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Application of remote sensing and geographic information systems in wildlife mapping and modelling

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ABSTRACT

Wildlife management requires reliable and consistent information on the abundance, distribution of species and their habitats as well as threats. This article reviews the application of remote sensing and GIS techniques in wildlife distribution and habitat mapping and modelling.

7.1 INTRODUCTION

The main purpose of wildlife conservation is to maintain maximum plant and animal diversity through genetic traits, ecological functions and bio-geo-chemical cycles, as well as maintaining aesthetic values (IUCN 1996). This has been achieved to a certain extent through the creation of parks and reserves in different parts of the world. These areas are set aside and managed to protect individual plant and animal species, or more commonly of assemblages of species, of habitats and groups of habitats. Different criteria are used in the establishment of parks and nature reserves. Ideally they should comprise communities of plants and animals that are in balance, and exhibit maximum diversity (Jewell 1989). However, some areas have been designated as parks or reserves based on high-profile species only or because they form a habitat for endangered or endemic plants or animals or are unique natural landscapes. Many parks are declared for purposes other than wildlife conservation.

For over a century national parks and reserves have been the dominant method of wildlife conservation (Western and Gichohi 1993). Because most of these areas are not complete ecological units or functional ecosystems in themselves, they have experienced a range of management problems. The main problem is the general decline in plant and animal diversity (Western and Gichohi 1993). A new approach is thus the ‘ecosystem approach’ to promote biological diversity outside the traditional protected areas (Prins and Henne 1998).

Wildlife, and its conservation, is in crisis. Unprecedented and increasing loss of native species and their habitats has been caused by different human activities. Management strategies have focused mainly on single species and protected areas.

Immediate conservation is required particularly for areas outside the protected area system, which have rich wildlife resources. However, this action is hampered by lack of information and knowledge about species abundance, species distributions and factors influencing their distributions in these areas. Also there is general lack of understanding about the ecological, social and cultural processes that maintain diversity in different areas or ecosystems, i.e. of wildlife conservation at a landscape scale.

In this chapter, the application of remote sensing (RS) and geographic information system (GIS) in the collection and analysis of wildlife abundance and distribution data suitable for conservation planning and management are examined. Section 7.2 briefly examines issues related to wildlife conservation and reserve management. Section 7.3 reviews the techniques used in mapping wildlife distributions and their habitats. Resources required by wild animals to fulfil their life cycle needs are described in section 7.4. Section 7.5 reviews the application of GIS in mapping and modelling suitability for wildlife and factors influencing their distribution. Modelling of species-environment relationship is discussed in section 7.6. A future innovative potential of the use of RS and GIS in the collection, analysis and modelling of wildlife abundance and distribution is briefly discussed in section 7.7.

7.2 WILDLIFE CONSERVATION AND RESERVE MANAGEMENT

With the exponential growth of human populations, and the consequent demand on natural resources, the Earth is being transformed from large expanses of natural vegetation towards a patchwork of natural, modified and man-made ecosystems. Faced with this reduction, fragmentation or complete disappearance of their specific habitat, many wildlife species have suffered reductions in their numbers or range, or have become extinct. The underlying factors responsible may be classified as those with a direct negative effect, such as hunting, fishing, collection or poaching, and those indirectly detrimental to wildlife through impact on their habitat. Among these, the alteration and loss of habitat is considered the greatest threat to the richness of life on Earth (Meffe and Carroll 1994).

Over the past century, conservation efforts have concentrated on the acquisition and subsequent protection of critical wildlife habitat. Today, approximately 7.74 million km^2 or 5.19% of the world's land surface is designated and protected as parks or reserves (WCMC 1992). Many of these parks and reserves, however, were created as attractions with geological or aesthetic appeal rather than for biological conservation. In general, they are remnants of lands with marginal agricultural value, while highly productive lands tend to be underrepresented (Meffe and Carroll 1994). The International Union for Conservation of Nature (IUCN) recommended the preservation of a cross-section of all major ecosystems and called for protection of 13 million km^2 of the Earth's surface (Western 1989).

Once established, reserves do not necessarily guarantee the conservation of wildlife, because various processes operating within their boundaries might negatively affect wildlife. In many cases, protection within reserve remains marginal at best, exposing wildlife to incompatible land uses such as livestock grazing, mining, agriculture or logging. Some species are vulnerable to poaching or over exploitation. In addition, exotic diseases or invasive species may impact wildlife populations (Prins 1996). Modification of environmental conditions including the availability of resources such as water points for livestock, may change the balance amongst native species, advantaging some and disadvantaging others. Visitors may exert a negative impact on wildlife or their environment, particularly in highly frequented areas or where sensitive species occur.

Traditionally, wildlife management focussed on the maintenance of some desired state of the resource base within the reserve, while controlling factors negatively impacted on wildlife and the resource base on which they depend. Such internal management does however not guarantee sustainable wildlife conservation. Biological and physical processes in the surrounding areas may negatively impact on populations residing in the reserve (Janzen 1986; Prins 1987). Fragmentation of wildlife habitat outside reserves for instance is considered a potentially important factor negatively affecting wildlife within (Meffe and Caroll 1994). Wildlife populations in reserves might be too small to persist on their own and depend for their long-term survival on interbreeding with other sub-populations inhabiting similar habitat outside. Fragmentation of the habitat outside would increase the isolation of the population inside the reserve and increase the probability that it will go extinct (Soulé 1986).

Nowadays many reserves are confronted with increased intensity of land use at their periphery. Therefore, successful wildlife management requires the provision and maintenance of optimal conditions both within and outside reserve boundaries. Species with large territories may be at risk when individuals cross reserve boundaries, e.g. grizzly bears may be shot by rangers when posing a threat to cattle. Successful wildlife management requires appropriate data on wildlife especially data on spatial and temporal abundance and distribution. Remote sensing and GIS techniques are increasingly being used in the collection and analysis of these data as well as the monitoring and overall management of wildlife.

7.3 MAPPING WILDLIFE DISTRIBUTION

Geographic information on the distribution of wildlife populations forms a basic source of information in wildlife management. Most commonly, distribution is derived from observations in the field of the animal species or their artefacts. Radio-telemetry and satellite tracking have been used (Thouless and Dyer 1992) to record the distribution of a variety of animal species.

Aerial survey methods based on direct observation augmented by use of photography have been used to map the distribution of various taxonomic groups such as mammals (Norton-Griffiths 1978), birds (Drewien *et al.* 1996; Butler *et al.* 1995) and sea turtles and marine mammals (Wamukoya *et al.* 1995). Aerial photography has been used to map the distribution particularly of colonial species such as birds (Woodworth *et al.* 1997) or mussels (Nehls and Thiels 1993).

GIS is increasingly used for mapping wildlife density and distribution derived from ground or aerial survey observations (Butler *et al.* 1995; Said *et al.* 1997). For example, Figure 7.1 displays the distribution of wildebeest in the Mara ecosystem in Narok district (Said *et al.* 1997). McAllister *et al.* (1994) used GIS to analyze the global distribution of coral reef fishes on an equal-area grid.

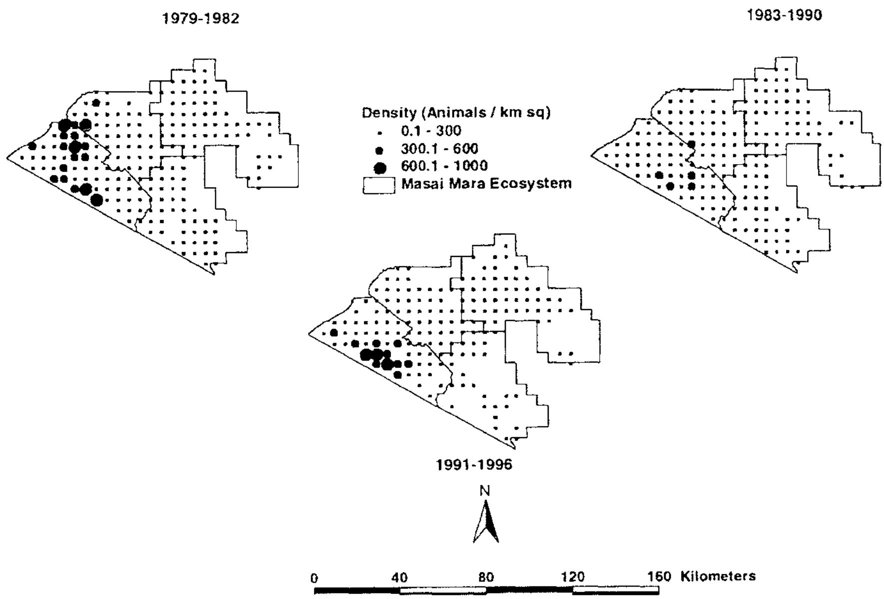

Figure 7.1: Spatial distribution and average density (N.km^2) of wildebeest in the Masai Mara ecosystem, Narok District, Kenya for the period 1979–1982, 1983–1990 and 1991–1996. The density was calculated on 5 by 5 km sub-unit basis.

Satellite remote sensing undoubtedly has a potential for mapping of animal distribution, but successful applications seem to be few. Mumby *et al.* (1998a) mapped coral reefs using aerial photography and remote sensing imagery. For mapping of nine reef classes, they reported an overall accuracy of 37 per cent for Landsat TM and 67 and 81 per cent with aerial photography and an airborne CASI hyperspectral scanner respectively. Mumby *et al.* (1998b) reported that classification accuracy could be significantly increased by compensation for light attenuation in the water column and contextual editing. Thermal scanners have been used to determine the presence and/or numbers of animals not readily observable, such as beavers and muskrats in their lodges during winter (Intera Environmental Consultants 1976). They have also been used in Canada to count bison, moose, deer and elk in comparison with aerial and ground counts (Intera Environmental Consultants 1976). The main drawback is error emanating from hot spots such as solar heated objects, vacated sleeping spots and non-target animals.

A number of species such as termites, earth worms, or shellfish increase the roughness of the substrate, either through their exoskeleton or through their impact on soil micro-topography. Radar, being sensitive to such micro-relief (Weeks *et al.* 1996, Van Zyl *et al.* 1991), could potentially be applied to map such animal populations.

Hence, successful satellite-borne remote sensing applications seem to be restricted to cases where species modify their environment to such extent that their impact on the environment can be detected by a sensor. It is envisaged that the ability to map animal distribution in this way will be greatly enhanced by the advent of high spatial resolution remote sensing platforms.

7.4 MAPPING WILDLIFE RESOURCE REQUIREMENTS

Resources used by animals include those material goods required to fulfil their life cycle such as food, drinking water, nesting sites, shelter etc. Vegetation maps tend to be used to map the spatial distribution of these resources (with the exclusion of drinking water) (Flather *et al.* 1992). In some studies, the distribution of a species has been related directly to the classes or map units of these vegetation maps (August 1983). Here it remains undetermined whether the animal is located in one vegetation class or another because of the availability of food resources, shelter, nesting or a combination of those. Researchers and managers have converted the information provided by a vegetation map into the spatial distribution of the individual resources. Pereira and Itami (1991) used prior knowledge on the feeding ecology of the Mt Graham squirrel and seed productivity for various conifer species, to derive a food productivity map from a land cover map containing information on dominant tree species.

Articles presenting vegetation maps[1] or describing the techniques to produce them frequently stress the utility of such maps for wildlife or faunal management. Typically, vegetation maps contain thematic information on physiognomy, species composition or some other vegetation attributes (see for example Loth and Prins 1986). A survey on the thematic content of a sample of 169 rangeland vegetation maps, mostly from the African continent (Waweru 1998), revealed that 115 (68 per cent) and 69 (40 per cent) maps included information on vegetation physiognomy and species composition respectively. Forty out of the 169 maps (24 per cent) provided information on vegetation biomass while only two maps (1.2 per cent) provided explicit information on vegetation quality.

Although they are the most frequently mapped attributes, one might question whether vegetation physiognomy and species composition would be the most appropriate ones from a wildlife management perspective. Wildlife managers might well prefer information on the quantity and quality of food resources, which are considered major factors determining the distribution of animals.

Remote sensing has been applied to quantify the spatial distribution of vegetation biomass (Box *et al.* 1989; Prince 1991; Hame *et al.* 1997). This quantification is mainly done by means of Normalized Difference Vegetation Index

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[1] For techniques for preparation of vegetation maps the reader is referred to Chapter 6. This section focuses on the application of vegetation maps to wildlife management.

(NDVI), or 'greenness index' (Tucker 1979) (see Chapter 4 for details). Annually integrated NDVI was shown by Goward *et al.* (1985) to be related to biome averages of annual net primary production (NPP). Prince (1991) demonstrated that there is a strong linear relationship between the satellite observation of vegetation indices and the seasonal primary production. Wylie *et al.* (1991) determined the relationship between time-integrated normalized difference vegetation index statistics and total herbaceous biomass through regression analysis. He concluded that availability of several years of data makes it possible to identify the temporal and spatial dynamics of vegetation patterns within the Sahel of Niger in response to year to year climatic variations. Although the NDVI appears to be a useful index of some surface phenomena, it is still not certain what biological phenomena the NDVI actually represents (Box *et al.* 1989). NDVI values based on the current NDVI products are not reliable in complex terrain (high mountains, coastal areas, irrigated areas in dry climates, etc.) due to mixed pixels. The NDVI values do not fall to zero in deserts or over snow cover, due to background effects (Box *et al.* 1989). However, current NDVI data seem reliable elsewhere, at least for annually integrated totals (Prince and Tucker 1986).

Many studies have been undertaken to relate NDVI to crop production (e.g. Groten and Ilboudo 1996) or grass biomass production (e.g. Prince and Tucker 1986). However, there are very few studies that have attempted to relate NDVI to animal distributions (e.g. Muchoki 1995; Omullo 1996; Oindo 1998).

Drinking water constitutes a critical resource to wildlife, particularly in arid and semi-arid zones. Hence, one would expect water dependent animals to be close to watering points. In studies in the Tsavo and Mara ecosystem of Kenya, Omullo (1995), Rodriguez (1997) and Oindo (1998) all reported significant relationships between the distribution of various wildlife species and the distance to permanent water points.

7.5 MAPPING AND MODELLING HABITAT SUITABILITY FOR WILDLIFE

In this section, habitats and habitat maps are described first. This is followed by a discussion about mapping of habitat suitability for wildlife, accuracy of the suitability maps and factors influencing wildlife distributions.

7.5.1 Habitats and habitat maps

Information and maps on wildlife distributions are essential for wildlife management. In many cases however, management interventions focus on the resource base on which the animals depend, rather than on the animals themselves as the vegetation or habitat is managed more easily than the animals themselves. Wildlife management organizations therefore traditionally displayed a strong interest in the mapping of resources relevant to wildlife. The underlying idea was that maps displaying the resource base could assist to identify areas suitable for wildlife.

Vegetation maps as well as so-called habitat maps have been used for this purpose. Traditionally, the term habitat has been defined either as the place or area where a species lives and/or as the (type of) environment where a species lives, either actually or potentially (Corsi *et al.* 2000). In all of the definitions reviewed by Corsi *et al.* (2000), the term habitat has been defined as the property of a specific species. Consequently, it can only be used in association with a name of a species, e.g. flamingo or tsetse habitat. This corresponds to the original use of the word, which was derived from *habitare* (to inhabit) in old Latin descriptions of a species. Hence, one would expect a habitat map to display information on the distribution of the habitat of a specific species. This, however, is not the case; habitat maps display information on the distribution of vegetation types or land units. For some intractable reason, these map units have been called habitats, e.g. a riverine or a woodland habitat, which is clearly a wrong but well-established terminology. In conclusion, habitat maps do not pertain to a specific species but refer to vegetation types or land units.

Use of the term habitat is not restricted to habitat maps. It has proliferated into the literature dealing with the assessment of suitability of land for wildlife. In habitat evaluation, habitat suitability index models and habitat suitability maps the term refers to units of land rather than to specific species.

The various meanings of the term habitat lead to ambiguity, for instance when used in the context of suitability assessment. According to the definition, above all habitat would by definition be suitable and unsuitable habitat would be a contradiction in terms. Areas unsuitable for a species would therefore have to be considered as non-habitat. When used in the second meaning, however, all land would be labeled as habitat, irrespective whether it would be suitable for a species or not. In this chapter, the term habitat is avoided whenever possible, and when applied it is used in relation to a specific wildlife species. The more neutral terms 'wildlife suitability model' and 'wildlife suitability map' are adopted.

7.5.2 Mapping suitability for wildlife

A wildlife suitability map is defined as a map displaying the suitability of land (or water) as a habitat for a specific wildlife species. Since the early 1980s, remote sensing has been used to localize the distribution of areas suitable for wildlife. Cannon *et al.* (1982), for instance, used Landsat MSS to map areas suitable for lesser prairie chicken. Wiersema (1983) mapped snow cover using Landsat MSS to identify snow free south facing slopes forming the winter habitat of the alpine ibex. Hodgson *et al.* (1987) used Landsat TM for mapping wetland suitable for wood stork foraging. More recently, Congalton *et al.* (1993) used a Landsat TM based vegetation map to classify the suitability of land for deer. Rappole *et al.* (1994) used Landsat TM to assess habitat availability for the wood thrush.

These studies depended on a vegetation map, derived from remote sensing, as the only explanatory variable. The assumption was that mapping units efficiently reflect the availability of resources and other relevant environmental factors determining suitability. However, the suitability of land for wildlife may be determined by more than one factor. A single explanatory variable, such as a vegetation map or a land-unit map, does not effectively represent such multiple factors, especially when they were poorly correlated with each other. This is frequently the case; the distribution of good quality grazing areas in arid zones, for instance, does not necessarily correspond to the availability of drinking water resources (Toxopeus 1998). In such cases, where factors are unrelated, GIS will be useful, since separate data layers may be combined in order to provide information on the distribution of independent landscape attributes.

In the second half of the 1980s, wildlife suitability maps integrating various explanatory variables were implemented in a GIS environment. Figure 7.2 shows a scheme of suitability mapping in a GIS context (see also Chapter 2 for a definition of model terms). Such a scheme consists of a suitability model that allows one to predict the suitability of land for a specific species, given a number of landscape attributes. Additionally, it contains a number of spatial databases describing the distribution of these landscape attributes. The suitability model is then used to process these spatial databases to generate a suitability map (Toxopeus 1996).

GIS-based habitat studies generally combine information on vegetation type or some other land cover descriptor, with other land attributes reflecting the resource base as well as other relevant factors. A model for Florida scrub jay developed by Breiniger *et al.* (1991), for instance, included vegetation type and soil drainage to discriminate primary habitat, secondary habitat and unsuitable areas. A more detailed model for the same species (Duncan *et al.* 1995) included seven attributes, all related to land cover.

Herr and Queen (1993) developed a GIS-based model to identify potential nesting habitat for cranes in Minnesota. A significant relation was observed to cover type, and two disturbance-related factors: distance to roads and distance to houses. Clark *et al.* (1993) included seven land attributes: land cover, elevation, slope, aspect, distance to roads, distance to streams and forest cover diversity to predict habitat suitability for black bear.

7.5.3 Accuracy of suitability maps

Wildlife suitability maps and their underlying suitability models have been criticized because of their assumed poor accuracy (Norton and Williams 1992). The maps produced by these models have rarely been validated (Stoms *et al.* 1992; Williams 1988), although this had clearly been advised in the habitat evaluation procedures (USFWS 1981). The accuracy of a wildlife suitability map depends on how well the output corresponds to reality (Figure 7.2). This accuracy is determined by two different sources of error. The first source of error is the spatial database, which comprise both geometric and thematic errors. The second source of error is the habitat suitability model. The accuracy of suitability models depends on the selection of the relevant variables and an unbiased estimation of the model parameters.

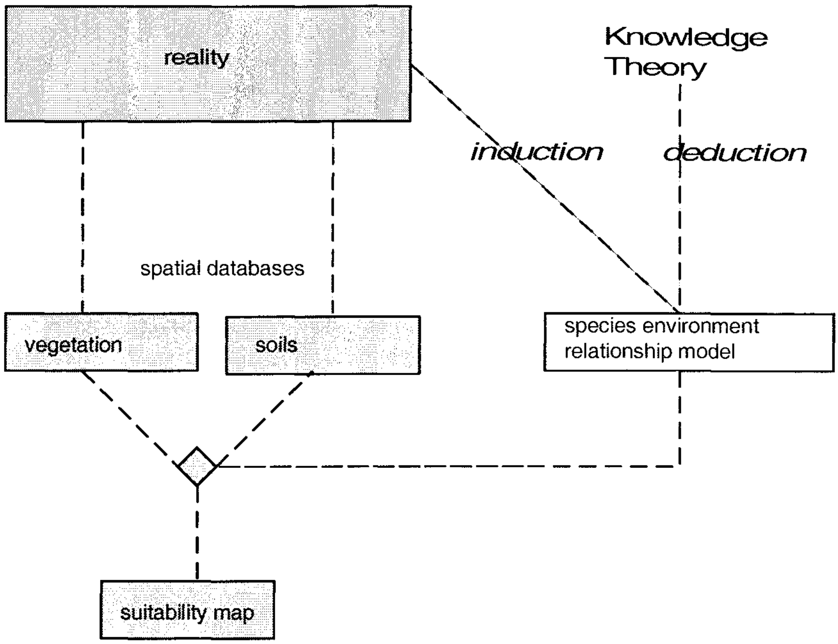

Figure 7.2: Scheme for GIS based suitability mapping.

Accuracy assessment of wildlife suitability models has been discussed in Morrison *et al.* (1992), while Corsi *et al.* (2000) provides a review of potential techniques to assess the accuracy of wildlife suitability maps. Skidmore (1999), Janssen and Van der Wel (1994) and Congalton (1991) give general discussions on techniques to assess map accuracy. These map accuracy assessment techniques require separate data sets for validation of the model developed. Verbyla and Litvaitis (1989) indicated that, in wildlife suitability studies, the number of samples may be too small and described resampling methods to overcome this problem.

In accuracy assessment, the predicted suitability is tabulated against observations on presence and absence of the animal species. Morrison *et al.* (1992) reviewed the reasons why animals would not be recorded in suitable areas (Type 1 error) or would be observed in areas considered unsuitable (Type 2 error). Most animal species are mobile, hence suitable land may not be temporarily occupied, while animals may pass through lands otherwise unsuitable to them. Furthermore, animals may be locally extinct. Animals differ in this respect from plant species or land cover and, because of this, accuracy matrices for wildlife-suitability-maps may yield relatively low accuracy values. We argue that such low accuracy values do not necessarily imply poor model performance. After all, the model predicts suitability rather than presence or absence. Besides, models with a low accuracy may still contain ecologically relevant information.

The potential of a vegetation map to explain the distribution of wildlife depends on its map accuracy. The accuracy of the map information depends on the level of thematic detail. Anderson (1976) distinguished three different levels in land cover maps: Anderson level I corresponds to broad land cover classes such as forest versus grassland; Anderson level II gives a further separation according to broad species groups such as broad-leafed versus pine forest; Anderson level III includes detail such as vegetation types defined by species composition. Accuracy obtained for Anderson level I and II vegetation maps tend to be above 80 per cent, while Anderson level III maps remain below this accuracy level.

7.5.4 Factors influencing wildlife distribution

The actual distribution of animal species may be determined by a variety of environmental factors (Morrison *et al.* 1992). We categorize these into three broad classes; those describing the resource base, physico-chemical factors and factors related to human activities (Figure 7.3). Physico-chemical and anthropogenic factors may influence the distribution of wildlife either directly or indirectly through their impact on the resource base.

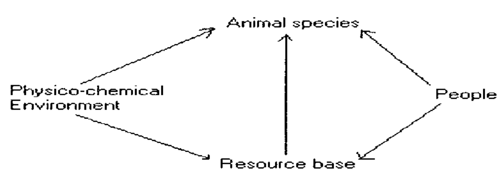

Figure 7.3: Scheme displaying the impact on the distribution of an animal species of three broad categories of environmental factors. People and the physical-chemical environment may exert a direct as well as an indirect impact through their influence on the resource base.

Johnson (1980) argued that selection of habitat by an animal species may occur at different spatial scales and proposed the following hierarchical order in the selection of habitat by an animal. First order selection corresponds to the geographic range of a species, second order selection to the home range of an animal or a social group, while third order selection pertains to utilization of resources within that home range.

It has been suggested by Diamond (1988) that different biophysical factors affect species richness at different scales. At the regional level, productivity and climatic zones determine species richness. This has been amply demonstrated in, for example, Rosenzweig (1995) but also by Veenendaal and Swaine (1998) in their analysis of the natural limits of the distribution of tree species from the West African rainforest. At the landscape level (or gamma level), productivity, climate (precipitation, temperature, growing season) play a role; this has been demonstrated for grazing herbivores in Africa (Prins and Olff 1998), but also for Gobi Desert rodents and even North Atlantic megafauna (fish, echinoderms and crustaceans) (Rosenzweig 1995). Even seasonality and plant phenological processes may play a role, for example, for primate assemblages in West Africa (Tutin and White 1998; see also Newbery *et al.* 1998). At the community level, the aforementioned factors play a role still, because the species assemblage at that level is a sample of the regional species pool. However, not all species of that pool will be found at the community level, often because of competition between species, and the smaller the area under scrutiny, the lower the number of species (Prins and Olff 1998). Lastly, at point or microhabitat level, the most important factors are soil moisture and soil nutrients, and, especially for plants, the light regime (Zagt and Werger 1998; Loth 1999). Especially at this level, chance effects, however, may dominate.

People and their associated activities may exert positive or negative influences on the distribution of wildlife. In the case of a negative impact, it may prevent the animals from occupation of otherwise suitable habitat. The potential number of human-induced disturbance factors is large and it would go beyond the scope of this chapter to list them all. However, most human-related disturbance factors do have one thing in common: their intensity or frequency diminishes with the distance from a human settlement or infrastructures used by people. Not surprisingly, therefore, distance has been used as an explanatory variable in many GIS-based wildlife distribution models. For instance, the areas mapped by Herr and Queen (1993) as suitable habitat for cranes were largely determined by distance to roads, buildings and agricultural lands. However, distance as such does not influence the distribution of the animals. Instead, an unknown variable (for instance, human disturbance) associated with distance would be the ultimate factor affecting the observed animal distribution (Prins and Ydenberg 1985). Distances should therefore be carefully interpreted and considered as factors reflecting associated human impact.

7.6 MODELLING SPECIES-ENVIRONMENT RELATIONSHIPS

The ability to model spatial distribution and change in distribution of wildlife is of considerable importance in wildlife management. Once spatial distribution can be adequately modelled, distribution and abundance may be monitored effectively over time. GIS can be effective in modelling animal distribution if the necessary data are available. However, data availability is currently the limiting factor in many areas.

Production of a suitability map requires a model to predict the suitability of land for a wildlife species given both a set of land attributes and also distribution of potential competitors. According to the source of knowledge on which they are based, such models may be classified as theoretical-deductive and empirical-inductive methods, based on the definitions in Chapter 2 (Figure 2.2). The former use theoretical considerations and existing knowledge to design a model, whereas the latter depend on knowledge on species environment relationships obtained through empirical research (Chapter 2).

Habitat suitability index (HSI) models, described by Atkinson (1985) as hypotheses about species-environment relationships based on the literature and opinions of experts, are an example of theoretical-deductive wildlife-environment relationship models. Hundreds of such models have been developed since the early 1980s (Atkinson 1985; Williams 1988) and several have been used to implement wildlife suitability maps in a GIS environment (Donovan *et al.* 1987; Duncan *et al.* 1995). Other deductive models have been presented by, for instance, Herr and Queen (1993) and Breininger *et al.* (1991) – see also Chapter 2. Deductive modelling, however, has severe drawbacks in wildlife ecology. For many species, knowledge about habitat requirements simply does not exist. However, expertise with respect to wildlife habitat requirements may be limited, biased or not be available (Kangas *et al.* 1993; Crance 1987).

Inductive modelling has been suggested to overcome these problems (Walker 1990; Walker and Moore 1988; Chapter 2). Inductive modelling is based on the analysis of data resulting in the generation of new knowledge and the formulation of new models. Here modelling goes from the specific case (field data) towards a generalization.

A variety of analytical techniques has been used to investigate species-environment relationships. These include logistic regression (Pereira and Itami 1991; Buckland and Elston 1993; Osborne and Tigar 1992; Walker 1990; Rodriguez 1997), discriminant analysis (Haworth and Thompson 1990), classification and regression trees (Walker and Moore 1988, Skidmore *et al.* 1996), canonical correlation analysis (Andries *et al.* 1994), supervised non-parametric classifiers (Skidmore 1998; Skidmore *et al.* 1996) and neural networks (Skidmore *et al.* 1997).

The distribution of a species may be related to many independent variables using a GIS. Initially this appears to be a panacea. However, one may become overwhelmed by the multitude of data layers available in a GIS. Many layers may be irrelevant to the problem at stake. The number of the independent variables included in the analysis could be reduced using *a priori* knowledge about the ecology of the species. Even then, however, many variables might be retained and frequently they will tend to be highly correlated. Such high mutual correlation is, for instance, a common phenomenon when using the various bands of a remote sensing image, and especially hyperspectral remote sensing (Skidmore and Kloosterman 1999; Van der Meer 1995) in wildlife suitability studies. Such collinearity may result in models that have a poor predictive power when extrapolated to non-surveyed sites. Osborne and Togar (1992) and Buckland and Elston (1993) used principal components analysis (PCA) and subsequently regressed the dependent variable against the principal components. Duchateau *et al.* (1997) used PCA and varimax rotation to reduce the dimensionality of the data and to identify a reduced set of climatic predictor variables. These were then regressed against the independent variable, the presence of outbreaks of a tick borne livestock disease. The reduction of the dimensionality was based on claims of superior performance when applied to an independent data set over models including a larger set of predictor variables. No attempt, however, has been made to verify this claim.

7.6.1 Static versus dynamic models

So far, wildlife suitability mapping techniques have been described in terms of static models, both for animal populations as well as for the environment (Table 7.1). In reality, both animal populations and resource bases tend to display highly dynamic behavior.

Table 7.1: Classification of GIS based models for wildlife management depend on whether a static or dynamic model has been used to map the resource base as well as whether the response of the animal population would be based on a static or dynamic model.

Resource base	Animal population	
	Static	**Dynamic**
Static	A	B
Dynamic	C	D

Breininger *et al.* (1998) studied the relationship between demographic characteristics to a habitat suitability index (HSI) map for Florida scrub-jay. Yearling production, breeder survival, demographic performance and jay density were significantly correlated to HSI. Pereira and Itami (1991) linked a static species-environment model to the current and an alternative state of the environment to assess the impact of the development of an astronomical observatory on the habitat of the Mt Graham squirrel. Such mapping of the suitability for wildlife does not capture the change over time due to succession, natural disturbances such as fire or storms, or human activities.

Prediction of the impact of human activities is relatively simple in case of such localized infrastructure projects. In many cases, it would be much more difficult to predict the location of future human impacts. Toxopeus (1995) predicted deforestation in Cibodas, Indonesia, using distance from settlements and accessibility of the terrain as predictor variables (Figure 7.4).

Kruse and Porter (1994) linked a dynamic resource base model to a habitat suitability model to evaluate the change in suitability of forest for wildlife over time under different management options.

A number of models have been published predicting the population dynamics of a wildlife species in response to a dynamic resource base. Two different types of models can be discerned, namely, non-spatially explicit models, and spatially explicit ones. For the spatially non-explicit models, we refer to those published on the reaction of wildebeest (*Connochaetus taurinus*) in the Serengeti in Tanzania to fluctuations in rainfall. Especially, variations in dry-season rainfall are important to understand population dynamics as rainfall determines the length of the growing season for the vegetation, which, in turn, determines animal condition and, thus, natality and mortality (Hilborn and Sinclair 1979; Mduma *et al.* 1998). Also the population dynamics of semi-wild Soay sheep on the Hebridian islands of Rhum and Hirta has successfully been modelled by Illius and Gordon (1999); fluctuations in rainfall and inclement weather determine the fluctuations and animal condition is linked to energy gains and, especially, losses.

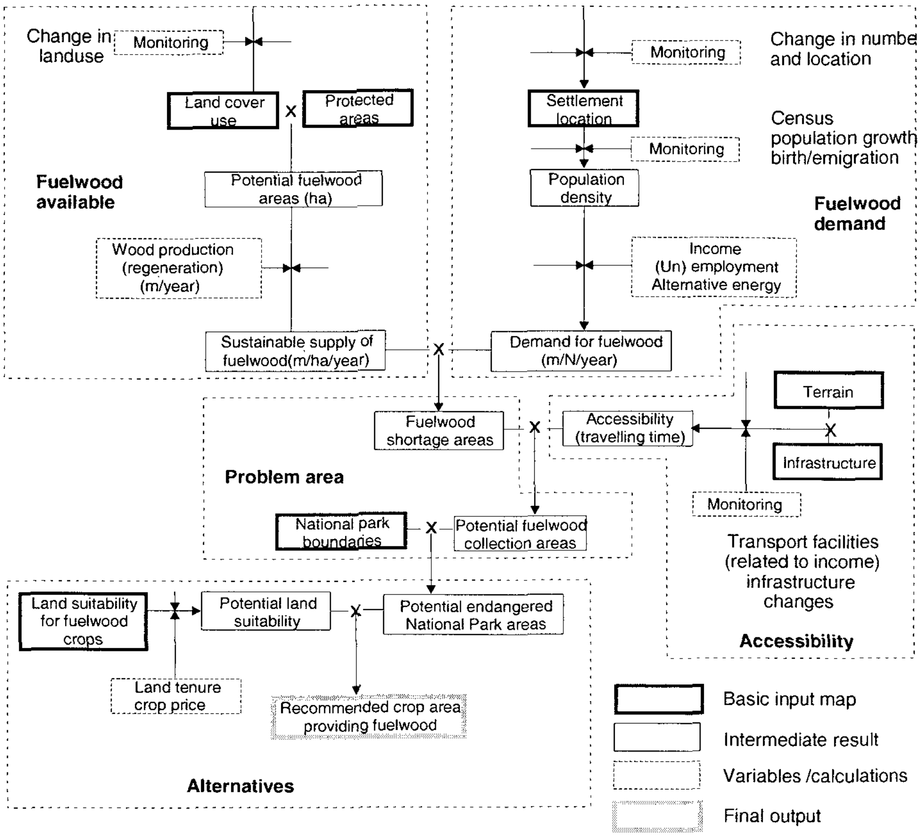

Figure 7.4: Scheme of a GIS model, applied to predict the fuelwood collecting areas in the Cibodas Biosphere reserve, West Java.

Dynamic wildlife-population models have also been linked to alternative states of the resource base. DeAngelis *et al.* (1998), for instance, predicted the reproductive performance of deer in the Everglades (in Florida, USA), based on the availability of resources as determined by the hydrological conditions under the current situation, and in an alternative scenario. Spatially explicit models linking the dynamics of the resource base to population dynamics are still rare. Central to these models are the feedback mechanisms between animal consumption, plant production and competition between plant species (Van Oene *et al.* 1999). Indeed, competition between plant species determines vegetation composition, and this, in turn, determines suitability for herbivores. Population dynamics of three large grazers (Red deer *Cervus elaphus*, Heck cattle, and Konik horses) and vegetation composition has thus been spatially modelled for the wetland 'Oostvaardersplassen' in the Netherlands (Groot Bruinderink *et al.* 1999).

7.6.2 Transferability of species – environment models

For a limited number of species, models have been developed for a particular site. However, the transferability of such site-level models to other areas remains unknown, and may lead to biased results. Consequently, application of a site-level model to a whole region is not recommended (Risser *et al.* 1984). So far, only a few papers have addressed the problem of transferability of wildlife suitability models, in geographically restricted areas. Thomas and Bovee (1993) investigated transferability of habitat suitability criteria between two rivers in Colorado. Homer *et al.* (1993) concluded that a model developed in a sub-area of a 2740 km^2 county provided reliable predictions when transferred to two other sub-areas in the same county.

Once established, a wildlife manager will be tempted to apply a suitability model and derived maps to the future. This assumes that the underlying species-environment model would remain unchanged through time. However, this will not be the case when relevant ecological factors affecting animal distribution have not been included in the model. Rainfall in arid and semi-arid zones, for example, is known to vary in both time and space. It is noteworthy that none of the cited articles related animal distribution to spatial pattern of rainfall prior to or during the study. Instead, animal distribution tends to be related to long-term averages of climatic variables. Oindo (1998) used NOAA-AVHRR NDVI instead of rainfall data to reflect spatial variation in vegetation phenology. He related NDVI, together with other landscape attributes, to the observed distribution of Topi around Masai Mara Reserve, Kenya. NDVI, however, did not significantly explain the observed distribution. Oindo (1998) reported that a suitability model for Topi, developed for a particular year, correctly predicted the distribution of the species in other years, indicating that the model may be transferred over time.

7.7 INNOVATIVE MAPPING OF WILDLIFE AND ITS PHYSICAL ENVIRONMENT

Various aspects of the physical environment have been used in wildlife suitability studies. It is beyond the scope of this chapter to review all of these and discuss methods to map them. Here we highlight a selection of factors for which innovative methods for mapping wildlife distribution appeared in the past few years.

Climatic databases, derived through interpolation from point-based observations, have been used to map distribution at continental scales (Prins and Olff 1998). At larger scales, micro-topographical climatic and soil variation becomes more prominent (Varekamp *et al.* 1996). Slope and aspect, which determine the local moisture regime through their impact on the solar radiation balance, have frequently been in suitability models. Nowadays, GIS-based models for mapping solar radiation in relation to topography are available (Kumar *et al.* 1997), but so far these have apparently not been used for wildlife suitability studies.

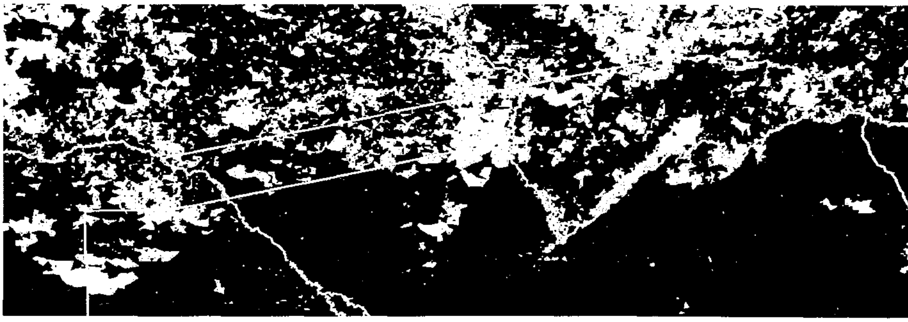

Figure 7.5: Map indicating bush fires (burn scars) in August, 1996, based on NOAA-AVHRR data, of the Caprivi region in Namibia (source: Mendelsohn and Roberts, 1997).

Fire occurs in many ecosystems (Huston 1979), and affects wildlife populations. Recent forest fires for instance exerted a negative impact on orang-utan populations in Kalimantan, Indonesia. Thermal remote sensing allowed mapping and monitoring the distribution of fire. Historic fire-maps based on NOAA-AVHRR thermal bands are available since the early 1990s (Figure 7.5).

Flooding may influence the distribution of wildlife habitat through its impact on the resource base animals depend on. Radar imagery has successfully been applied to map flooded areas (Richards *et al.* 1987; Pope *et al.* 1992; Imhof 1986), even underneath closed vegetation canopies. DeAngelis *et al.* (1998) used information on flooding derived from radar to predict the reproductive rate of deer in the Everglades.

A critical gap remains between satellite data and the many varieties of field observations (Miller 1994). Currently verification of broad-scale mapping efforts using field survey data is still a problem in our attempts to map natural and human induced features. A new approach that combines GPS and videography offers a practical method to validate and classify TM imagery to produce vegetation maps (Graham 1993).

Species distribution mapping is an increasingly important part of ecological science (Miller 1994). The equal-area grid arrangement is a useful framework for representing species presence/absence data and for analyzing species distribution patterns. This approach is suitable mainly for developing countries where the primary sources of data on species spatial distribution are very often presence/absence data. This approach is demonstrated in several recent studies of birds in East Africa (Pomeroy 1993).

7.8 CONCLUSIONS

The potential for the use of RS and GIS technologies in wildlife mapping, natural resource planning and management are large. These technologies are currently fully developed and they are increasingly being applied in natural resource mapping, planning and management. However, their application, particularly in developing countries, is still limited by lack of appropriate scale of data, hardware, software and expertise.

Future research in wildlife modelling should focus on developing more realistic dynamic models of wildlife in space and time. Since the ecosystems to be modelled can be significantly affected by stochastic events and the responses of wildlife are non-linear in form, models must be dynamic and aim to provide predictions of known precision that are testable.

Given that the two most common questions asked by wildlife managers are likely to remain 'where and in what abundance does it occur?' and 'what will happen to it if....?', continued research should be directed to refining models that can best answer these questions (Norton and Possingham 1993).

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