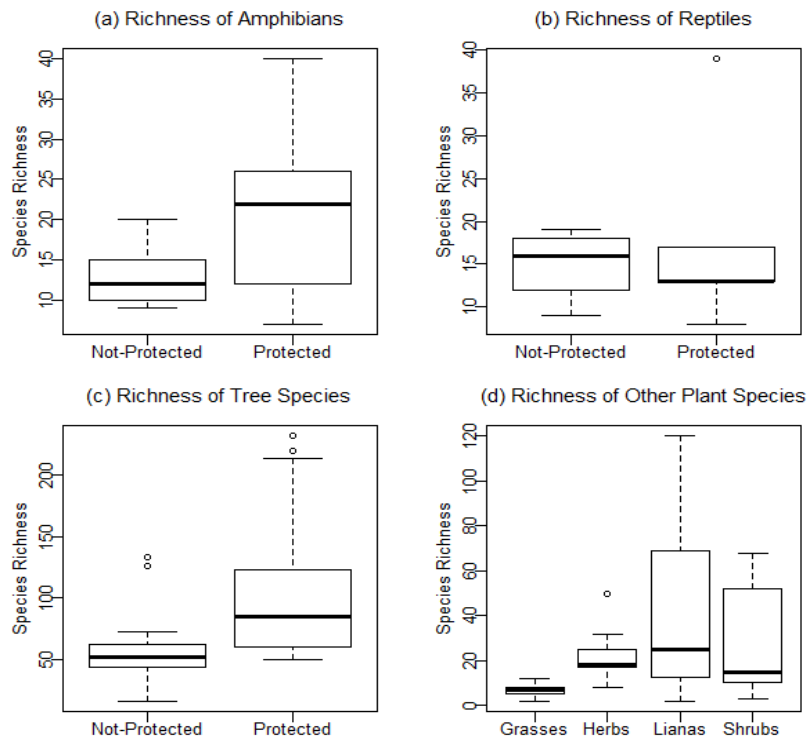


Figure 4. Richness of amphibians, reptiles and plants.

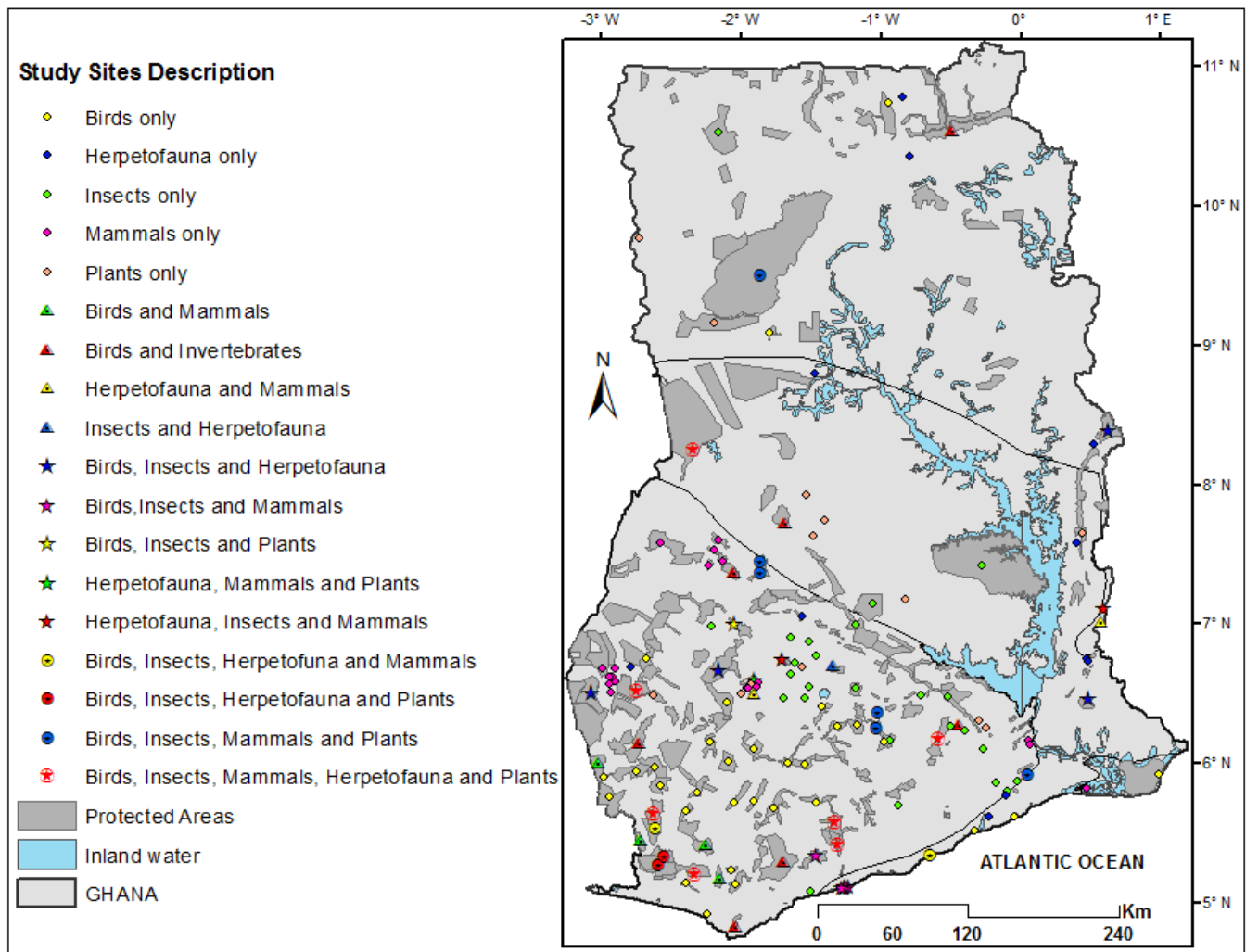


Figure 5. Spatial distribution of sites studied and the taxa of interest.

diversity and distribution in the forest zone have been well documented (Thompson, 1910; Hall and Swaine, 1976) even though the studies were carried out several decades ago. In recent years, studies have declined considerably in all vegetative zones. Tree and liana species richness of human-dominated land-use areas decrease with increasing years of continuous human activity (Asase and Tetteh, 2010; Anyomi et al., 2011; Anglaere et al., 2011; Addo-Fordjour et al., 2012). Secondly, perhaps due to the differences in structure and management, plant species diversity increases from seedling through sapling to tree stages in natural forests, while the reverse is true for agroforestry systems (Addo-Fordjour et al., 2009; Boakye et al., 2012). As a result, undisturbed forests in the country are characterized by high tree diversity, low herb diversity and medium sapling diversity; disturbed (logged) forests are characterized by low tree diversity, high sapling diversity and medium herb diversity; and disturbed-invaded forests are characterized by high herb diversity, low sapling diversity and medium tree diversity

(Fordjour et al., 2009). Thirdly, for highly degraded areas undergoing regeneration, plant diversity transitions in time is from grass-dominating through shrub-dominating to tree-dominating (Campbell, 2005).

Study coverage

The dataset on study sites comprises species richness data (mainly from 1994 – 2013) on single or multiple taxa covering 146 different locations scattered over the country (Figure 5). Of this number, 88 (~60%) sites are protected areas, 48 sites are off-protected (including wetlands) and 10 sites are sacred groves. With the exception of two sites (Muni-Pomadze Ramsar site and Wli/Agumatsa proposed protected area) which have had multiple taxonomic studies on multiple taxa, the remaining non-protected sites have been studied only once for single taxa. Eight out of 10 sacred groves were solely studied for butterflies, with Okyem and Prako

sacred groves being studied for small mammals. While acknowledging that several unpublished studies (for example, student thesis, etc.) may be available, this study reveals that only about 32% of the protected areas have had taxonomic studies over the past two decades. Fifty-four of the protected sites have had single taxa studies (33 birds-only, 11 insects-only, six small mammals-only and four plants-only). Atewa Range, Tano-Nimiri, Krokosua Hills and Draw River Forest Reserves, Bui National Park and Kakum-Assin Attandanso Conservation Area have had studies (multiple for except for herpetofauna) in all taxonomic groups and are thus considered the best known sites.

DISCUSSION

Knowledge gaps

Unlike moths which, despite the availability of large collections of incidence data, has received no systematic study in the country, butterflies in the country have been well-studied (examples are Larsen, 2005; Bossart et al., 2007; Merek and Doku-Marfo, 2009; Larsen et al., 2009; Bossart and Opuni-Frimpong, 2009; Nganso et al., 2012; Aduse-Poku et al., 2012). Butterfly studies have concentrated on only 42 protected areas and seven sacred groves (Table 1). Apart from the rough estimates of butterfly richness for different vegetation types (Table 3), the effects of many anthropogenic land-use systems on the diversity of butterflies still remain unknown.

Despite the new findings available, wide knowledge gaps still exist in all taxonomic groups as previously highlighted in the literature (Ministry of Environment and Science, 2002; Ghana's Report, 2009). In particular, non-lepidopteran insects need to be prioritized in biodiversity studies. These insects, apart from being good indicators of environmental change (Ansah, 2005; Larsen, 2005; Naskrecki, 2007; Kolo et al., 2011), play immense roles in nutrient recycling through wood and leaf litter degradation, soil turnover, dung and carrion disposal, as well as serving as primary food for many animals (Ansah, 2005). However, although a large collection of occurrence data is available in different museum collections outside the country, only a few studies have been conducted on these insects over the last two decades because of taxonomic difficulties (Ansah, 2005; Naskrecki, 2007; Oteng-Yeboah et al., 2009). The fact that approximately 84% of the occurrence records were made more than three decades ago indicates that these arthropods have generally become less popular in recent biodiversity studies. The Coleoptera (largest insect order), for example, has been studied only once over the last decade, and only the Ankasa Conservation Area was covered (that is, Davis and Philips, 2005). Similarly, for Orthoptera, the only recent studies are that of Naskrecki (2007, 2009) which recorded 61, 50 and 33 species of

Tettigonioidea from Atewa Range, Ajenjua Bepo and Mamang River Forest Reserves, respectively.

The order Hymenoptera has also had only two recent studies. Belshaw and Bolton (1994) recorded 197 species from 20 different sites with varying vegetation types throughout the forest and transition zones of the country, while Kolo and Hormenyo (2009) recorded 125 species comprising of 98 species from Ajenjua Bepo and 101 species from Mamang River forest reserves. In addition, Deikumah and Kudom (2010) sampled over 15000 individuals from forest patches in Cape Coast and classified them into 52 families, while Ansah (2005) recorded 60, 54 and 49 species from Boin-Tano, Krokosua Hills and Draw River forest reserves, respectively. The other taxonomic orders have not been studied in recent years.

Reptiles and amphibians have also recorded mostly single studies in very few areas of the country and thus need to be given more attention in future studies. Plant diversity, though considered well-known on the basis on extensive work done several decades ago, needs to be studied with particular attention to the savanna zone which is poorly known (Ayensu et al., 1996). With the exception of Asase and Oteng-Yeboah (2007) and Tom-Dery et al. (2012; 2013), no other recent, relevant studies on plant diversity in the savanna zone are available. Already, some studies have looked at the plant conservation roles of land-use systems such as mixed crop and cocoa agroforestry by comparing species richness of plants in these systems with species richness of plants in adjacent protected areas (examples are Asase and Tetteh, 2010; Anglaaere et al., 2011; Boakye et al., 2012). However, despite being limited to only few land-use systems, the proximity of these agroforestry systems to the protected areas enhances easy colonization of native species, and hence their outcomes may have been biased towards agroforestry systems since species richness of most widespread species increases significantly in disturbed land-use systems (Waltert et al., 2011).

Birds are also well-known, but unlike plants, knowledge regarding their diversities and distribution over human-dominated land-use systems is almost non-existent. Having lost virtually all forests outside the protected areas which, for centuries past, supported the traditional shifting cultivation system of farming, efforts towards documenting and managing biodiversity in off-reserve areas for continued provision of goods and services for human well-being has become necessary. This knowledge gap is also stated in Ghana Report (2009) as follows: "It is hoped that the other areas, such as in off-reserves where no management regimes or plans are in place, will receive adequate attention to become good sources for national indication of successful implementation of the convention. This achievement will clearly show how biodiversity has been mainstreamed into the society". At present, biodiversity knowledge in off-reserve

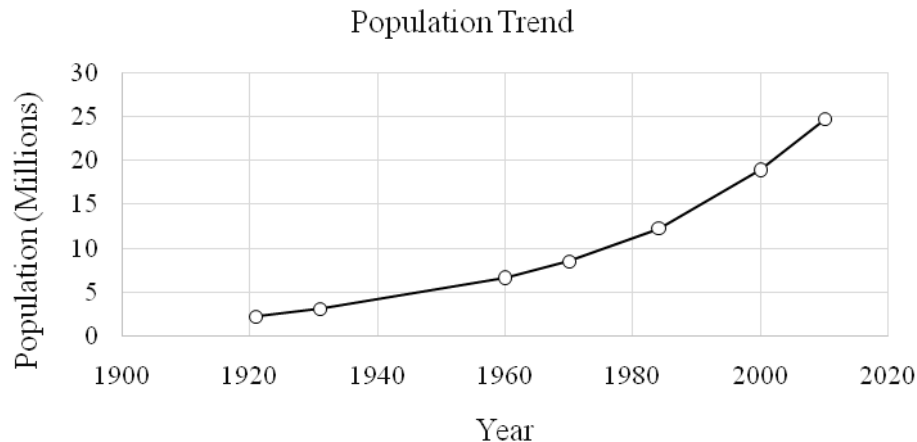


Figure 6. Ghana's population trend 1921–2013. Data Source: Ghana Statistical Service (2013).

Table 8. Land-use share of Ghana's land area.

Land-use	Share of area (km^2)	Cover
Protected Areas	35 814.14 [#]	15.0%
Area under cultivation	155 000.00 ⁺	64.9%
Inland water area	11 000.00 [§]	4.6%
Others (Fallow, pastures, sacred groves, etc.)	36 946.64	15.5%
Total	238 760.78 ⁺	100.0%

Sources: [^]CountryStat Ghana (2014); [#]IUCN and UNEP-WCMC (2013); [§]Ministry of Food and Agriculture (2011); ⁺Maplibrary (2013).

areas is too poor and needs attention.

Drivers of biodiversity loss

Ghana is no exception to the present global crisis of biodiversity loss (Myers et al., 2000; Brook et al., 2006). First of all, the population of the country has been increasing (Figure 6) and this has negative effects on biodiversity (Sarkar, 1999; Luck, 2007; Cincotta et al., 2000; McKee et al., 2003; Laurance et al., 2012; Cai and Pettenella, 2013; Bamford et al., 2014). Between 1960 and 2010, for example, the population of the country more than tripled with an average annual population growth between 1921 and 2010 of 3.3% (Ghana Statistical Service, 2013). Presently, the country's population stands at 25.9 million (CountryStat Ghana, 2014) and is fed mainly from food production on ~65% of terrestrial land (Table 8).

Agriculture is the largest and the most important sector of the country (Benhin and Barbier, 2001) and has been the main agent of deforestation and land-use intensification since the beginning of the 19th Century (Thompson, 1910; Dickson, 1969; Curry-Lindahl, 1969;

Benhin and Barbier, 2001). Food production was low and declining between 1960 and 1983 (Figure 7), and domestic food production was supplemented by food imports. Overall increase in the production of the seven major crops over this period was very small, a factor of 1.1. However, following efforts through the Economic Recovery Program to attain self-sufficiency in food production after the 1983 drought (Ofori-Sarpong, 1986; Attuquayefio and Fobil, 2005), the country has seen an increasing annual trend in food production (Figure 7). In contrast to the period prior to 1983, food production over the same number of years increased substantially by a factor 4.1. This increase in food production correspondingly resulted in the expansion of the agricultural land area which accelerated the rate of forest loss (Attuquayefio and Fobil, 2005). Twenty-eight years ago, land under cultivation of 30,000 km^2 (Ofori-Sarpong, 1986) which represented only 12.5% of the country's terrestrial land is today 5.2 times more. From FAO's 52-year time-series for the period 1960 - 2012, the country's annual crop yield and harvested area which indicate increasing trends are shown in Table 9, Figures 7 and 8.

Of the present land under cultivation, cocoa (24%),

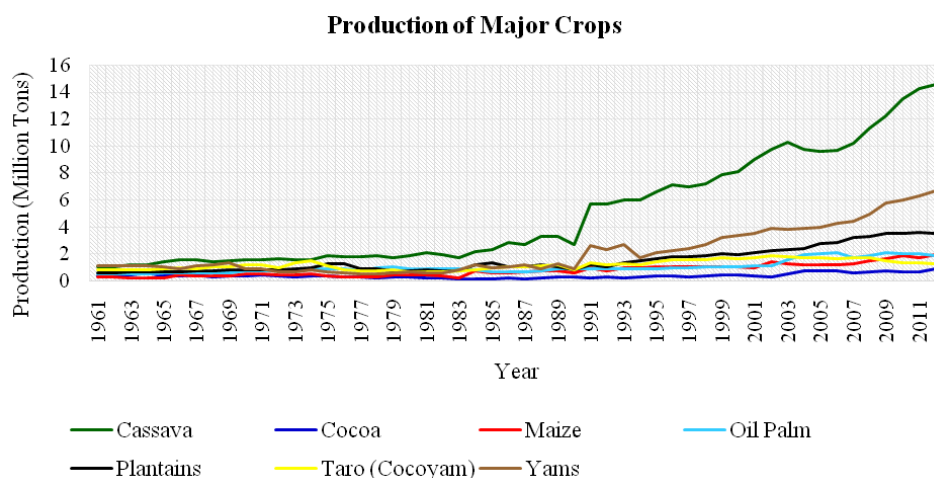


Figure 7. Trends in annual production of major crops in Ghana (Source of data: FAO 2014).

Table 9. Summarized statistics on major crops in Ghana

Crop	2012 Yield (tons)	2012 Harvested area (Square km)	Mean annual change (1961 – 2012) (%)	
			Yield	Harvested area
Cassava	14 547 279	8 685.50	+6.3	+4.4
Yams	6 638 867	4 263.43	+6.8	+4.2
Plantains	3 556 524	3 372.93	+4.5	+3.1
Maize	1 949 897	10 420.83	+10.3	+4.8
Oil Palm	1 900 000	3 500.00	+3.2	+4.0
Taro (Cocoyam)	1 270 266	1 963.28	+2.1	+5.5
Cocoa	879 348	16 003.00	+3.1	+0.6

Source: FAO, 2014.

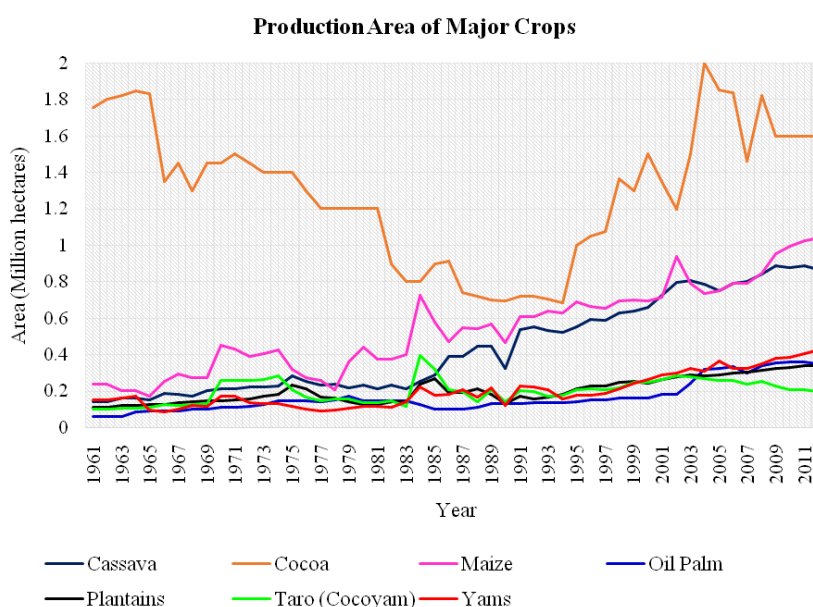


Figure 8. Trends in harvested area of major crops in Ghana (Source of data: FAO 2014).

maize (18%), cassava (13%) and yam (9%) alone occupy 63% (CountryStat Ghana, 2014). Despite the forest loss (Benhin and Barbier, 2001; Laurance et al., 2012), land-use intensification (Benton et al. 2003; Hansen and DeFries 2007; Cai and Pettenella 2013), bush burning (Birnin-Yauri and Aliero, 2008; Thompson, 1910), etc. associated with annual food crop cultivation, cocoa cultivation, the dominant agricultural activity in terms of land coverage, is problematic to biodiversity conservation because of its associated farming practices such as insecticide spraying (Padi and Owusu, 2003; Afrane and Ntiamoah, 2011; Tscharntke et al., 2012; Schiesari et al., 2013) and the elimination of other water-competing or pest-hosting plants (N'goran, 1998). Crop production has been restricted to the forest and the forest-savanna transition regions of the country due to land suitability. However, comparison between land under crop cultivation and suitable land indicates that in the future, production increases in crops cannot depend on area expansion (Breisinger et al., 2008). With no hope of expansion in cropland, further land-use intensification is expected if food sufficiency is to be attained.

Further land-use intensification could potentially have ecological (taxonomic and functional) decay consequences on both off-reserve biodiversity and on biodiversity inside protected reserves (Baillie et al., 2000; Hansen and DeFries, 2007; Laurance et al., 2012; Cai and Pettenella, 2013; Hamilton et al., 2013). Land-use effects on protected areas include reduction in effective size and alteration of ecological flows (Curran et al., 2004; Hansen and DeFries, 2007), and disruption of source-sink dynamics of ecosystems through the elimination of unique source habitats (Hansen and Rotella, 2002; Hansen and DeFries, 2007), edge effects (Hansen and DeFries, 2007; Bossart and Opuni-Frimpong, 2009; Laurance et al., 2012), increasing reserve isolation (Laurance et al., 2012) and reduction in habitat amount (Fahrig, 2013). In human-dominated land-use systems, ecological processes such as dispersal which naturally promote biodiversity could lead to population reduction and subsequently range contraction (Baillie et al., 2000; Benton et al., 2003) as well as landscape homogeneity. This is because dispersal could transport range-restricted species to poor-quality, unfavorable habitats (Benton et al., 2003) and widespread, invasive species to new, favorable habitats where they can spread rapidly. Thus, prioritizing studies that properly map land-use effects on biodiversity is necessary to ensure systematic conservation planning inside and outside protected reserves.

Restoration of degraded forest reserve through natural regeneration

It has been reported that biological recovery and regeneration is high not only for plants in highly degraded areas of the country (Lugo et al., 1993; Decher, 1997; Anyomi et al., 2011; Appiah, 2011; Boakye et al., 2012)

including areas that have consistently experienced annual bushfires (Tom-Dery et al., 2013), but also animals (Curry-Lindahl, 1969) in both the savanna and forest zones (Decher and Bahian, 1999). Within a period of 30 to 40 years, forests area which have completely degraded into savanna can be re-established provided those areas are protected from wildfires (Lugo et al., 1993; Decher, 1997). There is sufficient evidence that without further human interferences, degraded areas of the country's forest reserves can potentially undergo natural restoration into forest habitats. This notwithstanding, reforestation of degraded forest reserves is important to speed up biodiversity recovery. However, when exotic species are highly represented, it defeats the purpose of the reserves and violates some of the guidelines contained in the Forestry and Wildlife Policy (Ministry of Lands and Forestry, 1994) of the country.

McCullough et al. (2005) has already described as inappropriate, an attempt to replant degraded parts of the Krokosua Hills Globally Significant Biodiversity Area with *Cedrela odorata*. Larsen (2005) has also described this practice as a means of creating vegetative areas which are effectively biodiversity deserts, for which reason it should not be tolerated. While acknowledging that this practice has some historical support, it should not be considered at this time because there are several studies available on cheaper methods for restoring degraded tropical forests (Lamb et al., 2005; Shono et al., 2007; Holl, 2013).

Conservation campaigns should lead to voluntary participation

For over a century, the desire to preserve wilderness (Sarkar, 1999) and knowledge that loss of natural forest cover causes biodiversity loss, have driven conservation activities worldwide. Protected area systems of conservation are examples but their benefits are hardly noticed due to insufficient public knowledge, lack of alternative livelihood support for nearby communities, and the fact that there is almost zero interaction between a majority of the people and these reserves. Often, attempts to involve fringe communities in conservation efforts fail due to biased selection of co-management members, corruption in management and exclusion of locals from promised benefits (Songorwa, 1999; Sarker and Røskaft, 2011). Thus, residents of these communities usually engage in farming activities of varying intensities which gradually extend the agricultural frontier uncontrollably into the buffer zones (Bamford et al., 2014), leaving the protected areas as isolated islands (Sarkar, 1999).

Already, since the year 2000, The Rufford Foundation, a charitable organization based in the UK, has funded over 40, one-year duration biodiversity conservation projects costing over £260000 in the country (<http://www.rufford.org/projects/byCountry/GH>) (Table 10). These projects, together with several other short and

Table 10. Summary of conservation projects sponsored by The Rufford Foundation. The number of studies under each item is in brackets.

Target Audience	Conservation Target
Community conservation (24)	Habitat conservation/restoration (12)
General public (14)	Amphibians (11)
Students (6)	Primates (4)
Commercial drivers (1)	Medicinal plants (3)
	Bats (3)
Conservation Status	Conservation conflict resolution (2)
Threatened species (16)	Birds (2)
Non-threatened species (18)	Reptiles (2)
Not applicable (11)	Other plants (2)
	Fungi (1)
Study Area Type	Pangolins (1)
Reserved forests (26)	General (8)
Ramsar site (3)	
Plant garden (3)	
Others (13)	

long-term biodiversity conservation projects including the Northern Savanna Biodiversity Conservation Project, Ghana High Forest Biodiversity Conservation Project and the West African Primate Conservation Action, are reported to have achieved varying degrees of success. However, only a handful of them included alternative livelihood programs. Certainly, this deficiency makes them unsustainable because they would not get the continued local support (Songorwa, 1999; Sarker and Røskaft, 2011). In local communities, most people (perhaps greater than 90%) depend fully on farming for food, and resource extraction for other needs related to fuel, shelter and health. Therefore, running educational campaigns of “no-entry” into reserves contribute very little, if not nothing, to conservation because no one without the basic survival needs will truly show interest in such conservation messages (Sarker and Røskaft, 2011). There is the need to build a better conservation strategy which attracts the voluntary participation of the least-income earners (Kumaraswamy and Kunte, 2013).

As long as off-reserve biological resources remain inadequate to meet basic needs like meat, firewood, roofing materials, etc. of the poor, no amount of protection can prevent them from illegally operating within the protected areas as already observed (Belshaw and Bolton, 1994; Ntiama-Baidu et al., 2001; Decher et al., 2005; Rödel et al., 2005; McCullough et al., 2005; Abeney et al., 2008; Addo-Fordjour et al., 2009; Naskrecki, 2009; Larsen et al., 2009; Ghana Report, 2009; Demey, 2009; IUCN/PACO, 2010; Buzzard and Parker, 2012; The Proforest Initiative, 2012; Tollenaar, 2012; Tom-Dery et al., 2013). This should be expected because the people will naturally continue to eke out their living from nature in order to survive. The evidence in support of this is the scarcity of small and medium-sized

mammals within the country (Curry-Lindahl, 1969; Asibey, 1974; Ntiama-Badu et al., 2001; Larsen, 2005).

Given the present low protection of protected areas, alternative solutions which allow quota-based hunting (Songorwa, 1999; Kangalawe and Noe, 2012), firewood and other resource extraction in legally-designated areas (Sarker and Røskaft, 2011), as well as financial incentives for reserve protection (Sarker and Røskaft 2011) should be considered. The need to monitor and rid such systems of improper management and corruption (Songorwa, 1999; Sarker and Røskaft, 2011) is also essential. Moreover, there is the need to support studies which seek to identify threatened species and their distribution ranges so that location-based educational campaigns could be launched. Such campaigns should educate the target groups to properly understand the consequences of their continued dependence on such species using relevant research findings as well as similar cases from other locations. As a long-term project, primary, secondary and tertiary educational institutions should be considered effective vehicles for promoting the spirit of biodiversity conservation (Anyomi et al., 2008). Without these measures, it would be difficult to attract peoples' voluntary participation in conservation projects and all efforts will break down at some point as population increases.

Insects need to be prioritized in biodiversity studies

As discussed above and also in previous publications (Anyomi et al., 2008; Boshier et al., 2011), the situation of *T. gentii* and *S. fimbrisejala* brings into focus the thought that the country, like other tropical countries, could be experiencing several unknown losses, and perhaps mass extinctions (Sodhi et al., 2009), of several important species

because of the present gaps in relevant information. Already, Larsen (2006) has reported that one out of 10 insect species found in Ghana could be new to science, and thus significant effort need to be put into finding out the ecological status of the insect groups whose ecology and distribution across different land-use systems in the country is largely unknown.

The lack of insect taxonomists as already reported (Ansah, 2005; Naskrecki, 2007; Oteng-Yeboah, et al. 2009) deserves urgent consideration but should not be a setback to ecological knowledge acquisition. As Hobson (2014) points out “the first step to solving any problem is to not hide from it...” and besides, insects have a myriad of characteristics which make them strong candidates as species surrogates in studies that quantify the effects of environmental changes on biological systems. Insects are globally ubiquitous (Bossart and Opuni-Frimpong, 2009), are highly sensitive to anthropogenic and climatic disturbances (Kremen et al., 1993; Oliver and Beattie, 1996; Ward and Lariviere, 2004; Natural Resources Canada, 2010; Koch et al., 2013) and have large species assemblages (Beccaloni and Gaston, 1995; Natural Resources Canada, 2010) which allows them to broaden the scope of ecosystem factors that can be perceived (Kremen et al., 1993). They also contribute more than 90% of genetic variability (Duelli, 1997), they are tractable study species (Morris, 2010), they provide early warning of ecological changes (Kremen et al., 1993), they have high rate of endemism and speciation (Bossart and Opuni-Frimpong, 2009), and their diversity is highly correlated with that of vertebrates (Caro, 2010).

Besides the many desirable characteristics of insects, the taxonomic impediment could be overcome by identifying individuals to genus level (Caro, 2010). For rapid assessments (and in cases of financial constraints), careful identification to morphospecies by non-experts with basic training has been tested and proved to be sufficient (Oliver and Beattie, 1993, 1996, 1997; Ward and Lariviere, 2004; Obrist and Duelli, 2010; Derraik et al., 2002, 2010), even though this method has been criticized (Goldstein, 1997; Krell, 2004). Fortunately, countries such as Switzerland, Australia, New Zealand, Canada, etc., have already made great advances in using insects for biodiversity monitoring and their experience is an asset.

The need for data-driven conservation plans

The need to seek knowledge on the state of biodiversity in human-dominated land-use systems cannot be overemphasized (Kumaraswamy and Kunte, 2013), even though diversity is generally known to be low in these systems (Larsen, 2005; Davis and Philips, 2005; Bossart and Opuni-Frimpong, 2009). Landscape-scale studies which describe changes across different biodiversity groups in relation to land-use have become necessary in order to fill in knowledge gaps (Norris et al.,

2010; Kumaraswamy and Kunte, 2013). Because it takes years of observation to record a significant amount of species, focus should be on the effects of the interaction between these systems and biodiversity (Ntiamao-Baidu et al., 2000; Larsen, 2005). The outcome of such studies is the identification of a set of land-use activities for different areas which conserve different sets of ecologically-important species (Kumaraswamy and Kunte, 2013). This knowledge is necessary for off-reserve biodiversity conservation. Essentially, as areas get degraded some species adapt to the changes while others do not, and enrichment of these areas is possible under better land-use management (Tscharntke et al., 2012).

It is increasingly becoming clear that the protected area system of biodiversity conservation needs to be complemented by off-reserve conservation in order to make it more effective (Hansen and DeFries, 2007; Anand et al., 2010; Brussaard et al., 2010; Plieninger and Gaertner, 2011; Cai and Pettenella, 2013; Fahrig, 2013; Kumaraswamy and Kunte, 2013). Innovative land-use programmes for both crop yield improvement and farmland biodiversity conservation based on well-formulated policies should therefore be pursued (Benton et al., 2003; Anand et al., 2010; Kumaraswamy and Kunte, 2013; Baral et al., 2014). This naturally leads to a system in which food security for all people and biodiversity conservation are jointly achieved (Brussaard et al., 2010; Tscharntke et al., 2012). Thus, the problem of biodiversity conservation is a constrained optimization problem which cannot be solved in a vacuum, but must be linked with the food security of the lowest income earners (Tscharntke et al., 2012).

Adequate quality agro-ecological data is needed (Kumaraswamy and Kunte, 2013). Fortunately, the freely available high-resolution global land-use/cover products (for example, Gong et al., 2013; (<http://data.ess.tsinghua.edu.cn/>); Wang et al. 2014, etc) can be utilized in landscape-scale ecological assessments (for example, Fuller et al., 1998; Reidsma et al., 2006; Baral et al., 2014) to obtain the necessary ecological data, despite the belief that variability in farmland size make this pursuit difficult (Kumaraswamy and Kunte, 2013). Given data on agricultural variables, a locally-suitable, optimized land-use management solution can be developed. Already, the traditional systems of farming practiced predominantly in the northern regions of the country and Burkina Faso, in which native woody plants are interspersed with crops, and cocoa farming systems in the forest zone have ideal features that could facilitate easy integration into optimized land-use plans (also known as offset mechanisms; Kumaraswamy and Kunte, 2013). As Tscharntke et al. (2012) have indicated, biodiversity conservation and environmental cost minimization are achievable in the tropics amidst agro-ecological intensification.

Conflict of Interests

The author(s) have not declared any conflict of interests.

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Review

The effects of power lines on ungulates and implications for power line routing and rights-of-way management

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Thousands of kilometres of power lines exist and more are planned. Ungulates that range over large areas are likely to encounter power lines, but a synthesis of power line effects on ungulates is lacking. Reindeer (*Rangifer tarandus tarandus*) are suspected to avoid power lines up to distances of 4 km. In contrast, some forest ungulates preferentially forage in power line rights-of-way, cleared areas under power lines. We reviewed the factors that possibly influence avoidance and attraction effects of power lines on ungulates, construct a conceptual framework, and make suggestions on how to mitigate avoidance effects through power line routing and rights-of-way management. Power line construction, noise and electromagnetic fields are possible sources of disturbance, while rights-of-way management influences habitat use under power lines. Disturbance and altered habitat use can induce barrier and corridor effects, thereby influencing connectivity. Species-specific effects influence behavioural disturbance and habitat use. We found little evidence for behavioural disturbance of reindeer or forest ungulates under power lines. Forest ungulates could benefit from browsing in power line rights-of-way if they are managed to provide abundant and preferred forage as well as sufficient cover. However, power lines may facilitate access for hunters and predators. As a precaution, construction of power lines should be avoided in calving areas. To establish a causal relationship between the construction of power lines and potential avoidance, before-after-impact-control studies are recommended. More research is needed to make recommendations for the optimal design of power line networks.

Key words: Power lines, rights-of-way, ungulates, disturbance.

INTRODUCTION

The transmission network for central grid power lines of at least 220 kV covers about 300,000 km in Europe (European Network of Transmission System Operators for Electricity, 2012) and 250,000 km in the USA

(Abraham, 2002). Power lines above 220 kV may only constitute a small proportion of the total grid. About 200,000 and 450,000 km of overhead power lines carrying various voltages exist in Norway (Statistics

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Norway, 2011a) and Sweden (Grusell and Miliander, 2004), respectively.

In Norway, the central power line grid covers over 20% more surface than the central road network due to required corridor widths (740 versus 630 km²), even though it is only half as long (28,000 versus 55,000 km) (Appendix A). The footprints of the distribution power line grid and road network are similar (Appendix A). We estimated that over 60% of central grid power lines traverse forests, while circa 40% of the Norwegian land area is covered by forest (Appendix B). The Norwegian central grid operator Statnett is planning to increase the construction of power lines to 300 km per year by 2020 (Statnett, 2013). 3,600 km of new power lines are planned in Germany until 2023 (German Transmission System Operators, 2013).

An extensive body of research on potential effects of roads on wildlife is available (reviewed in: Forman and Alexander (1998), Spellerberg (1998), Seiler (2001), Coffin (2007), Fahrig and Rytwinski (2009), Benitez-Lopez et al., (2010)), but knowledge on power line effects is scattered. Power lines may influence wildlife through disturbance, clearing of forest habitat under power lines, edge, barrier and corridor effects (Willyard and Tikalsky, 2004; Ball, 2012). Fragmentation by power lines could induce genetic drift, as for roads (Epps et al., 2005, Kuehn et al., 2007), and reduce population productivity and persistence (Griffen and Drake 2008; Haanes et al., 2013). Wide-ranging and mobile species as some ungulates will likely encounter power lines within their home ranges (Tables 1 to 7).

If disturbance by power lines is analogous to predation risk, it can cost energy for fleeing, increased vigilance, lost resources in habitats associated with danger and impaired mating and parental investment (Frid and Dill, 2002). If ungulates are disturbed by power lines, they can be expected to avoid power lines to reduce these costs. However, cleared areas under power lines (rights-of-way or ROW) are maintained as early- to mid-successional habitats (Bramble and Byrnes, 1982) and provide benefits through additional browse for forest ungulates (Bramble and Byrnes, 1972). This may result in attraction effects towards power line ROW.

The aim of this article is specifically to review the factors that possibly influence avoidance and attraction effects of power lines on ungulates (Tables 1 to 7), set into a conceptual framework (Figure 1). The review is based on both peer-reviewed and grey literature found through the Web of Science (isiknowledge.com) and Google Scholar, proceedings from the symposia on Environmental Concerns in Rights-of-Way Management (<http://rights-of-way.org/1content.htm>) and reviews on road ecology (see above). We searched for the keywords power line or transmission line in combination with ungulate, deer, elk, reindeer, sheep or cow; and subsequently scanned the papers for relevant references

that did not show in the search engine. Research has mainly been undertaken on reindeer (*Rangifer tarandus tarandus*) in open alpine areas in Norway and on other ungulates in forests in Canada, USA and Norway (Table 1 to 7).

We identified proximate and ultimate causes of avoidance and attraction effects resulting from power line routing, construction, ROW management and species-specific effects (Figure 1). Power line construction may induce behavioural disturbance. ROW management and routing is expected to affect the use of ROW habitat. Sensitivity to disturbance and habitat preferences, which are species-specific, should further affect behavioural disturbance and habitat use. Both behavioural disturbance and habitat use determine avoidance and attraction effects. Therefore, power line routes could function as barriers and/or corridors, with consequences for connectivity and functional loss of habitats. Finally, we suggest how to mitigate avoidance effects through power line routing and ROW vegetation management.

EFFECTS OF POWER LINES ON UNGULATES

Behavioural disturbance from power line constructions

Power lines could disturb ungulates because they are artificial structures that can emit noise, light and electromagnetic fields. Frid and Dill (2002) reported that disturbance should be analogous to predation risk. Ungulates can be expected to alter their behaviour close to power lines if they are disturbed by power lines.

Noise

Electrical discharge by power lines produces crackling or hissing corona noise (Straumann, 2011). Wind can produce Aeolian noise through vibrations of the physical structure (Tsujiimoto et al., 1991). An audiogram suggests that reindeer can hear corona noise from power lines (300 and 420 kV) up to 79 m (Flydal et al., 2010). Although little is known on the effects of corona noise, noise of a 500 kV transmission line did not significantly influence cattle behaviour (Ganskopp et al., 1991).

Electromagnetic fields

Power line electromagnetic fields are suspected to disturb the hypothesized magnetic alignment of cattle and roe deer (*Capreolus capreolus*) (Burda et al., 2009). However, ungulates may align themselves in the direction of power lines (Burda et al., 2009), supposedly interrupted in their north south alignment (Begall et al., 2008), for other reasons that were not accounted for such as wind and solar conditions for thermoregulation (Hetem

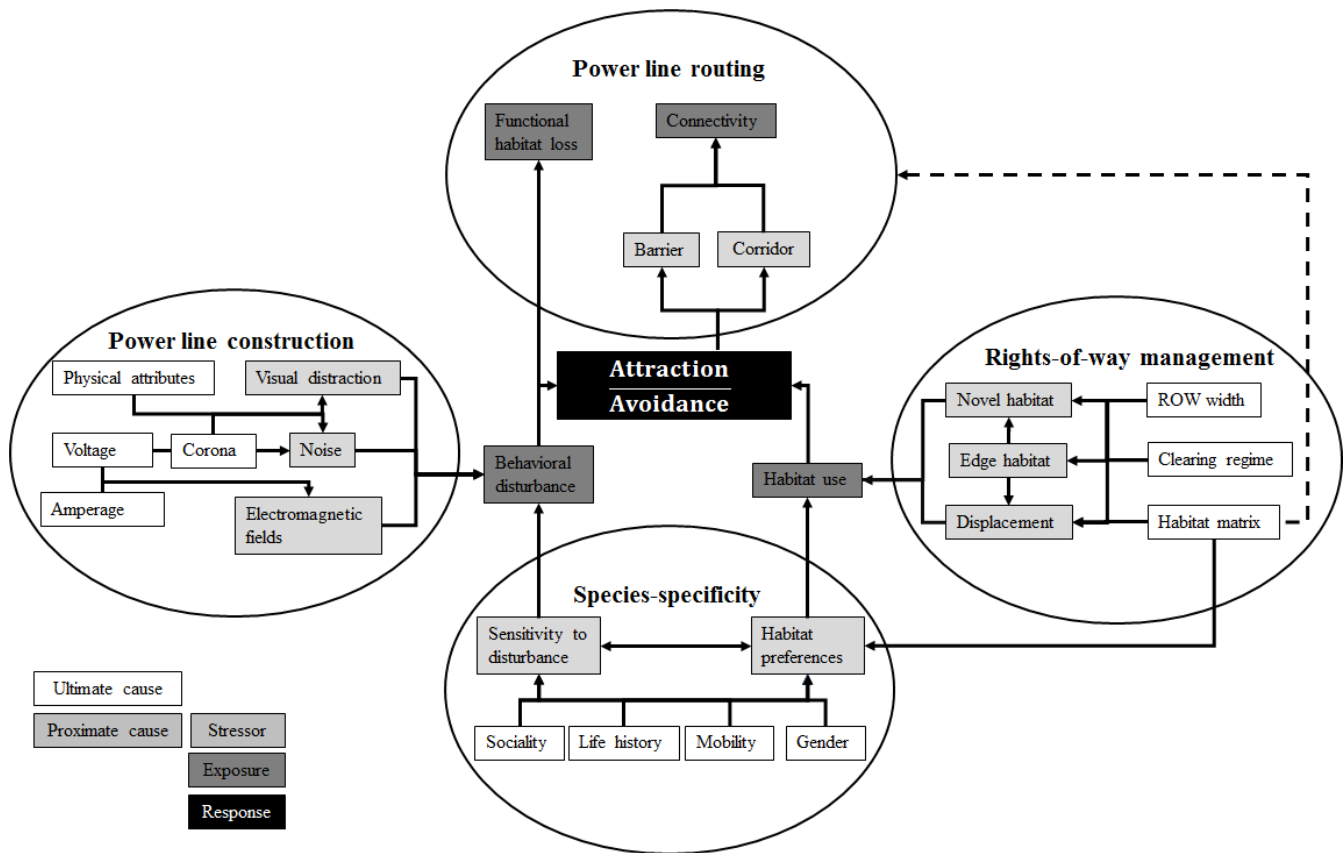


Figure 1. Possible factors influencing avoidance and attraction effects of power lines on ungulates.

et al., 2011). Fluctuations in electric fields of a 500 kV power line did not influence cattle behaviour (Ganskopp et al., 1991). Domestic-tame reindeer in enclosures became more restless and moved away from power lines (132 and 300 kV) when transmission load increased, although these results were ambiguous (Flydal et al., 2009).

Visual distraction

Visual distraction of power lines in the absence of noise and electromagnetic fields has apparently not been tested. Reindeer are suspected to see ultraviolet (UV) light (Hogg et al., 2011) and consequently corona flashes from power lines (Tyler et al., 2014). Reindeer's sensitivity to UV light is suspected to aid in detecting predators and forage in arctic environments (Hogg et al., 2011). The strongest emission of UV light by power lines was centred on 337 nm wavelength according to a patent application for a corona detector (Le et al., 1994). The eye lens of ungulates however blocked the largest proportion of light at this wavelength (Douglas and Jeffery, 2014). This may explain the lack of retinal

response towards UV light of 325 nm wavelength by other ungulates (Jacobs et al., 1994).

The lack of behavioural disturbance under power lines in general may indicate that the sight of power lines was not a source of disturbance. Deer (*Odocoileus* spp.), elk (*Cervus canadensis*) and other ungulates fed in a power line ROW (500 kV, 41 m wide) without signs of disturbance apart from a five-minute motionless period when entering the ROW (Goodwin Jr, 1975). Also semi-domestic reindeer in an enclosure experiment did not clearly avoid power lines (132 and 300 kV) (Flydal et al., 2009).

White-tailed deer (*Odocoileus virginianus*) (Bramble and Byrnes, 1972; Doucet et al., 1979; Doucet et al., 1983), moose (*Alces alces*) (unpublished data), bighorn sheep (*Ovis canadensis canadensis*) and elk (Goodwin Jr, 1975) bedded under power lines. Feeding positions and activity of cattle were similar in pens with and without power lines (Ganskopp et al., 1991). Other studies indicate that the vegetation in power line ROW rather than disturbance by power lines influences the use of those areas by ungulates (Goodwin Jr, 1975; Morhardt et al., 1984). Energy spent in response to indifferent stimuli may be wasted (Reimers and Colman, 2009).

Table 1. Possible disturbance effects of power lines on the behaviour of ungulates.

Species	Result	Reference	Location
Cattle Roe deer (<i>Capreolus capreolus</i>)	Cattle and roe deer align themselves in the direction of power lines, supposedly interrupted in their north-south alignment.	Burda et al. (2009)	Morocco South Africa, India, Australia, Belgium, Denmark, France, Germany, Ireland, Netherlands, Russia, UK, USA, Argentina
Semi-domestic reindeer (<i>Rangifer tarandus tarandus</i>)	Deer are suspected to hear power line (300 and 420 kV) noise up to 79 m distance.	Flydal et al. (2010)	Southern Norway
Reindeer (<i>Rangifer tarandus tarandus</i>)	Reindeer are suspected to see UV corona flashes from power lines.	Tyler et al. (2014)	

Although the given examples provide little evidence for the disturbance of ungulate behaviour by power lines, it does not necessarily mean that ungulates are not impacted by power lines. Human disturbance increased cardiac rates of bighorn sheep without changing their behaviour (MacArthur et al., 1979, 1982). Chronic stress can have adverse effects on reproductive, immune and neural systems and suppress growth in the absence of behavioural changes (Wingfield et al., 1997).

Altered habitat use at power line rights-of-way

Displacement from rights-of-way

Besides possible disturbance effects of power lines, forest ungulates may be displaced from cleared habitats under power lines because they lack canopy cover (Rieucan et al., 2007) and forage in the first years after clearing (Bramble and Byrnes, 1982; Lamothe and Dupuy, 1984; Garant and Doucet, 1995; Ricard and Doucet, 1999; Hydro-Québec, 2013) (Table 2).

Moose tracks and white-tailed deer tracks and pellets were less abundant in power line ROW (220 and 735 kV; 90 to 140 m wide) as compared to forests at 2 km distance (Joyal et al., 1984) and forests adjacent to ROW (120 - 735 kV, 30 - 150 m wide), respectively (Doucet et al., 1979; Lamothe and Dupuy, 1984; Jackson and Hecklau, 1995). White-tailed deer abandoned significantly more food provided in feeders in a power line ROW (30 m wide) as compared to adjacent forest when regeneration was absent (Rieucan et al., 2007). White-tailed deer browsed a smaller proportion of stems despite higher availability in power line ROW as compared to adjacent forest, except where the abundance of browsed stems exceeded that of the forest approximately six-fold (Mayer, 1976).

These results indicate that forest ungulates may be displaced by power line ROW (Table 2), especially when

food, cover or both are lacking (Joyal et al., 1984). Increased food abundance may however compensate for the lack of cover (Mayer, 1976; Rieucan et al., 2007).

Rights-of-way as novel habitat

Following regrowth, habitats in power line ROW can also create novel habitats for forest ungulates through the provision of attractive feeding opportunities (Bramble and Byrnes, 1979; Ricard and Doucet, 1999; Hydro-Québec, 2013) (Table 3). White-tailed deer deposited more pellet groups, foraged more intensely and left more signs in power line ROW as compared to forests adjacent to ROW or control forest (Bramble and Byrnes, 1972; Cavanagh et al., 1976). The ROW provided more stems for browsing. Black-tailed deer (*Odocoileus hemionus columbianus*) used a power line ROW significantly more than adjacent mature forest, indicated by pellet groups (Loft and Menke, 1984). Deer use increased with shrub and herbaceous cover as well as foraging plants. These results indicate that food availability in power line ROW habitat is important for the use of that habitat by forest ungulates.

Not only the amount of forage, but also its composition may influence the use of power line ROW for browsing (Milligan and Koricheva, 2013). Moose and white-tailed deer browsing intensity in power line ROW appeared to be influenced by the proportion of preferred browse species rather than browse availability (Garant et al., 1987; Ricard and Doucet, 1999).

Trees that have been cut in power line ROW could provide higher quality browse because they prioritize growth instead of defence against herbivore damage through secondary metabolites (Rea and Gillingham, 2001). However, the increased availability of light in power line ROW clearings may promote both growth and defence (Nybakken et al., 2013). Herbs in a power line ROW provided higher concentrations of protein and

Table 2. Displacement of ungulates from power line rights-of-way.

Species	Result	Reference	Location
White-tailed deer (<i>Odocoileus virginianus</i>)	Deer browsed a smaller proportion of stems in power line ROW compared to adjacent woods except where browsed stems were approximately six times as abundant as in woods.	Mayer (1976)	Eastern USA
	Significantly fewer deer tracks and fewer pellets groups were found in a power line ROW (120 kV, 30 m wide, cleared two years before the study) compared to transect in forest at 30 distances from the ROW except on one survey for tracks.	Doucet et al. (1979)	Eastern Canada
	Significantly fewer pellets were found in cleared areas of a power line ROW (twin power lines, 735 kV, 150 m wide) compared to lateral forest in three of four sites.	Lamothe and Dupuy (1984)	Eastern Canada
	Significantly fewer tracks and fewer pellet groups were found in a power line ROW (345 kV, 45.7 - 90 m wide) compared to adjacent forest. Fewer tracks and pellet groups were found in the ROW compared to the edge of the ROW.	Jackson and Hecklau (1995)	North-eastern USA
Moose (<i>Alces alces</i>)	Deer left more food inside feeders placed in a power line ROW (30 m wide) compared adjacent forest when regeneration was absent.	Rieucou et al. (2007)	Eastern Canada
	Significantly fewer tracks were found in power line ROW (220 kV and 735 kV; 90 - 140 m wide) compared to transects in forests at 2 km distance.	Joyal et al. (1984)	Eastern Canada

Table 3. Power line rights-of-way as novel habitats for ungulates.

Species	Result	Reference	Location
White-tailed deer (<i>Odocoileus virginianus</i>)	Deer deposited two to three times more pellets and browsed more intensely in a power line ROW (55 m (expressed in feet in Bramble and Bymes (1982); 1 foot = 0.3048) wide) compared to forest at 10 m (expressed in chain; 1 chain = 20.1 m) distance from edge of the ROW.	Bramble and Byrnes (1972)	North-eastern USA
	Signs of deer use were several times more frequent inside a newly cleared power line ROW (72 m (expressed in feet; 1 foot = 0.3048 m) wide) compared to control forest.	Cavanagh et al. (1976)	North-eastern USA
Black-tailed deer (<i>Odocoileus hemionus columbianus</i>)	Deer deposited significantly more pellets inside a power line ROW compared to adjacent mature forest.	Loft and Menke (1984)	Western USA
Moose (<i>Alces alces</i>)	Moose browsed seven times more intensely in a power line ROW compared to forests at 2 km distance. There was no significant difference in the count of pellet groups.	Ricard and Doucet (1999)	Eastern Canada

Table 4. Edge effects at power line rights-of-way on ungulates.

Species	Result	Reference	Location
	The proportion of browsed stems was in-between the proportion in power line ROW and adjacent woods in two study areas and lower than in both ROW and wood in another study area.	Mayer (1976)	Eastern USA
White-tailed deer (<i>Odocoileus virginianus</i>)	More tracks were found at the edge of a power line ROW (twin power lines, 735 kV, 150 m wide) compared to the ROW and lateral forest. Fewer pellets were found in the ecotones between cleared areas and lateral forests compared to lateral forest.	Lamothe and Dupuy (1984)	Eastern Canada
	Fewer tracks and generally fewer pellet groups were found along the edges of a power line ROW (345 kV, 45.7 - 90 m wide) compared to adjacent forest.	Jackson and Hecklau (1995)	North-eastern USA

minerals and contained less fiber as compared to woody browse (Bramble and Byrnes, 1972). Forbs in power line ROW contained more protein and minerals as compared to grasses and woody browse (Harlow et al., 1995). The quality of the forage can be expected to influence the attractiveness of power line ROW habitat for forest ungulates.

Edge habitat along rights-of-way

Forest ungulates can benefit from the increased availability of forage not only inside power line ROW but also along edges (Bramble and Byrnes, 1979). Stem availability within 3-10 m from power line ROW edges was elevated as compared to forests at further distances from the edge (Luken et al., 1991; Luken et al., 1992; Rieucan et al., 2007; Powell and Lindquist, 2011). 18 of 20 shrub species were significantly more likely to be found at the edge of a 60 m wide power line ROW as compared to its centre (Brisson et al., 1997).

Lamothe and Dupuy (1984) noted more white-tailed deer tracks along the edge of a power line ROW (twin power lines, 735 kV, 150 m wide) as compared to the ROW and lateral forest (Table 4). However, fewer pellets were found in the ecotones between cleared areas and lateral forests as compared to further inside the forests (Lamothe and Dupuy, 1984; Jackson and Hecklau, 1995), indicating that deer may have spent more time in areas of better cover. The abundance of stems along power line ROW edges as compared to ROW and adjacent woods and the proportion of those stems that were browsed by white-tailed deer did not follow a consistent pattern across study areas (Mayer, 1976). Shrub availability along power line ROW edges may favour ungulate browsing and habitat use along those edges but a link between the two has, as far as we know, not been established.

Functional loss of habitat

Disturbance by power lines may not only affect the use of areas directly under power lines but also habitats adjacent to it. Power lines contributed to a reduction in area use of wild female reindeer within 1 km from pitfall traps and hunting blinds (Panzacchi et al., 2013). The density of semi-domesticated reindeer was significantly (73%) lower within 4 km of a power line (132 kV) than further away during calving in areas of rugged terrain (Vistnes and Nellemann, 2001) (Table 5). However, more favourable snow conditions and lower predation rates at higher elevations further away from power lines may have influenced this result (Reimers and Colman, 2009).

Wild reindeer were significantly less abundant than expected within 2.5 km of power lines (300 and 420 kV) in six of eight sampling years (Nellemann et al., 2001) (Table 5). Areas transected by power lines (66 - 420 kV) were also used less than expected (Vistnes et al., 2001). However, the accessibility of lichen forage, provided by an index of snow depth and hardness, was approximately three times lower in areas transected by power lines and other infrastructure (Vistnes et al., 2001). The influence of forage accessibility, although not significantly different between areas, can be discussed.

Wild reindeer became less abundant within 4 km from power lines (300 and 420 kV) or roads after they were built and more abundant beyond this distance (Nellemann et al., 2003). However, the shift in abundance coincides with the flooding of an area close to power lines and roads following the construction of a dam (Nellemann et al., 2003).

In contrast, counts of wild reindeer were disproportionately high within 5 km of power lines and minor roads above 1,400 m a.s.l. in summer (Vistnes et al., 2008). There was no clear evidence for aversion by wild reindeer along a 66 kV power line indicated by lichen

Table 5. Functional loss of habitat surrounding power line rights-of-way for ungulates.

Species	Result	Reference	Location
Semi-domesticated reindeer (<i>Rangifer tarandus tarandus</i>)	The density of deer was significantly (73 %) lower below compared to above 4 km from a power line (132 kV) during calving in areas of rugged terrain.	Vistnes and Nellemann (2001)	Northern Norway
	Deer were significantly less abundant than expected in areas within 2.5 km from power lines (300 and 420 kV) in six of eight sampling years.	Nellemann et al. (2001)	Southern Norway
Wild reindeer (<i>Rangifer tarandus tarandus</i>)	Deer were less abundant in areas transected by power lines (66 - 420 kV) than expected.	Vistnes et al. (2001)	South-central Norway
	Deer became less abundant in areas within 4 km from power lines (300 and 420 kV) or roads after they were built and more abundant in areas above that distance.	Nellemann et al. (2003)	South-western Norway
	Power lines contributed to a reduction in area use of female deer within 1 km from pitfall traps and hunting blinds.	Panzacchi et al. (2013)	Southern-Norway

measurements (Reimers et al., 2007). Moose did not avoid moving towards central grid power lines except in certain habitats during autumn (Bartzke et al., in press). Hydro-Québec (2013) reported that the use of winter feeding grounds by white-tailed deer was not inhibited by power line ROW (120 - 735 kV, 30 - 164 m wide) close to them.

Connectivity and power line routing

Power line routes as barriers

Disturbance by power lines and the lack of canopy cover in power line ROW may prevent animals from crossing power lines. Vistnes et al. (2004) concluded that wild reindeer refrained from crossing power lines based on lichen measurements on two sides of parallel power lines (132 and 300 kV) indicating differential grazing. However, these power lines were routed along a dam in the northern part of the study area that could have impeded crossings. The side that was apparently less grazed in the southern part of the study area was closer to a main road and urban settlements at lower elevations. Reimers et al. (2007) suggested that harvesting along a summer open road close to power lines could have been another alternative explanation for reduced area use.

Wild reindeer crossed a 66 kV power line in 14 of 22 years according to aerial surveys (Reimers et al., 2007). Strand et al. (2001) hypothesized that wild reindeer cross barriers when the need to migrate is extra-large based on the difference in the availability and accessibility of forage, disturbances, predation risk and distance between alternative feeding areas. Moose did not refrain

from crossing power lines (735 kV) with ROW that were 90 m wide (Joyal et al., 1984). Neither did white-tailed deer refrain from crossing power line (120 - 735 kV) ROW of 30 - 146 m width (Hydro-Québec, 2013). Only two animals of 87 elk and nine deer (*Odocoileus* spp.) failed to cross a power line ROW (500 kV, 41 m wide) (Goodwin Jr, 1975) (Table 6).

However, white-tailed deer reduced crossings away from a planted area within a power line ROW (120 kV, 30 m wide) (Doucet et al., 1983). Moose refrained from crossing power line (230 - 735 kV) ROW that were 140 m wide (Joyal et al., 1984). The need to cross power lines, the size of the power line construction, transmission load, the width of the corridor and the availability of cover may influence the willingness of ungulates to cross power lines.

Power line routes as corridors

Food availability along power line ROW forest edges or routes for easy travel may encourage movement of ungulates along power lines. Moose increased movements along central grid power lines over movements towards and away from power lines when approaching them (Bartzke et al., 2014). However, when close enough to cross power lines (25 m), moose moved randomly with respect to the power line.

White-tailed deer were reported to start travelling along power line ROW (345 kV, 47.5 - 90 m wide) after construction (Jackson and Hecklau, 1995). Goodwin Jr (1975) observed an elk (*Cervus canadensis*) cow with two calves travelling along a power line ROW (500 kV, 41 m wide). Forman and Deblinger (2000) sighted a moose

Table 6. Power line routes as barriers or corridors.

Species	Result	References	Location
Elk (<i>Cervus canadensis</i> spp.) Deer (<i>Odocoileus</i> spp.)	Two of 87 elk and nine deer failed to cross a power line ROW (500 kV, 41 m wide).	Goodwin Jr (1975)	North-western USA
	Moose refrained from crossing power line ROW (230 - 735 kV, 140 m wide) significantly.	Joyal et al. (1984)	Eastern Canada
Moose (<i>Alces alces</i>)	Movements along central grid power lines increased over movements towards and away from power lines when getting closer.	Bartzke et al. (2014)	Central Norway
	Significantly different lichen measurements on two sides of parallel power lines (132 and 300 kV) indicated differential grazing on each side.	Vistnes et al. (2004)	South-central Norway
Wild reindeer (<i>Rangifer tarandus tarandus</i>)	Wild reindeer herds crossed a 66 kV power line in 14 of 22 years according to aerial surveys.	Reimers et al. (2007)	South central Norway

that travelled along a power line ROW and a railroad. We found no further evidence for the use of power line ROW as movement corridors. However, bison (*Bison bison*) were shown to move along roads (Bjornlie and Garrott, 2001; Bruggeman et al., 2007), although surrounding terrain can be confounding (Bruggeman et al., 2006).

Species-specific ultimate and proximate causes

Sociality

Reindeer in Norway may in general be more sensitive to power lines than other ungulates like moose or white-tailed deer because they live in large groups in open alpine habitat experiencing comparatively low human use but intense seasonal hunting. These attributes were shown to increase ungulate flight responses (Stankowich, 2008). In open habitats, ungulates should detect and react to disturbances at greater distances because there are no escape habitats to seek refuge (Stankowich, 2008).

In theory, ungulates in larger groups may spend more time being disturbed because they have a greater chance of detecting disturbances (Taraborelli et al. (2012) for guanacos (*Lama guanicoe*)), and disturbance might be transmitted between group members (Stankowich, 2008). Groups of West Greenland caribou (*Rangifer tarandus groenlandicus/tarandus*) became aware of humans at larger distances than solitary individuals (Aastrup, 2000). Although feral reindeer fled at shorter distances from humans in larger groups (Reimers et al., 2006) and larger groups of Svalbard reindeer (*Rangifer tarandus platyrhynchus*) did not discover observers earlier, they were reported to correspond cooperatively (Colman et al.,

2001).

The ability to quickly detect and react to disturbances could be an evolutionary advantage to protect against real predators but a disadvantage if the source of the disturbance is not lethal. Then animals lose time and energy in being unnecessarily disturbed. The degree of reindeer domestication may also influence their sensitivity to disturbances (Flydal et al., 2009; Reimers et al., 2012).

Mobility

The lack of power line avoidance by forest ungulates may also in part be explained by the mobility of the species of concern. Stationary species and species with small home ranges may not have alternative habitats available, or the costs of reaching those habitats outweigh the costs of remaining close to power lines (Gill et al., 2001). This could occur in times or areas of resource limitations. White-tailed deer increased stationary browsing time in a power line ROW from 7 to ~40% in cold winter as compared to a mild winter (Doucet et al., 1987). Possibly forest ungulates cannot afford to avoid power line ROW when overall food availability is insufficient in relation to their densities (Ytrehus et al., 1999; Lamoureux et al., 2001). Hagen et al. (2007) speculated that reindeer will also react less to disturbances when the population size increases.

Gender and life history

Throughout a species' life cycle, its disturbance tolerance towards power lines may vary with life history traits and states such as gender, age, reproductive status, social

Table 7. Potential demographic impacts of power lines and associated clearings on ungulates.

Species	Result	References	Location
White-tailed deer (<i>Odocoileus virginianus</i>)	Tracks and pellet groups increased significantly after the construction of a power line ROW (345 kV, 45-90 m wide).	Jackson and Hecklau (1995)	North-eastern USA
Wild reindeer (<i>Rangifer tarandus tarandus</i>)	Calf/cow ratio declined significantly with the construction of human infrastructure including power lines.	Nellemann et al. (2003)	South-western Norway
	Hunters harvested more, although not significantly more, moose in power line ROW sites compared to control sites.	Ricard and Doucet (1995)	Eastern Canada
Moose (<i>Alces alces</i>)	89 of 107 hunters said they were hunting in a power line ROW (500 kV, 41 m wide), control clearings or along roads.	Goodwin Jr (1975)	North-western USA
	A power line (500 kV) ROW was reported to be a traditionally preferred hunting area.	Perry et al. (1997)	Eastern USA

status and/or season (for example, rut) (Frid and Dill, 2002; Stankowich, 2008).

Several, although not all (Frid, 2003; Mahoney et al., 2011), studies indicate that groups with calves (Aastrup, 2000), females with young (Ciuti et al., 2008) and females in times of calving or raising calves (Maier et al., 1998; Vistnes and Nellemann 2001; Bartzke et al., in press) are most sensitive to human disturbance (Wolfe et al., 2000). However, no clear evidence for gender-specific effects towards power lines was found (Vistnes and Nellemann, 2001; Bartzke et al., in press). Possibly, power lines are not disturbing enough to promote gender-specific effects in contrast do direct harassment by humans or motorized vehicles.

Potential demographic impacts

Fragmentation effects by power lines have been argued to contribute to population decline (Nellemann et al., 2003), impair migration (Vistnes et al., 2004) and could induce genetic drift, as for roads (Epps et al., 2005; Kuehn et al., 2007). Yet we found no evidence that the construction of power lines alone influenced population dynamics of ungulates. Tracks and pellet groups of white-tailed deer increased significantly after the construction of a power line (345 kV, 45-90 m wide) (Jackson and Hecklau, 1995) (Table 7).

Although power line ROW can provide additional forage, they could also be an “ecological trap” (Battin, 2004). Ecological traps are thought to occur when the attractiveness of a habitat (through increased browse)

increases disproportionately relative to its value for survival and reproduction. An increased rate of disturbance in connection with an increased rate of predator encounters can result in a reduction of population size (Frid and Dill, 2002) due to increased antipredator investment (stress) (Ydenberg and Dill, 1986; Cassirer et al., 1992; Maier et al., 1998; Rumble et al., 2005), reduced net energy intake (Stockwell et al., 1991) and body condition (Bradshaw et al., 1998; Luick et al., 2011). Power line ROW along with access roads may provide access for hunters (Goodwin Jr, 1975; Ricard and Doucet, 1995; Perry et al., 1997). Natural predators were reported to travel along power line ROW (Foster, 1956 in Ball, 2012; Paquet et al., 1996; Gurarie et al., 2011). Power lines provide nesting and perching opportunities for golden eagle (*Aquila chrysaetos*) (Steenhof et al., 1993; Prather and Messmer 2010), which preys on ungulates (Nybakk et al., 2002; Norberg et al., 2006; Johnsen et al., 2007; Hamel and Côté, 2009; Nadjafzadeh et al., 2013).

Increased predation risk, coupled with disturbance and fragmentation induced by power lines, could affect the demography of ungulate populations. The availability of forage in power line ROW may however favour forest ungulate populations locally.

KNOWLEDGE GAPS

A more causal relationship between the construction of power lines and possible avoidance by ungulates may be established from before-after-impact-control studies

(Beyers 1998). Observing wild ungulate behaviour under power lines as compared to similar control areas would be necessary to find out if ungulate behaviour is disturbed by power lines. Wildlife cameras (Dunne and Quinn, 2009; Kuijper et al., 2009) or GPS collars with cameras can be used for this purpose. Measuring faecal glucocorticoid concentrations possibly in combination with other disturbance indicators (Tarrow and Blumstein, 2007) could help to show if power lines are anthropogenic stressors. Separating the causes of possible disturbances like visual distraction, noise or electromagnetic fields would require further experiments. Ideally, experimental and control animal populations should be established. Experiments using reindeer with different degree of domestication should be made as in Flydal et al. (2009).

A number of the reviewed studies lacked statistical analyses (Bramble and Byrnes, 1972; Cavanagh et al., 1976; Mayer, 1976; Doucet et al., 1983; Doucet et al., 1987; Garant et al., 1987), did not precisely report the methodology (Loft and Menke 1984), power line voltage and/or ROW width (Mayer, 1976; Ricard and Doucet, 1999; Rieucou et al., 2007; Burda et al., 2009; Panzacchi et al., 2013). There were also large differences in scale ranging from few metres (Lamothe and Dupuy, 1984; Jackson and Hecklau, 1995; Rieucou et al., 2007) up to several kilometres (Vistnes and Nellemann, 2001). The scale considered may invert the conclusions (Vistnes and Nellemann, 2008). The ability to see or hear power lines may be an important factor to be considered, as for roads (Montgomery et al., 2012).

Knowledge gaps exist on the extent of power line ROW edge effects on forest ungulates, the factors influencing barrier effects and the preference of human and natural predators for hunting near power lines (Table 7). Addressing these gaps would help to show if ungulate populations may be impacted by power lines.

Routing power lines along existing power lines and roads may reduce further fragmentation of ungulate habitat but at the same increases avoidance and barrier effects. Jaeger et al. (2005) concluded from a modelling exercise that bundling roads would have less negative impacts on population persistence than distributing them evenly across the landscape. However, female wild reindeer reduced area use within 1 km from ancient pitfall traps and hunting bows when accounting for the effects of power lines and roads more than roads or power lines alone (Panzacchi et al., 2013).

The food availability in power line ROW could attract forest ungulates away from roads and railroads, similarly to supplemental feeding (Wood and Wolfe, 1988; Andreassen et al., 2005), and reduce vehicle collisions. Power line ROW could also attract ungulates towards areas surrounding roads. Further studies are necessary to make recommendations for the optimal design of power line networks.

MITIGATION MEASURES

Implications for power line routing

The reviewed literature suggests that power lines induce minor negative behavioural responses to ungulates. Benefits from additional food in power line ROW for forest ungulates may be expected if they are routed through old coniferous forests with little food but good canopy cover (Bjørneraas et al., 2011). Moose browsed four times more intensely in power line ROW traversing coniferous forests as compared to those traversing mixed forests (Ricard and Doucet, 1999). When power lines traverse food-rich young, mixed or deciduous forests, the availability of browse may not compensate for the lack of cover (Rieucou et al., 2007).

In contrast to forest ungulates, several studies suggest that power lines may disturb reindeer inhabiting open alpine areas. However, the lack of clear evidence for the disturbance of semi-domestic reindeer behaviour in the proximity of power lines (Flydal et al., 2009) and the maximum distance at which reindeer may hear power lines (79 m) (Flydal et al., 2010) indicates that power lines themselves were not necessarily the main cause of the reduced area use close to power lines reported by Vistnes et al. (2001), Nellemann et al. (2001, 2003) and Panzacchi et al. (2013). Earlier, Vistnes et al. (2004) suggested that building power lines should be avoided in wild reindeer habitats. This recommendation could be debated.

To minimize inference with reproduction, the construction of power lines, which may require the use of helicopters and building access roads, should be avoided calving areas. Disturbance after the construction of power lines can be minimized by prohibiting hunting, vehicles and pets along them (Bergerud et al., 1984; Miller et al., 2001; Clair and Forrest, 2009). Power line corona noise, flashes and magnetic fields can be reduced through engineering solutions (Teich and Weber, 2002; Conti et al., 2003; Kalhor and Zunoubi, 2005; Semmler et al., 2005).

Implications for power line ROW management

It is possible to provide attractive power line ROW habitat for forest ungulates with appropriate management (Bramble and Byrnes, 1972; Cavanagh et al., 1976; Mayer, 1976). Experimental studies on the management of power line ROW have been undertaken in the USA, where the application of herbicides appears to be a common practice to reduce tree regrowth (Cavanagh et al., 1976; Mayer 1976; Bramble and Byrnes, 1982; Ballard et al., 2002; Haggie et al., 2008) in addition to mowing, burning and fertilizing (Harlow et al., 1995). This may not be an option in other countries.

A stem height of 4 m for birch and 3.5 m for pine may be

be the optimal size to provide maximum ungulate forage (Kalén and Bergquist, 2004). Felling trees that reach heights of 5 m may ensure continuous high browse availability without complete removal of cover. The removal of single large trees would also create gaps for the growth of forbs, ferns and bilberry (*Vaccinium myrtillus*), which can be an important part of forest ungulates diet (Hjeljord et al., 1990; Mysterud and Ostbye, 1995; Krojerová-Prokešová et al., 2010). Alternatively, trees could be cut at a height of 1 m instead of full removal to shorten the period of low browse availability after clearing. Adjustments can be made dependent on the ability to reach the vegetation by the respective species (Garant and Doucet, 1995). Cutting during winter instead of summer resulted in higher browse production in spring, supposedly because of a better ability of plants to allocate reserves for regrowth (Garant and Doucet, 1995).

During construction, hinge cutting, that is, cutting deciduous large trees only half way through, would maintain cover availability and facilitate regrowth of forage (Global Wildlife Management, 2013). Less preferred coniferous trees like spruce (*Picea* spp.) could be removed entirely in favour of deciduous trees. Stable scrublands are more resistant to tree invasion, potentially reducing the need for clearing (Niering and Goodwin, 1974). The applicability of the suggested management techniques would have to be tested in the field.

Disclosure or conflict of interest

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Appendix A. Land area traversed by power lines and roads in Norway. Line lengths were derived from official statistics (Brunvoll and Monsrud, 2011; Statistics Norway, 2011a, b). *Lengths were retrieved from a road database (Norwegian Mapping Authority, 2012). Corridor widths are provided following Bevanger and Thingstad (1988) for power lines and E Englien (Statistics Norway, pers. comm.) for roads. The width of roads consists of road width plus road edge.

Area	Type	Length (km)	Corridor width (m)	Tied-up surface area (km ²)
Central power line grid	220 - 420 kV	7,907	38	300
	110 - 145 kV	10,407	25	260
	33 - 66 kV	9,868	18	177
	Total	28,182		738
Distribution power line grid	0.2 - 24 kV	165,789	Variable (ca. 5 - 10)	829 - 1,658
Central road network	Highways	6,639*	17	113
	National roads	20,837*	13	271
	County roads	27,281	9	246
	Total	54,757		630
Distribution road network	Local roads	38,591	8	309
	Private roads	75,453*	7	528
	Forest roads	48,571	7	340
	Total	162,615		1,177

Appendix B. Habitat types and their re-classification of the Norut vegetation map (Johansen et al., 2009), percentage of centage of central power lines and road grid length traversing different habitat types and the percentage of those habitat types of the area in Norway. Power line routing data from the central grid operator Statnett was clipped with the contours from the Norwegian land area. We determined the percentage of the line length routed through different habitat types with the function isectlinerst in the program geospatial modeling environment (Beyer undated). For a comparison, we did the same analysis with data of the central road network (Norwegian Mapping Authority 2012) including highways, national and county roads.

To capture habitats surrounding power lines and roads, we resampled the 30 × 30 m habitat raster to a 60 × 60 m raster. Raster cells that overlapped urban settlement polygons were classified as settlements. The percentage of the different habitat types of the Norwegian land area was the percentage of the respective raster cells. No accurate power line routing data of lower voltage power lines was available. We used ArcGIS version 10 (ESRI, 2011) to handle and modify spatial data.

Habitat	Class of the Norut vegetation map	Percentage of central grid power line length	Percentage of central grid road length	Percentage of land area
Forest	Dense coniferous woodland.	61	45	39
	Open coniferous and mixed woodland.			
	Lichen-rich pine woodland.			
	Low-herb woodland and rich deciduous woodland.			
	Tall-herb and tall-fern deciduous woodland.			
	Bilberry and small-fern downy birch woodland.			
	Crowberry downy birch forest.			
Lichen-rich downy birch forest.				
Mire and freshwater	Ombrotrophic hummock bog and lawns.	7	5	11
	Minerotrophic flat fen.			
	Hollow mire and open swamp.			
	Freshwater.			
Alpine areas and ridges	Exposed ridges, scree, talus and rock.	14	8	31
	Gras and wood-rush ridges.			
	Heather ridges.			
	Lichens, heather and pigmy birch			
	Heather rich lee side.			
	Grass and dwarf willow snow patch.			
	Late snow patch vegetation.			
Glaciers, permanently snow-covered areas and extreme snow patch plant communities.				
Grass and heather	Heather and fresh brushwood (lowland and mountain areas).	11	10	15
	Herb-rich grassland (lowland and mountain areas).			
Agriculture		5	21	3
Towns and villages		<1	11	<1
Not classified		<1	<1	<1

A photograph of two giraffes in a savanna landscape, used as a background for the journal cover. The giraffes are in the foreground, with one slightly behind the other. The background shows a grassy plain and some trees under a clear sky.

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