\\ LAND USE AND LAND COVER CHANGES IN THE GREATER AMBOSELI ECOSYSTEM, KAJIADO DISTRICT, KENYA BETWEEN 1988 AND 1998.//

BY

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DECLARATION

This thesis is my original work and has not been presented for a degree award in any other University.

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DEDICATION

This thesis is dedicated to my late mother Mama Margaret Atieno who despite her lifetime commitment to my welfare, could not live to see me through this study.

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Abstract.

For millenia, Greater Amboseli ecosystem of Kenya has had a central role in subsistence pastoralism and wildlife conservation by providing vast biological resources for pastoralists and their livestock; and habitat for wildlife. Recently, with the creation of Amboseli National Park and rapid changes in land tenure systems, the human use of the ecosystem has intensified. This change has shaped a pastoral landscape composed of livestock grazing, wildlife conservation, rain-fed and riverine crop cultivation and permanent settlements. A study was undertaken to map land use over time, and document the causes and consequences of these changes for land cover, vegetation species diversity and distribution within the 3,112 km^2 of Amboseli ecosystem. Remote sensing, Geographic Information Systems and ground-based techniques were applied. Land-use and land-cover maps were produced with an accuracy of 85.7% and changes analysed for the years 1988 and 1998. During the ten-year period, changes in natural vegetation cover and cultivated fields were significant ($p<0.05$) with cultivated fields replacing 8% of natural vegetation. Cultivation increased along the mountain slopes, rivers and swamps — areas formerly used by pastoralists and wildlife for dry season grazing. Land-cover types changed significantly with bushed grassland and cultivated fields increasing by 10% and 8% respectively; while grassland and bush land decreased by 6% and 11% respectively. The overall result is a change in landscape structure with increasing patch diversity as number of patches increased from 157 in 1988 to 250 in 1998. Likewise ecosystem complexity declined with mean cover type area decreasing significantly ($p<0.05$) from 1,989 ha in 1988 to 1,247 ha in 1998. Mean cover type perimeter and dominance value both changed significantly ($p<0.01$) from 234 meters to 211 meters and 2.09 to 2.04 respectively. The emerging land-use and land-cover types can be attributed to changes in land tenure policies and pastoralists' socio-economic lifestyles. These on-going trends pose greater threat to survival of pastoralism and wildlife conservation within East African rangelands as pastoral lands and wildlife habitats continue to be fragmented and fenced off. The results of this study reveals the need for integrated research that would address policy issues, pastoral welfare, benefit sharing, resource accessibility and utilisation, and wildlife conservation so that landscape fragmentation and resource degradation can be managed.

CHAPTER ONE: INTRODUCTION.

1.1.0 Background information

In recent years, pastoralists in much of the world have faced gradual loss of rangelands, which are converted to other uses (WRI, 1997; Ellis et al. 1999; Wu Ning, 1999). The ecological, economic and social integrity of rangelands throughout the globe thus continue to decline (Oldeman et al. 1991; Foran et al. 1999). Rapidly changing land tenure, land-use systems and expanding human populations are causing dramatic changes in vegetation cover, structure and composition in the rangelands of sub- Saharan Africa including Kenya (Behnke, 1985; Grandin et. al., 1987; Bourn and Wint, 1994; Smith, 1996). Large and extensive areas of native vegetation are being fragmented into smaller blocks of introduced small-scale rain-fed agriculture, settlement schemes and individual ranches (Gichohi et al., 1996; Foran et al. 1999). Land-cover change has immediate environmental consequences (such as loss of soil cover, soil nutrient change, habitat fragmentation, diversity loss) that feedback on land-uses as well as on the biophysical and human driving forces. The insurgence of immigrants into rangelands, sub-division of pastoral lands, socio-economic developments and changes in local-scale decision-making have exacerbated changes in land-use and land cover-types thereby further altering landscape structure and diversity (Kituyi and Kipuri, 1991; Campbell, 1999; Igoe et al., 1999). Despite its importance in global climate change, our understanding of changes in land-cover is inadequate owing to several reasons including: lack of accurate measurements of its rate, geographic extent, and spatial pattern, and overall poor capability of modelling from the empirical observations (Burke, 2000; Crist et al., 2000).

Several scientific studies have been carried out on the effects of fragmentation and overall human influence on forests and high potential areas. However, very little information exists for native grasslands and rangelands (Risser 1990; Samson and Knopf 1994). Grasslands have very high community diversity and provide valuable insights into the complex ecosystem dynamics including temporal and spatial changes in landscape structure and function within rangelands (Risser, 1990; Turner, 1989; Johnston et al. 1992). In other ecosystems such as savannah, their integrity is shown to be dependent on the maintenance of key ecological processes (Gichohi et al. 1996). Therefore, removal or replacement of natural vegetation alters the land cover structure, which often reduces ecosystem diversity on both regional and local scales (Krummel et al 1987; Burke, 2000). Such anthropogenic changes within rangelands cause concern over preservation and management of biological diversity for sustainable pastoral production (Urban et al. 1992; West 1993; Swift et al., 1996).

The effects of the degree to which land-use changes affect vegetation in rangelands are therefore critical aspects of rangeland ecosystem management the world over in various ways (Lambin, 1994; Crist et al., 2000). These effects are relevant to the ecological principles of community stability, diversity and dynamics as well as ecosystem structure and function under varying degrees of use or disturbance. Since the highly diverse herbivore communities found within rangelands derive their livelihood by utilising the basic vegetation resource either directly or indirectly, many scientists are expressing increased concern about vegetation resource degradation within rangelands and in the tropics in general (Child, 1986; Roy et al. 1991; Foran, 1999; Ellis et al., 1999).

In conclusion, changes in land cover and vegetation structure affect a wide variety of ecological processes in different ecosystems. Yet, in spite of the growing need for precise estimates of rates of land-cover change to support basic scientific research, comprehensive and systematic information is not available on global, regional or local bases (Behnke and Scoones, 1992; Crist et al., 2000). Assessment and description of changing land cover and land-use patterns therefore form an important component in our understanding of ecological dynamics necessary for policy formulation and integrating the often conflicting demands of wildlife habitat, recreation, livestock grazing, agriculture and development within rangelands (Milne, 1992; Scoones, 1992; Reid et al., 1997). It is with this view that this research study was conceived.

1.2.0 Problem statement

During the past few, land-cover change has been accelerating in the tropics as a result of population expansion and economic development (Campbell, 1999; Igoe et al. 1999). The growth in population especially in rangelands and the demand for rangeland products places direct pressures on ecosystem integrity. In Kenya alone it has been projected that human population will increase from a current 29 million people to 66 million people in the year 2050. This represents a 147% increase at a rate of 3.5% annually (UNDP, 1998; Foran, 1999). It has likewise been shown that the greatest irreversible change associated with land cover change is the loss of biodiversity from habitat destruction and fragmentation of landscapes (Reid et al., 1997; Crist et al., 2000). In East Africa, high rates of human population increases coupled with changes in people's expectations over the past few decades have resulted into unprecedented conflicts between humans and biodiversity (Western, 1982; Reid et al. 1997; Ellis et al., 1999). Research has revealed that increased population growth leads to an imbalance between animal and human numbers especially within subsistence pastoralism due to reduction in available grazing land (Ngethe, 1992; Prins, 1992; Foran, 1999). In general, this presents a situation of declining land resources relative to current and future demand for food and high lifestyle expectations leading to increase in livestock numbers and expansion of cultivation into the already contracting rangeland ecosystems (Foran, 1999). Due to changes in tenurial arrangements from communal to individual land holdings, the present market oriented economy tends to promote increased expansion of crop-agriculture and intensification of livestock production systems within rangelands. Resultant effects include fragmentation of geographic ranges into isolated ecological islands due to extensive expansion of agriculture and permanent settlement encroachment into rangelands the world over (Gichohi et al. 1996; Wu Ning et al. 1999).

Kajiado district located to the southern part of Kenya has been inhabited by the Maasai pastoralists for decades but has undergone tremendous anthropogenic and climatic changes over time, which has impinged differently on its natural resources (Bekure et al. 1991; Campbell, 1999). The district has experienced tremendous changes in land tenure systems from communally owned to individually owned land parcels. The results include land sales, which have led to high rates of immigration into the district from the neighbouring communities, introduction of small-scale rain-fed agriculture and permanent settlements (Kajiado ASAL Atlas, 1990; Campbell, 1999; Igoe et al., 1999). As early as 1960s widespread speculations regarding vegetation destruction and deterioration of pastures in Kajiado district had already started emerging (Bekure et al., 1991). Yet up to now, systematic, comprehensive and objective studies on rates and patterns of resource degradation as well as the impacts of land-use change, loss of primary productivity and vegetation cover are scanty or non-existent for the whole district (Kituyi, 1990; Bekure et al., 1991).

The greater Amboseli ecosystem within the south eastern part of Kajiado had been home to the Ilkisongo Maasais until 1974 when the area covered by the Amboseli national park was carved out of the entire ecosystem (Smith, 1996). The ecosystem is inhabited by numerous wildlife species and has been a high-income earner to the Kenyan government through tourism activities. The Amboseli Park itself covers less than 10% of the ecosystem approximately 392 square kilometres and is surrounded by a dispersal area of about 3,740 Km^2 into which many of the wildlife disperse from the park at certain times of the year (KWS, 1991). The dispersal area around the park belongs to Ilkisongo Maasai members of four group ranches viz. Olgulului, Mbirikani, Eselenkei and Kimana (Kajiado ASAL Atlas, 1990). This area covered by the park in the past consisted of swamps, which were often used by the Maasai for dry season grazing and watering. Therefore, the eviction of the Ilkisongo Maasai from the park region contributed to their loss of access to dry season grazing zones, watering points and likewise reduced mobility due to creation of park boundaries and fences (Smith, 1996; KWS, 1991). Likewise the development of the pipeline road linking Mombasa road to Loitokitok town has led to emergence of small town centres which have been expanding as the population increase and settlement expand. These, amongst other changes in land tenure have led to shifts in land-uses such as small scale agriculture around Namelok, Kimana and Noolturesh swamps, subdivision into individual ranches as well as increased conflicts between the people and wildlife (KWS, 1991; Campbell, 1999). Lack of mobility, which has led to sedenterization of the pastoralists, has been identified to be a key factor resulting in degradation of rangelands throughout the region (Igoe et al. 1999). This is because pastoralist's ability to track environmental conditions and mobilise herds to seek areas of good forage has been eliminated as more areas become partitioned and fenced off. This turn of events has led to negative vegetation degradation especially around the watering points and bomas (*manyattas*) where widespread overgrazing occur. This human-wildlife conflict has led to confinement and congestion of wildlife within the protected/park areas thereby destroying the vegetation within these areas (Western and Gichohi, 1993). As intensification of use continues within rangelands, more land is cleared for agriculture and wildlife on the other hand is seen to be competitors against livestock for forage resulting into exacerbated conflicts with pastoralists. Therefore, this region formed a representative study site to investigate the impacts of changing land-use and land tenure on land cover as well as their effects on the vegetation. This would provide necessary information to enable stakeholders better reconcile the often-conflicting demands of wildlife habitat, livestock grazing, agriculture and other developmental issues within the region.

1.3.0 Study aim, objectives and hypotheses

1.3.1 Study aim

The study aim was to map out land-use over time, and document the causes and consequences of these changes for land cover, vegetation species diversity and distribution within the Greater Amboseli Ecosystem in Kenya. Remote sensing, geographic information systems and field techniques were applied.

1.3.2 Objectives

1) To stratify land-use and vegetation based on cover, structure and composition using field and digital techniques.

2) To analyse changes in land cover over time using Landsat imagery and Geographic Information System techniques.

3) To investigate changes in land-use systems and subsequent effects of land-use on vegetation cover from 1988 to 1998.

1.3.3 Null hypotheses

1) Changes in land-use and land tenure systems do not lead to changes in vegetation structure and cover.

CHAPTER TWO: LITERATURE REVIEW

2.1.0 Role of vegetation cover in Rangelands

Natural vegetation is the principal resource base for pastoralists within rangelands. The direct and indirect contributions of natural vegetation include cover for the earth; forage for both wildlife and livestock; habitat for wildlife and other organisms; physical environmental stabilisation attributes such as enhanced micro-climatic conditions, water and moisture regimes (Pratt and Gwynne, 1977; Goudie, 1983;). Given changes in rates of natural resource exploitation, human population density, land-use and ecosystem integrity, various natural habitats especially vegetation types have been altered to different degrees by human uses (Urban et al. 1987). This land cover alteration is manifested in various ways. However, the primary and usually the most visible are through changes in various characteristics of vegetation, for example, productivity, complexity (composition and structure) and cover (Goudie, 1983). Loss of biological productivity and complexity of ecosystems or vegetation communities is, therefore, an important indicator of land degradation.

The widespread and increased frequency of accelerated run off, soil erosion, nutrient losses and floods are consequences of reduced vegetation cover, which are exacerbated primarily by the impact of human activity. Therefore, land degradation arises from activities in deforestation, fuel wood utilisation, poor cultivation practices and overstocking. Loss of diversity *or* change in composition of plant species in any area gives an indication of severity of pressures at that particular site (Olsson et al., 2000). Grazing pressures lead to change in the ratio of desirable and undesirable plant species; likewise selective exploitation of woody plants by humans either for construction or fuel leads to reduction in vegetation cover, structure and diversity. Changes in vegetation structure, especially the loss of vegetative cover are a result of a combination in varying proportions, of overgrazing, cultivation, fires and wood exploitation for construction and fuel. These changes lead directly to loss of woody canopy and herbaceous cover; shifting the dominant life form, e.g, change from a forest to a grassland or woodland to a shrubland. With the eventual loss of soil fertility, structural changes may also be manifested in average heights of the different life forms on the site and therefore biomass or woody volume (Sinange 1997).

Since vegetation acts as indicator of the physical and biological attributes of an area and is usually used for evaluating for ecosystem conservation; mapping of vegetation therefore becomes a basis for most ecosystem management (Austin, 1991). Methods of mapping vegetation vary and selection of a specific mapping method is usually goal specific. In any description of a vegetation community within rangelands, the dominant life form and cover are, more often described or implied. Therefore cover, which can be measured and assigned a numerical value, becomes very important for assessing and monitoring the vegetation status for any given site.

2.2.0 Human Impacts on rangeland vegetation

The human proximate causes of land-cover change are the immediate land management strategies that convert cover from one type to another or modify an existing cover type (Urban et al., 1987; Turner et al., 1989; Gichohi et al., 1996). Likewise landscape fragmentation is a land-use question; the use to which a landscape is subjected results in different patterns and geometry of land cover since lands are managed differently by different classes of human agents (Turner et al. 1993). For example, small cover types of cleared land are created by activities of small-scale farmers, while large-holder cattle ranchers create large rectilinear cover types. This shows that different land-uses influence the pattern and geometry of cover-types, and different patterns and geometry have different effects on the environment as a whole. With the continued growth of human population, competition for limited land resources has steadily increased over recent years with most countries in sub-Saharan Africa experiencing progressive expansion of their agricultural land and rural settlement (Wint and Bourn, 1994). In the marginal areas these have been compounded by the transition of certain communities from purely pastoral and nomadic existence to a semi-pastoral and agro-pastoral settings (Campbell, 1999; Igoe et al., 1999). All these changes and demands have subjected marginal lands to continuously increasing ecological stress (Sinange, 1997). The cumulative impact of human activity on the environment has far reaching consequences such as converting potentially good quality grazing lands into areas of low vegetation cover thus reducing soil fertility and increased susceptibility to wind and water erosion; changes in forage quantity and quality as well as species composition. The study by Bourn and Wint, 1994 in Nigeria reveals that vegetation destruction continues as land-use systems get transformed . Within the Amboseli ecosystem, changes in land use have led to various changes in vegetation cover as well as reduction in wildlife habitats. This has further resulted into conflicts between humans and wildlife leading to wildlife concentration within the protected areas, which reduces their vegetation and land-cover status (Smith, 1993; Western, 1982).

2.3.0 Effect of climatic fluctuations within rangelands

The climate of the world is now known to fluctuate frequently and extensively over the years that humans have inhabited the earth (Goudie, 1983). Climatic changes can be attributed to a wide range of different natural factors that operate over a variety of time scales. Human influence has become a significant factor in the variations of the world climate that take place due to the increasing human population and rising level of technology in natural resource exploitation.

Most rangelands are characterised by low erratic rainfall with high coefficients of variation that are often over 50% and evaporation potential that is as high as $2.6m^3$ p.a in some areas. The higher the co-efficient of variation, the more total or seasonal rainfall varies from year to year and season to season. This constitutes the greatest source of risk and uncertainty in planning and management of rangeland ecosystems (Ellis et al. 1993). These phenomena calls for proper planning of range resource use, which should be in response to the sporadic nature of the rainfall and the extent of natural resource, cover types. This variability drives the pastoralists into adapting mobility strategies so as to exploit the spatially and temporally scattered resources coupled with serious seasonal shortages

2.4.0 Land tenure and Land-use changes in the area.

Land-use practices in Kenyan rangelands have undergone numerous changes caused by both natural and human-made events. The marked changes are exhibited by loss of land and loss of traditional pastoralist mobility and flexibility in resource use strategies (Bekure et al., 1991). Traditionally, land ownership in the pastoral areas is a common property regime as resource tenure. In this regime access to resources was determined by membership to a given social unit owning those resources, such as a clan or community or by permission of the social unit (Mugerwa, 1992). This system was also typical to the Maasai community until they were disrupted and transformed by factors that weakened the indigenous pastoral institutions. The National Park Ordinance of 1945 led to loss of access to Nairobi National Park, Tsavo National Park and Amboseli Game Reserve areas thus restricting the use of these areas. The Swynnerton plan of 1955, set conditions for sound and productive use of rangelands which required a limit to numbers of resident stock and control of grazing at productive level with owners maintaining their grazing areas (Campbell, 1981). This plan paved the way for the creation of group ranches.

In 1965 the Government of Kenya proposed to the World Bank a variety of organisational structures for the different social and ecological systems within pastoral areas. In the rangelands, the recommendations were to change pastoral production from subsistence to commercial-oriented production through major changes in land tenure and organisation (Republic of Kenya, 1974). Security of tenure was advocated as a key instrument in promoting the development of pastoral rangelands with a belief that tenure would reduce the tendency to overstock. Thus group ranches were created and the Land Act of 1968 stated, "each member shall be deemed to share in the ownership of the group ranch in undivided shares" (Migot-Adhola et al., 1980). This objective was to be realised through adjudication of the trust lands into group ranches, characterised by registration of permanent members of each ranch, freehold title deeds held by group members, allocation of grazing quotas to members to limit the carrying capacity, development of shared ranch infrastructure and use of loans with members managing their own stock numbers (Kituyi and Kipuri, 1991). The group ranch arrangement was operational till the early 1980s when the clamour for subdivision emerged due to a myriad of reasons. The breakdown of the group ranches and the loan burden saw the majority of members calling for subdivision into individual parcels (Grandin, 1987). The subdivision led to issuance of title deeds to individuals thus transferring exclusive ownership rights to the individual owners to enable them use their title deeds as collateral for loans. Subdivision of land has led to several changes in land-use given variations in individual goals in the use of rangeland, such as crop agriculture, pasture rentals, sale of land parcels to outsiders and settlements.

Within Amboseli ecosystem, the driving of Maasai community from the park area interfered with their style, which had been that of solely livestock production. They were actually cut off the natural watering and dry season grazing zones (Smith, 1993; Kajiado ASAL Atlas, 1990). There have been gradual changes in land tenure systems with group ranches being subdivided into individual holdings. Changes in pastoral lifestyles into a more market oriented approach with the emergence of small towns, as well as introduction of small scale irrigation and rain-fed agriculture within and away from the swamps to the east of the region. Likewise the increasing population and expansion of cultivation fields within these areas surrounding the park, human-wildlife conflicts has been precipitated. The wildlife is now largely confined within the protected areas thus impacting differently on the park vegetation.

2.5.0 Loss of Biological Diversity in rangelands

Diversity within flora and fauna continue to erode steadily throughout the world rangelands and populations of many individual species are decreasing as well (WRI, 1997; Foran, 1999; Ellis et al., 1999). It has been established that habitat loss is the main cause of all these. Given that almost 40% of the net productivity of all plants is appropriated for human use, the result is usually extreme simplification of ecosystems as a whole (Krummel et al 1987; Crist et al., 2000). When wild rangelands are converted for agricultural, industrial, and domestic purposes, a complex ecosystem capable of supporting many plants and animals is often turned into one that can only sustain far fewer wildlife and plant forms (WRI, 1997). Until recently, the human population was distributed sparsely enough that it had little effect on the prevailing background rate of extinction estimated at an average of 90 species disappearing each century (WRI, 1997). The demand for food and material goods has increased commensurately, to the detriment of flora and fauna habitats thus raising the rate of extinction.

2.6.0 Indirect methods of assessing vegetation cover

Assessment of cover loss on large pieces of land with diversified activities can be accomplished by analysis of remotely sensed data alongside field checks on the ground (Lillesand and Kiefer, 1994). This involves the use of satellite images, topographical maps, soil maps as well as aerial photographs. Assessment of changes in land cover over a given period of time is thus best accomplished by analysis of satellite images captured at different times.

Remotely sensed data of the earth surface is acquired in digital format. Spatially, the data are composed of discrete picture elements called pixels. Each of these pixels represents distance in meters on the earth surface, which is referred to as spatial resolution. The pixels are quantified each into discrete brightness values. These values represent the amount of light reflected back to the sensor by the surface feature. The brightness values are described in radiometric resolution. The radiometric resolution is usually expressed in binary digits, or bits necessary to represent the range of available brightness values. Thus data with an 8-bit radiometric resolution has $2^8 = 256$ levels of brightness. The sensor records the amount of light reflected by features on the earth surface in values format ranging from 0 to 255. The 0 value implies no reflectance while 255 represents maximum reflectance (Lillesand and Kiefer, 1994).

Detailed digital image processing can be used to extrapolate the assessment of cover loss and mapping of vegetation beyond study areas and in future times (Lillesand and Kiefer, 1994; Turner et al., 1985). Important factors in this analysis include a correct mix of data acquisition, interpretation and conventional data analysis, which can be accomplished by applying computer-based GIS programmes on the satellite images.

2.7.0 Advantages of using remote sensing and GIS techniques in studying rangeland resources

Most experiments carried out in rangelands usually have been short term and concentrated within narrow experimental plots due to high associated costs (Palmer et al. 1999). This may not satisfactorily explain the overall characteristics of the entire rangeland as such, extrapolation of results beyond study areas is often limited. The heterogeneity and complexity of rangelands require methods that would give extensive, long-term data and results at reduced overall costs to enable sustainable development decisions.

For a long time the principal source of vegetation data has been; vegetation mapping by ground survey. Given the large extent of rangelands, the observations are often made by many people and this requires synthesis, which introduces the problem of reconciling disparate observations made by several people (Tucker et al. 1985). Likewise, Lobo et al. 1998 states that an important practical problem in the analysis of spatial-temporal patterns in ecological systems is that of requisition of intensive data with both fine resolution and large extent, which is often difficult to obtain from field-measured variables only. At the same time, determination of net primary production requires frequent measurement of biomass changes by destructive harvesting which are time consuming and expensive. These also limit repetitive sampling and can't be extended to cover larger areas because of logistics and huge number of samples required to cope with increased sample variation in rangelands (Prince, 1991). At the same time visual interpretation has inherent drawbacks of subjectivity and does not provide technology for regular monitoring on operational basis (Roy et al. 1991). It is therefore necessary to develop techniques based around digital processing too. Use of Remotely sensed data and aerial photographs alongside digital image processing techniques thus present a more appropriate method in vegetation resource mapping and long term monitoring (Tucker et al. 1985; Lillesand and Kiefer, 1994) as opposed to localised ecological studies. Ease of extrapolation of results, ability to detect changing conditions over time at significantly reduced costs all gives credence to the technique (McCloy, 1995). This study has been conceived with these facts in view, since digital imagery can offer a valuable alternative source of information in the analysis of ecological patterns. The remotely sensed data can be used to provide links between field-based information and spatially explicit modelling of ecological processes (Lillesand and Kiefer, 1994; McCloy, 1995).

2.8.0 Wildlife issues and habitat destruction within rangelands.

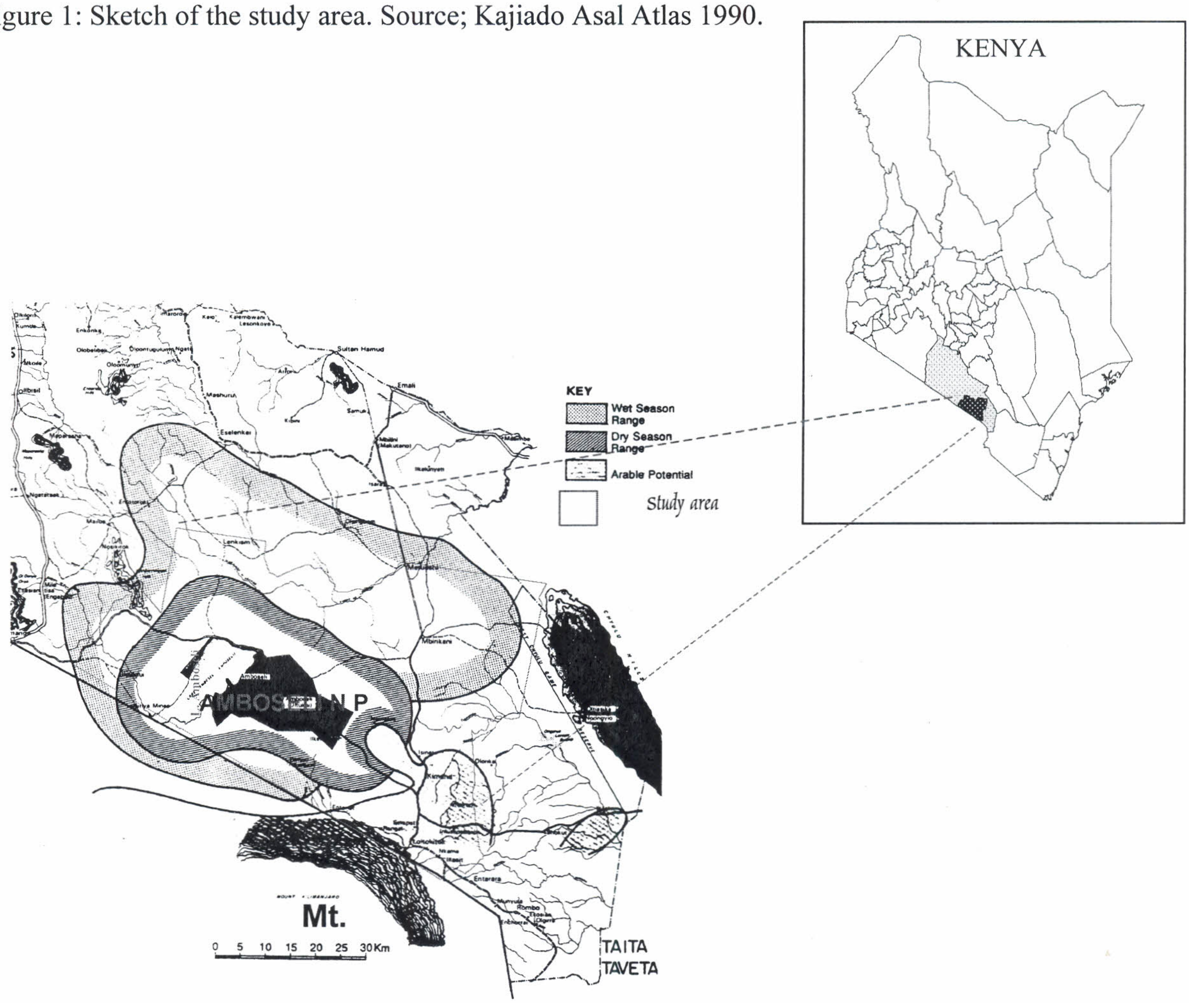
Most rangelands provide habitat for wildlife species all over the world (Child, 1986, 1988). In East Africa, the regions with abundant and diverse wildlife communities are the same ones occupied by pastoralist communities (Smith, 1996). In Kajiado district, wildlife is mostly found in Amboseli and Chyulu game conservation areas as well as defined dispersal areas that consist of group and individual ranches (Kajiado ASAL Atlas, 1990). Earlier on, the low population of the district had favoured wildlife movements with little restrictions (Barnes, 1998). Smith, 1996 stated, " while livestock herding people have coexisted with wildlife for thousands of years, the potential for conflict over land-use has increased in recent decades mostly following the intervention of modern governments in pastoral lifestyles." Currently the changing land tenure and land-use patterns put wildlife and humans at conflict due to decreasing home range for wildlife (Campbell, 1999; Igoe et al., 1999). There is intensification and expansion of agriculture in and around the swampy areas that provide dry season grazing zones for wildlife and replacement of wildlife with livestock or any other land-uses (Barnes, 1998). Since wildlife are vectors for livestock diseases, compete with livestock for forage and predating on them as well as destroying crops and other structures, the conflict has intensified thus reducing wildlife populations when their ranges are fenced off. This also leads to changes in vegetation cover thereby threatening wildlife survival through the destruction of their habitats and blocking of migration corridors (Child, 1988; Barnes, 1998).

CHAPTER THREE: MATERIALS AND METHODS.

3.1.0 Study area and site

The Greater Amboseli Ecosystem covers an area of about 4,132 square kilometres of southern Kenya within Kajiado district, running along Kenya-Tanzania border at the foot slopes of Mt. Kilimanjaro. Amboseli Park itself covers less than 10% (392 km^2) of the area and is surrounded by a dispersal area of about 3740 km^2 into which many of the wildlife disperse from the park at certain times of the year (Policy Framework, 1991). The dispersal area around the park belongs to Ilkisongo Maasai members of four group ranches viz. Olgulului, Mbirikani, Eselengei and Kimana (Kajiado Atlas, 1990).

Figure 1: Sketch of the study area. Source; Kajiado Asal Atlas 1990.



The ecosystem supports many wildlife species and has been a high-income earner to the government through tourism. Traditionally, wildlife and livestock co-existed and moved within this area unrestricted until the time of the park declaration of Park status in 1974 (Smith, 1996). Most group ranches around the park derived their livelihood from rearing livestock. Recently, small-scale irrigated agriculture has been established to the east of the park (KWS, 1991). Some of the surrounding ranches have also undergone spectacular changes through subdivision into individual ranches and subsequent conversion into other uses along the slopes of Mt. Kilimanjaro.

3.1.1 Human population trends and migrations in Kajiado District.

The overall population of the district has been rising steadily over time. In 1969 the total population was 85,903 persons; in 1979, it was 149,005 persons and 1989, 258,659 persons (CBS 1994, Republic of Kenya, 1990). This rapid population increase could be attributed to population in-migration and to a slight increase in fertility levels (Kituyi and Kipuri, 1991; Campbell, 1999). This is exhibited in the falling proportions of the Maasai population in the whole district, which was about 68.6% in 1969 and dropped to 62.8% in 1979. Jacobs (1984) in his analysis of population growth within rangeland districts of Kenya between 1969 and 1979 showed that the population of Kajiado had increased by 74%. However, only half of this growth was due to the increase in Maasai population, the remainder was caused by in-migration of agricultural people (Kikuyu and Kamba ethnic groups) from surrounding districts. The population of non-Maasai groups is steadily increasing as exemplified by the Kikuyu population, which increased from 18.9% in 1969 to 22.6% in 1979. Other significant ethnic groups include the Luo, Kamba and Luhya from Kenya and Chaga from Tanzania. Most of the in-migration has been taking place in major urban centres, but also in marginal areas especially after the subdivision of group ranches (Republic of Kenya, 1990).

In-migrating agricultural communities have occupied the high potential areas meant for dry season grazing such as the Ngong Hills and the Loitokitok region, Namelok and Kimana swamps (Bekure et al. 1991). This has led to reduction in accessible grazing lands and overall pastoral mobility. The practice of settling and cultivation of these grazing areas are speculated to lead to overgrazing and overall vegetation destruction within the area (Campbell, 1981).

3.1.2 Transport and communication

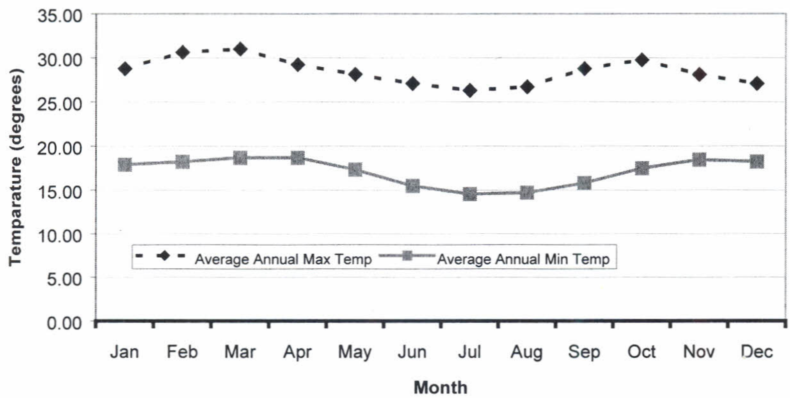
This area is situated about 300km south east of Nairobi served by the Nairobi-Mombasa road via the Emali-Loitokitok pipeline road, which is the main connection with the agriculturally important areas at foot slopes of Mt. Kilimanjaro. The Namanga-Amboseli-Kimana-Tsavo road serves tourism sector while the road from Kajiado town via Mashuru linking up with pipeline road and Sultan Hamud-Kibini road serves the Kibini marble quarry. There are several murram roads that serve as feeder roads within the study area especially within the Amboseli National Park. Some roads have not been well maintained and are only motor accessible during the dry season. In cases of heavy rains the Olkejuado River usually swells making the pipeline road impassable.

3.1.3 Climatic conditions

This ecosystem falls in both ecozones V and VI (Jaetzold Schimdt, 1983) and is described as a rangeland (semi-arid to arid zones). It has a bimodal pattern of rainfall with the short rains falling between October and December while the long rains fall between March and May (Kajiado ASAL Atlas, 1990). The average annual rainfall ranges between 475mm and 750mm approximately in the major parts of the study area. Rainfall is strongly influenced by altitude as such heavy rains (about 1300 mm p.a) are realised around Loitokitok due to the influence of Mt. Kilimanjaro, and around Chyulu hills, whereas the areas around Lake Amboseli receive less than

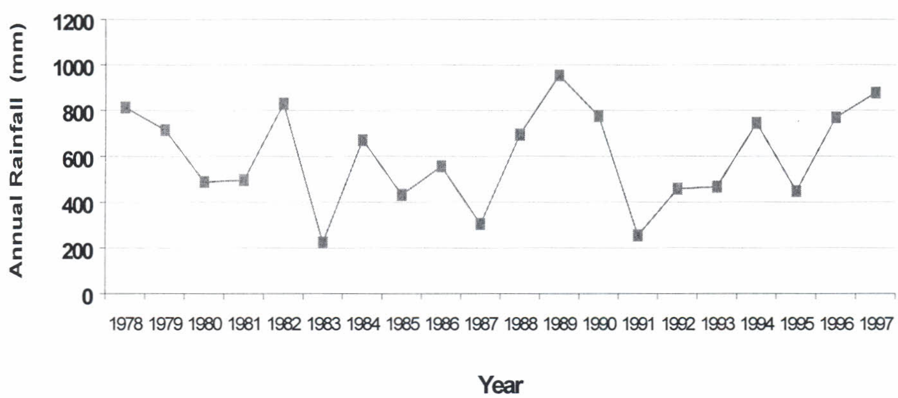
500mm p.a (FAO-KARI report, 1978; Jaetzold, 1983). Temperatures also vary with altitude and range from a mean maximum of about 34^{0}C around Lake Magadi to a mean minimum of 22^{0} C on the slopes of Mt. Kilimanjaro.

Figure 2: Annual Average Monthly Temaparature 1978- 1999



Rainfall and temperature data from Makindu rainfall station, Kenya.

Figure 3: Annual Total Rainfall



3.1.4 Geology, Soils and Drainage

The rocks of the study area can broadly be subdivided into basement system rocks, volcanic and superficial deposits (FAO, 1978; Saggerson, 1962). Recent volcanic activity has significantly enriched large areas of basement system rocks with volcanic material. This enrichment has taken place either directly through deposition or re-deposition of volcanic materials. The recent volcanic ash is represented by strings of ash and cinder cones of the Chyulu range. The surrounding lava flows consist of olivine basalts while superficial deposits are present in the form of various lacustrine, colluvial and aeolian deposits. The lacustrine deposits, which are re-deposited volcanic ashes, dominate the Amboseli basin and their consolidation is attributed to the calcerete and fragipan formation. The alluvial deposits consist mainly of clay and sandy loam sediments derived from basement system rocks. Aeolian deposits are likewise well developed along the western edge of Lake Amboseli.

The soils of Amboseli ecosystem are therefore based on altitude, landforms (their shape, stability and age), geology and climate (Republic of Kenya, 1990). On the open plains there are typically heavy clays, of the type known as "black cotton" soils, which are shallow, poorly drained and extremely difficult to work, often waterlogged in the rainy season and very hard/cracking type in the dry season. These heavy soils are formed by alluvial deposits from seasonal streams and past lakebeds and are of medium to high natural fertility. Where the plains are derived from recent volcanic activity they are usually shallow, very porous and therefore have little water holding capacity. Most of the remaining regions consist of loose volcanic soils and rocks (Saggerson, 1962). It is believed that the first lava flows of Kilimanjaro obstructed the formerly west-east running Namanga River, causing a topographical depression in which the Lake Amboseli was formed. Subsequently sedimentation has filled up the lake and a lacustrine plain formed the present Amboseli basin.

The physiography of the area is influenced by geology and therefore the basement landscape systems comprise hills, foot slopes, uplands and erosional plains. The hills constitute the oldest landscape (Saggerson, 1962) and are surrounded by long, slightly concave foot slopes built by colluvium of eroded material from the steep hill slopes. A vast area is covered by gently undulating erosional plains, which are a result of long undisturbed periods of weathering and erosion (Saggerson, 1962). The volcanic systems are grouped as the Kilimanjaro and Chyulu systems. Kilimanjaro covers a large area and consists of volcanic uplands, foot slopes and piedmont plains. The uplands comprise a vast undulating area, densely dotted volcanic vents that occur as small, rounded, rocky hills. At the margin of lava flows, piedmont plains of flat, gently undulating topography have been formed. These are the result of colluvial and alluvial deposition of material eroded from higher positions.

The study area covers some of the water catchments areas such as the Eselengei and the Olkejuado/Kiboko seasonal rivers. The south-eastern area is drained by a considerable number of small perennial rivers, running from the Kilimanjaro slopes and fed by the melting snow-cap most of which join together to form the Noolturesh swamp. Boreholes are scarce and many of the ones constructed earlier have broken down. Numerous wells are dug in beds of the seasonal rivers for water during the dry season especially near Eselengei.

3.1.5 Vegetation of the area.

According to Pratt and Gwynne, 1966 in their classification of East African ecological zones, the vegetation of Amboseli ecosystem falls into zone V. It is described as thorn bush land and thicket and dominated by *Acacia* and *Commiphora* woody species and grasses such as *Cenchrus ciliaris* and *Chloris roxburgiana.* The vegetation of the area is mainly determined by altitude, soil type, extent of human utilization and livestock grazing. Woody cover is found on the units over basement complex, while bush lands are confined to the basalt outcrops on volcanic uplands

(Bekure et al. 1991). At lower altitudes, vegetation is scarce and herbaceous layer is found in areas where both basement and volcanic ash mix. Ground cover throughout the area varies seasonally with rainfall and grazing intensity. Woody canopy cover ranges from less than 1% on heavily settled areas to about 30% on a few steep slopes where no settlement is possible. A large portion of the area consists of semi-desert type of vegetation. The dominant tree species are species of *Acacia, Commiphora* and *Balanites aegyptiaca* while grass species include species of *Themeda triandra,* species of *Digitaria*, *Pennisetum* and *Sporobolus* genera. There are about six vegetation communities found within the area *Commiphora-Acacia, Balanites-Acacia, Pennisetum-Cenchrus, Sporobolus-Acacia* and short grasses (Kiunsi, 1993).

3.2.1 **Approach to the problem.**

Since an ecosystem is a complex system of interacting variables, this study employed the synecological approach that combined both descriptive, pattern and systems analysis methodologies (Kershaw, 1985). For this study, the parameters of immediate concern were those deemed to be of most consequence to the present and future management of resources within the study area including: -

i. Land-use, land cover-types and their distribution within different land-use systems in the region.
ii. Floristic composition and diversity within the different cover-types found in the region

3.2.2 **Methodology**

Both computerised and field techniques were used for image classification and data collection. After image processing, strata were chosen and sample plots for detailed study were randomly located within each strata depending on accessibility in the field with the aid of topographic maps (McCloy, 1995). Land cover and land-use maps were created for 1988 and from 1998 from Landsat-5 Thematic Mapper images and changes were analysed using geographic information systems, ARCVIEW and ARCINFO (ESRI, 1992). The ten-year period was selected based on image availability as well as the assumption that a decade was long enough to exhibit changes in vegetation cover. At the same time, according to available literature this period represented an era during which there occurred numerous changes in human population, land tenure policies, external influence, and socio-economic lifestyles as well as climatic changes within the study area (KWS, 1990; Bekure *et al.* 1991; Campbell, 1999).

(a) Field measurements and Ground truthing

Field data are defined as verified or verifiable data collected using proven, relatively close-range techniques such that the resultant data are much more accurate, consistent and at a higher resolution than the information derived from remotely sensed data (McCloy 1995; Lillesand and

Kiefer, 1994). For this study, this was done by the stratified random access method, i.e, strata were delineated from the image and then test plots were randomly located within the strata depending on ground accessibility (McCloy 1995; Reid et al. 1997). The test plots on the ground were located using a Global Positioning device (Garmin GPS) and reference points spread out over the study area were mapped and recorded during the entire fieldwork. Location of sampling plots depended on changes in landscape and land cover-types as derived from the image as well as observations on the ground at the time of sampling. At every test plot ground cover was surveyed in a 2500m^2 plot and the following data were recorded: -

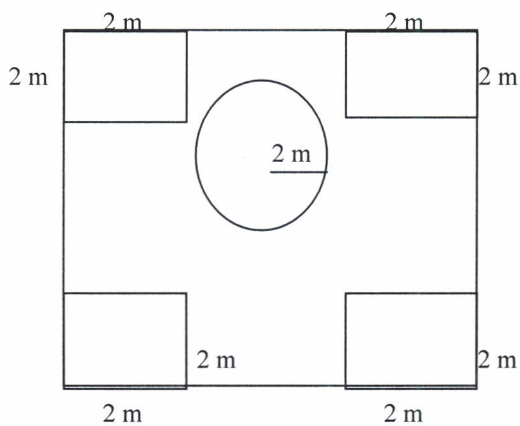
(a) Position (UTM co-ordinates and reference points/benchmarks).
(b) Percentage cover of grass, shrubs, crops and trees, using the Braun-blanquet method
(c) Dominant tree, shrub and grass species
(d) Approximate height of grass, shrubs and trees using a clinometer
(e) Soil type, colour and other field characteristics such as slope and moisture status
(f) Striking disturbance features such as erosional features, felled trees, heavily trampled areas by humans, livestock or wildlife
(g) Any geological features such as rocky outcrops, hills etc.

In addition to the above data, photographs of the plots were taken to verify the classification process. The photographs represented a survey revealing different aspects of the landscape and showed in detail the different types of land-use/land cover as observed in the field. After the survey, all the sample points were transferred from the topographic maps to a paper copy of the satellite image and compared with the results of the earlier preliminary classification for final map output.

(b) *Vegetation cover, species composition and abundance measurements.*

Several characteristics of the vegetation and the surrounding landscape were measured at five points within each of the 50 x 50-m sample plot. The five points were laid out in a regular grid to provide maximal plot coverage (Fig. 4). For all plots, the five points were averaged to derive plot mean that was used in analysis.

Figure 4: Sketch of ground sample plots (50 m by 50 m) during fieldwork.



The vegetation was divided into three strata: herbaceous plants, bush/shrubs (woody plants <6 m tall) and trees (> 6 m tall). For the herbaceous layer, the maximum height was measured at the point and percentage cover estimated visually within a 2-m diameter ring centred on the point. Shrub and tree canopy cover was quantified differently using a densiometer and recorded according to the *Braun-blanquet* method (Kershaw, 1985). Visual estimates were made for those plots covered by cultivated crops and grassland to enable comparison at different locations (Reid et al. 1997). The types of crops cultivated in each plot were also recorded.

(c) *Interpretation of Landsat TM images and production of Land-cover maps*

For the investigation of landscape changes, interpretation of Landsat images for the years 1988 and 1998 at a scale of 1:250 000 were used. Land cover maps for 1988 and 1998 were created so that the relative distribution of each type in each year could be calculated and compared. The 1998 map was created first because it could be easily ground-truthed against the then current conditions in the field. The map was created in three steps: field verification, classification and accuracy assessment. Field verification was conducted in April-June 1999 by locating sample plots in the study area and collection of data which was later on divided into two groups: one group was used for training sites during supervised image classification and the other set used to assess the accuracy of the final classification results. The final map was produced with an accuracy of 84%. With the help of ground truthing data, vegetation/land cover types were visually interpreted from the images for each year and features were traced and co-registered to a transparent sheet. Landscape pattern information was then digitised from the transparent sheets, edited, processed and labelled using geographical information system (GIS) program packages, Arc/Info and ArcView (ESRI 1992) and IDRISI (Eastman, 1995). At this stage the study area was described digitally through land cover maps for 1988 and 1998 respectively and several aspects of temporal changes quantified. The relative area of each land cover type in 1988 compared to 1998 was measured to show overall changes.

(d) *Land-use, Land Tenure and pastoralists' perceptions on change*

Past and present land tenure and land-use as well as people's perception on ongoing changes were investigated by means of informal group interviews during fieldwork supported by available secondary literature. For every location several ad-hoc informal group interviews were carried out. Informants were asked about past and present use of pastoral lands, brief information on population trends, socio-economic lifestyle changes and details on past and present grazing patterns and ecological trends. The questions asked could be summarized under the following broad headings:

- ③ *Type of land ownership:* Focused on membership of either a group ranch or an individual ranch. If individual, opinion was sought on what could have been the possible causes of subdivision, size of land per household, adequacy of land owned for livestock production, role of subdivision on immigration of other non-Maasai tribes due to land sales. If still members of group ranches, then one was asked about the present inherent difficulties they are facing given the ongoing land tenure changes within the neighbouring group ranches.
- ③ *Land use practices:* These targeted past and present land use activities such as livestock production, crop agriculture (cash/subsistence), wildlife tourism and commodity trade. Reasons that could have led to the revolution of the present activities and how profitable they are compared to the past traditional livestock production systems.
- ③ *Resource use strategies:* Focussed on current and past resource use patterns given the ongoing changes in land tenure regimes and land use activities. People's perception on use and accessibility of pasture, water and dry season grazing areas as opposed to the past regulated communal usage. The effects of these changes on productivity levels and overall livelihoods, for example, effects on animal numbers per household and levels of family income.
- ③ *Ongoing ecological changes and perceived causes:* These focused on people's perception on the evident ongoing changes in vegetation species, cover, structure and patterns. Their opinion on the role of the ongoing changes in land tenure and land use systems onto the observed vegetation changes.

3.3.0 Data Analysis.

a. *Measured field vegetation parameters*

Multiple linear regression and correlation analyses were done for the different vegetation parameters measured at a set of data points on the ground. This was in view of the fact that environmental or spatial variables are often correlated with each other. The data therefore was simplified by looking at the sets of related variables e.g. vegetation cover-types with soil type, elevation, climate among other factors. The factors affecting species composition other than anthropogenic influences such as human settlements could therefore be positively identified (Reid et al. 1997). Correlation analysis was also applied in vegetation pattern analysis to show the association patterns of vegetation species within different cover-types (Greig-Smith et al., 1963).

b. *Vegetation species and cover types diversity.*

This was calculated based on the number of species/cover types within the sample plots/locations. The Shannon-Weiner ($H\chi$) index was applied.

$$H\chi = -6P_i \log P_i$$

Where, H' = Shannon-Weiner diversity index (Simpson, 1949).

$P_i = n_i / N_i$

N_i = Total number of species or cover types

n_i = Number of individuals or cover types of the i^{th} species/cover types in the sample/landscape

The diversity index H' equals zero whenever there is only one species or when the landscape contains only one cover types (i.e no diversity). The index increases as the number of different species of cover types (i.e species or cover types richness) increases and/or the proportional distribution of the number of species or area among cover types becomes more equitable.

c. Change detection and analysis.

To analyse land cover change in the study area, the vegetation map for the year 1988 was compared to the present static map for 1998 and areas of change delineated. The annual rate of change was calculated using the relative growth rate formula (Harper, 1977) which takes into consideration the area coverage in the base year and in the final year together with the time frame between the two years.

$$\ln x = \frac{\ln Q - \ln P}{t_2 - t_1}$$

Where x is annual rate of change in ha; P is the land-use/land-cover area in ha for year t_1; Q is the land-use/land-cover area in ha for year t_2; t_1 is the initial year and t_2 is the final year considered.

d. Indices of landscape pattern (dominance, diversity, fractal and heterogeneity)

These were quantified for each year to give an insight on the processes of landscape pattern changes- using FRAGSTAT software (McGarical and Marks, 1994). The ability to quantify landscape structure is a prerequisite to the study of landscape function and change. For this reason, much emphasis has in the past been put in developing methods to quantify landscape structure and pattern. Most of these efforts have developed user-generated programs which allowed for the inclusion of customized analytical methods but generally lacked the advanced graphics capabilities to link to the applied remote sensing and GIS programs used herein to produce the land use/land cover maps for the area. Likewise the inherent capability of FRAGSTATS to offer a comprehensive choice of landscape metrics and its versatility in handling both vector and ASCII image files, 8 or 16 bit binary image files, Erdas image files, and IDRISI image files made it more appropriate for this kind of analysis.

i. Mean Cover types size MPS

$$\text{MPS} = \frac{\mathbf{A}}{\mathbf{N}}\left[\frac{1}{10{,}000}\right]$$

A is landscape area (m^2), N is total number of cover types.

ii. Dominance index

$$\mathbf{D = \ln}\ \boldsymbol{n}\ \mathbf{+}\overset{m}{\underset{i=1}{\mathbf{6}}}\ \boldsymbol{P_i}\ \mathbf{\ln}\ \boldsymbol{P_i}$$

Where P_i is the proportion of the landscape in habitat i, ln n is the maximum diversity when habitats occur in equal proportions and m is the total number of habitat categories.

iii. Fractal index (Turner, 1994)

$$\textbf{FRACT} = \frac{2 \ln \mathbf{p}_{ij}}{\ln \mathbf{a}_{ij}}$$

Where **a** is the area (m^2) of each polygon, **p** is the perimeter (m).

A fractal index greater than 1 for a 2-dimensional cover types indicates an increase in shape complexity. For example, FRACT approaches 1 for shapes with very simple perimeters such as circles or squares while it approaches 2 for shapes with highly convoluted perimeters (McGarical and Marks, 1994).

CHAPTER FOUR: RESULTS AND DISCUSSION

4.1.0 Field vegetation attributes

A total of 117 species were identified during fieldwork (April-June, 1999), which included 83 woody species (trees and shrubs) and 34 grasses/sedges/herbs. The woody species were more abundant compared to the herbaceous species. Land use types and human activities influenced plant species distribution and life forms. The three land-use types identified during fieldwork differed widely in species numbers and distribution, especially the herbaceous and shrub species as shown in Table 1. For example, commercial wildlife-livestock land-use type supported more dwarf shrub and herbaceous species than all the other land use types.

Table 1: Vegetation species life forms and diversity from $50m^2$ test plots within land-use types.

Vegetation Life forms / Land use types	Trees	Bush	Shrubs	Herbaceous	Species diversity index
Wildlife	4	6	9	12	3.56
Livestock & crop agriculture	7	7	10	13	3.28
Commercial wildlife & livestock	7	13	25	19	3.70

The variability in vegetation species composition from one land use type to another as shown in Table 1 above could be attributed to the fact that some species have a greater ecological amplitude than others across different locations while other species exhibit local dominance. For example *Acacia mellifera* and *Acacia xanthophloea* could be seen to have a higher ecological amplitude given their spread over both the plains and hilly regions while, *Salvadora persica* and *Salvadora monoica* had localised dominance within salt plains in the lake bed. Such species with wider ecological amplitudes are referred to as ecological dominants (Odum 1971).

To test the influence of environmental parameters on vegetation species, the distribution of plant species and their diversity were tested within broad soil classes. The results are shown in the Table 2 below.

Table 2: Number of species, total number of individuals and species diversity within soil types.

Soil type	Number of species	Total number of individuals	Species diversity
Bottom lands	16	19	2.79
Dunes	6	6	1.79
Erosional plains	25	42	3.06
Flood plains	29	52	3.23
Foot slopes	79	231	3.98
Hills	9	12	2.25
Lacustrine plains	27	51	3.19
Swamps	17	24	2.80
Uplands	57	107	3.88
Volcanic plains	4	4	1.38

From Table 2 above, foot-slope soils supported the highest number of species while volcanic plains supported the least number of species. Dunes were completely dominated by grass species of *Sporobolus* genera and *Cenchrus ciliaris*; erosional plains hosted a mixture of vegetation classes but was dominated by bush species such as *Acacia mellifera, Acacia nubica, Balanites aegyptiaca* and some dwarf shrubs mainly *Sericocomopsis hildebrantii.* The main grass genera in this soil type were species of *Pennisetum* and *Cenchrus* genera.

From field data, an attempt was made to classify vegetation cover of the study area based on the percent cover values within the sampled test plots as well as their species composition. This gave the following results as shown in Table 3.

Table 3: Field based vegetation classification showing cover values within each vegetation class.

% cover / Cover type	Whole plot	Tree	Bush	Dwarf shrub	Herbaceous	Bare ground	Cultivated field
Bare ground	0	0	0	0	0	100	0
Bushed grassland	65+	<10	20+	<20	20+	<35	0
Dwarf-shrub grassland	85+	<10	<10	65+	<10	<15	0
Agriculture	80+	<10	<10	<10	<10	<20	80+
Grassland	75+	<5	<15	<15	55+	<25	0
Thicket	75+	<10	70+	<20	<10	<20	0
Wooded grassland	90+	30+	<10	<10	50+	<10	0

Plant species diversity was tested within derived cover type classes, revealed the following diversity indices (Table 4) across different classes.

Table 4: Vegetation species diversity within vegetation classes derived from field data.

Cover type	Species diversity index
Bush land	3.78003
Bushed grassland	3.81255
Agriculture	0.3446
Dwarf shrub grassland	2.88822
Grassland	3.64221
Thicket	3.4187
Wooded grassland	2.80968

The vegetation cover-types differed in species composition and diversity. Table 4 shows that bushed grassland had the highest species diversity while areas of intensive agriculture had the lowest. From this it can be concluded that changes in land-use will have a great effect on species diversity even though the overall ground cover may not change.

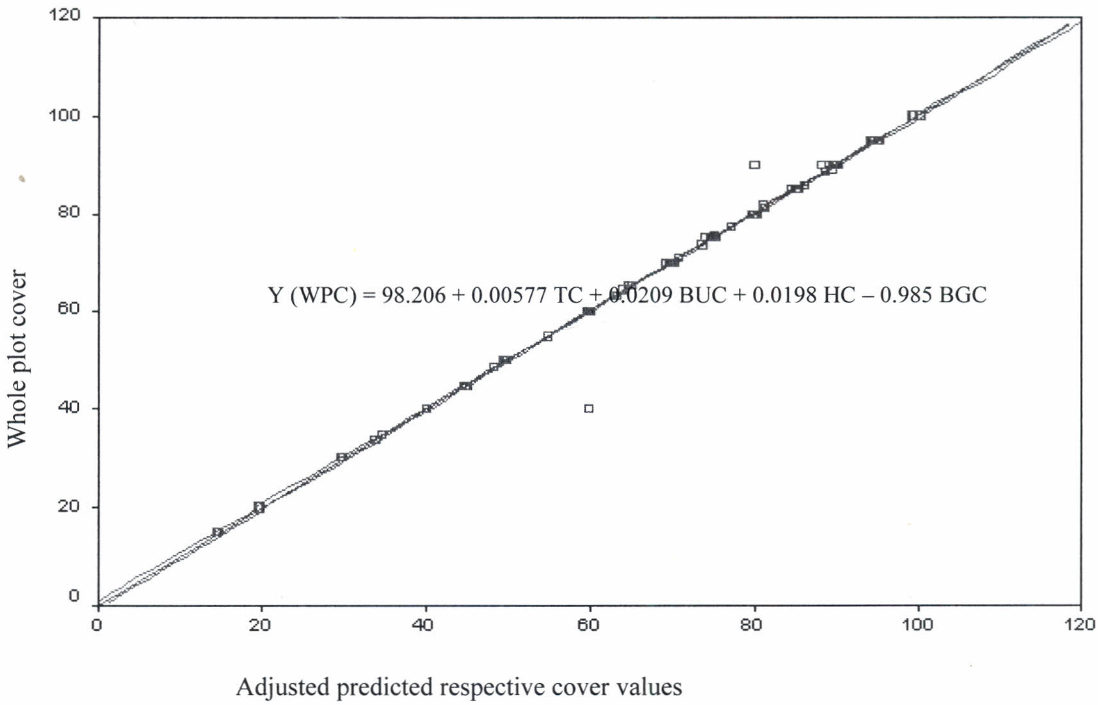
Since different classes of vegetation life-forms and species composition determine the structure and composition of different cover-types, a regression analysis was done to test their contributions to the estimated whole plot cover values. Multiple regression analysis of different vegetation cover attributes revealed that whole plot cover could only be explained by all the vegetation parameters measured at that particular sampling plot including bare ground cover (Table 5 and Figure 5). Single attributes (figure 6) could not explain the whole plot cover, which was the basis of assigning classes during image classification.

Table 5: Multiple regression analysis of vegetation class cover percentages within plots.

Model	R	R Square	Adjusted R Square	S.E
1 (Woody cover)	.058[a]	.003	.003	18.7356
2 (Tree & Bush/shrub cover)	.209[b]	.044	.042	18.3593
3 (Tree, Bush/shrub & Herbaceous cover)	.896[c]	.802	.802	8.3515
4 (Tree, Bush/shrub, Herbaceous & Bare ground cover)	.998[d]	.996	.996	1.2520

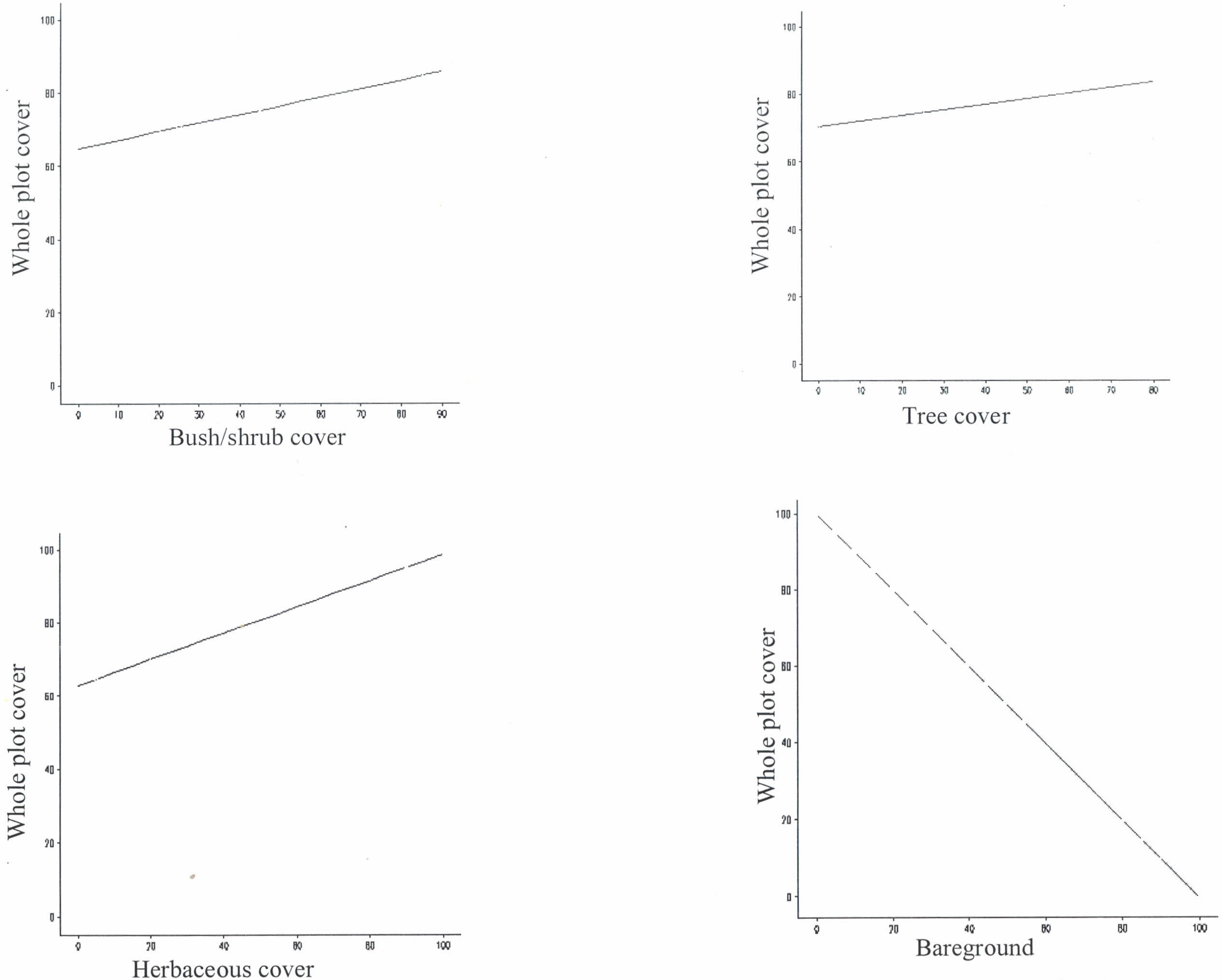
a. Predictors: (constant), Woody cover
b. Predictors: (constant), bush shrub cover
c. Predictors: (constant), Woody cover, bush-shrub cover, herbaceous cover
d. Predictors: (constant), Woody cover, bush-shrub cover, herbaceous cover, bare ground

Figure 5: The combined effect of cover % for all the vegetation classes in explaining whole plot cover (at 95% confidence level).



From the equation on the graph; WPC = Whole plot cover, TC = Tree cover, BUC = Bush cover, HC = Herbaceous cover and BGC=Bare ground cover.

Fig. 6: Relationships between whole plot cover and its individual component cover values



According to the whole plot cover model summary results in Table 6, the influence of bare ground cover in the model was found to be significant since R^2 value increased from 4.2% accounted for by tree and bush cover to 80.2% accounted for by all the vegetation cover types revealing a highly fitting model in explaining whole-plot cover. Analysis of variance results in Table 7 below, likewise reveals how well the model fitted in predicting whole plot cover with a P-value of 0.000 at $p<0.05$.

Table 6: Model Summary for whole plot cover.

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	.058[a]	.003	.003	18.7356
2	.209[b]	.044	.042	18.3593
3	.896[c]	.802	.802	8.3515
4	998[d]	.996	.996	1.2520

a) Predictors: (Constant), Tree cover

b) Predictors: (Constant), Tree cover, Bush shrub cover

c) Predictors: (Constant), Tree cover, Bush shrub cover, Herbaceous cover

d) Predictors: (Constant), Tree cover, Bush shrub cover, Herbaceous cover, Bare ground cover

Table 7: Analysis of Variance for the whole plot cover model.

Model		Sum of Squares	df	Mean Squares	F	Sig.
1	Regression	1564.537	1	1564.537	4.457	.35[a]
	Residual	471074.581	1342	351.024		
	Total	472639.118	1343			
2	Regression	20634.328	2	10317.164	30.609	.000[b]
	Residual	4520004.79	1341	337.065		
	Total	472639.118	1343			
3	Regression	379178.004	3	126392.668	1812.157	.000[c]
	Residual	93461.114	1340	69.747		
	Total	472639.118	1343			
4	Regression	470540.289	4	117635.072	75048.210	.000[d]
	Residual	2098.828	1339	1.567		
	Total	472639.118	1343			

a) Predictors: (Constant), Tree cover

b) Predictors: (Constant), Tree cover, Bush shrub cover

c) Predictors: (Constant), Tree cover, Bush shrub cover, Herbaceous cover

d) Predictors: (Constant), Tree cover, Bush shrub cover, Herbaceous cover, Bare ground cover

e) Dependent Variable: Whole plot cover

Table 8: Whole plot cover model coefficients

Model	Unstandardized Coefficients		Standardized Coefficients	t	Sig.
	B	Std. Error	Beta		
1 (Constant)	72.033	.640		112.547	.000
Tree cover	8.254E-02	.039	.058	2.111	.035
1 (Constant)	65.837	1.035		63.591	.000
Tree cover	.127	.039	.088	3.267	.001
Bush shrub cover	.179	.024	.203	7.522	.000
2 Constant)	12.522	.880		14.227	.000
Tree cover	.852	.020	.594	41.900	.000
Bush shrub cover	.823	.014	.936	58.553	.000
Herbaceous	.853	.012	1.199	71.698	.000
3 Constant)	98.206	.379		259.364	.000
Tree cover	5.766E-03	.005	.004	1.242	.214
Bush shrub cover	2.094E-02	.004	.024	5.321	.000
Herbaceous	1.976E-02	.004	.028	5.085	.000
Bare ground	-.985	.004	-.979	-241.427	.000

Dependent Variable: Whole plot cover

From the model, the whole plot cover at times may not add upto 100% (+/- 5%) due to errors in estimating respective cover percentages within each plot. These errors usually arise from the fact that in any given plot which is composed of tree or bush species it is often difficult to estimate the under-storey cover values for herbaceous species or bare ground, thereby resulting in either underestimation or overestimation. Despite these errors, the general statistical tests revealed the suitability of the model equation in predicting whole plot cover.

Multiple regression analysis results gave a higher value for regression co-efficient (R) when all plot parameters were combined. This necessitated a further analysis to test for any collinearity between those parameters that could be responsible for the high R-value. This test was done using Pearson's correlation analysis, which revealed the following results in Table 9.

Table 9: Pearson correlations between vegetation classes cover percentages within sample plots.

		Woody cover	Bush shrub cover	Herbaceous cover	Bare ground
Woody cover	Correlation		-.151**	-.329**	-.068*
	Sig. (2-tailed)		.000	.000	.013
	N		1344	1344	1344
Bush shrub cover	Correlation	-0.151		-.547**	-.186**
	Sig. (2-tailed)	0.000		.000	.000
	N	1344		1344	1344
	Correlation	-0.329**	-0.547**		-.489**
Herbaceous cover	Sig. (2-tailed)	0.000	0.000		.000
	N	1344	1344		1344
Bare ground	Correlation	-0.114	-0.158*	-0.404**	
	Sig. (2-tailed)	0.000	0.000	0.000	
	N	1344	1344	1344	

**Correlation is significant, 0.01 level (2-tailed).

*Correlation is significant, 0.05 level (2-tailed).

To test the significance of bush-herbaceous cover relationship, an analysis of variance was performed which gave the results shown in Table 10. The tests revealed a high significant relationship between bush and herbaceous species at $p<0.05$.

Table 10: Analysis of Variance, relationships between bush and herbaceous cover values.

Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	279613.84	1	279613.844	573.158	.000[a]
	Residual	654691.94	1342	487.848		
	Total	934305.78	1343			

a. Predictors: (Constant), Bush-Shrub

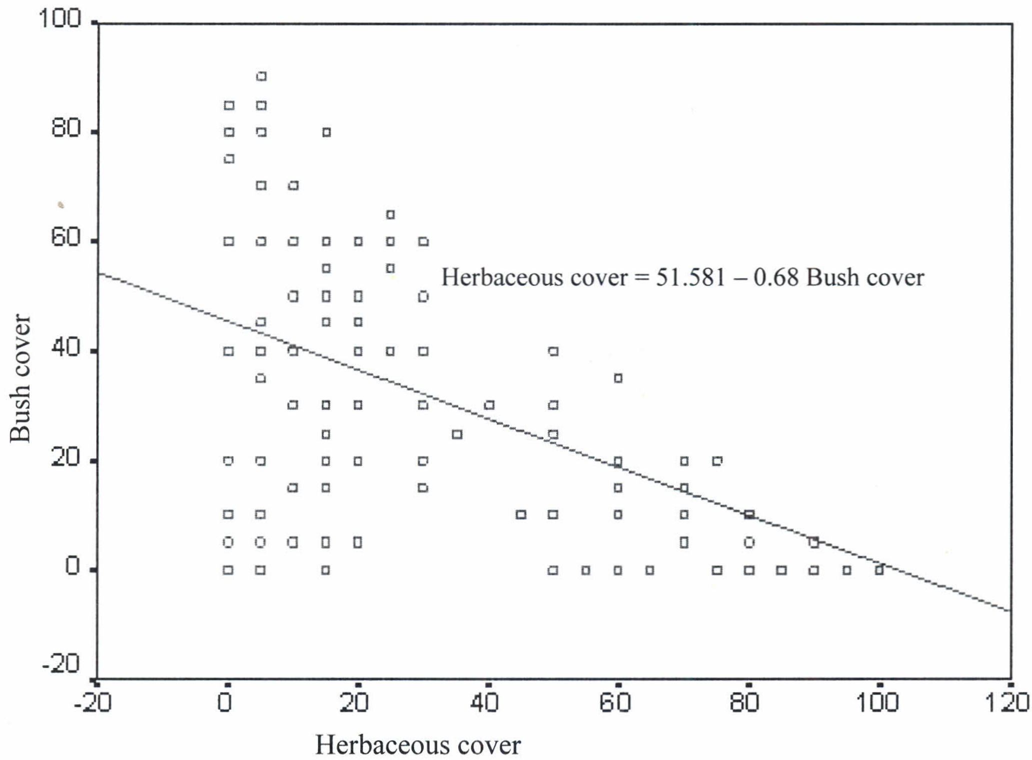
b. Dependent Variable: Herbaceous

A general regression model was generated to help further explain the relationship between bush and herbaceous species cover. The results in Table 11 show the model coefficients for bush-herbaceous cover relationship, which also explains the graph in figure 7.

Table 11: Coefficients for the relationship between bush and herbaceous species

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
4	(Constant)	51.581	1.092		47.230	.000
	Bush-Shrub	-.677	.028	-.547	-23.941	.000

Figure 7: Relationship between herbaceous and woody vegetation species.



The above statistical analyses have revealed a negative relationship (-0.677) between bush and herbaceous species. This can likewise be supported by observations in the field, which revealed an invasion of bush species into the former grazing areas. This was prevalent in areas where large ungulate grazing pressure had reduced either due to lack of watering facilities or by being fenced off. This is especially the case within the northern part of the study area towards Eselengei group ranch. Other similar studies elsewhere by (Norton- Grifiths, 1979; Dublin et al. 1990; Skarpe, 1992) indicated that changes from mixed savannah to grassland and vice versa are highly dependent on fire and large ungulate effects especially within African rangelands. They attributed this to excessive killing of trees by elephants and the slow plant regeneration due to fires and browsing. A general conclusion would be that, the observed dynamics in vegetation cover-types within rangelands could therefore be part of natural, large-scale oscillations, involving fire, elephants, other browsers and grazers, and coupled by human land use activity. This seems to explain in part the situation within the study area in which large ungulate migratory routes are being blocked by crop agriculture and fencing coupled by breakdown of watering facilities which reduces grazing pressures and the possibilities of opening up of thickets and woodlands to enhance grass regeneration. However, this process is likewise occurring within those areas open to grazing as a result of overgrazing by cattle due to reduced mobility.

4.2.0 Land-use-land cover map of the study area

The land cover map for the study area was produced through a combination of classification parameters derived from both Landsat Thematic Mapper (TM) image for 1998 and field ground truth data. This map provided the best and most accurate information available at the level of 50m by 50m resolution grids on the ground. The Landsat TM image was first classified by an unsupervised method called MAXLIKE operation in IDRISI (ESRI 1992), which statistically separated the entire population of pixels into homogenous clusters in terms of their spectral characteristics (Plate 1). This was then followed by field ground truthing to collect data from training sites for validation of the interpreted information from Landsat TM image. The clusters were then labelled into classes based on available collected field and ancillary information such as existing topographic maps and visual analysis of spatial vegetation cover distribution patterns.

The second map was produced from supervised classification combining both field ground truth data and inferred information from the image to enable accurate classification of every pixel into their respective classes. During field verification the resultant clusters were found not to be of a clear representation of the real cover-types as observed in the field. This was attributed to the confusion which often occurs between reflectance characteristics of objects within satellite images. Therefore, the final map was produced through image digitisation by means of visual interpretation and classified using collected field data. From Landsat TM image of 1988 and 1998 six land cover-types were classified for the study area (Plate 2 and 3).

Plate 1: 1998 Land cover types derived from fine unsupervised classification

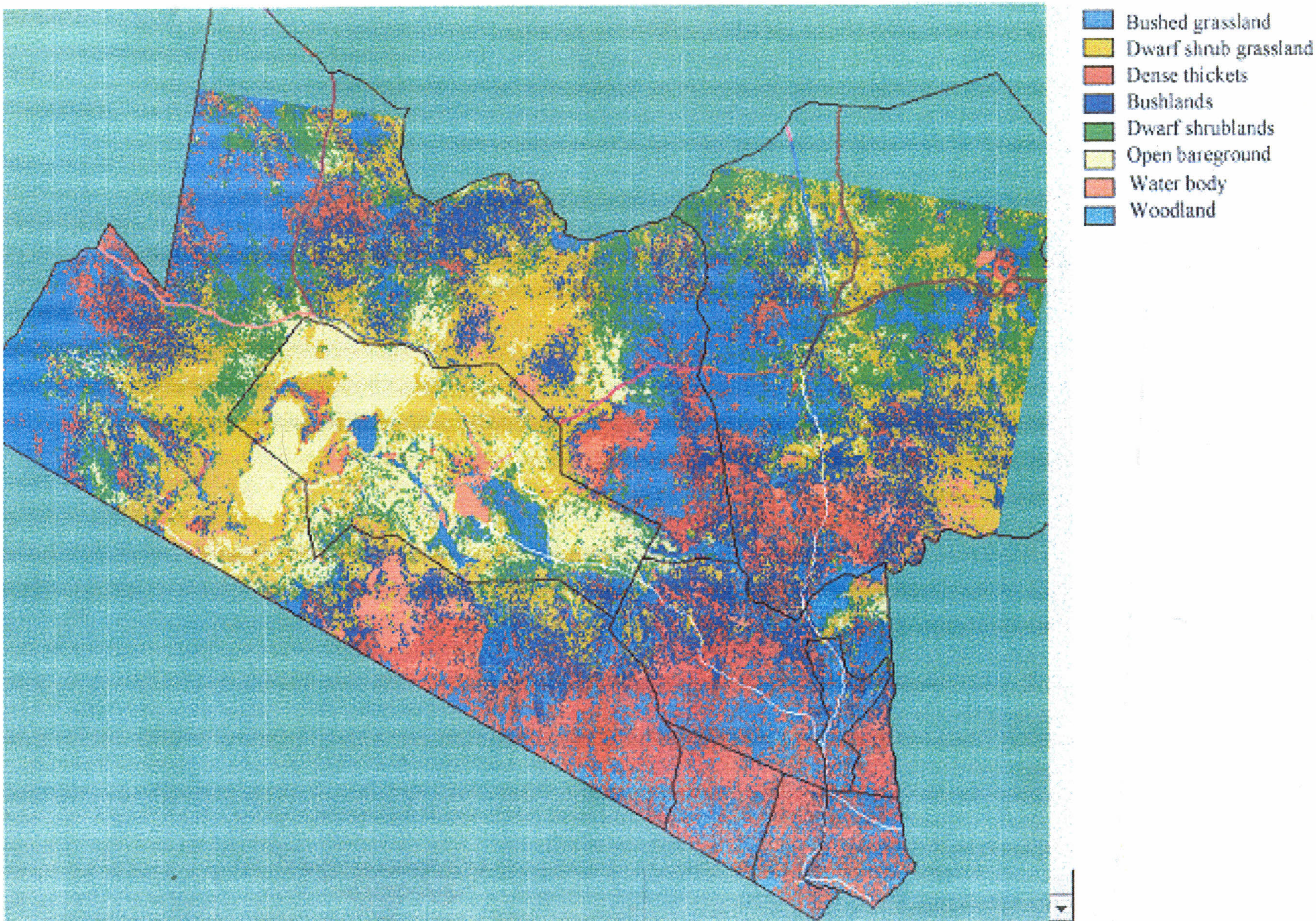


Plate 2: 1998 Land use/Land cover types of Greater Amboseli Ecosystem

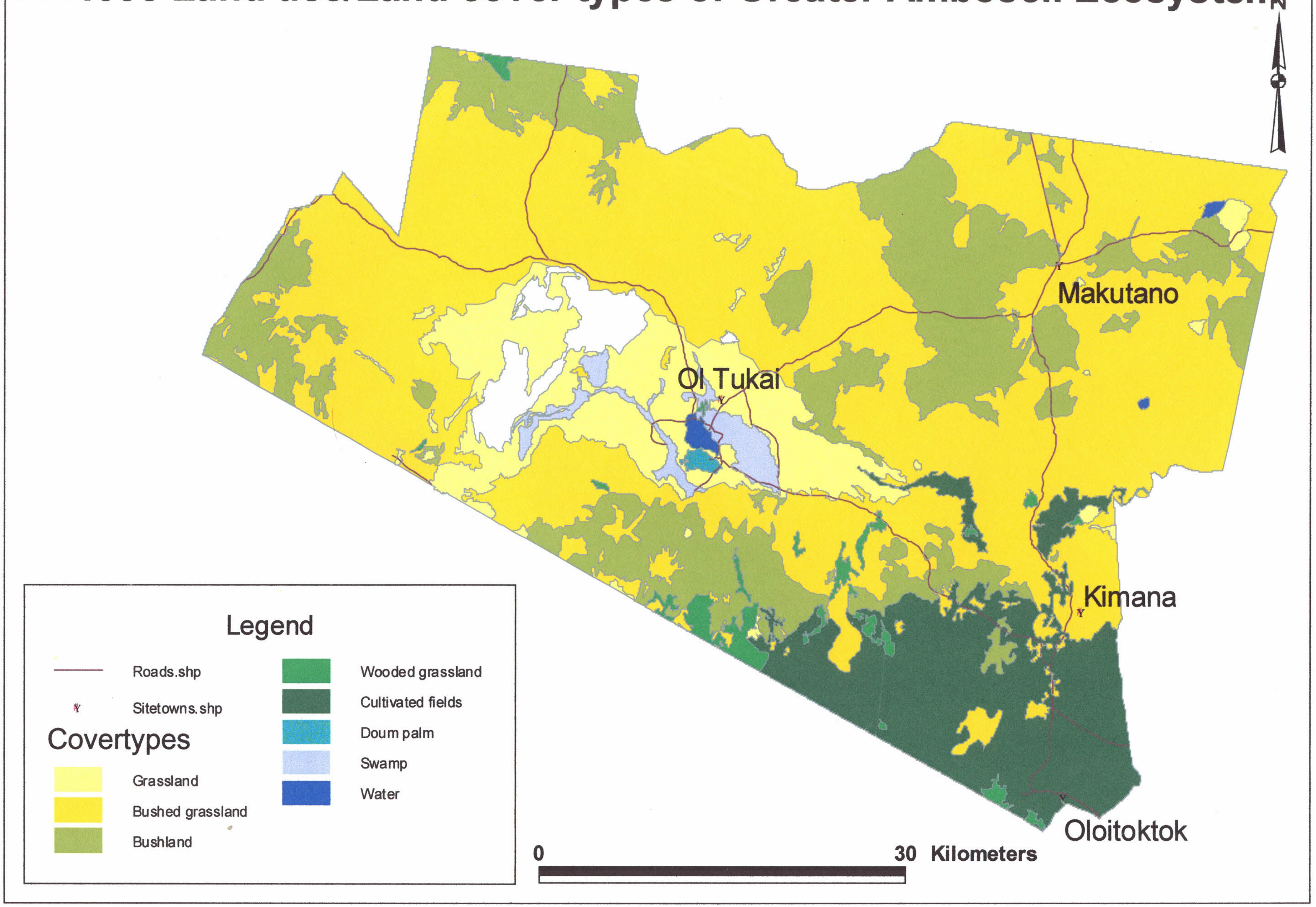
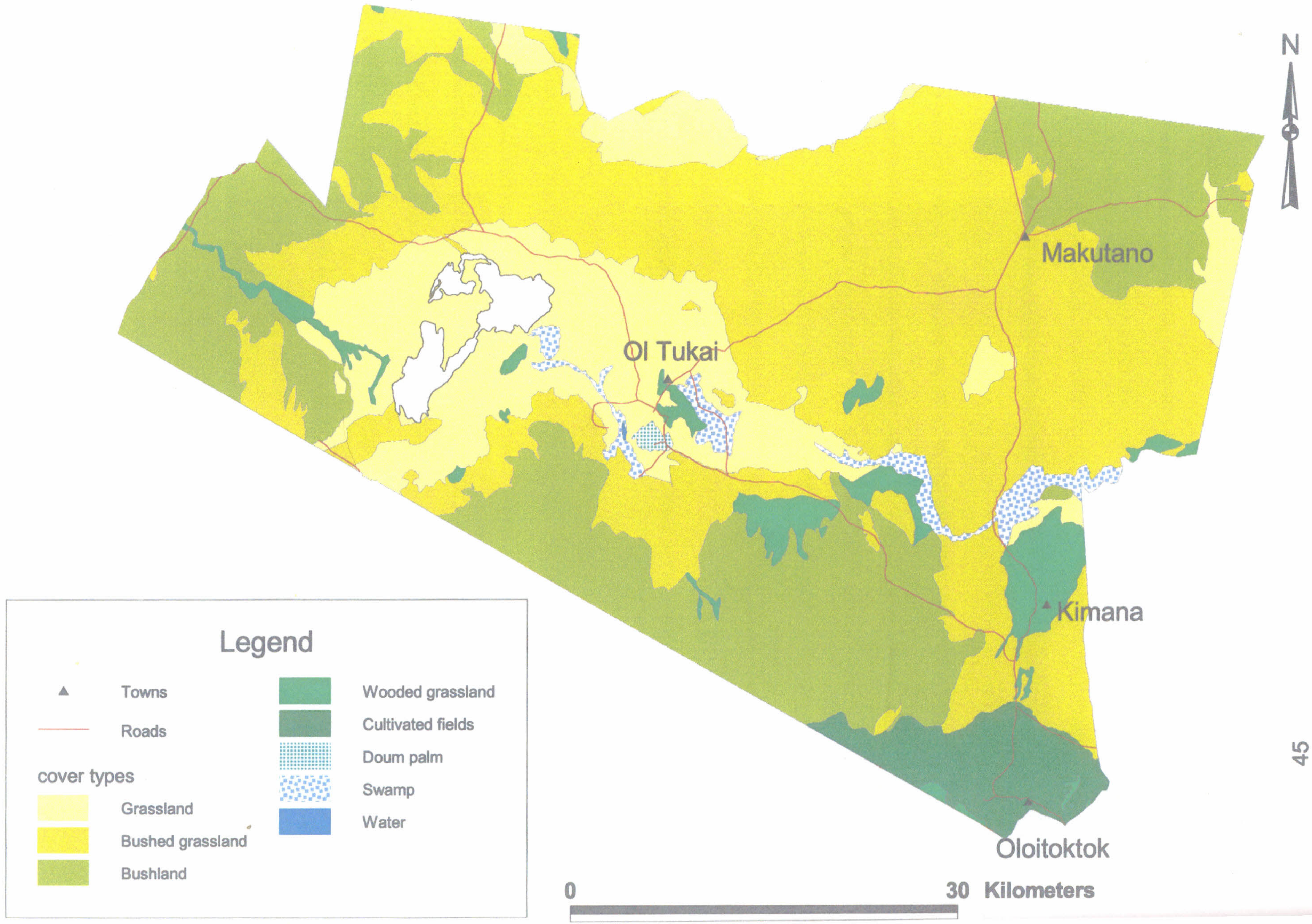


Plate 3: Land use/Land cover types of the Greater Amboseli Ecosystem 1988



4.2.1 Classification procedure

The principles that underlie the classification criteria used in this study evolved from earlier classifications, including Pratt and Gwynne (1966); Western (1973); Whites (1982) and FAO (1978). These studies, used direct reference to most of the visible vegetation attributes such as height and forms of plants, species composition and cover values. Pratt and Gwynne (1966) stated a general rule for vegetation classification, which indicated that any criteria should use observable attributes so that the results become more objective and unequivocal. This study therefore used physiognomic attributes to classify the vegetation by subdividing cover types according to vegetation species composition and their ground cover percentages. Species abundance and canopy cover were valid measures of species contribution within each cover type. However, some of the classes overlapped with each other or appeared as mosaics thereby giving mixed vegetation cover-types.

It has been noted that within the tropics, rangeland ecosystems exhibit marked resource variation and instability in both time and space with the former being their most important feature (Behnke and Scoones, 1992; Ellis et al. 1992). Therefore, rarely do we find pure vegetation stands; rather there exist a gradient of related cover-types ranging from woodlands to almost treeless grasslands (Swift et al. 1996; Mayaux et al. 1999). In the light of this, it is essential to develop generalised models and or description of cover-types for rangelands, including the study site. This study revealed that cover-types within rangelands could be described in relation to their modifying factors such as soil types, grazing pressure, available soil moisture, topography and human land use. Most of the earlier classifications applied different criteria such as floristic and physiognomic composition (Pratt and Gwynne, 1966; 1977), climatic or edaphic characteristics (FAO, 1978). Nevertheless, there is a great need to try and integrate multiple factors so as to come up with a more valid classification criterion. Different vegetation mosaics can be attributed to different modifying factors implying that within local scales, changes in vegetation cover-types may occur from year to year or spot to spot depending on climatic conditions and land-use practices. For these reasons, it might be concluded that there is no single classification procedure that can be considered absolutely valid or sufficiently detailed for all ecosystems. Therefore, there is a need for modifications or additions of classification criterion especially for those dealing with local or transitional vegetation types such as the Greater Amboseli ecosystem.

4.2.2 Classification Accuracy Assessment

Vegetation cover classes interpreted from the image were compared to the actual field collected data so as to assess their classification accuracy. Since surveying and monitoring of natural resources is necessary for their management, there is an absolute need to test the accuracy of derived information from remotely sensed sources to the actual information on the ground. Likewise, accuracy assessment is necessary for revealing the errors that occur during extraction of such information from remotely sensed data. For this study, a matrix was constructed which compared the counts of both relevant field and derived classes from the image (Table 12).

Table 12: Classification accuracy matrix.

Field data / ***Image data***	Bare ground	Bushed grassland	Bush land	Cultivation	Grassland	Wooded grassland	Total
Bare ground	4	0	0	0	0	0	4
Bushed grassland	2	78	0	3	1	1	85
Bush land	0	4	16	0	0	0	20
Cultivation	1	4	2	21	0	2	30
Grassland	0	6	0	0	33	0	39
Wooded grassland	0	0	0	0	0	5	5
Total	7	92	18	24	34	8	183
Accuracy	.57	.84	.88	.87	.97	.62	.85

The accuracy of classifying any given class was calculated by the formula:

Pr (no. of times a class is classified correctly)

For example the probability of classifying bushed grassland,

Pr(78/92)=0.8478 ~85%

Table 13: Probability of occurrence of a class in relation to total actual occurrence.

Field data / ***Image data***	Bare ground	Bushed grassland	Bush land	Cultivation	Grassland	Wooded grassland
Bare ground	.57	0	0	0	0	0
Bushed grassland	.28	.84	0	.125	.0294	.125
Bush land	0	.043	.88	0	0	0
Cultivation	.14	.043	.08	.80	0	.25
Grassland	0	.39	0	0	.97	0
Wooded grassland	0	0	0	0	0	.62
Total	1	1	1	1	1	1

The error of omission is the error that occurs whenever a particular class based on field data is classified as another different class on the image. For example, the error of classifying bushed grassland as bush land:

E_o *(Bush land/Bushed grassland) =4/92 =0.04347.*

The errors that occurred during accuracy assessment could have resulted from inaccuracies in boundary delineation, which could have been due to a combination of interpretation, identification and cartographic errors. This was amplified by the fact that object boundaries within images were poorly defined; a scenario common in land cover-types in which boundaries are never linear thereby not forming perfect shapes for easy delineation.

From this study, accuracy levels increased with decrease in number of cover classes, revealing that, the more detailed the classification, the less accurate the vegetation cover classes became. This is because satellite images at times tend to give different reflectance values to similar objects at different locations either due to the physical features on the ground, geomorphology, surrounding objects or physiological stages of growth in case of vegetation species. For example, there was an enormous error in classifying thicket cover type based on field data because its reflectance was quite similar to bushed shrub grassland in the image given their similarity in species composition. This also applied to the misclassification of dwarf shrubland as bushed shrub grassland (mixed vegetation). Wooded grassland had the lowest accuracy due to fragmentation, which had resulted in an interspersion of other cover-types within the former woodland cover types thereby giving mixed reflectance within satellite images.

4.2.3 Description and characterization of derived cover-types.

i. Grassland

This is land covered by grasses and other herbs but with grasses being more dominant. Sometimes this cover type is practically devoid of woody plants while at times widely scattered or grouped trees and shrubs with a canopy cover that does not exceed 10% occur. Within the study area, this cover type was assigned to those areas with over 40% herbaceous cover but less than 10% woody cover. This cover type is dominant within Amboseli National Park, Lolarash and Mbirikani group ranches. The most dominant vegetation species included *Pennisetum mezianum* and *Pennisetum straminium* outside the park while *Sporobolus* species covered over 75% within the park area. The swampy areas comprised of *Cyparacea* and *Juncacea* genera, and *Digitaria macroblephera* species. The diversity of species within this cover type was 3.6 but, it also had the highest number of bare ground cover types compared to other cover types. The height classes ranged from very short species (0.25 m) like *Sporobolus spicatus* to the tall species (1.2 m) such as *Sporobolus robustus*. Species distribution was rather even with more or less the same species occurring within every location except the S*porobolus* genera that seemed to be more localised on saline soils within Amboseli and Kimana swamps. There was an observed high correlation between grass species and soil types, for example salt tolerant species were found to be dominant over the others within the saline regions. This cover type likewise showed a remarkable diversity of growth forms and species composition in terms of perennial and annual species.

Although there is some evidence that pure grassland stands without an admixture of woody plants represent zonal climax vegetation (White, 1983), observations in this study area indicate that woody plants invasion is on the increase thus altering the grassland climax. However, in Eastern Africa, focus has been on woody/bush climax; therefore this observation on changing grassland climax gives an insight on the ongoing vegetation climax changes within pastoral areas of Eastern Africa (Fig. 5) which needs to be understood.

Plate 4: A *Sporobolus robustus* grassland cover types within the Amboseli National park.

ii) Swamps

This is an area characterized with seasonal or permanent water logging. These sites usually occur in depressions, which receive more runoff or water than that supplied by the incident rainfall within the study area. It has been shown that impeding and reversal of drainage due to warping and tilting of the earth's crust has greatest effects on the distribution of waterlogged grasslands (Saggerson, 1962). Within the study area, the influence of the parent material and changing climatic and geomorphology conditions have contributed greatly to the formation of waterlogged regions. The results include swamps that occur within recently deposited soils such as recent volcanic soils and flood plains as is evident in Amboseli, Kimana and the agricultural areas. This vegetation type varies greatly in the ease with which it can be replaced by secondary succession following human interference such as after cultivation or grazing on the edges. The dominant species in this cover type were *Cyperus papyrus* and *Cyperus latifolia*, a few cover types of *Digitaria scalarum* and *Cynodon dactylon* on the swamp edges.

iii) Bushed shrub grassland

This refers to an assemblage of woody plants mostly of bush-shrubby habit together with occasional emergence of grasslands and a woody canopy cover of not more than 40% scattered within a continuous sward of grass. In this study, bush species are defined as woody plants intermediate in habit between shrub and tree, are usually multi-stemmed and the main stem is often 10cm or more in diameter at the base. This cover type was the most common within the study area. The cover percentages of different classes of vegetation consisted of a balance between bush and shrub species, which was over 40% and less than 30% grass cover. Almost all locations had cover types of mixed vegetation and this also had the highest vegetation species diversity of about 3.8. This cover type consisted of *Acacia mellifera, Acacia nubica, Salvadora persica, Capparis tomentosa, Grewia bicolor, Grewia similis, Commiphora africana* and *Cadaba farinosa* as main bush species; the dwarf shrub species included *Sericocomopsis hildebrantii, Ocimum americana, Hibiscus ludwigi* and *Solanum nicotianum;* while grass species included

Pennisetun mezianum, Pennisetum straminium Cenchrus ciliaris and *Sporobolus spicatus.* Bush species varied in height from 2m to 7m while the dwarf shrubs were mostly the woody species below 1.2m in height. This cover type was observed to be replacing the former grassland cover types caused either by overgrazing then abandonment or complete non-utilization. Lack of water resulting from collapsed borehole projects is a major factor in vegetation cover type transitions within these ranches. Local informants noted that the insurgence of mixed vegetation into former grazing areas results in a reduction in overall grassland cover thus reduced available forage for their livestock. The dwarf shrub-grassland cover types seemed to occupy regions where the bush species had been excluded, possibly due to poor soils, low rainfall or extreme oligotrophy as is evident in some areas within Lolarash and Mbirikani group ranches.

Plate 5: *Acacia mellifera-Pennisetum mezianum* bushed grassland within Lolarash group ranch.

iv) Bush land

This cover type comprised over 40% bush cover but the bushes were denser and formed an impenetrable association. Within the study area, this cover-type was composed of the *Acacia* *mellifera and Acacia nubica* stands. The floristic composition was more or less similar to that of bushed grassland except that there occurred differences in plant density and undergrowth. This cover type was scattered all over the wildlife-livestock land-use systems especially in regions where livestock grazing was minimal or excluded. Informants reported that lack of grazing was closely associated with lack of watering facilities and or inaccessibility to the limited grazing areas. Within this cover type, herbaceous undergrowth is minimal due to dense cover by the woody species.

Plate 6: *Acacia mellifera* bush land within Mbirikani group ranch.

v) Wooded grassland

This was land consisting of an open stand of trees of about 8-20m in height and a ground cover of at least 30%. The crowns of adjacent trees were often in contact but not densely interlaced. The crown cast little shadow below and the under-storey was mostly composed of the herbaceous layer, which at times grew to a height of 2m. This cover type was common within the wet regions and mostly occurred as riverine vegetation or in areas of higher altitude and rainfall. The most dominant tree species included *Acacia tortilis, Acacia xanthophloea, Acacia etbaica, Croton megalocarpus* and artificially planted tree stands of *Eucalyptus* and *Cupressus* species. The distinct dominant sub-classes identified within the study included: *Acacia xanthophloea-Sporobolus robustus* and *Acacia tortilis-Pennisetum mezianum. Acacia xanthophloea* species had greater ecological amplitude ranging from saline to non-saline soil habitats and within cultivated fields to along the swamps; the *Acacia tortilis* class showed signs of degradation especially within the eastern part of Amboseli national park while outside the park, the under-storey was sparse due to overgrazing by livestock and at the same time over utilisation of the trees by people for firewood and settlement constructions.

Plate 7: *Acacia tortilis- Pennisetum mezianum* wooded grassland within Lolarash group ranch to the eastern part of the park.

vi) *Cultivated fields*

These were areas under crop agriculture that occurred especially in regions along the swamps and the mountain foot slopes. The original natural vegetation has been cleared for cultivation so that within the swamps, natural vegetation such as sedges and papyrus only remain along the watercourses. The expansion of cultivation within the area is in response to the soils, climate and topography. The main types of crops grown are subsistence crops such as maize, beans, bananas and vegetables while in the irrigated fields; people grew cash crops including flowers, bananas and vegetables.

4.3.0 Trends in land-use and land cover changes within the study area.

4.3.1 Temporal changes in landscape patterns 1988-1998

Landsat TM images for 1988 and 1998 were used to create land-use/land cover maps for the study area from which changes could be calculated. The selection of the ten-year period was a factor of image availability as well as the belief that a decade was a period long enough to exhibit changes in land cover. At the same time, this period represented an era during which there occurred changes in human population, land tenure policies, external influence, and socio-economic lifestyles within the study area (Western 1982; Bekure et al. 1991; Campbell 1999).

Table 14: Land cover types and changes within the study area between 1988 and 1998.

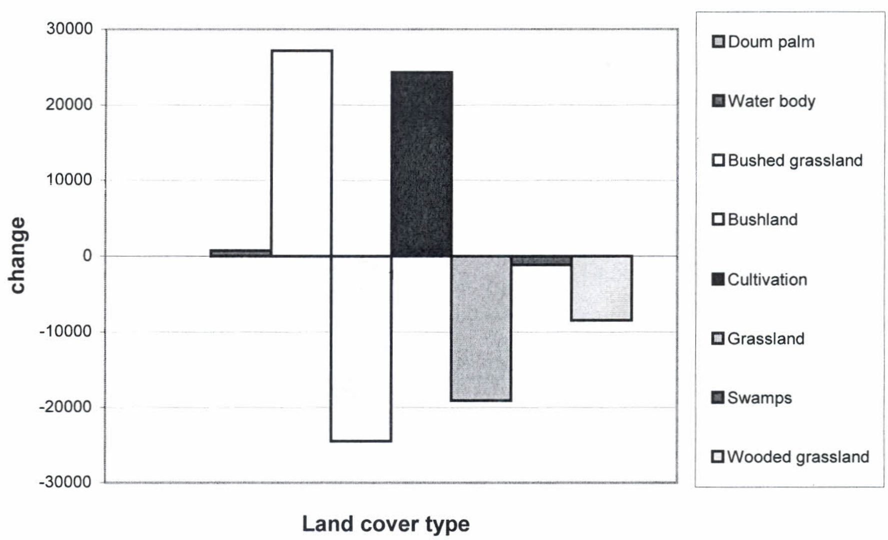
Cover type	Area (ha) 1988	Area (ha) 1998	Total change (ha)	Annual rate of change '000 (ha)	% change (Base year 1988)
Bushed grassland	140409.18	167571.52	27162.33*	2.71	119.3
Bush land	85791.13	61320.74	-24470.39*	-2.44	71.5
Cultivation	11469.04	35766.07	24297.03*	2.43	311.8
Doum palm	438.34	406.35	-31.99	-0.00	92.7
Grassland+bareground	55788.18	36172.42	-19615.75*	-1.96	64.5
Swamp	5696.51	4510.10	-1186.40*	-0.11	79.2
Water body	31.20	755.52	724.32	0.07	243.7
Wooded grassland	12670.54	4247.72	-8422.82*	-0.84	33.5
TOTAL	311279.65	311281.67			

*Significant at $p<0.05$

From Table 14, it can be seen that the total area of the two maps (1988 and 1998) has a difference of 2.02 ha which indicates that the two maps do not exactly overlay onto each other. This was attributed to cartographic errors during digitising or missing polygons in the resultant map. The percentage change in the table is calculated using 1988 as the base year with a 100% coverage for each cover type. This implies that values below 100% in 1998 reflect decrease in coverage while values above 100% reflect increase in coverage.

As depicted in Table 14 above, conspicuous changes have occurred in land cover and land-use between 1988 and 1998 within the study area. It also reveals that bushed grassland, cultivated fields and water bodies increased in coverage from 45%(140,409ha) to 54%(167,572ha), 3.7%(11,469ha) to 11.5%(35,766ha) and 0.01%(31.2ha) to 0.24%(756ha) out of the total land area respectively. However, vegetation cover generally decreased from 96.3% to 88.2% during the ten-year period.

Figure 8: Relative Land cover area changes in hectares between 1988-1998.



Areas covered by cultivated fields increased markedly around the swamps and along mountain slopes between 1988 and 1998 as a result of increased settlement and influx of agricultural communities into the area. The swamp areas reduced in size outside the park but increased in size inside the park. This can be explained by the fact that most of the swampy regions which were not yet under extensive agricultural use in 1988, had now been converted to crop cultivation by 1998.

4.3.2 Measure of changes in landscape pattern and structure.

Landscape structure analysis involved the study of landscape patterns, interactions among cover types and how those patterns and their interactions changed over the ten-year time period. Landscape pattern and structure indices were calculated using FRAGSTAT analysis package (McGarical and Marks, 1994). Specific indices were selected to measure cover types diversity, mean cover type size, dominance and complexity. Cover type diversity was quantified using Simpson's diversity index (Simpson, 1949), because of its superiority over other indices and the fact that it has been widely used in other similar landscape ecological studies (Kershaw and Looney, 1985).

Table 15: Mean size and number of cover types: an indication of habitat fragmentation

	1988		1998	
Cover type	Mean (ha)	No. of cover types	Mean (ha)	No. of cover types
Bushed grassland	3416.832*	41	1351.383*	124
Bush land	438.345	22	2114.509	29
Cultivation	2060.448*	3	3251.462*	11
Doum palm	3876.277*	1	406.352*	1
Grassland+ Bare ground	3820.23*	27	1310.844*	28
Swamp	1423.607	4	1503.37	3
Water body	31.205	1	251.842	3
Wooded grassland	218.184	58	202.2725	21

*Significant at $p<0.05$

The mean cover-type area (polygon size) did not change during the ten-year period, although some of the individual cover-types changed significantly as shown in Table 15 above. There was however, a statistically significant difference in mean cover type during the ten-year period with smaller-scale landscape pattern in 1998 compared to 1988 (Table 16).

Table 16: Changes in mean area of polygons (ha)

Statistic	***1988***	***1998***
Sum	312294.15	311724.05
Count	157	250
Mean	1989.13*	1246.89*
Standard deviation	5532.01	4657.68

*Significant at $p<0.05$

The overall mean cover type perimeter of identified cover-types did not change significantly over the period, although there was a trend toward decreased cover type perimeter in 1998 as shown below (Table 17). However, the mean cover type perimeter was significantly smaller in 1998 hereby again reflecting the small-scale landscape pattern in 1998 compared to 1988.

Table 17: Mean perimeter (m) of polygons and changes between the years

Statistic	***1988***	***1998***
Sum	36768.04	52795.09
Count	157	250
Mean	234.19**	211.18**
Standard deviation	463.78	573.88

** Significant at $P<0.01$

The actual trends have already been depicted in the tables 14 and table 15, except that it is interesting to note that the number of cover types of bushed grassland increased while the average size of those cover types decreased by approximately 55% (Table 15). The doubling in the number of cover types of continuously increasing cultivated fields within former swamps, bush land and grassland areas further reflect this trend. Therefore, changes in land-use within the study area are reflected by a significant increase in the number of cultivation cover types from 3 to 11 over the period 1988 to 1998 (Table 15). Consequently, changes in land cover have inevitably resulted into more changes in landscape structure within cultivated areas than wildlife-livestock production areas.

Dominance indices did not show any significant changes though the values were lower in 1998 indicating evenness in size among polygons of different cover type categories unlike in 1988 as shown in table 18.

Table 18: Index of dominance in cover-types between 1988 and 1998.

Cover type	Dominance index (***D***)	
	1988	***1998***
Doum palm	0.74	0.70
Bushed grassland	0.73	1.03
Bush land	1.09	1.01
Cultivation	1.08	0.94
Grassland	0.85	0.92
Swamp	1.04	0.75
Water body	0.81	0.71
Wooded grassland	0.86	0.75
Total	2.09**	2.05**

Fractal dimension values were relatively low in 1998, which implied that cover-types were less complex in shape in 1998 than in 1988 (table 19).

Table 19: Changes in complexity of landscapes within Amboseli Area.

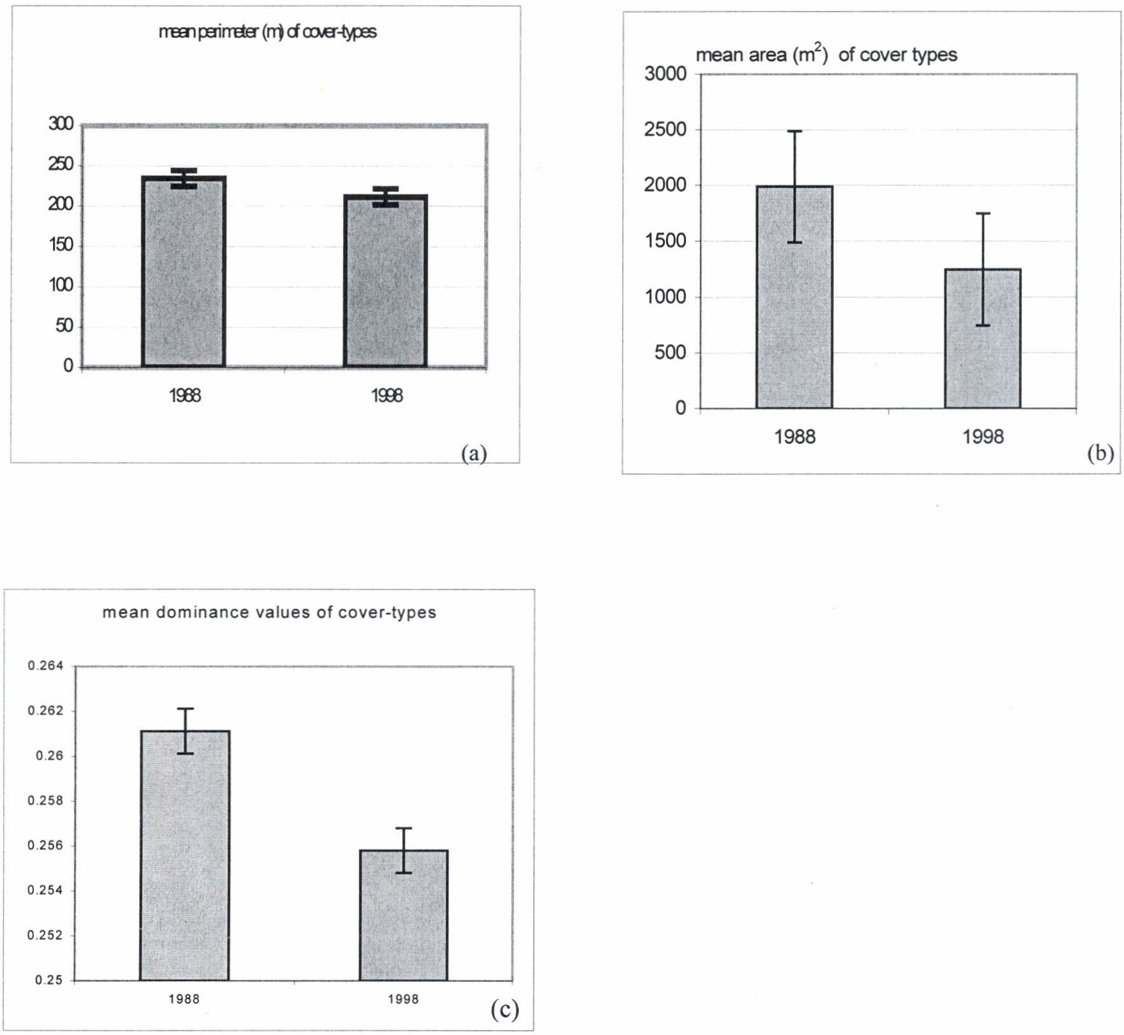
1988 Fractal indices

COVER TYPE	AREA	Log A	PERIMETER	Log P	FRACTAL INDEX
Doum palm	4383451.2	6.6	9781.2	3.9	1.6
Water	312049.9	5.4	3161.3	3.4	1.5
Bushed shrub grassland	1400901216.7	9.1	771368.5	5.8	1.5
Scrubland	852780961.5	8.9	549658.1	5.7	1.5
Sparse agriculture	114606890.2	8.0	74935.2	4.8	1.6
Grassland	556320980.0	8.7	478797.1	5.6	1.5
Swamp	56944283.8	7.7	149132.5	5.1	1.4
Wooded grassland	126546696.6	8.1	291550.4	5.4	1.4

1998 Fractal indices

COVER TYPE	AREA	Log A	PERIMETER	Log P	Fractal index
Doum palm	4063524.9	6.6	10881.2	4.0	1.6
Bare ground	66861087.7	7.8	100323.5	5.0	1.5
Bushed grassland	1675715208.7	9.2	1384399.9	6.1	1.5
Bush land	613207473.5	8.7	922590.5	5.9	1.4
Cultivation	357660771.6	8.5	389803.2	5.5	1.5
Grassland	300175085.4	8.4	504412.7	5.7	1.4
Swamp	45101092.4	7.6	134160.0	5.1	1.4
Water body	7555262.7	6.8	20327.0	4.3	1.5
Wooded grassland	42477228.0	7.6	208322.2	5.3	1.4

Figure 9: Changes in landscape structure parameters



Dimensions of change in landscape patterns. These figures give means for the ten-year period. The bars indicate standard errors. (a) Mean habitat area (b) mean habitat perimeter (c) Mean habitat dominance

Table 20: **Transitions in land cover types between 1988-1998**

1998 land cover types

1988 land cover types

	Bushed grassland	Bushland	Cultivation	Doum palm	Grassland	Swamp	Wooded grassland
Bushed grassland		10.90%	25.00%	.0.00%	15.00%	0.10%	20.00%
Bushland	14.10%		44.00%	.0.00%	4.00%	0.00%	9.00%
Cultivation	0.20%	.0.00%		.0.00%	.3.50%	.12.30%	20.00%
Doum palm	.0.00%	.0.00%			2.00%	.0.00%	.0.00%
Grassland	6.40%	12.00%	10.0%	0.10%		7.00%	3.00%
Swamp	0.40%	.0.00%	60.00%	0.00%	5.00%		0.00%
Wooded grassland	3.00%	2.00%	40.00%	.0.00%	2.00%	0.20%	

The information on this table is derived from table 14 detailing changes in land cover types. For example, bushed grassland increased by 19.34% and 14.10%, 6.4% and 3% of this change through transitions came from Bushland, Grassland and Wooded grassland respectively. This process indicates that transition doesn't necessarily mean loss but can be either a loss or a gain in cover type coverage i.e it is a give and take scenario upon which the degree of each dictates the direction of the process.

4.3.3 Changes in land-cover types within the study area

These changes clearly indicate a progressive expansion of agriculture to the limited favourable areas, formerly available for livestock grazing. This can be attributed to the increasing human population, changing land tenure, land use and changing pastoralist lifestyles within the study area during the past decade. Other similar studies have also indicated that most pastoralists in sub-Saharan Africa are losing their prime grazing lands particularly within the low lying drainage areas so as to make room for small-scale rain fed, flood and riverine irrigated agriculture (Keya 1991; Hadley 1993; Wint and Bourn, 1994). The greater Amboseli ecosystem as a study area was therefore a case in point where dry season grazing lands within the flood plains and swampy regions are being cleared for crop agriculture.

Natural vegetation, which covered 96% of the total area in 1988, subsequently decreased to 88% in 1998. This decrease could be attributed mainly to, clearance of natural vegetation especially wooded grassland, woodland, bushed grassland and bush land including swamps for crop agriculture and settlement. Bushed grassland and cultivated fields have increased in 1998, while the bush lands, grasslands and wooded grasslands decreased compared to 1988. An increase in woody vegetation species at the expense of herbaceous species has been commonly attributed to overgrazing in other areas (Pratt and Gwynne, 1977; Le Houerou 1980; Skarpe 1991; Coppock 1994). For example, Coppock (1994) reported that ecological sustainability in Borana, Ethiopia was being threatened by increased cultivation, settlement and encroachment by woody vegetation as well as by soil erosion, which could be attributed to heavy grazing by livestock. The results of these related studies reveal that, replacement of herbaceous vegetation species by woody species often occur as a result of heavy ungulate utilization over some time period or may be due to major climatic changes. It was difficult however from the present study to ascertain the climatic causes of land cover changes due to lack of long-term climatic and vegetation data. From this study it was not possible to detect the effect of other factors such as fire on vegetation cover changes due to lack of consistent data. In general, results from image interpretation provided information on patterns of land-use and land cover changes and an overall view of vegetation status within the study area. This is in agreement with Hientz et al, 1979; Owens et al. 1985; Turner et al. 1998 who reported that in semi-arid rangelands where sampling requires a substantial amount of financial and logistical resources, satellite images of the same area taken at different times usually contain valuable information on natural resource bases and dynamics.

4.3.4 Main trends in landscape changes and their causes.

Informants within the study area pointed out that the ongoing changes in land tenure have had tremendous impacts on land-use and resource utilization. Therefore, the emergence of landscapes in the study area reflects both land-use changes and environmental influences during the ten-year period. Generally, livestock grazing that was once widespread in the area has presently declined in most parts of the ecosystem especially within those areas suitable for crop agriculture. However, for commercial wildlife and livestock land use systems, this decline has been slightly less pronounced but the number of cattle per household continues to decline as grazing land areas reduce over time. This has been accompanied by a remarkable increase in the number of small stock (goats and sheep) within the study area as pointed out by the interviewees. Reduced access to forage or grazing areas has led to introduction of fewer improved milk cows especially in those areas undertaking crop agriculture. Within wildlife-livestock production systems, it can be speculated that, the breakdown in watering facilities and reduced access to some grazing areas has had a two-fold effect. The abandoned areas due to lack of watering facilities has led to invasion by woody species into the former grazing areas a result of reduced intensity of use; likewise, reduced pastoral mobility has resulted in localised overgrazing thus vegetation destruction and woody species invasion within particular sites. These in part may explain the widespread expansion of bushed-grassland from 1988 to 1998 as revealed in the cover type transition maps.

In general, a suggestive explanation for the observed landscape changes might be as follows: Subdivision of group ranches has led to expansion of crop agriculture at the expense of grazing lands; sale of land and subsequent sedenterization has further fragmented the landscape and; erection of fences which blocks livestock movement into suitable grazing areas and wildlife migratory corridors, has led to concentration of wildlife within parks, and overgrazing within the remaining communal grazing lands. These processes have led to modification of vegetation as well as transitions in land cover-types within the park and the surrounding communal ranches. The initiation of tourism activities within the surrounding group ranches have hade adverse modification effects on the landscape. For example, sedenterization of pastoralists and their livestock within the established cultural *bomas* has led to overgrazing while the creation of tourist concession areas, which excludes livestock utilisation, has further increased grazing pressure on the remaining areas as transformation of one cover type to another occurs due to either lack of use or over use.

It can be observed that, any changes occurring within a given tenure regime affects all the remaining ones thereby influencing landscape development either directly or indirectly. For example, land tenure change from communal to individual land holdings have encouraged sale of land, initiation of crop agriculture together with fencing off of crop farms to protect them from wildlife destruction which results into increased people-wildlife conflicts. This indirectly has led to vegetation destruction within the still intact state property (the Amboseli Park) due to wildlife concentration within the protected area. Similarly when cultivation cuts off pastoralists from dry season grazing areas, overgrazing occurs within the enforced grazing boundaries, as is the case for Lolorash and Mbirikani group ranches.

The interplay between land-use and environmental factors that relates to landscape pattern must also not be overlooked. For example, expansion in cultivated fields can be seen to be responsive to soil types, rainfall regimes and topography. Therefore, most cultivation takes place along the mountain slopes, recently deposited soil types and around swamps in the lowland areas, which exhibit fragility with less capacity of resilience upon degradation. Similarly, the increase in overall swamp area can be attributed to the increased drainage in the lowlands for example within the park; while an increase in bushed grassland corresponds to areas with low rainfall and fewer watering points for cattle (Western, 1975). The *Sporobolus* grasslands and *Sueda monoica* bush lands often occur in saline soils indicating the influence of soil type on vegetation distribution as well as respective cover-types.

4.3.5 Ecological relevance of landscape pattern and structure indices.

Great deals of spatial area changes have been recorded for the cover-types of bushed grassland, wooded grassland, bush land and cultivation. However, these changes are equalled by changes in other cover-types through habitat transformations, for example, decrease in swamps and bush land were balanced by increase in cultivation fields. As land-use increased in intensity and extent, complexity and diversity decreased, while heterogeneity increased. In Table 15, landscape structure is more heterogeneous due to increased number of cover types in 1998 compared to 1988. Fragmentation of cover-types by human activities into smaller cover types has contributed to increased heterogeneity but reduced complexity. All these can be attributed to changing land-use and pastoralists lifestyles. The expansion of cultivated fields, fencing of farms and their coalescence have resulted in formation of geometrically simple polygons like squares and quadrates with smaller fractal dimension and therefore reduced complexity (Turner, 1989). However, it should be noted that these indices were insensitive to underlying changes in ecological content of the contributing cover types; except, by comparing the different index values for the study area during the ten-year period, the differences in scale and integrity over time is nonetheless well illustrated.

The broad consequences of the present landscape development within the study area was revealed through initiation of succession processes and subsequent changes in habitat distribution, habitat areas, and transitions from one habitat to another and therefore, implied changes in their ecological contents of the habitats. The expansion of cultivation into natural vegetation habitats has led to fragmentation of those specific habitats both by a decrease in cover types sizes and by splitting of the existing cover types. This change in land-use today implies a break in a long-term cultural tradition of pastoralists and a break in the disturbance regimes that has contributed to the present landscapes. This may be termed as *a break in the continuity of a long-term pastoral land-use system.*

The widespread maintenance of livestock production in the still intact group ranches has led to the development of specific plant communities of high species diversity, which contain a mixture of rangeland species such as *Acacia* and *Commiphora* genera as is the case in Lolorash and Mbirikani group ranches. Past research had indicated that such dominant species were formerly widespread in the whole of the earlier uncultivated fields all over the study area (Western, 1973; Bekure et. al., 1991) but are nowadays reduced in coverage due to changes in land-use and emergence of agricultural practices and other land uses that lead to habitat destruction, transformation and fragmentation. However, remnants of these vegetation types can still be found in areas under cattle and wildlife grazing especially in the much drier parts. The observed processes such as ongoing colonisation by bushed grassland will in the near future transform the present grazing areas into bush land. Most of the natural vegetation cover types will therefore disintegrate and some may subsequently disappear- a process that might include extinction of a variety of biological diversity (Olsson et al. 2000).

In conclusion, populations of endangered plant and animal species may become vulnerable due to the ongoing fragmentation of rangeland vegetation, which include reduced cover type sizes and increased distances to similar sites due to isolation (van Dorp et al., 1997; Ellis et al., 1999). This pattern is particularly evident in the agricultural areas within the study area.

4.3.6 Changes in landscape pattern and structure.

Pastoralism as noted by Swift et al. 1996 is neither a high technology land use system, nor is it an energy subsidized form of resource use. Direct and complete replacement of native plant communities with other domesticated plants as occurs in agricultural systems is therefore not quite common. Nevertheless, the changing pastoral land-use systems within the study area have had important effects on the structure and dynamics of the rangeland ecosystems they inhabit by the pastoralists. During fieldwork, it was evident that few exotic/domesticated vegetation species had been introduced into the study area but the cover and structure of the natural vegetation has been altered over time due to changing land-uses systems. Even though the structure of vegetation communities within rangelands is almost universally modified by pastoral activity, neither is the degree nor direction of change the same in all of the pastoral systems. This scenario can be likened to the different driving forces of economic activities and land-use practices undertaken by pastoralists within rangelands.

It can be concluded that; the changes in number, diversity and density of land cover types could have resulted from changes in land-use activities within the study area. Land clearing for cultivation, integration of livestock and wildlife production systems and breakdown of livestock watering facilities have all contributed to the fragmentation of large areas of the greater Amboseli ecosystem. The swamps have reduced, grassland cover types are diminishing and bushed grassland are expanding with signs of overgrazing in several places. Cultivated fields have increased at the expense of grazing fields, swamps and bush lands. The rapidly growing population within the study area has also led to the expansion of agricultural activities along the slopes of the mountain and widespread sparse cultivation on the suitable regions on the lowland grazing areas. With continuous clearing of more land for agriculture by crop farmers, smaller fields coalesce to form larger homogeneous cover types of cultivated fields. Over time, the variability in cover types within crop production areas decrease thereby indicating that the landscape is becoming more and more homogenous as human land-use intensifies. The increase in edge effects in 1998 have reduced shape complexity to create more regular cultivated fields thereby replacing the less regular shapes seen in the 1988 land cover map. Land cover types were more mixed and interspersed in 1988 than in 1998. As land-use intensifies, interspersion decreases, while cultivation replaces small cover types of natural vegetation and grass fields continue to diminish.

Heavy utilisation of herbaceous vegetation is leading to replacement by woody vegetation species, which reduces fire risks and available forage coverage. Within the study area, young woody shrubs dominate those places currently enduring heavy grazing pressure while those that have undergone heavy utilisation in the past have been invaded and are dominated by mature woody shrub and bush species. In conclusion, the past methodology of resource use strategy within pastoral areas, which involved cover types dynamic utilisation through mobility is now being replaced by a sedentary resource use strategy that is responsible for continuous natural resource destruction.

4.4.0 Land tenure and land-use systems

4.4.1 Land tenure systems

Interviews with pastoralists revealed that recent land losses in the study area follow the long history of land tenure changes. For resident pastoralists, these primarily resulted from establishment of protected areas for wildlife conservation (Amboseli National Park) and alienation of land for crop agriculture along the slopes of Mt. Kilimanjaro and swampy areas of Noolturesh, Kimana and Namelok. In the past, Maasai people traditionally moved their livestock from place to place, depending on the availability of pasture, water and incidences of diseases. The Kenya government introduced group ranches in Kajiado district in 1960's aimed at addressing the problem of overgrazing and land degradation then said to be occurring in pastoral areas. This change was expected to encourage the Maasai people to limit their livestock numbers to match the group ranch grazing resources. Since its implementation in 1960's, the group ranch system had been the main operating land tenure arrangement within the study area. The policy on which the group ranch programme was based was the ideological assumption that resources, when held in common, led to overgrazing, resource degradation and a low level of investment by pastoralists. Development and policy formulation experts called this "tragedy of the commons". In real sense, they confused collectively owned resources such as communal grazing areas with open access in which resources are unrestricted with equal privilege of use to all but with no obligations. Unlike this belief, Maasai grazing systems were traditionally far from open access but resources were collectively managed under a set of clear rules and regulations. The spread of "open access" view led to and continues to tragically contribute to the massive condemnation of pastoralism as a production system within Kenyan rangelands including the study area.

Generally, the group ranch arrangement within the study area as reported by informants continues to fail because of the inherent difficulties within the group ranch concept, related to frequent droughts and the disparity between government's objectives and the Maasai's traditional reasons for keeping livestock. From the late 1970's many Maasai people began calling for group ranch subdivision due to inefficiency of the management committees, threat to their land by increasing immigrants, and desire for title deeds to secure individual loans. With the changing government policies on land tenure, some of the group ranches within the area have been undergoing rapid subdivision into individual land holdings as is the case in Entonet, Enkariak and Emperon areas. The division of land into private holdings has resulted in small parcels of land being put to a myriad of uses other than livestock production. The results have been; sale of land to non-Maasai members allowing non-Maasai immigrants and foreign landholders, introduction of rain-fed agriculture, fencing of farms and clamour for individual control of wildlife tourism activities. The present land tenure systems within the study area presently include individual, company, communal and trust tenure systems. Subdivision has also cut off many pastoralists from access to critical dry season grazing zones, such as, mountain slopes and swamps.

A number of people who have realised that their resulting land sizes are unable to support viable livestock production, have formed companies or associations for collective resource use, for example, within Mbirikani and Lolarash ranches. In this arrangement the individuals have their respective titles but they form resource use groups with their neighbours in which the group members use the available resources without restriction.

Table 20: Changes in numbers and area of group ranches in Kajiado District, which had been sub-divided between 1984 and July 1996

Year	1984	1990	1996
No. of subdivided group ranches	7	12	22
% of original total group ranch area	5.1	10.2	20.3
Approximate area (ha)	77,453	154,845	315,956

Source: Pasha (1986); Rutten (1992); Kajiado District Land Registry data

4.4.2 Land-use systems within Greater Amboseli Ecosystem.

The following land-use types were derived from informal oral interviews and observed ongoing activities supported by documented evidence in available literature. Amboseli rangeland had for long been used solely for subsistence livestock production. Therefore, to understand the effects of current tenure and use changes, it was important to examine the history of landscape within which the observed changes were taking place. Due to rapid population growth, changing economic situations, external influence and changing climatic conditions, the present use of rangelands within the study area is dynamic and represents totally different economic and social lifestyles compared to the past traditional pastoral settings. Livestock herding amongst the Maasai pastoralists is still widespread and extends over much of the rangeland, however, small-scale irrigation and rain fed agriculture; market-oriented production and wildlife-based eco-tourism activities are replacing traditional livestock production systems. These changes in land-use can likewise be attributed to the low returns to livestock production due to low prices as indicated by some interviewees. Within the study area, three distinct land-use types were identified: (1) wildlife conservation (2) livestock production (3) intensified rain-fed agriculture on the mountain slopes, and on the edges of swamps and rivers

4.4.3 An overview of the operating land-use types

i. Wildlife conservation and tourism

National policies have increasingly turned towards controlled non-consumptive commercial wildlife use so as to justify the allocation of land to wildlife within the study area. This land-use is characterized by conservation and full utilisation of wildlife through eco-tourism and is mainly found within the restricted park areas (Amboseli National Park). In this case there is restricted access into these protected areas by outside communities so as to maximise profits and to conserve the available biodiversity. The management of the park area differs from the surrounding ranches in the sense that the park perimeter is guarded to preclude influx of livestock and people. Hunting and killing of game is prohibited with a view of sustaining and improving the existing diversity in order to boost the tourism industry. The main wildlife species within the park are: Elephant (*Loxodonta africana*), Buffalo (*Syncerus caffer*), Giraffe (*Giraffa camelopardalis*), Zebra (*Equus quagga*), Hippo (*Hippopotamus amphibious*), Thomson's gazelle (*Gazelle thomsonii*) and Grant's gazelle (*Gazelle granti*). Expansion of lodges within the park affects vegetation negatively as well as overcrowding of tourist vehicles within the park since off-road driving by tourist vehicles destroys the vegetation within the park. Informants reported that this kind of land-use is not of direct benefit to the local residents since the profits and employment from tourism sector rarely reach the local population. Due to conflicts with the people living around the park, most wildlife is nowadays confined within the protected areas, especially the large ungulates. This confinement has been claimed by others (Western and Gichohi, 1993) as being responsible for over-utilisation and destruction of the vegetation within the park turning what used to be *Acacia* woodland into an open bushed-grassland cover type (Plate 8). At the same time, tourism has led to sedenterization of local communities within cultural bomas around the park perimeter as they seek jobs from tourist hotels and entertainment tourists. This sedentarization impacts negatively onto the vegetation since livestock get concentrated around settlements due to reduced mobility ranges thereby destroying the natural vegetation.

Plate 8: Degraded former Acacia woodland within the Amboseli National park

ii. Mixed Commercial use of wildlife and livestock

Pastoral ranches surrounding the Amboseli Park serve both as seasonal wildlife dispersal areas as well as a pastoral livestock production rangeland. The existence of wildlife within these ranches present opportunities both for economic benefit and a potential for serious conflicts between pastoral livestock and wildlife because of competition for grazing resources. Given the opportunities and constraints presented by both the economic, social and climatic conditions, the people within the study area have responded by diversifying their activities to include both non-consumptive exploitation of wildlife as well as livestock keeping. This land-use type is characterised by an integrated use of both wildlife and livestock. Informants maintained that tourism enterprises (wildlife viewing and safari camping) on these public ranches is more financially and economically profitable than the standing alone livestock production. Rental fees from eco-tourism are realised through provision of tented campsites on group ranches while the presence of tourism offer a market outlet for locally made jewellery by the locals.

This kind of land utilisation exists within the Lolarash, Kimana and Mbirikani group ranches and in other areas where local communities have been sensitised and appreciates the importance of wildlife conservation. Such land use systems are found in Kimana and Eselengei sanctuaries. Plans to develop stronger community wildlife management groups are underway to eliminate open-access problem and to enable better common property management to better integrate both wildlife and livestock for conservation and a better economy. This will enable the communities to optimally capture the economic benefits of commercial wildlife use found within their ranches. A community appointed committee, whose mandate is to oversee the management of campsites and collection of rental fees, manages the proceeds realised from these instituted wildlife enterprises. The collected fees are used towards overall community development. This is quite well co-ordinated within Kimana and Lolarash group ranches. Ongoing projects observed within the study area included drilling of bore-holes to provide drinking water for both livestock and people, establishment of a school bursary fund to cater for school-going children, establishment of local schools and small enterprises such as shopping centres. Interview with KWS and the local community members revealed that these initiated projects have been moderately economically successful.

iii. *Livestock and Crop agriculture*

This land-use pattern is as a result of external influence, which has led to sedenterization of pastoralists. Most elders interviewed described this land-use pattern as the greatest transformation of pastoralism, both as a production system and a way of life. This is true because it has led to the displacement of native Maasai pastoralists, loss of critical dry season grazing lands, erosion of socio-economic and ecological adaptations necessary for pastoralist survival within the study area. The people are involved in both rain-fed and irrigation agriculture including horticulture along the rivers, swamps and mountain slopes. This occurs within the areas bordering the Noolturesh swamp and extends towards the foot slopes of Mt. Kilimanjaro where the climate favours rain-fed crop agriculture. The main crops grown include maize, beans, bananas, vegetables and flowers. Cultivation by non-Maasai farmers is expanding due to substantial immigration into the locations where irrigation agriculture is possible and to the wetter slopes of Mt. Kilimanjaro. Sale of food crops is a major source of cash income. Those ranches with crop and livestock land use pattern include Endonet, Emperon, Enkariak and smallholder farms in Kimana/Tikondo group ranch. The existence of small town centres such as Kimana and Loitokitok provide market outlets for the farm produce as well as livestock products such as milk and meat from these regions. These agricultural regions receive slightly higher moisture either from run-on due to geomorphology causing swampy areas or due to the high elevations on slopes along mountainous and hilly regions. As such the distribution of this land-use type within the study area shows a strong correlation with climate, soils and topography. The soils mostly associated with crop agriculture are the recently deposited alluvial and volcanic soils, well drained and rich in minerals. The community here are dependent on crop production for subsistence and are more market-oriented economy than members of the other group ranches within the study area. Informants report that this agricultural activity has encouraged the ongoing subdivision of the group ranches, which has further led to immigration of agricultural communities including the Chaga, Kikuyu and Kamba. Even though there is great emphasis on farming, the local pastoral Maasai community are still involved in some livestock keeping as part of subsistence for their livelihood indicating a strong attachment and belief in their tradition of livestock keeping as a means of survival within the harsh rangelands.

5.1.0 Overall conservation aspects on landscape changes within the study area.

As the Amboseli landscape get fragmented and vegetation within the park is destroyed, the remaining natural vegetation within the surrounding areas becomes the only refuge for wildlife species outside the park and only source of forage for pastoral livestock. It is thus of critical importance that the current development trends such as agricultural expansion, erection of fences and settlements be managed. To ensure the sustenance of species and communities within the study area, it is important to maintain the ecological processes that created them. This means continued grazing by ungulates and other measures to curb disintegration. The present grazing pressure should be regulated to attain conservation goals and arrest the continuing destruction of natural vegetation habitats within the study area. Considering the fact that this landscape has been used for pastoral production for a long period and with the context of dynamic changes in its natural ecosystem, the past prevailing sustainability of the production system should therefore be enhanced so as to preserve the long-term landscape climax.

5.2.0 Role of Land-use and tenure changes

The study findings were based on: interpretation of Landsat Thematic Mapper images taken at two points in time (1988 and 1998); a field study and analysis of vegetation datasets and; an informal group interview of the local people to identify land tenure systems and land-use changes. This approach has less bias than others compared to pure ecological or socio-economic studies where there occur very few options for data integration. Therefore, the combination of the three approaches used here furnishes us with adequate information on the trends of natural vegetation cover, land-use change and the overall environmental status.

One of the major conclusions from this study is that there has been significant land-use change over the past decade in the greater Amboseli ecosystem. Large portions of the land have been converted to small-scale agriculture and natural vegetation has been destroyed as a result of overgrazing and human use. The Maasai pastoralists are now expanding their small-scale agriculture into the swamps for their livelihood. However, this may not be sustainable because the swamps will continue to diminish and dry up with continued cultivation. In general, the ecological sustainability in Greater Amboseli ecosystem is being threatened by increased cultivation, settlement and encroachment by woody vegetation as well as by soil erosion most likely attributed to heavy grazing by livestock and human land use changes. This trend in ecological changes will certainly have negative impacts on the existing biodiversity, which may in turn, negatively affect pastoral grazing strategies and overall national income from tourism activities. The evidence from the study area therefore points to the non-sustainability of agro-pastoral land use system in the face of increasing demand for more food, changing land tenure and land uses.

The above factors have put pressure on the study area resulting in range resource use conflicts and vegetation resource destruction. Finding lasting solutions to these challenges require systematic studies and in-depth analysis that will assess the ways on how to best manage pastoral resources. Future plans for sustainable rangeland resource utilisation within the study area is absolutely necessary. Poor resource management and over-use is evident in all the different land tenure regimes within the study area. Overgrazing and vegetation resource degradation is noticed on state owned land (the Amboseli National park), as well as on private and communally owned land. It is therefore necessary to identify the shortcomings in each type of land ownership regimes in order to evolve appropriate natural resource use and management policies.

Traditionally, protected area management stops at the park or reserve boundary yet movement of wildlife do not. The importance of the role played by the surrounding landowners in conservation of biodiversity can thus not be overlooked. Therefore, when introducing policy changes and development projects, the role of indigenous pastoralists should be taken into account. There is a crucial need for integrated landscape management through involvement of pastoralists into the ongoing conservation and economic development schemes within the Greater Amboseli Ecosystem.

In view of the hostile climatic environment within rangelands influences ensuing from external pressures within the study area, pastoralism in its own traditional form will merely persist on the basis of its own intrinsic ability to respond to changes in its internal or external structure and dynamics. For this reason, it is necessary to identify appropriate policy and technological interventions for sustainable development within the study area.

5.3.0 Recommended areas for future research:

a) There exist very minimal data on how the emerging land tenure regimes affect pastoral resource use strategies with respect to conservation practices, livestock and agricultural productivity, economic efficiency, and equity within the study area and other rangelands as well.
b) Major priority is to understand the effects of external factors on fragmented landscapes and to investigate changes in internal ecological processes as fragmentation occurs over time.
c) Investigation of the isolation effects on rates of genetic exchange and understanding the role of corridors in allowing biotic movement between fragmented landscapes.
d) Investigation of the impacts of landscape fragmentation on pastoral livestock production and income, and formulation of ways of integrating pastoralists' into overall conservation and development issues.

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Appendices

Frequency and diversity of all plant species identified within the study area.

SPECIES	FREQUENCY	DIVERSITY
Abutilon grandiflorum	2	0.0127
Abutilon hirtum	16	0.0677
Abutilon longiscupe	1	0.0071
Acacia brevispica	1	0.0071
Acacia drepanolobium	7	0.0356
Acacia etbaica	2	0.0127
Acacia hockii	1	0.0071
Acacia mellifera	43	0.1380
Acacia nilotica	17	0.0708
Acacia nubica	11	0.0508
Acacia tortilis	37	0.1245
Acacia xanthophloea	22	0.0858
Achyranthes aspera	11	0.0508
Aerva persica	6	0.0314
Aeschynomene schimperi	1	0.0071
Amaranthus hybridus	3	0.0127
Aristida adoensis	1	0.0071
Aristida kiniensis	15	0.0644
Aspilia mossambicensis	1	0.0071
Atriplex muelleri	1	0.0071
Aystasia chairmani	1	0.0071
Balanites aegyptiaca	40	0.1314

Barleria eremonthoides	15	0.0644
Bidens pilosa	2	0.0127
Boscia coriacea	1	0.0071
Cadaba farinosa	11	0.0508
Capparis tomentosa	19	0.0770
Cassia didymobotrya	1	0.0071
Cassia occidentalis	3	0.0179
Cenchrus ciliaris	45	0.1424
Chloris pichnothrix	5	0.0271
Chloris roxburgiana	7	0.0356
Commicarpus helenae	2	0.0127
Commiphora africana	16	0.0677
Cordia ovalis	2	0.0127
Crotolaria deflersii	1	0.0071
Crotolaria mauensis	1	0.0071
Cyathula polycephala	4	0.0226
Cynodon dactylon	9	0.0434
Cynodon plectostachyus	1	0.0071
Cynodon plectostachyus	14	0.1200
Cyperus papyrus	1	0.0071
Dactyloctenium aegyptium	2	0.0127
Dasysphaera prostrata	6	0.0314
Datura stramonium	5	0.0271
Digitaria blepharis	1	0.0071
Digitaria scalarum	11	0.0508
Drake brockmania	4	0.0226
Duosperma sp.	18	0.0739

Species list cont'

Dyschrosite hildebrantii	1	0.0071
Eragrostis cilianensis	3	0.0127
Eragrostis tuneifolia	1	0.0071
Glinus lotoides	1	0.0071
Grewia bicolor	7	0.0356
Grewia similis	14	0.0611
Grewia villosa	7	0.0356
Heliotropium steudneri	11	0.0508
Hermania alhensis	2	0.0127
Hibiscus ludwigi	4	0.0226
Hibiscus micranthus	13	0.0577
Hibiscus trionum	3	0.0179
Hypoestes aristata	6	0.0314
Hypoestes hildebrantii	1	0.0071
Indigofera spinosa	7	0.0356
Indigofera volkensii	1	0.0071
Phoenix reclinata	4	0.0226
Justacia hildebrantii	1	0.0071
Kalanchoe schweinfurthii	1	0.0071
Lantana camara	2	0.0127
Leonotis africana	7	0.0356
Leonotis nepetifolia	2	0.0127
Leucas deflexa	3	0.0179
Leucas pratensis	17	0.0708
Leucas pruensis	2	0.0127
Linarrifolia persica	2	0.0127
Lintonia nutans	1	0.0071

Species list cont’

Maytenus arbutifolia	1	0.0071	*Species list cont'*
Monechma debile	2	0.0127	
Nicotiana tabacum	1	0.0071	
Ocimum americana	19	0.0770	
Panicum coloratum	1	0.0071	
Pavonia zeylanica	2	0.0127	
Pennisetum masaicum	1	0.0071	
Pennisetum mezianum	57	0.1664	
Pennisetum stramonium	24	0.0915	
Pupalia lappacea	3	0.0179	
Rhus natalensis	1	0.0071	
Salvadora persica	20	0.0800	
Sansevieria intermedia	3	0.0179	
Sansevieria suffruticosa	2	0.0127	
Typha angustifolia	1	0.0071	
Sericocomopsis hildebrantii	33	0.1149	
Sericocomopsis pallida	13	0.0577	
Sida acuta	4	0.0226	
Sida ovata	12	0.0543	
Solanum coagulans	2	0.0127	
Solanum incanum	35	0.1198	
Sphaeranthus napierae	1	0.0071	
Sporobolus fimbriatus	18	0.0739	
Sporobolus pyramidilis	12	0.0543	
Sporobolus robustus	10	0.0471	
Sporobolus spicatus	23	0.0887	
Sporobolus tuneifolia	6	0.0314	

Species list cont'

Sueda monoica	12	0.0543
Tagetes minuta	6	0.0314
Tephrosia holstii	1	0.0071
Tetrapogon spatheceous	1	0.0071
Thunbergia holstii	1	0.0071
Tragus grass sp.	1	0.0071
Tribulus terresteris	1	0.0071
Triumfetta brachyceras	4	0.0226
Triumfetta flavascens	14	0.0611
Vernonia galamensis	3	0.0179
		4.2702