

Chapter 2

Literature Review

2.1 Biodiversity and ecosystem function

Human activities, which cause modification of the earth's surface, are leading to global changes, including, amongst others, accelerated climate change, soil and water perturbations, nitrogen deposition, erosion, land use change and the introduction and extinction of species. As a consequence, all ecosystems are experiencing directional changes with positive feedbacks that are leading to modified ecosystems (Chapin *et al.*, 1996). Natural ecosystems are sustainable over time and space when the system maintains its characteristic diversity of major functional groups, productivity and rates of bio-geochemical cycling (Chapin *et al.*, 1996).

Species composition and structure of any ecosystem, including its successional development, are not indefinitely sustainable and are therefore transitory assemblages (Chapin *et al.*, 1996). Time is a factor of sustainability and should be viewed over time scales of many generations and be relevant to human interactions with ecosystems. Ecosystems are not static entities of plant and animal species composition, ecosystem productivity or nutrient cycling, all of which will change in response to stochastic events and successional change. Ecosystems are governed by internal and external forces, which interact with four common factors of all ecosystems, namely local climate, soil, major functional organisms and disturbance regime (Chapin *et al.*, 1996).

The concerns of species loss and reductions in biodiversity become more apparent as the world population grows. Chapter 1 touches briefly on the effects of desertification, with particular reference to climate change and livestock farming. The question is at what point will the inherent structure and ecosystem function be reached when there is an unsustainable decline in the resilience of any ecosystem and, in this case, the desert margin ecosystems?

Walker (1992) suggested that the concern of biodiversity reductions observed in many regions of the world is focused on "charismatic megavertebrates", rather than lesser known and understood organisms, such as fungi or nematodes. Although the concern and conservation of the panda or elephant are laudable, these efforts are generally emotional or

moral and therefore not scientific. A further frequently referred to reason for concerns in biodiversity reduction is the commodity value of possible benefits from the pharmacological perspective, which have yet to be discovered. Walker (1992) appreciates there could be a possible hidden value somewhere in the biodiversity of an area, but without appropriate information these values cannot be given credibility.

There needs to be a change in the approach to managing and conserving biodiversity as in the past the emphasis has been placed on all species having a similar value. An understanding of the functional approach to biological composition is required. Walker (1992) suggested that biodiversity should be seen as the integration of biological variability across all levels of genetic and species composition within landscapes. The biodiversity declines that are currently being experienced worldwide should be seen in light of the reduction or simplification of biological heterogeneity from individuals to regions. The biological heterogeneity or, as Walker (1992) proposed, the “ecological complexity”, includes *inter alia* a combination of changes, some of which are listed below:

1. Genetic variation within a population, allowing for a wide range of genotypic responses to environmental conditions
2. Genetic variability between populations within a species
3. Species richness within a community
4. Species diversity, which is the number of species and the number of individuals per species
5. The abundances of functionally different kinds of organisms
6. Diversity as a consequence of speciation of ecological equivalents
7. Community diversity, which pertains to the numbers, sizes and spatial distribution of communities
8. Landscape diversity

The objectives of many current conservation efforts are to minimise the reduction of biodiversity, which includes the loss of species. However, the challenge is to understand the kinds of biodiversity that are most relevant to the functioning of the ecosystem. Alternatively, as Walker (1992) suggested, it is the amounts and types of biological simplification which will result in significant and irreversible changes in the inherent structure and function of an

ecosystem. Put more simply – which aspects of diversity and which kinds of species are most important to ecosystem function?

An understanding of ecosystem function was explained by Erhlich and Erhlich (1981) by being likened to the rivets on an aircraft, with each rivet contributing to the ability of the aircraft to fly. A limited reduction in the numbers of rivets will not prevent the plane from flying. The question is at what stage will the reduction of rivets eventually lead to the plane not being able to fly? This analogy has created much debate, as the implication is that all species are regarded as equal in an ecosystem's function. However, Walker (1991) referred to the main categories of species in an ecosystem as,

- ecological indicator
- keystone
- umbrella
- flagship
- vulnerable species

It is these categories of individual species that have been used to denote their importance in maintaining biodiversity.

When suggesting that some species are more important than others, Walker (1992) noted that ecologically some species are “drivers” while others are “passengers”. The level of importance of species is justified by observing the detrimental cascade effects as a result of the removal of a “driver” species and the apparent unchanged results to the ecosystem at the removal of a “passenger” species. The distinction between the importance of different species to ecosystem function is the critical point made by Walker (1992), who stressed that allocating levels of importance for different species does not imply that the less important species can be removed and are not of consequence to the system.

Another important point made by Walker (1992) was that there will be redundancies of species and by understanding this, the decline of biological diversity can be more clearly understood. Conservation efforts should be aimed at critically maintaining ecosystem resilience, which is the ability of the ecosystem to keep its patterns and functional processes. Rather than focusing on species in the efforts to maintain biodiversity and ecosystem function Walker (1992) suggested that the approach of conserving species functional groups, or guilds, should be the aim of conservation efforts. The proportional mix of species, which,

when combined with the functional diversity of biota, are an integral part of ecosystem function which when imbalanced will result in a deterioration of ecosystem processes (Fleischner, 1994). The consequences of biological simplification can lead to significant and / or irreversible changes to the inherent structure and function of an ecosystem, which leads to the unsustainable decline in resilience (Walker, 1992).

The use of avian guilds, which include feeding and nesting guilds, will be used in conjunction with individual species distribution when discussing the results of this study in Chapter 5.

2.2 Habitat heterogeneity and disturbance

The influence of disturbance, which is part of most, if not all communities, will result in these communities being in a state of disequilibrium (Connell, 1978). Disturbance creates an availability of new resources that will create competition between species for the new resource. Disturbance could also remove some resources, which could result in the demise of species, with the net result that there will be a progression of species invasions and replacement after a disturbance (Miller, 1982).

The size and frequency of disturbance are important when assessing the results of disturbance (Connell, 1978) and will have a significant role to play in the structuring of some communities (Miller, 1982). The Intermediate Disturbance Hypothesis (IDH), as suggested by Connell (1978), is a level of disturbance in a community that will produce the highest levels of species diversity (Connell, 1978). The disturbance frequency and intensity are factors of importance when assessing the IDH (Connell, 1978).

The current study analysed the effects of grazing intensity in north-eastern Botswana by assessing the impacts on four land use types during which the IDH was also considered. Miller (1982) took the IDH, as noted by Connell (1978), further by suggesting that the size of the disturbance will have a bearing on species diversity. Miller (1982) categorised groups of species within a community as competitive species and colonising species. Competitive species will eliminate other species in their quest for resources, but they will have a slow rate of growth and dispersal. Colonising species, on the other hand, will have high rates of dispersal and reproduction, which will allow them to take advantage of new resources made available by the disturbance. The division of species into these groups by Miller (1982)

suggests that these species have different life history strategies when confronted by disturbance. The disturbance that creates new resources will favour colonising species, but when a community is in a state of relative equilibrium, the resource availability is constant and the competitive species will have the edge over colonising species. The patch size or area of disturbance plays a role in the different categories of species put forward by Miller (1982), as the larger the patch of disturbance, the greater the perimeter of disturbance and the greater availability of the new resource. If the disturbance is evident in many smaller patches, the combined perimeter will be greater than the perimeter of a large patch of disturbance, with the net result being that there will more resources for the colonising species (Miller, 1982).

Ecologists have studied natural dynamics in ecosystems with the focus on successional development and the equilibrium of communities. The consequences of disturbance and the results on the evolutionary process have attracted attention since the early 1980's. The definition of a disturbance, provided by Pickett and White (1985), is "any relatively discrete event in time that disrupts ecosystem, community or population structure and changes resources, substrate availability or the physical environment". This gives readers an understanding of the meaning from an ecological perspective.

The five key factors of ecological disturbance described by Miller *et al.* (2011) are the intensity, timing, duration, extent and interval between disturbances. Miller *et al.* (2011) suggested a flood as a practical example of disturbance, with the depth of the flood being the intensity, the date of the flood is the timing, the inundation period as the duration, the maximum area being the extent of the flood and the time since the previous flood of this nature as the disturbance interval.

The current study assessed the impacts of land use, with particular reference to grazing intensity and its effects on the vegetation of the different land use types. The intensity, timing, duration, extent and interval between disturbances, as suggested by Miller *et al.* (2011), play a pivotal role in the results of the study. Todd and Hoffman (1999) believed that livestock can and will have a detrimental effect on the biodiversity of a region, while overgrazing can be regarded as the dominant factor that reduces biodiversity.

2.3 The effects of land use, in particular grazing, on vegetation

Disturbance is inherently a multi-dimensional and multi-faceted phenomenon (Miller *et al.*, 2011) and although disturbance regimes can have many effects on community diversity, these effects can only be understood by studying the different aspects of disturbance. Community dynamics, as suggested by Savory and Butterfield (1999), reveal a kaleidoscope of changing patterns within a supposedly stable community. These communities can be altered by disturbance over a short period of time. This phenomenon is most evident on islands with the introduction of predators or grazers, which have disrupted the community dynamics in a short space of time (Savory & Butterfield, 1999). The intensity, timing, duration, extent and interval between disturbances, as suggested by Miller *et al.* (2011), play a pivotal role in the results of the disturbance. The effects of land use, in particular grazing, are addressed as the major form of disturbance in the study area. Rapid changes in time and space undermine the concept of equilibrium between available fodder and grazing pressure (Rowntree, *et al.*, 2004).

Vegetation response to disturbance is complex and does not follow the pattern of reduced cover or diversity (Abel, 1993). In their book on Holistic Management, Savory and Butterfield (1999) suggested that, when confronted with a problem, in this case the impacts of grazing intensity on vegetation, the cause and effect is never a simple chain of events, but a mesh extending infinitely in all directions. Tainton (1999) proposed that the savannah vegetation of southern Africa, for example, has a unique set of circumstances, which interact with one another. These include physical determinants, biological interactions and individual species properties that are affected over space and time. The interplay between these factors during a disturbance will determine the equilibrium reached after the disturbance has passed.

The disturbance in this study is the intensity of grazing on the vegetation, particularly grasses. The consequence of leaf removal on grasses is central to this study, as the effects on plant growth will depend on whether the whole or only part of the plant is removed by grazing and whether the basal meristematic tissue remains intact. The method, timing and intensity of defoliation will determine the effects of the defoliation on grasses and the grass sward as a whole. Repeated removal of leaves can reduce the capacity of the grasses to produce roots and root penetration. The removal of the basal meristematic tissue will result in the termination of growth of the grass (Tainton, 1999). The removal of the apical meristem, on the other hand, can result in more light reaching the lower basal nodes of the grass and also making carbohydrates available that would have been used by the apical meristematic tissue.

The consequence will be that grasses will then produce tillers from basal nodes. Although the removal of the apical meristem will result in the prevention of further growth of the grass, there will be an improved quality of the leafy structure of the grass, compensating for the reduced yield. The removal of leaves by grazing will reduce the capacity of the grass's ability to photosynthesise, which will reduce the carbohydrates available for the growing and storage parts of the grass plant (Tainton, 1999; Savory & Butterfield, 1999).

Different grasses react differently to defoliation and defoliation rates, but they are adapted to a degree of leaf removal. Grasses need a certain amount of grazing as they are inclined to become moribund, reducing new growth. Leaf removal results in a slowing down or complete halt to root growth, depending on the severity of defoliation. The activity of the grass roots will increase as the removal of leaves is increased in height from the root zone and as the removal intensity increases (Tainton, 1999). Not all grasses react in the same manner to grazing intensity. Perennial grasses are inclined to be grazed first as they are more palatable, leading to increased grazing intensity. The consequences of intensive grazing will lead to a reduction of these perennial grasses' ability to recover, with annual grasses replacing them after their eventual demise. Annual grasses, although less palatable, are inclined to replace perennial grasses that are overgrazed (Abel, 1993; Abel & Blaikie, 1989; Du P. Bothma, 1989; Tainton, 1999).

Although trees and shrubs do not constitute a significant part of cattle diet, they do play a role in the maintenance of livestock during the winter months when most trees are in their annual dormant state and when grass cover is generally low (Tainton, 1999; personal observation). Little research has been conducted on the effects of browsing when compared to the research that has been conducted on the effects of grazing by domesticated livestock on herbaceous cover (Savory & Butterfield, 1999). Unlike grasses, which have the ability to protect their basal meristematic tissue from being grazed, the meristematic tissue of trees and shrubs is apical and susceptible to grazing pressure. However, like grasses, the method, timing and intensity of browsing are important when assessing the effects on trees and shrubs, which have defence mechanisms, such as producing tannin and thorns, to reduce the browsing intensity of cattle. The tree height also protects trees from being over-browsed, with the browse lines being very evident in wooded areas where livestock are farmed in marginal regions (Savory & Butterfield, 1999).

2.3.1 Overgrazing

The carrying capacity of vegetation and overgrazing are controversial topics with many different views on the topic: some authors assess the carrying capacity from an environmental aspect (Rowntree *et al.*, 2004; Savory & Butterfield, 1999) and others from a social and economic perspective (Abel, 1997; Barnes *et al.*, 2008).

Overgrazing is regarded as the position where grazing rates of intensity are in excess of the production rates of forage vegetation production (Rowntree *et al.*, 2004). Degradation of grazing areas, as suggested by Abel (1997), is the irreversible decline of productivity of vegetation and should be considered, not only from the number of cattle utilising an area, but in conjunction with rainfall. Abel (1997) continues by suggesting that if the effects on the vegetation are reversible, degradation has not taken place, but when the degradation has reached a level where loss of soil through sheet and rill erosion has been reached, then degradation has taken effect. Abel (1997) used a model to test his hypothesis regarding overgrazing and suggested that, although models are simplifications of reality, they can be used to assist in forming a conclusion.

There are many variables to consider when assessing the effects of grazing, particularly in marginal areas. Drought is a factor that is unpredictable, yet a certainty in marginal areas, which, when combined with grazing intensity, can have devastating impacts on the vegetation. When a drought breaks, the exposed land is susceptible, as rains often fall erratically and at high energy with erosion being considerable. The impacts of erosion are far reaching on future plant growth (Abel, 1997). Abel and Blaikie (1989) pointed out that calculating a stocking rate for marginal areas with erratic rainfall would be difficult, as when the rains fall in summer, the stocking rate will be higher than in winter when there is no rain, with the ability of the vegetation to support grazing animals being reduced.

Abel (1997) concluded that, when managing a complex and unpredictable system, the policies and resource management need to be flexible when addressing stocking rates – these policies should be socially and ethically based, involving stakeholders and not simply technically determined. Light grazing, as an alternative to heavy grazing, could have a positive effect on herbaceous species richness, but as the intensity increases, the species richness will be reduced (Joubert & Ryan, 1999; Nsinamwa, 2005).

Savory and Butterfield (1999), on the other hand, analysed overgrazing from a holistic viewpoint and suggested that overgrazing cannot be attributed to one factor, as each environment has a different set of circumstances, including soil, weather pattern, vegetation cover and history of land use. Environments were classified by Savory and Butterfield (1990) on a scale from brittle to non-brittle, with the most brittle environments having the lowest humidity and precipitation with a long period of non-active growth, while the most non-brittle environments had the highest humidity and precipitation, with bare ground becoming more evident as brittleness increases. The more brittle the environment, the more susceptible it is to overgrazing. Stocking rates had little bearing on what happened to individual plants as grazers, cattle in particular, are selective feeders, grazing the most palatable grass repeatedly. The net result is the sward will appear patchy, with areas of ungrazed grass between bare areas where the palatable grasses had been completely eaten (Savory & Butterfield, 1999). Savory and Butterfield (1999) confirmed the findings of Miller *et al.* (2011) that timing is critical when assessing overgrazing in general. Timing includes the length of time cattle graze an area, as they graze different plants at different times of the year and grasses recover at different rates on different parts of the land. Time, rather than cattle numbers, governs the ultimate impact of overgrazing (Savory & Butterfield, 1999).

Grazing pressure and overgrazing, as an ecological disturbance, should be assessed within the parameters of grazing intensity, timing, duration, extent and interval between the grazing disturbances (Miller *et al.*, 2011). The value and understanding taken from Miller *et al.* (2011) is central to the current study. Timing, duration and interval are all factors of time relevant to the grazing disturbance.

2.3.2 Bush encroachment

The increase in woody plant density in southern Africa has resulted in there being less grazing land available for livestock, with the reasons for the increase being many, diverse and complex (Tainton, 1999). Some of the reasons for the increase in woody vegetation include climate change, involving changes in temperature, rainfall and soil types, which Tainton (1999) regards as determinant factors, while the other factors include fire, the impact of herbivores and human activities. The warmer, drier climates that have been experienced over the last century have also favoured woody vegetation (Smiems, 1983). The impacts of

secondary determinants influenced by human impacts can be addressed in the short term, while climate change and soil types will take generations to change (McClellan *et al.*, 2005). The transition from grassy to a shrubbier ecosystem induced by overgrazing is known as bush encroachment (van Vegten, 1983)

There is a balance between the grasses and woody vegetation component of the vegetation in the natural savannah bushveld areas of southern Africa (Du P. Bothma, 1989). This delicate balance was maintained by a moderate grazing regime that was in place before domesticated stocks were introduced. Wild herbivores roamed the grassy plains, grazing as they went. As the grazing condition was reduced, the animals moved to alternate areas, leaving grazed areas to rest and recover (Savory & Butterfield, 1999). The introduction of domesticated stock, and subsequent creation of fenced paddocks and artificial waterholes, has upset the balance between grazers and grasses (Savory & Butterfield, 1999). Grazing movements are now restricted with intensity, timing, duration, extent and interval between the grazing disturbances, all factors that contribute to changing vegetation structure (Miller *et al.*, 2011).

When overgrazing upsets this balance, there is a shift towards woody vegetation and bush encroachment becomes evident. If the grazing is subsequently reduced, the vegetation does not recover to the original grass sward (Walker *et al.*, 1981). There may also be a shift to woody vegetation when there is a change in rainfall, with years of good rain favouring the grasses and years of low rainfall favouring the woody vegetation (Du P. Bothma, 1989). There is no standard rule for the changes in vegetation in the semi-arid areas as there are structural and environmental differences in these areas, which, when combined with the different management practices, can lead to different results. With the increase in domestic stock, the grass cover is significantly reduced in these semi-arid areas. At the onset of the rains, the earth is exposed as a result of the overgrazing and water infiltration is reduced to such an extent that there is extensive water run-off and sheet erosion occurs (Kelly & Walker, 1976).

Nsinamwa (2005) noted that grazing in excess of 80% of perennial grasses resulted in a significant reduction in the basal cover of these perennials, while at 40% the basal cover was not adversely affected. Moderate grazing, on the other hand, could have a beneficial effect on the biodiversity of the area concerned. Nsinamwa's (2005) results therefore support the IDH suggested by Connell (1978), as discussed earlier in this chapter. Dominant species could be defoliated, creating an environment suitable for other species to become more

evident and therefore increasing the diversity of the moderately grazed area. However, it is not grazing alone that has an effect on the biodiversity of the managed area. Rainfall, and particularly the lack of rainfall, combined with grazing can have a collective effect on the vegetation and, subsequently, the biodiversity of the area (Abel & Blaikie, 1989; Maitima *et al.*, 2009; Nsinamwa, 2005; Rowntree *et al.*, 2004).

The net result of overgrazing will be less water infiltration into the top soil as grasses are removed. Future grass growth will be reduced as top soil will contain less water, while vegetation with a deeper root system, such as trees and shrubs, will have access to subsoil water and will thrive, resulting in bush encroachment (Walker *et al.*, 1981).

In his study in eastern Botswana, van Vegten, (1983) found that in a relatively short time bush encroachment changed the landscape significantly as a result of overgrazing, with an increase in woody biomass tripling in the space of 25 years.

The study by Buffington and Herbel, (1965) in southern USA noted that there was very little woody vegetation, called mesquite, on the open grasslands prior to grazing by livestock in the early 1880's. Intensive grazing has resulted in a considerable increase in the woody vegetation and a reduction in grass cover.

The most important factor when ascertaining the impacts of soil erosion is vegetation cover, which includes plant species and biomass with a 30% ground cover often being suggested as the threshold between erosive and non-erosive conditions (Thornes and Francis, 1990). One of the crucial factors that affect water infiltration is the biomass of grass at the onset of rains. As the grass cover is removed, there is less water infiltration into the top soil, which then affects the growth of grass (Walker *et al.*, 1981). Kelly and Walker (1976) calculated that the infiltration rate is reduced tenfold on bare ground when compared to grass covered ground.

Trees and bushes, on the other hand, act as traps for water in the same semi-arid areas. As the rain falls onto the trees, the impact of the raindrop is lessened and the water is then filtered down the branches and stem, into the soil. Pressland (1973) studied the effects of tree cover on the infiltration of water and found that there was a six-fold increase in the infiltration of water below a tree when compared to open ground.

The CO₂ increase in the atmosphere has been an important factor in the increased rate of climate change (Gonzalez, 2002), but the increase also has an effect on the problem of bush encroachment. Grasses use different chemical pathways to access CO₂ for

photosynthesis, while trees and bushes use a chemical pathway, which permits them to access more of the available CO₂, providing a positive feedback. Trees and shrubs are able to access more CO₂ to their benefit, giving them a further the edge over grasses (Tainton, 1999).

Reference is made to studies that specifically considered the effects of agricultural land use on the vegetation in Africa. In their study on the effects of cattle ranching in the sandveld region of Botswana, Perkins and Thomas (1993) noted that the impact on semi-arid environments was significant, as these areas have a high sensitivity to disturbance with a very low recovery rate after the grazing disturbance (Perkins and Thomas, 1993). However, Shackleton (1993) found that once the grazing pressure has been removed, there is a certain amount of resilience of the vegetation of heavily grazed areas in South Africa

Perkins and Thomas (1993) stressed that generalisations regarding semi-arid areas should be avoided as many of these marginal semi-arid areas have differing climates, history and vegetation communities. Secondly, they emphasised that the physical environment, which includes soil quality, soil erosion rates and community structure, in marginal regions will possibly respond to disturbance and degradation at differing rates with differing sensitivities. The study was conducted in the sandveld region of Botswana and concluded that the extent of environmental change as a result of excessive grazing in the Kalahari would increase with the exposure to grazing. These changes would include reduced nutrients in the soil, increased soil compaction and reduced water filtration, increased water and wind erosion with a decrease in grass cover and an increase in woody vegetation (Perkins and Thomas, 1993).

Nsinamwa (2005) also conducted an influential study in central Botswana, noting the results of grazing intensity on the vegetation. Much of the area studied was unfenced, with cattle movements restricted only by the distance they had to travel from the nearest watering point. Of the four land use types in the current study, the CLM and the RLM have a similar management approach when compared to the Nsinamwa (2005) study, as both areas have no fences and only the distance the livestock have to travel to water every day restricts livestock movement. The findings presented by Nsinamwa (2005) are summarized below:

- The intensity of grazing will have an effect on bush encroachment (Abel & Blaikie, 1989; Nsinamwa, 2005) although there is some doubt as to the timing and causes of this ecological transition. Intensive grazing, whether controlled or uncontrolled, will

have an effect on the fodder availability, which can lead to bush encroachment. The net result would be an increase in the ratio of bush to grass, with bush species, such as Sickle-bush (*Dichrostachys cineria*), and Black thorn (*Acacia melifera*), being very common encroachers, while palatable perennial grasses would decrease and be replaced by less palatable annual grasses. The change in vegetation structure as a result of intensive grazing will have an effect on the ecology of the area and ultimately affect the natural fauna (Nsinamwa, 2005). Encroachers, such as Brandybush (*Grewia flava*), Bastard brandybush (*Grewia bicolor*), Shepherd's tree (*Boscia albitrunca*), and Sickle-bush, can have a beneficial effect for some animal species as they provide valuable feed in the form of leaves, seeds and nectar (Pulgrave, 1981).

- Grazing pressure can affect the spectrum of the herbaceous layer in three characteristic ways. Nsinamwa (2005) classified the herbaceous layer as decreasers, increasers and invaders when he assessed the use of these plants from a grazing perspective. Decreasers initially make up a large portion of the herbaceous layer and will be heavily grazed, but respond differently to grazing as a result of the defoliation tolerance and the level of utilisation. Increasers may be more tolerant to defoliation and/or be less frequently grazed as a result of being less palatable. The grazing pressure on the decreasers leads to reduced basal area, root biomass, mulch levels and leaf area, which ultimately leads to the creation of establishment gaps benefiting other growth and life forms (Nsinamwa, 2005). These growth forms are called invaders. They are typically annuals, which are less palatable, being less favoured by stock as they are generally lower in nutrient value, having low, erratic or highly seasonal productivity.

The effects of land use change on the biodiversity in many parts of East Africa have been significant. Maitima, *et al.* (2009) studied the effects of pastoral and crop farming on the biodiversity of rural areas of East Africa. This is relevant as much of the farming practiced in the current study area involves both livestock farming and the growing of crops. Maitima *et al.* (2009) concluded that as the demand for space increases for agriculture, the biodiversity of these areas is under threat. Agricultural land changes the land use from natural vegetation to crop and livestock farming. Crop farming in East Africa had a more significant effect on biodiversity than livestock farming as much of the natural vegetation is removed and replaced

by monocultures. However, crop farming when practiced with a variety of crops has less of an impact on the biodiversity of the area. Livestock farming, on the other hand, had less of an impact on biodiversity in East Africa as much of the natural vegetation is maintained. Nevertheless, there was a shift in the vegetation species (Maitima *et al.*, 2009). When crop and livestock farming were farmed in parallel, the inputs of livestock farming through the use of dung reduced the impacts of crop farming and contributed to maintaining biodiversity. The reduction of tree cover for crops, fencing and fuel had a negative impact on bird species diversity in the study area which was confirmed in studies by du Plessis, (1995) and Maitima *et al.* (2009). Well-wooded areas harbour more bird species than do open grasslands (Maitima *et al.*, 2009). As land use intensified to moderate levels, there was an increase in small mammals and birds as a consequence of increased habitat diversity, but as the land use intensity increased further, the habitat diversity decreased (Maitima *et al.*, 2009), which was confirmed by Connell's (1978) IDH.

The impacts of arable and pastoral farming on the soil have been noted by Maitima *et al.* (2009). He found that soil productivity declined as farming intensity increased, with the most important impacts being:

1. Reduced organic matter and soil biological activity
2. Degradation of soil structure
3. Reduction of the macro-nutrients and micro-nutrients
4. Increased toxicity due to acidification and salinisation

2.3.3 Soil erosion

The current study analysed the effects of agricultural land use on birds, with a particular reference to the impact of livestock on vegetation. The consequences of cattle grazing on the vegetation are significant, with increases in soil erosion most evident. There are two types of soil erosion, the first being natural or geological erosion and the second being manmade or accelerated erosion. The former is a slow continual process, while the latter is a much faster process, which has become the dominant form of erosion (Tainton, 1999). There are a complex set of inter-related issues, including economic and social, that contribute to high

rates of accelerated soil erosion. Amongst these issues suggested by Taiton (1999) that contribute to soil erosion are some that are particularly relevant to the current study:

1. Degradation of plant cover and composition as a consequence of overgrazing
2. Highly erodible and shallow soils in many of the degraded areas
3. Over-exploitation by farmers as a consequence of poverty

As a consequence of land use, soil erosion is important because, as the levels of soil loss increase, the recovery of the soil potential decreases (Abel, 1997; Savory & Butterfield, 1999; Tainton, 1999).

There has been much debate regarding the results of soil erosion as a consequence of overgrazing. It has become widely accepted that livestock farming in the sensitive areas of the Kalahari in Botswana has led to severe range degradation and desertification (Cooke, 1985). Tainton (1999) believes that throughout South Africa plant cover has declined under pressure of the traditional stock farming systems. Poor grazing practice, overstocking and the dominance of small stock have been responsible for this decline in plant cover. The primary concern of managers of natural vegetation is its nutritional value, which is a function of species composition and biomass (Abel, 1993). Tainton, (1999) confirmed that the primary aim of land use managers was the production of livestock, but added that this objective should also include the development of a dense and stable plant cover to effectively control soil loss. Grazing areas, particularly communal areas, are often perceived to be vulnerable to overgrazing and degradation as a result of poor management (Abel & Blaikie, 1989).

Rowntree *et al.* (2004) suggested that, when assessing the impacts of livestock on the vegetation, the historical land use should be assessed as a contributor to soil impact. Vegetation cover is the result of a complex set of interactions between climate, soil, topography and herbivore usage (Rowntree *et al.*, 2004). Small fluctuations in rainfall and grazing, with a suitable time lapse, can lead to variations in fodder availability. Rowntree *et al.* (2004) concluded that although there appears to be a correlation between grazing and erosion, this was not always the case; when analysing the history of the eroded areas, it is evident that many of these areas of erosion were abandoned cultivated fields prior to being put to grazing. Abandoned cultivated fields were largely unvegetated, with reduced bio-physical attributes, which predisposed the soil to severe erosion at the onset of the rains. Soil

erosion cannot be simply assigned to overgrazing. Although it is often blamed for widespread and severe erosion in dry land areas, it should be assessed by recognizing the dynamic nature of the relationship between grazing pressure, vegetation cover, soil moisture and the historical pattern of land use (Rowntree *et al.*, 2004).

It has been noted previously that the effects of livestock farming as a land use cannot be seen in isolation, particularly in marginal areas, such as is the case in the current study area. Drought, as a contributory impact on the condition of the vegetation in marginal areas, is significant and has been compounded by injudicious grazing practices (Tainton, 1999; Tietema *et al.*, 1991).

As the condition of the natural vegetation deteriorates and soil erosion increases, there is an increased risk of “manmade” drought. Man-induced droughts are evident when rainfall recorded during the rainy season is normal, and yet the condition of the vegetation deteriorates to such levels that there appears to be a drought (Tainton, 1999). Droughts caused by repeated years of low rainfall contribute to reduced plant cover and are compounded by heavy grazing pressure (Abel, 1993). Droughts in southern Africa generally end with severe thunderstorms in the summer rainfall areas. The impact of rainfall on bare earth has been noted previously (Elagib, 2011), with positive feedback increasing the risk of man-induced drought. The erosion potential of water after heavy storms is significant. When combined with grazing pressure during the recovery period of grasses, this further intensifies the erosion effect. Hoof action and grazing prevents surviving plants from reaching leaf canopy (Tainton, 1999). The maintenance of plant cover and condition of the vegetation appear to be the most important factors controlling water run-off and soil erosion (Elagib, 2011; Tainton, 1999).

2.3.4 Fuel wood removal

Grazing is the primary land use of the current study area, with differences in intensity and timing being most evident. However, there is a possibility of a further impact on the vegetation not directly related to livestock grazing. Many rural pastoral farmers depend on the woody vegetation for cooking, heating, construction and fencing material, which is particularly evident in the ILM, RLM and CLM areas.

Although few studies have been made on the effects of fuel wood removal, the impact on bio-diversity is significant (Joshua & Johnsingh, 1994). Mwampamba (2007) conducted a

study on the manufacture of charcoal and the effects on the forests of Tanzania and concluded that scanty data on forest loss and forest recovery were the biggest impediment to understanding the impacts on bio-diversity.

African rural communities obtain a large portion of their energy requirements from fuel wood collection in the area they live. The figure for South Africa is 12% of total energy requirements, while the figure for other African countries, such as Tanzania, Mali and Zambia, is as high as 90% (Kalapula, 1989). Not all forest resources are under threat as a consequence of wood collection; some authors suggest that the demise of forests in Africa is primarily the cause of agricultural land expansion rather than from the collection of wood for fuel and construction (Foley, 1985). However, Kalapula (1989) concluded that the results of uncontrolled fuel wood collection will lead to a chain of events resulting in soil erosion, and sedimentation of dams and rivers, which will lead to flooding and drought, with wood for fuel thus being in short supply, ultimately causing widespread malnutrition.

Thirty five to 50% of birds and other animals in Africa rely on cavities for roosting and nesting, with these cavities being adversely affected by the reduction of habitat as a consequence of fuel wood collection (Du Plessis, 1995). When comparing a riverine forest that was protected from fuel wood collection to a riverine forest where fuel wood collection occurred daily, Du Plessis (1995) noted some marked differences. Both vertebrate and invertebrate species were affected adversely as a result of fuel wood collection, which could lead to localised extinction of some species (Chadwick *et al.*, 1986; du Plessis, 1995).

Birds, particularly cavity-nesting birds, are seriously compromised by the reduction of deadwood, but there was a further effect on the foraging substrata which compromises the bird species that glean from this stratum. Primary cavity nesters, which excavate the cavities, as well as secondary cavity nesters, which use excavated cavities, were adversely affected. Jackson and Jackson (2004) studied woodpecker nests, active and abandoned, and noted that they provided a suitable environment for secondary nesters and many other organisms, thus making these nests an important part of the dynamics of the environment. Woodpeckers can and do make a number of attempts when excavating a nest as not all trees are suitable for nesting sites, with dead trees or dead limbs of living trees more suitable than living trees and living tree limbs. All attempts at creating nest sites create opportunities for other animals for nesting and roosting sites. The excavation attempts will also attract fungi, which will inhabit

these moist cavities. This could in turn cause limbs of living trees to die (Jackson and Jackson, 2004).

Not only were birds adversely affected by fuel wood removal, but also other vertebrate and invertebrate animals that depended on dead wood. Natural cavities provide valuable nesting and roosting sites for many species of birds which when included with the birds that feed amongst the dead wood strata, sum to a significant number of bird species that can be adversely affected by the collection of wood for fuel and construction (du Plessis, 1995). The results of the Du Plessis's (1995) study showed that fuel wood collection will have a significant negative impact on the dead wood availability, not only as substrata for cavity nesting, but also as foraging substrata.

Sustained heavy utilisation of fuel wood collection can result in the local extinction of some species that depend on dead wood for their survival (Maurer, 1984). This was confirmed by Du Plessis (1985) who recorded an extinction of Cardinal Woodpecker (*Dendropicos fuscescens*) and Southern Black Tit (*Parus niger*) from the utilised forest when compared to the unutilised forest. The study by Chadwick, *et al.* (1986) in New England, USA suggested that the time lapse between the fuel wood removal and recovery in forests would permit different bird species to take advantage of different emerging vegetation strata, resulting in a positive effect on species diversity and a significant increase in bird species composition.

Du Plessis (1985) concluded that removal of dead wood as a stratum for foraging will affect a wide variety of species found in a range of taxa. Invertebrates, in particular, may be directly affected by the removal of dead wood, but the indirect impact will be felt by a host of other species that feed on these invertebrates. Vertebrate cavity nesting species will be adversely affected by the removal of dead wood, impacting the availability of food, foraging micro-sites, nest and roost cavities and ultimately residency of these species (Du Plessis, 1985).

2.3.5 Habitat Fragmentation

The edge effects that result when there is a transition from one ecological type to another have for many years been encouraged by wildlife practitioners as areas of high biotic diversity (Harris, 1988). Transitional areas between biomes will yield species diversity greater when compared to individual biomes since the transitional zone will be visited by species from both biomes (Harrison & Martinez, 1995). Different land uses will inevitably lead to edges being created as a result of significant vegetation changes, caused in this case by grazing as the major land use.

The increase in animal species can be attributed to a combination of the species of each of the two plant communities, included with the species that use the ecotone as their preferred habitat (Gates & Gysel, 1978), with the net result that these ecotones will have a higher species richness. However, induced fragmentation as a result of human activities may not have the desired effect of creating edges that are suitable for increased biodiversity. Created edges may result in decreased species composition and richness (Malan, 2001).

Harrison and Martinez (1995) found that it was the core areas of biomes rather than edges that provided the selection pressures, which were necessary to maintain gene pools peculiar to that biome. Population density was inclined to drop off towards the edges of biomes as species range of the required habitat decreased, due to deterioration, and competition increased.

The more heterogeneous a vegetation, the greater is the species diversity (McGarigal & McComb, 1995). A heterogeneous landscape implies an area which is made up of interacting patches which affect the distribution of a species, in this case birds (McGarigal & McComb, 1995). Habitats found within this heterogeneous landscape will have a strong influence on the numbers and distribution of species. When habitats are fragmented, there is an alteration in the patterns of distribution of the vegetation, which can lead to a dividing of the populations of animals. This division of the population may affect the viability of the population being examined (McGarigal & McComb, 1995).

Cary and Temple (1988) used a computer model in their study on the effects of fragmentation on bird populations. They worked with the variable fecundity of bird populations as a function of the distance from the ecological edge. As the landscape became more fragmented, the breeding success of the bird populations dropped significantly. There was a

geometric increase in population in the proportion of forest habitat that is near the ecotone. This is a direct result of increased fragmentation. The consequence of continued fragmentation is that in time there will be little or no habitat interior. The effects of predation, competition and parasitism that were more evident at forest edges become more meaningful as they have a direct effect on the dynamics of bird populations. The results of the computer model compared favourably with actual results made in fragmented forests in southern Wisconsin (Cary & Temple, 1988).

Laurance (1991) studied the edge effects in forests in Queensland, Australia, with reference to the changes in vegetation in newly fragmented tropical forests, and found that disturbance or change in the original vegetation was observed at least 500 metres into the forest, with the most marked differences being within the first 200 metres of the created edge (Laurance, 1991). Laurance (1991) recorded marked changes in edges created in forests, with relative humidity being reduced between 5 and 20%, as well as significant changes in air temperature. These microclimate changes near induced edges created suitable conditions for pioneer and edge species, which ultimately could have resulted in under-story birds being less evident in the first 50 metres of the edge and light-loving butterflies being found 200 - 300 metres in the interior of the forest where they are not generally found (Laurance, 1991). Induced edges result in a change in the forest canopy, changes in the microclimate with significant increases in temperature, lower soil moisture content, as well as an increase in the penetration of light. These alterations in normal interior forest conditions can affect under-story birds, bats, rodents and other small mammals and butterflies (Laurance, 1991). Laurance concluded by confirming that the disturbances resulting from created forest edges were a function of distance from the edge, with different species being affected at different distances from the created edge (Laurance, 1991).

Andren and Angelstam (1988) tested predation of bird nests and artificial bird nests relative to forest edges and noted predation decreased the further away from the forest edge nests were found (Gates & Gysel, 1978). If, as a result of reduced area, the forest fragment becomes so small that the centre of the forest is regarded as part of the edge, there would be a decrease in successful bird nests in the induced forest edge and an extinction of interior forest bird species (Andren & Angelstam, 1988).

Rare bird species, and particularly forest-interior species, are most detrimentally affected by forest fragmentation. The reason could be that interior-forest bird species are less

prone to cross open patches than forest-edge species. As a forest becomes reduced, forest-interior birds tend to become extinct, while forest-edge species are more inclined to survive. Tropical forest under-story bird species are susceptible to extinction relative to their abundance and distance from the forest edge, with forest-interior birds being the most vulnerable (Krauss *et al.*, 2003; Newmark, 1991).

2.4 Requirements of birds

Birds are confronted by many factors that influence the environments in which they live. When combined with climate change and habitat modification, these create an array of possible impacts that affect their lifestyle. If these factors are significant and influence the bird's ability to survive in that environment, they have the ability to leave an area. Species composition of the area will change as a consequence of these impacts (Hockey, 2003).

However, alterations to the environment need not necessarily be to the disadvantage of all species as changes create new resources, providing the opportunity for different species to utilise these new resources (Hockey *et al.*, 2011). Reference is made to the IDH, as suggested by Connell (1978), who concluded that intermediate levels of disturbance could result in increased species richness as new resources become available because of disturbance.

Birds have certain requirements in order to survive and reproduce. These include water, food and nesting sites, which are encompassed in vegetation structure. Predation and competition will influence survival and reproductive success (Evans, 2004). These conditions of survival and reproductive success can be referred to as the bird's 'ecological niche', which relates to the organism's tolerances and requirements. The niche describes 'how' as opposed to 'where' the organism, in this case birds, live (Begon *et al.*, 2006). The niche requirements for different bird species are varied, but most species require common inputs, such as nesting sites, food, water, suitable vegetation structure, intraspecific and interspecific competition, as well as predation, all of which influence the distribution of birds (Hockey *et al.*, 2005).

The niche requirements listed below should not be regarded in isolation as a combination of these requirements form the niche.

- The increase in complexity of vegetation structure, particularly with the number of trees, creates an environment with more opportunities for bird species (Lack, 1986; Willson, 1974). Penry (1994) refers to the vegetation structure as 'physiogamy', which can be influenced by rainfall, drought, fire and flood, and is the shape and character that will determine the avifauna found in the area. As the vegetation structure is modified, different bird species will be affected in different ways (Hockey *et al.*, 2011; Penry, 1994) with possible redistribution becoming more evident.
- Nesting sites will influence the reproductive success of different birds, which have different nesting requirements (Hockey *et al.*, 2005). Habitat modification will impact on breeding success, as predation rates can be influenced by creation of artificial edges. Food items could be reduced, compelling parents to travel further to find suitable chick food or substituting inferior food quality to the detriment of the chick. Nests could be unguarded while parents are foraging for longer periods, exposing chicks to predators and also possible abandonment (Evans, 2004).
- Water availability in marginal areas is important to many bird species that need to balance water loss with water intake, with many species requiring water on a daily basis. There are bird species that do not require water on a regular basis, while there are others, such as insectivores, that do not need water at all as they obtain their requirements from their prey (McKechnie, 2007). Granivorous species of birds, rodents and ants are an important part of the marginal faunal community as many of these species receive sufficient water from the seeds they consume. MacMillan (1990) suggested that granivorous birds in marginal areas, particularly those of low mass, achieve a water balance from their seed diet. Water requirements will influence different bird species and therefore influence their distribution (Hockey *et al.*, 2005).
- The availability of food will impact on the distribution of different feeding guilds with some guilds influenced more than others. For example, carnivores that have a large percentage of flesh as their diet will have larger home ranges than carnivores that consume less flesh in their diet (Gittleman & Harvey, 1982). Little is known about the influence of arthropods in the insectivorous birds' diets

(Moorman *et al.*, 2007). Insectivores and granivores could be influenced by food availability, but as insects and grasses are widely distributed in marginal areas, the influence may not be as evident. Fruit-eating birds, however, are more specialised and it has been noted that frugivores will be influenced by fruit availability (Doherty *et al.*, 2000; Lack, 1986).

- Predation will influence avian distribution. Evans (2004) addressed the impacts of habitat modification with particular reference to predation on bird nests. Reduced availability of chick food will cause parents to travel further and expose chicks to predation for longer periods (Evans, 2004). However, Doligez *et al.* (2003) noted that there was a reduction in clutch size of some species of birds when confronted by increased rates of predation. This would permit a quicker turnover of chick rearing, contributing to better breeding success (Doligez *et al.*, 2003). The exposure of chicks to predators was increased when the parents were foraging and the chicks begged for food (Evans, 2004). Briskie *et al.* (1999) observed that chicks' begging time was reduced and their begging tone was altered in such a way as to be less detectable by predators.
- Intraspecific and interspecific competition influenced the distribution of birds in the Andes Mountains of South America (Terborgh & Weske, 1975). This was confirmed by Svanback and Bolnick (2007) in their study on intraspecific competition where it was noted that competition increased the dietary range, promoting niche variation within a population. Weiner (1994) also suggested evolutionary adaptations in birds when testing the intraspecific competition between the finches on the Galapagos Islands when there were changes in food availability.

2.5 Indicators of land use change

2.5.1 Biological indicators

The task of monitoring changes in the environment presents land managers with a multitude of potential variables to measure. The literature cited suggests that there will be a mosaic of factors confronting birds when attempting to choose a habitat in which to breed. The effects of

land use on the environment can be considerable and when considering wildlife, it is clear that land bird monitoring is an economical, flexible and effective monitoring tool (Alexander, 2002). Indicators can quantify complex phenomena in a simple manner and can be developed and updated easily without much cost (Gregory, *et al.*, 2005).

Birds have the ability to change their distributions and abundances rapidly and, as a vertebrate group, will be valued indicators of environmental change (MacDonald, 1992). When compared with mammals, particularly large mammals in South Africa, birds are valued indicators as most mammals are confined to game parks (Siegfried, 1989). Birds on the other hand, have a good historical record (Siegfried, 1989) and, when compared to mammals over the same time period, birds have shown reduced reductions (MacDonald, 1992). Lawton, *et al.* (1998) noted that the use of a single and popular group of animals, such as birds and butterflies, as bio-diversity indicators could lead to a highly misleading result. The use of some groups of animals in predicting diversity on a large geographical scale is acceptable, while the use on local patterns of land use gradients could produce conflicting results (Schulze, *et al.*, 2004). Although birds are generally less specialised within micro-habitats when compared to insects and plants, they are valuable indicators on a larger geographical scale (Schulze, *et al.*, 2004).

The use of diversity indicators can be regarded as a legitimate tool if a detailed study of whole species assemblages is not possible, as in the case of tropical ecosystems, which require resources in the many different fields of taxonomy and ecology that are not always currently available (Lawton, *et al.*, 1998). Schulze, *et al.* (2004) analysed the effects of land use in the tropical forest of Indonesia, using trees, under story plants, birds, butterflies, dung beetles and endemic species as indicators of land use modification. The results confirmed that there were decreases in species richness of both plant and animal groups as modification to the landscape increased (Lawton, *et al.*, 1998; Schulze, *et al.*, 2004). The study did, however, show that species richness and diversity did not always decrease for all species with increased modification. There was an increase, for example, of insects after forest disturbance on a small scale. This was confirmed by Davis, *et al.* (2001) who found that dung beetles species richness increased when primary forests were disturbed on a small scale. The increase was attributed to overlapping of species ranges, which were usually spatially separated in an undisturbed forest (Davis, *et al.*, 2001).

Few studies have been conducted where a diverse set of taxonomic groups have been used to assess a decrease in diversity (Schulze, *et al.*, 2004), but a study conducted by Lawton, *et al.* (1998) in the Cameroon produced a similar set of results when compared to the Schulze, *et al.* (2004) study, which showed decreasing diversity with increasing modification of the landscape. The interesting difference between these two studies was that Schulze, *et al.* (2004) found only a small positive correlation between species richness of taxa that were examined, while Lawton, *et al.* (1998) found a 25% correlation.

Schulze, *et al.* (2004) concluded that the modification of diversity can not be assumed to impact different guilds with the same intensity when comparing for example insectivorous birds and insectivorous mammals. Although both guilds were insectivorous there was not necessarily a correlation between these guilds as a consequence of habitat modification. Schulze, *et al.* (2004) found there was little correlation when comparing similar guilds in different animal groups, although data of the study did confirm that there was a general decrease in species richness of all taxa studied as landscape modification intensified. However, there was a correlation between a few species of the same feeding guild but of different taxa such as frugivorous birds and fruit-eating butterflies for example. When assessing the impacts of land use on bio-diversity, the use of biological indicators can be used, but a multi-species approach would produce a more realistic result (Bock, *et al.*, 1984; Wallgren, *et al.*, 2009).

It is worth noting at this stage that a number of authors have recorded the change in species diversity of other vertebrates and invertebrates as a result of heavy grazing pressure. Glendenning and Paulsen (1955) recorded an increased rodent population associated with high stocking rates. Barker (1985) found that increased grazing pressure reduced the herbage biomass, which impacted negatively on some species of grasshoppers (*Acridoidea*), while there was a positive correlation to grazing intensity on other species of grasshopper. Churchill and Ludwig (2004) found in their study on the effect of a grazing gradient on spiders that some spiders were highly sensitive to disturbance, particularly when perennial grasses were reduced and replaced by annual grasses as a result of heavy grazing pressure. Cagnolo, *et al.* (2002) found that the lower structural complexity of a habitat, impacted by heavy grazing, could result in an impoverishment of the insect assemblage as a consequence of reduced food availability, refuges, oviposition sites and micro-climates. Most insect groups may have been unable to cope with the conditions created by heavy grazing. Seymour and

Dean (1999), however, found a contrasting result, with greater invertebrate abundance of some species in areas of less vegetation cover, possibly because soil temperatures were higher due to more exposure of the ground to direct sunlight.

Allan, *et al.* (1997) found that birds and bird communities reflected similar distribution patterns when compared to mammals, reptiles, butterflies and amphibian and could therefore be used as surrogates to assess bio-diversity. Knowledge of bird species ecology will assist when assessing the reasons why certain species are present while others are absent in different sized forest patches (Blake & Karr, 1984).

2.5.2 Birds as indicators

It should be remembered when using birds as indicators of land use that they may use one or more habitat types over space and time. Birds could use different habitats for different reasons and at different times and seasons, and it is the interaction of these different reasons that create the birds' profile, which can differ from the clear-cut distribution patterns found in some literature (Bouwman, *et al.*, 2009).

However, birds are possibly the most obvious and widely appreciated component of wildlife. Conditions that benefit birds will more than likely provide benefits for bio-diversity in general (O'Hallaran, *et al.*, 2002). Birds, and particularly bird species, can be seen as providing different benefits when used as indicators. O'Hallaran, *et al.* (2002) suggested the following benefits:

- Umbrella species are those species that use a habitat that will be beneficial to other taxa.
- Flagship species are noticeable with a high profile, creating an awareness that promotes their conservation.
- Some species of birds, such as game birds, will generate income and be of economic value.
- The dispersal of seeds by some bird species contributes to their ecological importance.
- There are species of birds that are intrinsically important, irrespective of their value as suggested under previous headings.

The use of birds as a reference for changes in habitat use and distribution is valuable as they are a part of the environment that is well known. Birds have a number of positive attributes that enhance their use as indicators, including:

1. Knowledge about birds is extensive and their biology and life-histories are well known (Alexander, 2002; Gregory, *et al.*, 2005)
2. Birds are widespread throughout the world (O'Halloran, *et al.*, 2002)
3. Birds are mobile and responsive to environmental change (Hockey, *et al.*, 2011; MacDonald, 2002)
4. Bird families and genera have wide geographical ranges, unlike many individual species that have narrow distributions (Schulze, *et al.*, 2004)
5. There are many birds that can show meaningful patterns of distribution (Gregory, *et al.*, 2005)

Because of their rapid metabolism, and generally high position on food webs, land birds are good indicators of the effects of environmental change (DeSante, 1989). Birds have the ability to quickly change their range and will be a valuable vertebrate group that can be used as indicators of climate and environmental change (Hockey, *et al.*, 2011; MacDonald, 2002). Birds and bird communities reflect similar distribution patterns when compared to mammals, reptiles, butterflies and amphibians. It is therefore valuable to use birds as surrogates to assess bio-diversity (Allan, *et al.*, 1997; Gregory, *et al.*, 2005). Indicators, in this case birds, should not be seen as substitutes for detailed knowledge, but should be used as a proxy for ecosystem function and health.

Although birds are regarded as being effective indicators of land use, it should be remembered that different species react differently to disturbances, such as predation and the influence of man. Birds that have differing flightiness will therefore have an effect on the collection data. Birds that are more flighty will be observed less than birds that are more tolerant to disturbance. Because species flightiness is fairly constant, survey data may provide skewed results.

Blumstein (2005) was interested in the disturbance tolerance in birds. Blumstein's (2005) results concluded that carnivores and omnivores are more prone to flightiness than vegetarian species, as carnivores and omnivores have senses more alert to catching prey

and these detect prey or predators at greater distances, thus initiating flight sooner than vegetarian species which don't have as finely tuned senses (Blumstein, 2005). Body size was another trait of birds that affected their flightiness as smaller birds have a higher metabolism per kilogram and are therefore less reluctant to escape when approached. Larger species will escape sooner than smaller species which have a lower cost of flight (Blumstein, 2005).

The measurement of species richness, birds in the instance of this study, must be treated with caution according to Harrison and Martinez (1995). It would be expected that species numbers would increase as the sampling effort increased and then slowly level off, but in practice this appeared not to happen. As the numbers of samples increased, so the number of species increased, which was a result of rare visitors and vagrants being recorded (Harrison & Martinez, 1995). These were not true members of the community. Birds are very mobile, which only accentuates the problem. Over-sampling can, therefore, produce species richness in excess of what would have meaning and was as much a problem as under-sampling. Combined with the possibility of over- and under-sampling, abundance was a further problem as each species was given the same weight, regardless of whether they are common or rare (Harrison & Martinez, 1995). As an alternative to comparing species individually, it would be more valuable to compare communities or groups of birds, which would be particularly useful when assessing environmental impacts and bio-diversity (Harrison & Martinez, 1995; Blake & Karr, 1984). The selection of the correct biological species as indicators for monitoring purposes is important (MacDonald, 2002).

The impact on vegetation by different grazing intensities has implications for animal communities, including birds, both directly and indirectly. Rodents are effective predators of bird nests and compete for food, such as seeds and fruit. Grasshoppers and other insects provide insectivorous birds with part of their diet and also compete for food, while many bird nests are reinforced with spider web (Hockey, *et al.*, 2005)

There have been numerous studies on the effects of land use on birds, other vertebrates, invertebrates and plants. It is worthwhile noting that the natural environment and changes in climate and local weather patterns can have a direct effect on birds (Hockey, *et al.*, 2011).

Brown (1987) studied two populations of Cape Vultures (*Gyps coprtheres*), one population in the Drakensberg Mountains of Kwa-Zulu Natal in South Africa and the other population in Zimbabwe. When comparing the two populations, Brown (1987) noted a

difference in the size of birds, with the Drakensberg birds being heavier than those of Zimbabwe. Brown (1987) attributed the size difference to the colder, windier conditions of the Drakensberg, which had resulted in these birds being heavier, giving them a greater ability to withstand the low winter temperatures and providing them with winter energy reserves to survive longer periods between meals. The larger nesting chicks of the Drakensberg birds had more reserves, which, when combined with increased body surface area reduced their heat loss and gave the chicks a better chance of survival (Brown, 1987).

Weiner (1994) recorded some of the findings of the Grant family and their research on Daphne Major, an island of the Galapagos Archipelago. The remote island was the scene of extreme weather conditions, with times of severe drought and times of rainfall that promoted the growth of a numerous forms of vegetation not recorded during the dry years. The Grant family studied “Darwin’s” Finches and the effects of changes in vegetation on the size and shape of the finch beaks. The changes were marked and synchronised with changes in vegetation (Weiner, 1994). Genetic variation within the finch population permitted selection for the most adaptable trait of beak size and shape, which could be utilised for the food available. As the climatic conditions changed, the availability of food, types of food and nesting sites changed and it was these changes that influenced the adaptation and modification in the beak sizes and shapes of the finches (Weiner, 1994).

What is evident from the literature cited is that there could be a multitude of factors contributing to bird distribution patterns, which is part of what the current study is testing. The physiology of the birds could also change as a result of vegetation and climatic changes; these changes, however, are beyond the scope of this study.

Many studies on the impacts of environment on bird redistribution have been conducted in the Northern Hemisphere where changes in vegetation have stabilised after many years of agriculture, while there is a significant amount of natural vegetation in the Southern Hemisphere that has not been changed by agriculture (Hockey, *et al.*, 2011). Ambrosini, *et al.* (2002) and Britschgi, *et al.* (2005) both refer specifically to the impacts of land use on avian distribution patterns in the Northern Hemisphere.

Livestock farming can have a positive effect on some species of birds, as noted by Ambrosini, *et al.* (2002) in the Northern Hemisphere. Livestock farms had a 91.4% chance of having Barn Swallows (*Hirundo rustica*) breeding on the farm while farms without livestock had only a 43.9% chance. The association was not as a result of the livestock itself, but as a

result of the structures built to house the livestock and the hayfields cultivated for the livestock. The structures housing the livestock provided a stable environment for the breeding and rearing of fledglings, while the fields provided a suitable environment for insects on which the swallows could feed. Fledgling success in this study was, however, not as successful as anticipated as predators congregated where there were large numbers of breeding swallows (Ambrosini, *et al.*, 2002).

Britschgi, *et al.* (2005) studied the Whinchat (*Saxicola rubetra*) which is an insectivorous passerine found in mountainous areas of Europe. The lifecycle of its main prey item had resulted in the Whinchat synchronising its breeding in such a way that the prey breeding peaks coincided with their chick feeding. Different land management practices of the Whinchat habitat had affected the arthropod abundance available for chick rearing. The more intensive management practice involved mowing, which resulted in a marked reduction of arthropods when compared to the traditional farming methods where mowing was not part of the management practice. Britschgi, *et al.* (2005) recorded that Whinchat parents had to travel further to find suitable prey items in intensively managed fields and also substituted less favourable prey items when feeding their chicks. The net result was that fledgling mortality was more evident in intensively managed fields.

The decline in bird species need not necessarily be as a result of the reduction of a particular plant food, but as a result of an overall reduction of the diversity and abundance of food plants, which has been caused by intensive farming techniques, particularly arable farming (Arroyo, *et al.*, 1999). Invertebrate population declines are associated with the application of insecticides and herbicides, which, when combined with increasing specialisation of farming, loss of field margin habitats and increased ploughing, will have a negative effect on invertebrate populations. Intensively managed grasslands will result in the reduction and loss of some species of grasshoppers, ants, spiders and butterflies, all of which are food sources for young chicks of a number of birds (Arroyo, *et al.*, 1999). However, the increased plant material, which results from managed land, could have a positive effect on some phytophagous taxa, but the general pattern of reduced invertebrate species diversity with increased intensification and specialisation of arable farming was evident. The impact of intensive arable farming can be reduced significantly by encouraging uncultivated field margins, ditches and road verges, thus providing diverse food resources for many granivorous and insectivorous bird species (Arroyo, *et al.*, 1999).

Taylor (1986) showed that cattle grazing had a significant effect on passerine birds with the intensively grazed areas having the most marked negative impact. The numbers and species of birds decreased as grazing increased (Taylor, 1986). Birds use different levels of the vegetation structure, from the canopy to the lower levels, all of which can be effected by grazing intensity. The lower levels of vegetation structure were affected almost immediately by grazing as grasses formed an important supply of food through seeds, nesting material and cover, and also through invertebrates that feed on the grasses and are part of the diet of insectivores. Intermediate and upper levels of vegetation were affected by the longer periods of grazing as young trees were either trampled or grazed inadvertently, resulting in fewer young trees growing to replace the older trees (Taylor, 1986).

Todd and Hoffman (1999) studied fence line contrasts between communal and commercial land use in the Namaqualand region of South Africa, with particular reference to the impacts of grazing of livestock on the vegetation. Livestock did have a detrimental effect on the bio-diversity of a region, while overgrazing can be regarded as the dominant factor that reduces bio-diversity (Todd and Hoffman, 1999). Light grazing could have a positive effect on species richness and diversity of vegetation, but as the intensity of grazing increases, the species richness was reduced. Increased grazing pressure will lead to assemblages of toxic and spinescent woody vegetation, moving away from palatable perennials to unpalatable annuals (Todd and Hoffman, 1999).

Todd and Hoffman (1999) did find that when comparing the species richness of grasses of the communal and commercial land use there was a similar species richness which was attributed to a change in species composition from perennials to annuals which has been confirmed by others (Abel & Blaikie, 1989; Barker, 1985; Moser & Witmer, 2000; Tietema, *et al.*, 1991).

Over-utilisation of vegetation in a natural environment was studied by Herremans (1995) who analysed the effects of land use on the natural vegetation of the Chobe Game Reserve in Botswana. The change in vegetation as an impact of heavy elephant utilisation resulted in the loss of tree canopy species as elephant numbers increased, with the only watering point during the dry months of the year being the Chobe River. The impact of large numbers of elephants reduced the mature tree canopy to shrub vegetation with scattered large trees. When comparing bird species of the mature woodland area with the modified scrub, it was found that there was more of a replacement of bird species resulting from a

change in composition of the woodland bird community. Specialist species, such as the Arnots Chat (*Thamnolaea arnoti*), were found in mature woodland, but were absent where elephants have reduced tall woodland to low scrub. Herremans (1995) compared the bird species of the mature woodland and the modified scrub and noted a replacement of canopy species with species associated with the lower strata of vegetation. However, 22 species of birds were unique to the mature unmodified woodland, while only three species were unique to the modified woodland scrub.

Hockey, *et al.* (2011) suggested that the impacts of climate change and land use will have more significant effects on birds of the Southern Hemisphere than those of the Northern Hemisphere. The reason was that land use in the Northern Hemisphere changed many years ago and has stabilised, while land use in the Southern Hemisphere is still in the process of being changed.

The impacts of climate change and changes in land use, with particular reference to the effects of livestock on the environment, have been presented earlier in the chapter. Birds are confronted with these changes and have reacted differently to these different drivers. Hockey, *et al.* (2011) proposed that both land use and climate change impacted on the distribution of birds, but isolating the impacts of these two drivers was a challenge. Climate change, especially increased temperature and resultant changes in rainfall, will affect the distribution of birds, but not as anticipated (Hockey, *et al.*, 2011). Bird distributions were expected to have a general pattern of moving southwards and eastwards towards the poles as temperatures increased in central Africa. The southward movement away from rising temperatures is understandable, but the anticipated easterly movement is not as apparent, with a westerly movement of some bird species being more evident.

In light of this, Hockey *et al.* (2011) suggested that separating the impacts of the drivers of range alteration was difficult, and this difficulty was evident too when trying to isolate which driver influenced the range extension of some bird species. The apparent westward redistribution in some southern African bird species was possibly the result of impacts of land use contributing greater influence to bird species redistribution than the impacts of climate change. Land use changes included the construction of water impoundments and the planting of woody vegetation as wind breaks around dwelling places. Hockey, *et al.* (2011) suggested that perhaps the modelled impacts of climate change may

have been overstated or that the models may have over-estimated the sensitivity of birds and their ranges to climate change as a driver of bird redistribution.

The important point taken from this article in relation to the current study is that birds are confirmed indicators of change, whether it is climate change, land use change, or a combination of both.

2.6 Conclusion

What is most evident from the literature review is that there are a considerable number of possible reasons that will influence bird distribution and redistribution patterns. These include climate and climate change, vegetation structure, and land use which create a mosaic of possible impacts influencing bird distribution patterns.

The drivers of bird distribution patterns are complicated, with possible feedback mechanisms influencing these patterns either directly or indirectly. Climate, and consequently climate change, will impact on temperature and rainfall with vegetation distribution patterns changing as a result (McClean, *et al.*, 2005). The resultant changes in climate and vegetation will impact on avian distribution patterns (Hockey, *et al.*, 2011). Impacts of climate can contribute to physiological changes in birds, as noted by Brown (1987) and Weiner (1994).

Different forms of land use, with particular reference to farming, whether arable or pastoral, impact on bird requirements by altering the vegetation structure (Acocks, 1975), which leads to changes in food availability, nesting and roosting sites, predation, as well as competition between species and within species (Evans, 2004).

The impact of livestock farming and its direct effect on the herbaceous component of the vegetation is evident in the literature review, while the indirect impacts of habitat fragmentation, bush encroachment and soil erosion also contributing changes in biodiversity (Elagib, 2011; Tainton, 1999). Fuel wood collection, as a consequence of a rural lifestyle, will impact on the vegetation and avian distributions (du Plessis, 1995).

There has been some debate regarding the effectiveness of birds as indicators of land use, with some authors suggesting the use of number of taxa, both animal and plant, when assessing land use impacts (Lawton, *et al.*, 1998; Schulze, *et al.*, 2004). There is, however, a consensus that birds can be used as indicators of land use (Hockey, *et al.*, 2011), but a knowledge of bird ecology is a prerequisite (Blake & Karr, 1984).

2.7 Aims of this study

The aims of this study are to assess:

1. What effects livestock farming have on avian diversity when comparing species richness and numbers of birds of four different land-use grazing intensities?
2. What effects livestock farming intensity would have on avian nesting and feeding guilds?
3. Whether certain bird species could be used as indicator species for livestock farming intensity?

2.8 Hypotheses

Hypothesis 1: **Bird numbers and species richness will be affected by different land uses.**

Hypothesis 2: **Avian guilds will respond to different land uses**

2.9 Objectives

After having achieved the aims of the study and tested the hypotheses the objective would be to make suggestions regarding the effect of grazing intensity on bird communities. These objectives would involve social, economic, management and conservation goals with the aim of reaching a better understanding of the complexity interactions between and within these fields.