2 Literature review

This chapter opens with a discussion of the terms criteria and indicator, using the meanings attributed to them in the so-called Helsinki Process (Granholm *et al* 1996). Then other approaches to the indicator concept are presented, such as the CIFOR definitions (Stork *et al* 1997). Direct assessment and quantification of biodiversity is a large and complicated task, which requires intensive fieldwork, often by researchers with specialised knowledge. It was therefore considered outside the scope of this and previous projects to devise methods for quantifying on-the-ground biodiversity with values derived *only* from EO data. However, it was found important to provide an overview of how (and if) biological diversity can be measured and quantified - and how precise and reliable the results are - in order to find the extent to which the use of remote sensing can contribute to or supplement conventional (labour intensive) methods of environmental monitoring.

Spatial metrics derived from digital EO data are more valuable, and applicable for (ecosystem/conservation) management purposes, when there are solid theoretical links between the biological processes and properties of land cover maps (Haines-Young and Chopping 1997, McCormick and Folving 1998, Gustafson 1998). Thus a section of the literature review is devoted to outlining basic ecological theories with spatial aspects and discussing how they relate to and incorporate statistical measures of diversity and of landscape geometry. The nature of natural (forest) ecosystems, in that they are complex and nested systems makes it relevant to look closer at scaling issues, as done in section 2.3.3. The potential relationships between spatial metrics from land cover maps and results from numerical modelling of meta-populations in real and simplified landscapes are addressed, and the use of “neutral” models discussed, i.e. assessment of metrics values from artificially generated ‘images’ of ideal landscape where the properties under investigation can be controlled (Gardner and O’Neill 1991, With and King 1997). This can help select a group of spatial indices to be used in assessment of sustainable forestry at landscape or regional level.

Working with EO data poses some practical problems during the process of moving from raw sensor data to reliable land cover or habitat maps. It is not within the scope of this thesis to review the wide range of possible image processing techniques, to that end ´standard approaches' based on recommendations found in the literature will be used, and examples of their implementation are shown in subsequent chapters.

2.1 Sustainability and Biodiversity in environmental policy

In this section, a summary will be made of how the concepts of criteria and indicators, sustainability and biodiversity are defined and applied in environmental sciences, policy and management.

2.1.1 The need for definitions

For the purpose of protection and planning of Europe's forests at inter-national and continental level, a strong interest exists in getting a broad view of their state, be it in terms of vegetation health, species composition or environmental conditions in general (Granholm *et al* 1996, European Commission 1999, Duniker 2000). In particular, it has been considered worth investigating the potential of Earth Observation and Geographical Information System (GIS) techniques for characterising and monitoring forests and their stability as habitats (Scott *et al* 1993, Haines-Young and Chopping 1995, Jones 1998, Hansson 2000).

The spatial structure of forests, and knowledge of the processes that it reflects, can be used to derive some of the criteria and indicators that are needed for monitoring of forest sustainability. Thus, one of the intentions of this review is to examine and describe how the spatial structure within forests influences biological diversity. This implies identifying methods for (a quantitative) description of the shape or outline the forest elements and their position relative to other land-cover types (typically expressed in terms of connectivity and/or fragmentation) – and an assessment of whether quantitative measures of spatial structure can be used as indicators of sustainable forest management or naturalness. It must be stressed here, that these indicators are tools for the assessment of the sustainability of forest- and landscape-management, their numerical values are not goals in themselves. Thus this review will not go into detail with the precise definitions, but rather look at the link between what should be indicated (level of sustainability) and the available remote sensing based techniques to monitor forested landscapes.

However before doing so, some definitions and concepts must be clarified. Standardised, operational definitions are essential if different persons are to make similar measurements of similar entities in order to be able to analyse and compare the results (Morrison and Hall 2002). What is for example meant by this much talked about "landscape level" at which we aim to do our analyses? What do we understand by a "habitat" – perhaps the spatial expression of (the presence of) a niche – depending on the species? How are ecosystems defined and delimited? What actually are "Core Areas" and "Hot Spots" – and to what degree do these concepts depend on the context in which they are used? And finally, what do we mean by words such as "criteria" and "indicator"? (ibid.) The following section provides some material to address these questions.

2.1.2 Criteria and Indicators

The concepts of Criteria and Indicators (C&I) are widely used, and their use as parts of systems for environmental assessment is a special case of their general use – the specification and/or selection of C&I for specific uses, such as assessing the sustainability of forestry being far from simple or without conflicts (Stork *et al* 1997, Mosseler and Bowers 1998, Hansson 2000).

According to Stork *et al* (1997) a <u>criterion</u> is a standard that a thing is judged by. In the forest context it can be seen as a state or aspect of the dynamic process of the forest ecosystem, or a state of the interacting social system, which should be in place as a result of adherence to a principle of sustainable forest management (or well managed forest). The way criteria are formulated should give rise to a verdict on the degree of compliance in an actual situation (van Bueren and Blom, in Dobbertin 1998). In the framework of the 'Montreal process' (ref. Section 2.1.3) a criterion is characterized by a set of related indicators which are monitored periodically to assess change (Granholm *et al* 1996) – thus a criterion can be seen as a category of conditions or processes by which sustainable forest management may be assessed.

An <u>indicator</u> is a measurable attribute of a system component (Duinker 2000), that can ultimately be expressed as a number, i.e. quantified. An indicator is a quantitative or qualitative parameter, which can be assessed in relation to a criterion. It describes in an objectively verifiable and unambiguous way features of the ecosystem or the related social system, or it describes elements of prevailing policy and management conditions and human driven processes indicative of the state of the eco- and social system (van Bueren and Blom, in Dobbertin 1998).

In the Montreal Process (see section 2.1.3), an indicator is a measure (measurement) of an aspect of the criterion, a quantitative or qualitative variable which can be measured or described and which, when observed periodically, demonstrates trends (Granholm *et al* 1996).

In a CIFOR working paper, Stork *et al* (1997, Box 1, p.3), note that C&I form indispensable parts of a hierarchy of assessment tools:

<u>Principle:</u> A fundamental truth or law as the basis of reasoning or action.

<u>Criterion:</u> A standard that a thing is judged by.

<u>Indicator:</u> An indicator is any variable or component of the forest ecosystem or the relevant management systems used to infer attributes of the sustainability of the resource and its utilisation.

<u>Verifier:</u> Data or information that enhances the specificity or the ease of assessment of an indicator.

These definitions are good for theoretical considerations, but in disagreement with the definitions given above following Duinker (2000). According to the CIFOR definitions, the word 'indicator' is often used when it should rather be verifier, the border between these concepts will in practice be hard to define. A review of the different meanings of criteria and indicators can also be found in Granholm *et al* (1996, report 1). Accepting the definitions in the Helsinki process of a **criterion, as something describing the different sides of sustainability *on a conceptual level*** (Ministry of Agriculture and Forestry 1994), the goal of *developing* criteria is clearly outside the scope of this thesis – which will instead look more into how indicators can be defined or selected and calculated. This is in line with the Helsinki process definition of **indicators as typically quantitative measures of change**. Thus an important criterion for selecting an indicator based on EO data is that it is sensitive to environmental changes as manifested in spatial structure at the landscape level.

2.1.3 Sustainability – the concept applied to forestry

The definitions found indicate a close relationship with management, which is reasonable, as the concept of sustainability generally refers to processes and (land use) practices. Following the resolutions from the Ministerial Conference on the protection of forests in Europe, Helsinki, June 1993 (Finnish Ministry of Agriculture and Forestry 1993): *"sustainable management means the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economical and social functions, at local, national and global levels, that does not cause damage to other ecosystems".* The last part indicates the awareness that no part of the landscape can be monitored in isolation. Just as we can not ignore the forested parts when examining agricultural landscapes, we can not leave out the surrounding "matrix" consisting of land used for agricultural, urban or other purposes, when we examine the structure of forests in order to monitor their environmental status, for nature protection and conservation purposes. Meanwhile, we can not leave out the processes related to the human use of forested lands, be they driven by social, economic, practical or even aesthetic motives (Haines-Young and Chopping 1996). Thus, criteria for sustainable forest management should not only focus on maintaining production capacity, nor on the actual biological diversity, but also on the structure and dynamics of forest in relation to the surrounding landscape and the people that inhabit it. This point of view is reflected in the six criteria agreed upon at European ministerial level through the decisions of the ministers at the Helsinki meeting (Finnish Ministry of Agriculture and Forestry 1993). The criteria for sustainable forest management are:

> **1. Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles;**
>
> **2. Maintenance of forest ecosystem health and vitality;**
>
> **3. Maintenance and encouragement of productive functions of forests (wood and non-wood);**
>
> **4. Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems;**
>
> **5. Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water);**
>
> **6. Maintenance of other socio-economic functions and conditions.**

The follow up on these decisions and the reporting from the countries is are often referred to as ‘the Helsinki Process’. A ‘liaison unit’, since 2004 situated in Warsaw (before that in Vienna), manages service to the member countries and the exchange of information, and amongst other activities information is shared at the web site: http://www.mcpfe.org/. Worldwide, several established international initiatives to develop criteria and indicators for sustainable forest management (the Montreal Process, Helsinki Process, the International Tropical Timber Organization (ITTO) Process) are now reaching an implementation stage (United Nations 1998). The Montreal process is concerned with the temperate and boreal forests outside Europe, and thus includes North America and Australia. The Tapparo protocol is concerned with protecting Amazon forests through development of C&I for sustainable management, while the ITTO has produced guidelines on sustainable management of tropical forests (Granholm *et al* 1996, United Nations 1998). According to the Subsidiary Body on Scientific, Technical and Technological Advice to the Convention on Biological diversity (UNEP 1997, annex III), C&I provide a conceptual framework for forest policy formulation and evaluation. Criteria define the essential elements of SFM while Indicators provide a basis for assessing actual forest conditions. C&I, when combined with national goals are also useful for assessing progress towards SFM and they can play an important role in defining the goals of national forest programmes and policies.

2.1.4 Biodiversity – definitions and assessment

According to the convention of biological diversity (CBD 1992): *"Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems."*

2.1.4.1 <u>The value of biodiversity</u>

The economic value of biological diversity and possible future benefits, for instance in the medical field, is being recognised, along with the realisation that the more diverse an ecosystem is, the better equipped it is to withstand and recover from disturbance. In a strategy paper from the European Commission (European Commission 1998, p. 1), the importance of biological diversity is outlined as: *"Biological diversity (biodiversity) is essential to maintain life on earth and has important social, economic, scientific, educational, cultural, recreational and aesthetic values. In addition to its intrinsic value biodiversity determines our resilience to changing circumstances. Without adequate biodiversity, events such as climate change and pest infestations are more likely to have catastrophic effects. It is essential for maintaining the long term viability of agriculture and fisheries for food production. Biodiversity constitutes the basis for the development of many industrial processes and the* *production of new medicines. Finally, biodiversity often provides solutions to existing problems of pollution and disease."*

With the growing awareness at global and continental political decision making level (internationally and within large countries such as USA, Canada, Brazil and Australia) it is becoming clear that the relation between sustainable development and the maintenance of biological diversity is becoming increasingly important, as well as the growing awareness of the interactions between ecosystem composition, structure and functioning (EWGRB 1998, part A, chapter 1). In the proceedings from the first expert meeting of the European network for forest ecology (EFERN), Oswald (1996) states that: *"The conservation of 'biodiversity' is considered today as a major and integrated part of sustainable forest management. But, as biodiversity can concern different levels of appreciation, i.e. populations, individuals and genes, several often quite diverging definitions are used"*.

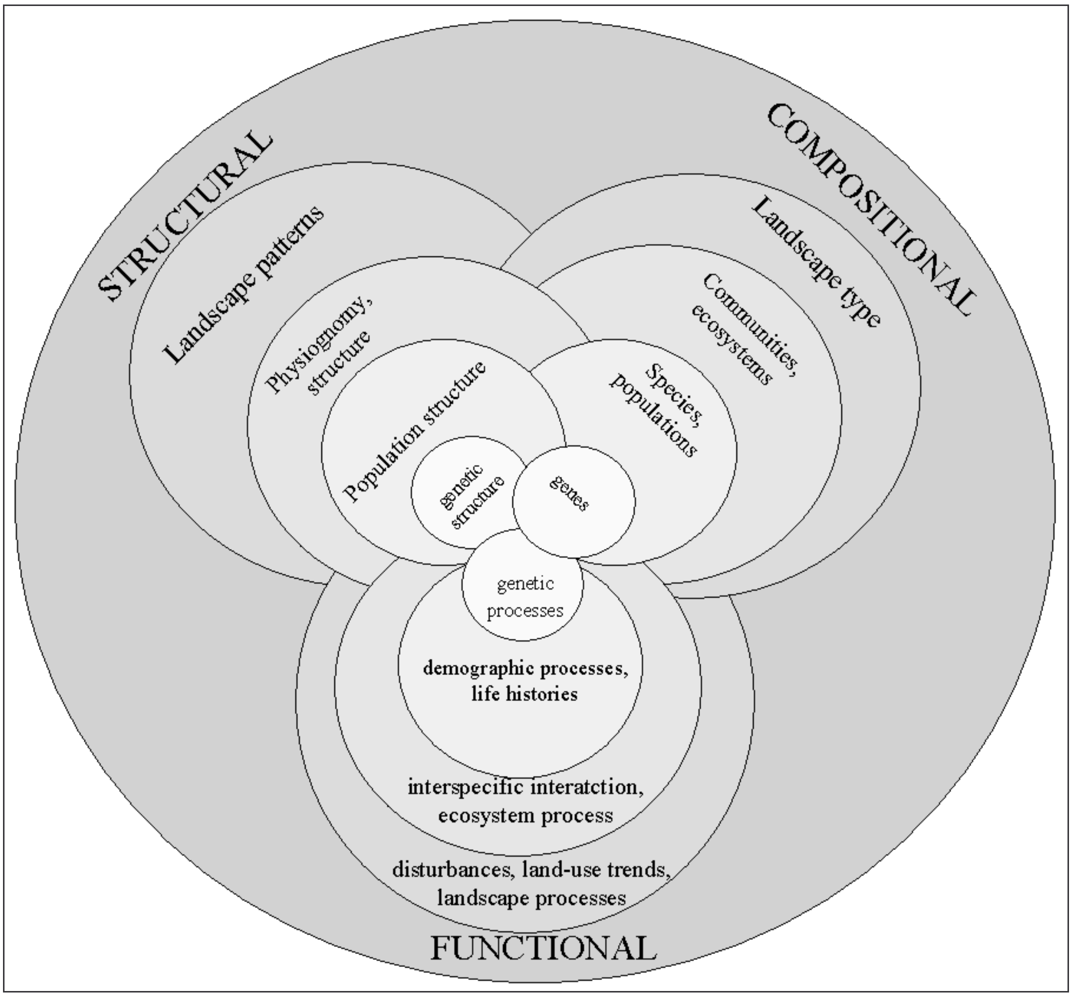

Figure 2.1. Compositional, structural and functional biodiversity, after Noss (1990).

2.1.4.2 Types of biodiversity

It has become a widespread practice to define biodiversity in terms of genes, species and ecosystems, corresponding to three fundamental and hierarchically-related levels of biological organisation (WCMC 1995). In the context of this discussion focus will be placed not so much on the species diversity, but more on the ecosystem 'domain' when it overlaps (spatially) with concepts such as habitat and landscape.

Hierarchy theory shows that higher levels of organisation incorporate and constrain the behaviour of lower levels (King 1990, Marceau 1999). Thus, knowledge of structures and processes dominant at one level – coinciding with a certain spatial scale - will allow us to infer the processes that can take place and which species that will 'fit in' at lower levels or 'smaller' or more restricted spatial scales (Mackey 1996, Mackey and Lindenmayer 2001), as utilised by McGarigal and McComb (1995) and by Rolstad *et al* (2000) in a study of woodpeckers in a mosaic of forests and cultivated land. In a report on ecological conditions of old-growth.Douglas-fir forests in the North-western United States, Franklin *et al* (1981, referred in Noss (1990)) distinguished between compositional, structural and functional biodiversity, as illustrated in Figure 2.1, see also Table 2.4, page 56. This approach has since been applied intensively in ecological research, where 'function' sometimes is replaced by 'development', indicating that this is the component of biodiversity with the strongest temporal dependence, or sensitivity to temporal scale when it comes to observation of parameters. For a recent review of concepts, terms and applications, see Puumalainen (2001).

2.1.4.3 Spatial levels of biodiversity

Whittaker (1972) defined and discussed a selection of diversity metrics. He introduced the measures of Alpha, Beta and Gamma diversity, to be used along with the concepts of niche and hyperspace (of niches). The definitions below are taken from Gale (1996), but are commonly accepted.

Alpha diversity is the variety of the organisms that occur in a particular place or habitat, this is often also called the 'local diversity'.

Beta diversity is defined as

a) The diversity between or among more than one community or along an environmental gradient, or

b) The variety of organisms within a region arising from turnover of species among habitats.

Beta diversity can thus be considered the change rate of the Gamma diversity, which is the Landscape-level or regional diversity. Clearly what should be aimed at and focused on, when investigating the applications of remote sensing techniques, is whether and how it is possible define some links between the Gamma diversity and the spatial structure of forests and wooded lands.

The terms Epsilon and Delta diversities are used to respectively denote inventory or area diversities and gradients of Alpha and Gamma diversity across regions and continents (Stoms and Estes 1993), thus making comparisons possible on a global scale. The concepts are illustrated in Figure 2.2.

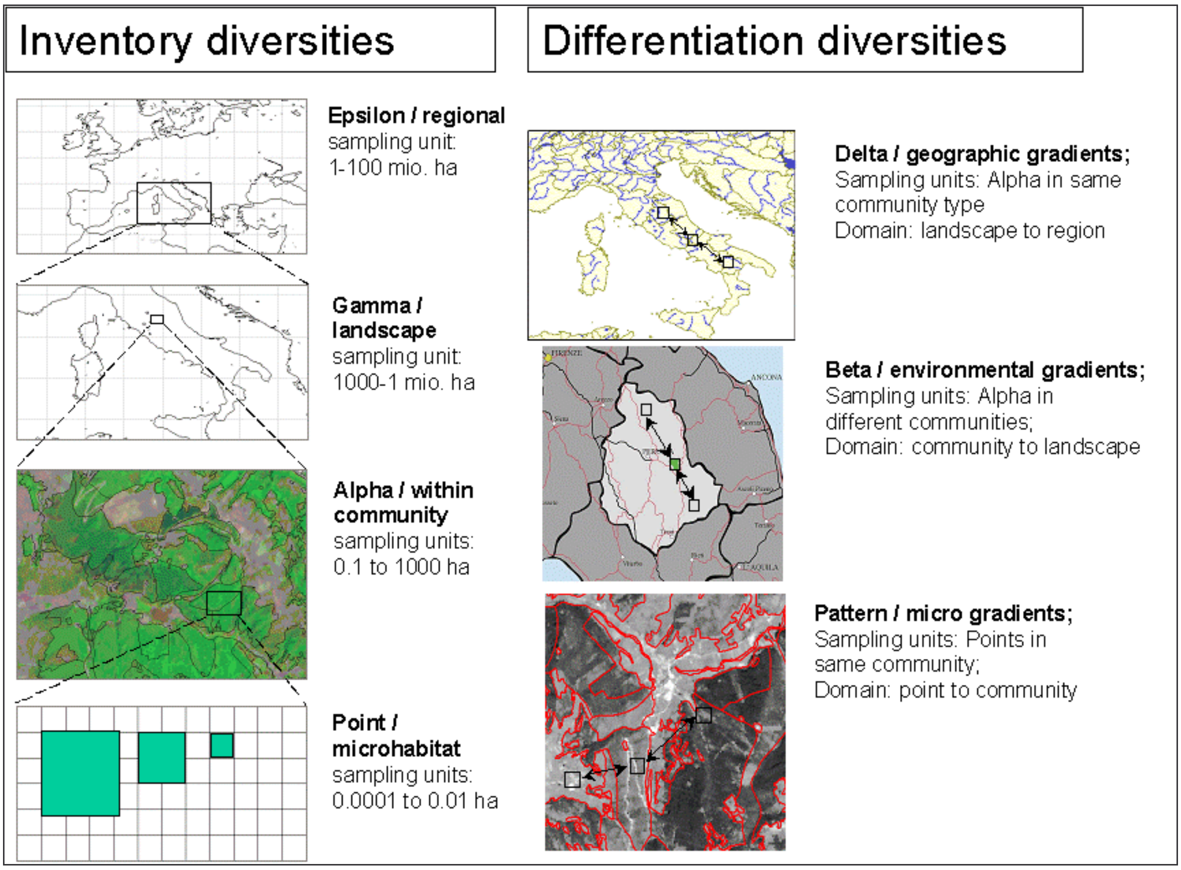

Figure 2.2 Levels of biological diversity as defined by Whittaker (1972). The maps sketches to the left represent inventory levels of richness; those on the right show differentiation levels or changes in composition across gradients. Sampling unit sizes indicate approximate spatial dimension for each ecological scale

2.1.4.4 Demands for indicators of biodiversity

At the European level a project was initiated by the European Environment Agency (EEA) to define criteria and indicators of forest diversity. The object of the BEAR project is to "*formulate an integrated system of indicators of forest biodiversity that are applicable over a wide range of European biogeographic regions, and at regional, landscape and stand levels.* (Hansson 1998, p. 2). It is further stated (ibid. p. 4) that ideally an indicator should be:

• relevant to ecologically significant phenomena,

• able to differentiate between natural cycles/trends and those induced by anthropogenic stress,

• capable of providing continuous assessments over a wide range of stress,

• sufficiently sensitive to provide an early warning of changes,

• distributed over a broad geographical area, or otherwise widely applicable,

• easy and cost-effective to measure, collect, assay and/or calculate.

The work done in this project is presented in Larsson *et al* (2000) and partly at the project web site: http://www.algonet.se/~bear/. Among the main achievements of the project was the agreement on a common scheme of key factors of biodiversity applicable to European forests. These factors are divided into structural, compositional and functional factors. There are different factors for the structural (physical characteristics) and compositional (biological component) types at different spatial scales while the functional key factors, which relate to natural disturbances and human influence are the same across the scales (Larsson *et al* 2000, chapter 3.1). The main recommendations from the project are as follows:

1) to introduce the key factor approach in monitoring of forest biodiversity, and
2) to make a further division into different 'forest types for biodiversity assessment' in the reporting of the key factors (in all 33 different forest types were identified, they mostly correspond to the national classification schemes), and finally
3) to standardise indicators, methodology and protocols.

2.2 Use of landscape ecology concepts in forest and landscape assessment and monitoring

This section is intended to provide conceptual links between the types of information needed by forest managers at different levels and the tools provided by landscape ecology in terms of understanding processes and identifying and quantifying patterns that are of relevance. Important concepts in this context are habitat and habitat quality, structure and scale, which are introduced and reviewed in separate sub-sections.

2.2.1 Forest management information use and needs

Rural landscapes, of which forest and woodland are important parts, need protection and careful management. This is reflected in the principles outlined in the declarations from the European ministerial conferences in Helsinki 1993 (Forest) and Sofia 1995 (The Pan-European Biological and Landscape Diversity Strategy, PEBLDS (Smith and Gillet 2000))[1]. Sustainable management includes preservation of the structural and biological diversity of the agricultural and forested landscapes. In order to develop such management practices, an understanding of the landscapes spatial and temporal dynamics is needed, as stated by Stoms and Estes (1993), Turner *et al* (1993), European commission (1999).

According to Köhl and Päivinen (1996), remote sensing has the potential to act as an instrument to provide harmonised European forestry statistics. Lin and Päivinen (1999) list five user groups for forest information:

- International organisations, NGO's and environmental organisations
- National ministries
- Research and academic institutes
- Forest Industry
- Forest owners

These groups obviously have different information needs, which are only to a certain degree to be fulfilled using EO techniques, as illustrated in Lin and Päivinen (1999) and discussed by Köhl and Päivinen (1996) – refer Table 2.1, see also Table 2.5, on page 58.

[1] The full text of the strategy is available at http://www.strategyguide.org/fulltext.html (accessed 22/2 2004)

Function, type and level of information	Variable / data type
Forest protection	
Stand	Forest area (actual/potential ratio) Species Composition Structure (horizontal, vertical)
Site	Soil Vegetation types Topography (elevation, aspect, slope) Climate
Stability	Forest condition, Quality, health
Management	Value of protected infrastructure Water resources Objectives
Ecosystem / environment	**Variable / data type**
Carbon Cycle	Woody and herb biomass Soil organic matter Climate
Biodiversity - Ecosystem	Vegetation type Vegetation cover Pattern of vegetation Naturalness; management history, age, exotic species Management objectives Forest condition (rate of change)
Biodiversity - Species	Species composition (including rare species) Species richness (indicator species) Pattern (corridors / networks) Threats to sp. diversity; human disturbance, pollutant deposition, exotic species
Sustainability	Management objectives / history / planning and Land use change

Table 2.1 Forest management information needs as function of forest use, issues related to ecological functions – from Lin and Päivinen (1999), based on Kennedy and Luxmoore (1994).

Forest owners and the wood/paper industry will typically have an interest in maintaining resources for production, while environmental organisations and other NGO's are concerned with the biodiversity aspects. Thus, there is a challenge to define the correct level on which to monitor forest conditions and ecosystem parameters. Often, much information can be found and (perhaps just as important) changes be made in current practices at the Forest

Management Unit (FMU) level (Duinker 2000) – even if the size of a typical FMU will vary from country to country depending on tradition and geographical conditions.

A European Forest Information and Communication System (EFICS) has been proposed, (McCormick el al 1995) in which EO data would have a central role and contribute to monitoring and management of rural environment in general (Estreguil *et al* 2001, fig. 2). This project currently continues as the European Forest Information System (EFIS)[2]. Different NGO's and parts of the forest industry have during the last decade been working on developing various certification initiatives. These obviously need and do use some criteria for sustainability (Baharuddin 1996). Such an 'ecocertification' procedure focuses on the quality of forest management and thus requires a prior definition of the criteria and indicators to be used as a basis for the guarantees that buyers are expected to demand (Berthod 1998).

In Europe, 'old growth forest' is the closest we come to 'natural' forests, and special attention is given to them, as it has become clear that they have a higher number of species, many of which can only live only under the special conditions found there, (Diamond 1988, Davis *et al* 1990, Spies 1998). The particular information needs of such special forest types, that typically serve as important habitat for specialised species were discussed as part of the BEAR project (Hansson 1998, Larsson *et al* 2000).

2.2.2 A biotope approach: Habitat quality

There is a knowledge gap – a lack of precise 'laws of nature' between the levels of individual organism behaviour (movement) and the one of spatial dynamics of ecosystems that should be protected (Karieva and Wennergren 1995, Mann and Plummer 1995). As ecosystems and their dynamics *per se* can not be directly observed, they are either represented by some 'indicator species' or substituted by features such as habitat, guild, vegetation type and disturbance and

[2] Information on project status, data and software development at http://www.ec-gis.org/efis/ (accessed 24/2 2004)

guilds, which are then used to make possible assessments of biological diversity and naturalness. One promising approach is assessment of habitat quality, for which some approaches are presented in this section.

In terrestrial environments, plants form a structured environment that provides the habitat for the diversity of animal species (Franklin, 1995, May 1988). Forests are unique amongst ecosystems in the degree to which a certain type of vegetation, i.e., trees modify the environment, and so to say define the available niches. It follows that in forests the habitat quality or naturalness will vary according to management practices, ownership status and history, as human intervention in forests normally consists of planting and removing trees of certain species at certain times, often done in specific non-random spatial patterns (Franklin and Forman 1987, Borgesa and Hoganson 2000).

It is beyond doubt that the biological diversity of an area depends on environmental factors. The most basic of these are geological and climatic factors that follow geographic position and topography (Nichols *et al* 1998, Griffiths *et al* 1999). Since trees are able to alter the local microclimate, it follows that in forests and woodlands the diversity of the fauna depends strongly on the compositional, structural and developmental diversity of the vegetation (McCormick and Folving 1998). This in turn altered by faunal activity ranging from insects, harmful or just pollinating, to human settlement and forestry practices. Thus any quantification or description of biological diversity in forested areas will, to some degree, be a ‘snapshot’ of many dynamic feedback processes, and only sustained monitoring can reveal the dynamics and thus the functional diversity of the area. Another important factor determining how many species a given patch of land, landscape or island can host is its area. The use and reliability of area-species curves are described by e.g. May (1975) and later reviewed and discussed by Reid (1992) and recently by Lomolino (2001).

Diamond (1988) provides an interesting conceptual framework for assessing species diversity with the QQID concept: resource Quality and Quantity, Interaction and Dynamic processes. Quality is here to be understood as the habitat and resource factors that determine the 'number of niches' or habitat diversity. Quantity represents the availability of area and productivity. Interaction represents the complex issue of species interactions, be it predation or plant community successions, while finally D denotes the spatial dynamics including immigration, extinction and in the long-term speciation. Roughly, Quality and Quantity correspond to the structural and compositional aspects of biodiversity, while Interaction and Dynamics correspond to the functional aspect. Stoms and Estes (1993), in a review of what types of biological diversity that can be monitored, and at what scales, argue for QQID as a useful approach, although in practice the Structure-Composition-Function(Development) framework is generally used. Wilson (1992, chapter 10, pp.171-199) mentions some factors of importance for establishment and maintenance of biological diversity (species richness): climatic stability, energy availability and area extent, and Griffiths *et al* (1999, table 1) provide a list of factors thought to influence species richness, including habitat heterogeneity (diversity/complexity) and disturbance, where moderate disturbance is seen as positive for maintenance of high biodiversity - as competitive exclusion is thus prevented. These factors obviously have to be incorporated in sustainability assessment at landscape and regional levels – perhaps more than has previously been done in biodiversity assessments. Along the same lines, Angermeier and Karr (1994) recommend using the concept of 'biological integrity' in environmental and conservation policy, in order to rethink prevailing views of land stewardship.

The EU-level report to the CBD (European Commission 1998) mentions that for "Woodlands", there are several threats to biodiversity, amongst these are, listed by the sectors from which they stem:

- Agriculture: neglect of small woodlands,
- Forestry: Logging of old-growth forests, management intensification (and exotic species),
- Transport and energy: fragmentation and acidification,
- Tourism: forest fires,

i.e. largely threats that are eventually reflected in land cover changes, and thus can potentially be monitored using earth observation and GIS techniques (Firbank *et al* 1996, Gallego *et al* 2000, Mucher *et al* 2000).

EEA has established a European-wide nature information system (EUNIS)[3]. A central part of this system is habitat definition and classification, with the aim of providing a common and easily understood language for the description of all marine, freshwater and terrestrial habitats throughout Europe (Davies and Moss 2002). The EUNIS definition of habitat is “plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors (soil, climate, water availability and quality, and others), operating together at a particular scale.” For the purpose of categorising habitats *sampling sizes* ranging from $1m^2$ to $100m^2$ are found adequate, – at the smaller scale, still, microhabitats are found, at larger spatial scales the EUNIS habitats can be grouped to “habitat complexes” – of which estuaries are used as an example, but which also will be the case for many woodland types. The EUNIS habitat classification system has been used for designation of NATURA 2000 sites (European Commission 1999, Estreguil *et al* 2001, see also section 2.3.1.1). Thus, a prerequisite of this project is the ability to map *relevant habitats types* using RS data – at a spatial resolution that requires high-resolution input imagery, refer section 2.3.2.

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[3] The portal to background information and data is at http://eunis.eea.eu.int/index.jsp (accessed 24/2 2004)

2.2.3 Approaches to spatial structure in ecology – the landscape perspective

In landscape ecology, *landscapes* can be considered as *mosaics* of natural and managed *patches* that vary in size, shape and arrangement. The pattern that this arrangement forms is not only reflecting the processes going on, but also influencing a variety of ecological phenomena (Franklin and Forman 1987, Forman 1995, chapter 9). Combined with the notion of corridors, typically strips of land with a composition and structure that differ from the surrounding (Forman 1995, p. 145) and may enhance flow of resources and movement of plants and animals between patches, the patch-corridor-matrix model emerges. In this conceptual model patches are seen is habitable 'islands', where the distance (difficulty of movement) between them can be modified by the presence and quality of corridors (for instance hedgerows or strips of riparian forest, ref. Hanson *et al* (1990), Petit and Usher (1998), Brooker *et al* (1999). A similar concept, or just corridors with a 'negative' function is the one of barriers, ref. Robson (1996).

The theoretical foundation for these assumptions is to be found in the ecological sub-discipline of island biogeography, or the island theory by MacArthur and Wilson (1967), as referred by Delbaere and Gulinck (1994). Basic assumptions of this theory are that the number of species will be found in a spatial entity (island, forest, habitat type) will depend on

- the area of the entity as well as the
- number of ecological niches available (habitat quality) and the
- distance to and number of similar entities (other islands or mainland)

The underlying theories of island biogeography have been hard to test in practice (see e.g. Simberloff and Abele, 1976, Karieva and Wennergren 1995, Petit and Burel 1998), and Griffiths *et al* (2000) observe that only few studies have used explicitly landscape ecological approaches for biodiversity monitoring. Recent advances in computing capacity have however made it possible to model individuals' movements, breeding patterns and survival/extinction across landscapes and the consequences for species under consideration (Green 1994, Verboom 1996, Firbank *et al* 1996, Petit and Burel 1998). The use of island biogeography concepts and species-area relations in design of protected areas has led to the discussion about "few large or several small" wildlife preserves – sometimes referred to as the SLOSS dilemma, see e.g. Simberloff and Abele (1976), Andren (1994), Haines-Young and Chopping (1996).

Meta-population theory is a further development and sophistication of the Island Biogeography approach (Wu and Vancat 1995), and appears to be the best model for understanding species dynamics in the context of landscapes made up of habitats that are distributed as discrete patches (Hanski 1998, Hanski and Ovaskainen 2000). This theory describes species and guilds of species as being in a dynamic equilibrium or metastability within the landscapes they inhabit (Wu and Loucks 1995); where local extinctions are compensated by immigrations from nearby patches (Hanski 1999, chapter 8).

In cases where entire landscapes of a given scale have been distinguished and mapped, the LUC map itself provides a visual estimate of ecosystem type richness and homogeneity/heterogeneity (evenness). A clear and useful introduction to the links between landscape structure and ecological processes, with the intention of applying quantitative, spatial methods for analysis are provided by McGarrigal and Marks (1994) in the manual and background document for the Fragstats software (for description, see appendix 3), but see also Noss(1990), Hansson and Angelstam(1991), Kupfer (1995), Dreschler and Wissel (1998). Table 2.2 represents an attempt to outline the various concepts of diversity and the spatial scales at which they operate or at which they can be observed, compare also Figure 2.2.

Concept for diversity mapping/monitoring	Type	Area extent (scale) for observation/ mapping	Objects
Bio diversity	Compositional (Genetic)	1m^2 - 1 ha	All plants, animals
Habitat diversity	Compositional	1 ha - 100 km^2	Ecosystems (internal structure) Forest internal structure
Habitat structure and (biotope) distribution	Compositional (Populations)	1 ha - 100 km^2	Ecosystems Forest internal structure
Landscape diversity	Structural	1 ha - 100 km^2	Tree species, crops, Other land cover types
Forest or landscape Structure	Structural	100 – 10000 km^2 (possibly continental scale)	Distribution of Forest stands / patches Forests (outline/shape)
Forest diversity	Compositional & Structural	100 km^2 – 1000000 km^2 (entire continent)	Broad land cover classes

Table 2.2. Summary of concepts for diversity mapping / modelling - the area extent is somewhat arbitrary and is based on currently available satellite data, partly based on table 1 and 2 in (Stoms and Estes 1993).

According to Forman and Godron (1986), structure analysis in Landscape Ecology is defined as setting the distribution of energy, materials and species in relation to sizes, shapes, numbers, kinds and configurations of landscape elements or ecosystems. **The *structural* component of forest diversity thus, in this Landscape Ecology-context, refers to the spatial pattern of the forest blocks and patches that are identified in a forested area** **(**McCormick and Folving 1998), see also Figure 2.1, on page 23 and section 2.2**.** Structure is the one component of forest diversity that can most easily be analysed using Remote Sensing and GIS applications (McGarigal and McComb 1995, Ricotta 2000). Furthermore, since it is assumed, that the structural diversity of forested landscapes is an indicator of biological diversity in general, assumptions have been made that statistical relations can be found at the landscape level between some spatial metrics and e.g. species richness – and thus the comparison of structural diversity of different areas with objective methods is made possible, see e.g. Turner (1990), Wrbka *et al* (1998), Jensen at al (1998), Häusler *et al* (2000).

2.2.4 Scale issues in landscape ecology

Scale can be defined as **the resolution at which patterns are measured, perceived or represented** (Morrison and Hall 2002), in landscape ecology scale primarily refers to grain (resolution) and extent in space and/or time (Wu and Qi 2000). In cartography, scale denotes the ratio between pairs of point on the map and distance as measured between the corresponding pair of points on the Earth's surface (Goodchild and Quattrochi 1997, p. 2). This is somehow similar to the way the term is used when dealing with data in vector format, then scale normally denotes the cartographic scale at which it will be feasible to display the data or at which to print them as a map – "scale" is actually used to describe the accuracy of the data (Goodchild and Quattrochi 1997, p. 4). The concept of scale is also related to sampling issues, as in biology/ecology (Carlile *et al* 1989, Noss 1990, Bowers and Dooley 1999) and soil science (Oliver and Webster 1986).

The variogram, sometimes mentioned as the "semivariogram", is a tool that has been proposed and commonly used for description of spatial structure and charcteristic scale, where variance between point measurements is plotted against distance (Curran 1988). According to Curran and Atkinson (1998), one may use variograms not only to estimate summary statistics such as the dispersion or sample variance, but also to design optimal sampling strategies before the actual survey takes place.

In Landscape Ecology, the concept of scale is closely related to the concepts of grain and extent. Grain here means the spatial and temporal resolution of observations; the smallest resolvable unit of study (Morrison and Hall 2002), technically often identical to the size of the basic/atomic picture elements – in Remote Sensing terms referred to as the pixel size. This is in line with the notion of grain as the resolution of an image or the minimum area perceived as distinct by an organism (Farina, 1998, in Dobbertin 1998). Grain size can also be seen as an inherent property of a landscape: it then is defined as the average, and the variability in, diameter or area of the landscape elements present (Forman and Godron, 1986, p. 216).

Extent is the area over which observations are made and the duration of those observations. (Morrison and Hall 2002), often used in the meaning of the geographical size of map or an image scene under analysis. In an ecological sense, extent is the coarsest scale of heterogeneity, or upper threshold of heterogeneity, to which an organism responds (McGarigal and Marks 1995, p. 5).

The term 'scale' is often used as synonymous with 'level', ie. 'the landscape scale', or even in the resolution domain with 'grain', ie. 'coarse-scale' pattern. Throughout this thesis, I will try to avoid confusion, using scale as describing only *spatial scale*, thus more or less synonymous with *resolution*. It follows from this, that a central problem of this thesis, the *scaling* issue is actually an investigation of the behaviour of spatial metrics (see section 2.3.1.3), for the same landscape imaged/mapped at different spatial resolutions, corresponding to different grain sizes and extents of the representations.

2.2.5 Application of landscape ecology in landscape monitoring

Before applying land cover information derived from remote sensing or land cover data in general for the assessments of sustainability and biodiversity, it is important to know the causative relations between landscape structure and biodiversity. For instance, it is widely recognised that in natural systems, the number of species are in dynamic equilibrium, local extinctions being matched by immigration (Saunders *et al* 1991, Hanski 1998) – but how should a natural system, within which these processes are taking place, be delimited, the administrative borders relevant for land managers might not fit with ecological units or regions. Furthermore, if we look only at the forested part of landscapes, is it then relevant to apply landscape ecological analysis to these areas in isolation from the surrounding agricultural and urban areas – which in Europe are never far away? Saunders *et al* (1991) claim that research in "Island biogeography" has provided only little valuable information to forest managers and decision makers. On the other hand there is no doubt that optimised forest management can contribute significantly to the overall biological diversity of landscapes, though there is also no doubt that this diversity can be further enhanced by "good", environmentally friendly or even "organic" agricultural practices (Kutzenberger and Wrbka (1992), van Mansvelt and van der Lubbe (1998)).

O'Neill *et al* (1997), in accordance with the recommendations given by Noss (1990), outlines a useful approach for analysing landscapes in relation to habitat requirements of a given species. Consider a "window" the size of an organism's home range. Within the window are found a variety of habitat requirements, such as vegetation mixture, edge, and available water. By placing the window over a corner of the landscape map, it is possible to determine whether the land covers that are within the window meet all habitat requirements. The window could then be moved systematically over the map, yielding an overall indicator of the status of the landscape for this organism. In digital image processing terms, such a moving window is similar to a filter kernel, this facilitates implementation in GIS and software for processing of Remote Sensing data. A suite of windows for individual species, guilds, or populations could be designed by adjusting the resolution of the data, the size of the home range window, and the habitat requirements. This approach provides a simple indicator of the impact on wildlife of a change in landscape pattern. Häusler *et al* (2000) demonstrated an implementation of moving-windows for assessment of structural diversity of European forests and change detection (see ibid. fig. 9-11), and concluded that it was possible to make local and regional scale comparison of forest (tree) species diversity, making possible also detection of temporal trends. A functioning system however, must be flexible regarding species and their respective "ranges" of occupation and movement.

Wrbka *et al* (1998) describe different aspects of the Austrian SINUS (Study of Structural Features of Landscape Ecology as Indicators for Sustainable Land Use) project. Landscape structure was characterised using a hierarchical theory approach, focusing on the relation between pattern and intensity of land use. Field work was done in 140 quadrates of 1*1 km, which were also mapped from aerial photos. The sampling design for the selection of the test areas was a 'stratified random' approach. A similar approach has been used for measuring the 'Hemeroby' ('cultural influence' or lack of naturalness) of Austria's forests (Grabherr *et al* 1995). These approaches seem to assume that the feature of Hemeroby or 'un-naturalness' for a landscape is the directly opposite of 'sustainable', as also seen from the nomenclature used in Table 2.3, something professional foresters would surely not agree to. Steinhardt at al (1999) proposed a Hemeroby index for landscape monitoring, and demonstrated the application using land cover data from eastern Germany from 1944 and 1989 respectively, finding significant changes. Brentrup *et al* (2002) use the Hemeroby concept for Life Cycle Impact analysis of LUC, through definition of a Naturalness Degradation Potential (NDP) applied corresponding to different degrees of Hemeroby, which again can be assigned to land use classes in map data such as CLC.

Degree of Hemeroby	**Degree of Naturalness**	**Human Impact**
Ahemerobe	Natural	None
Oligohemerobe	Close to natural	Limited removal of wood, pastoralism, limited emissions from through air and water
Mesohemerobe	Semi-natural	Clearing and occasional ploughing, clear cut, occasional slight fertilisation
β-euhemerobe	Relatively far from natural	Application of fertilisers, lime and pesticides, ditch drainage
α-euhemerobe	Far from natural	Deep ploughing, application of pesticides and intensive fertilisation
Polyeuhemerobe	Strange to natural	Covering of biotope with external material
Metahemerobe	Artificial	Total

Table 2.3 Levels of Hemeroby for description and evaluation of biotopes, from Steinhardt *et al* (1999).

However, one must be aware that the spatial arrangement of landscape elements cannot explain everything happening in forest landscapes, neither in terms of mass- and energy-flows nor absence or presence of species. Also the total forest area of a country or region and the physical conditions determine forest structure, function – and diversity. In countries where forests only occupy a few percent of the surface area, they play a proportionally larger ecological role, as they host a larger number of species than agricultural land - and often function as a refuge, corridor or feeding area for species normally dwelling somewhere else (Oswald 1996, European Commission 1998, 1999). These countries are among the most densely populated, and thus where we can expect to find the strongest pressures on the environment and biodiversity in general. In such countries, forest cover is then found to be already fragmented and is continuously being threatened by expanding transport networks, urban sprawl and intensification of agricultural practices. In countries with high forest covers, in Europe typically found in the Boreal and the Alpine zone, the structure and naturalness of the forest itself is of the greatest interest, such as variance between and shape of patches, managed as well as natural.

Research in densely forested countries tends to have focused on management applications such as forest mapping and timber volume estimates, but fortunately the methods developed for these ends can also be used for land cover mapping. What is now needed in terms of monitoring for assessment of sustainability (mostly from an ecological point of view) of forest and land management is methods and systems that for a given selection of land cover data can answer questions like:

- Does this landscape have a sound structure (promoting/inhibiting natural processes)?
- How far is the structure of this landscape from its natural state?
- Has it become better or worse during a certain period?

The answers (in terms of indicator values) should allow decision makers to evaluate whether the principles and criteria for sustainable land use are being followed and fulfilled. The biggest challenge in application of landscape ecological concepts is now to link the various levels of diversity with spatial scale for practical applications (Kareiva and Wennergren 1995, Firbank *et al* 1996, Blaschke and Petch 1999), thus finding methods to quantify the concepts shown in Figure 2.2 - or as was one of the initial objectives of this project: find surrogate parameters, derived from EO data, that correlate with (measures of) the biological diversity in the forested landscape.

2.3 Spatial approaches to analysis of structure and diversity at landscape level

This section will present some promising approaches to spatial analysis of ecological conditions and processes, especially biological diversity as expressed through species richness – and provide an assessment of their applicability for larger-area monitoring. The methods presented and discussed in the following sections are all based on the fact that land cover maps at various spatial and thematic resolutions can be derived from Earth Observation data (section 2.3.2), and the observation that the precision of these is mainly a technical problem and dependent on available data sources – and not least cost (and to a lesser extent time) considerations.

2.3.1 Use of Geographical Information in environmental management

A Geographical Information System (GIS) is a suite of computer software used for the capture, storage, manipulation, display and analysis of spatial data, describing physical properties of the geographical world (Sparks *et al* 1994, Elmasri and Navathe 2000, p. 891), developed for a particular set of purposes (Burrough 1986, p. 6). In studies on the ecology of separate landscape components, typically carried out by organisations such as research councils, government bodies, conservation groups and university departments – GIS has helped integrate the findings and making better use of the results. Meanwhile GISs are increasingly being used in forest mapping and for organisation of and data management in National Forest Inventories (Nel *et al* 1994, Pitt *et al* 1997, Blaschke 1999), as well at international level (Lund and Iremonger 1998) and have potential for use in monitoring of deforestation (Skole and Tucker 1993, Mertens and Lambin 1997) or for verification of national commitments to the Kyoto protocol (Goodenough *et al* 1998). According to Dykstra (1997), GIS represents a tremendously powerful tool that has the potential to enhance greatly the capabilities of forestry organisations in tactical planning – although he warns that "*GIS will be useful for forestry analysis only if the foresters use it*".

2.3.1.1 Gap Analysis

An approach for the analysis of the effects of land use and land cover changes for larger regions, typically loss of natural habitats, is the so called "Gap Analysis", an approach to "optimise" networks of natural and protected areas. In this context, Remote Sensing has been seen as a useful tool (Davis *et al* 1990, Scott *et al* 1993). Forman (1995, p. 312), explains how, in Gap Analysis a map of species-rich spots is superimposed onto a map of existing protected areas, and then the difference between the maps indicates the areas or 'gaps' that need protection based on species rich sites. Gap Analysis can thus be seen as a way of combating habitat fragmentation, or at least as a way of finding ways to relieve the effects of processes that lead to habitat loss such as (sub)urbanisation or intensification of agriculture and forestry. Monmonier (1994) raises the issue of weighting species against each other for their protection value, and points to the limitations of regional Gap Analysis when data availability is limited by for instance state borders. Seen from a management point of view Geographical Information Systems show great promise, perhaps most consistently demonstrated in the American 'Gap Analysis Project' (Scott *et al*. 1993, Jennings 2000)[4].

Gap Analysis is normally carried out for large areas of natural land cover, so that this approach is probably not directly transferable to the cultural landscapes of Europe, where natural and uninhabited areas are scarce and limited by pressure from human activity and population density. Still relevant, however, is the multi-layer approach to identify, if not gaps, then at least areas with over- and under-representation of species relative to what is expected from environmental and topographic (and geological etc.) parameters.

The European Commission (1999) introduced a common framework for preserving biodiversity within the "Natura 2000" network, stressing the need for urgent measures to be taken. It refers directly to the obligations following from the birds and habitat directives – and

[4] The US National GAP web site at: http://www.gap.uidaho.edu/ (accessed 21/2 2004)

from the United Nations Conference on Environment and Development (UNCED, the 1992 "Earth Summit" in Rio de Janeiro).

2.3.1.2 Modelling ecological processes in a landscape framework

There are several reasons that it is practical to use GIS for modelling ecological processes with a spatial aspect. Firstly, it allows establishment of general relations between the structure of certain landscapes or some special features within them and the potential for certain species to maintain a population there (Herr and Queen 1993, Sparks *et al* 1994, Westervelt and Hopkins 1999). Secondly, it easily allows testing of models by verification using geo-referenced field data (Davis *et al* 1990, Verboom 1996, Scott and Jennings 1998). Kareiva and Wennergren (1995) reviewed current research in the field and identified two types of ecological models for population dynamics:

1) occupied - un-occupied patches
2) dynamics within patches

They found that given the practical aspect of these investigations, it was time to ask whether any general principles were emerging from the explosion of spatially explicit theories. For instance, cellular automata models suggest a stabilizing effect only on the scale of landscapes orders of magnitude larger than the lifetime dispersal of the organisms under study. Finally, GIS naturally form an integrated part of the landscape assessment projects mentioned in section 2.3.2, and thus modelling of the historical processes that have shaped the current landscape or prediction of the effects (e.g. on biodiversity) of future developments of the landscape structure can easily be integrated in GIS analyses (Vasconcelos *et al* 1993, With 1997, Borgesa and Hoganson 2000, Petit and Lambin 2001). The role of RS data in this context is to provide the structural and compositional framework for models of environmental functions. Also the visual consequences of landscape modifications, which can be very important, can now be modelled using GIS techniques (Weidenbach and Proebstl 1998, Hunziker and Kienast 1999).

2.3.1.3 Calculating spatial metrics

A spatial or landscape metric is a numerical value describing a property of a map or an image, or an object contained therein, utilising the spatial heterogeneity that is ubiquitous in nature across all scales (Wu *et al* 2000), in line with Pickett and Cadenasso's (1995) recommendation of using spatial heterogeneity in ecology to perform valuable and predictive functions rather than excluding it as a troublesome source of error.

In this context it is assumed that maps or images represent landscapes, as when McGarigal and Holmes (2000) use the term 'landscape pattern metrics'. Fortin (1999), in Leitão and Ahern (2002), specifies the difference from spatial statistics, which are tools that estimate the spatial structure of the values of a sampled variable, while landscape metrics area tools that characterise the geometric and spatial properties of a patch or a mosaic of patches. McCormick and Folving (1998) use the concept of 'landscape structural parameters', thereby implying that differences in these metrics across a landscape or between landscape units will reflect 'structural diversity'. Fry (1996) provides some clear definitions of the goals and methods of landscape ecology, with a relevant discussion of how and when spatial metrics can be applied. Fry (1996) further argues that landscape metrics are needed in order to investigate the role of landscape in determining ecological processes, and compares these metrics to the parameter that we lack to place on the x-axis of a graph of landscape versus biodiversity. Spatial metrics can be added *ad infintum*, many of them being redundant and, see e.g. Riitters *et al* (1995). The capacity to generate information about spatial properties of landscapes generally exceeds our ability to apply or interpret such information ecologically (Griffiths *et al* 2000), and according to McGarrigal and Marks (1994), the task is not so much to define metrics, but rather to find out how to interpret them. That is also what Häusler *et al* (2000) conclude from a study, where spatial metrics are derived semi-automatically from EO data and applied in forest monitoring. One of the challenges to environmental scientists ranging from entomologists to physical geographers is thus to find ways of combining models based on individual or sub-population behaviour with quantitative metrics of landscape structure.

In this thesis 'spatial metrics' is used to mean quantitative description of spatial structure as it appears in land cover maps. These metrics can be simple, geometric information that can be obtained from most GIS programs, such as patch area or edge length or more complicated metrics defined from information theory and/or landscape ecology, where special software is required for their calculation.

Spatial metrics can be calculated on at least three levels (McGarigal and Marks 1994):

- Patch: a spatially and functionally coherent object (ideally a forest stand or biotope)
- Class: the set of (functionally) similar objects in the scene/on the map, typically the same as a land cover class, vegetation or habitat type.
- Landscape: the entire image/scene, possibly excluding a class defined as 'background'. Metrics of compositional diversity can only be calculated at the landscape level.

Spatial metrics can be seen as belonging to one of the types listed below and illustrated in Figure 2.3 (McGarigal and Marks 1994, Häusler *et al* 2000):

- **Area** metrics describe the extent of patches, classes or the total landscape. This can be done in absolute values, as mean values or in percentages.
- **Edge** metrics describe the amount of occurring edges between patches or classes. This is done by perimeter calculations of each patch. In that way, these indices can give information about the spatial variance of an area. A high number of edges can indicate variable ecological conditions, which is e.g. necessary for the occurrence of specific species. Low edge frequencies typically indicate monotonous conditions for the subject/species of interest. It is possible to assign different weights to certain edge-types, e.g. if forest-agriculture edges are considered more drastic than forest-natural grassland edges (McGarigal and Marks 1994, p. 30 ff.).
- **Shape** metrics are based on perimeter-area relationships of the patches, where e.g. the perimeter of a patch is compared to the perimeter of a square with the same area (such as done by Frohn (1998, p. 17)). High values may indicate the occurrence of many patches

with complex and convoluted shapes, while low values represent the dominance of simple geometric shapes, like rectangular or circular shapes. Fractal metrics are also shape metrics, since they can the calculated from information of patch area and perimeter, although in this case the value characterising the landscape is based on a regression between single patches surface area and their perimeters (Olsen *et al* 1993, see also this reference for definitions and discussions of alternative fractal metrics).

- **Core Area** metrics. Core area is defined as the area within a patch beyond certain edge distance or buffer width. Core area metrics compute statistics regarding the inner/central parts of patches in relation to the total patches. These metrics can give information about habitat quality for certain species. For instance, some species might not be able to exist within narrow forests like riparian forests, even if the total forest area was sufficient (following simple species area relations).
- **Patch** metrics describe the total number of patches and their relative proportion (if more classes are present) in a given area.
- **Nearest-Neighbour** metrics are based on the distances from patches to the nearest neighbouring patch of the same type/class. These indices are calculated by using the minimum distance measured as edge to edge distance from one patch to the nearest neighbouring patch of the same class type. They thus quantify landscape configuration. These measures can be used for describing migration possibilities of species or species interaction of separated populations. This type of indices clearly describes the spatial configuration of landscapes and of the different land cover classes.
- **Diversity** metrics measure landscape composition and are function of the richness and evenness of the patch types in the landscape. The simplest diversity metric is the one of *richness* i.e. the number of different species or land cover types found within a certain area, but as illustrated in Figure 2.3 on page 48 where the three example landscapes (in the bottom right) have the same number of classes but do become more diverse from right to left, this number can be misleading, or at least not sufficient. However, more advanced metrics do exist. Most of these diversity measures are originally developed for

information theory, such as the Shannon-Wiener index, (ref. O'Neill *et al* 1988) or for biology with no spatial dimension in mind (Simpson 1949). Dependent on the probability of the occurrence of all cover types these is a measures indicate to which degree all cover types are evenly proportioned in terms of their spatial extent. Vice versa, this index measures the extent to which one or a few class types dominate the landscape. A prerequisite to meaningful application of diversity measures although is the existence of a number of land cover types that are well defined, functionally and physically separated (also spectrally/texturally), preferably equidistant - as far as it is possible to measure distance in terms of functionality.

- **Contagion** and **Juxtaposition** metrics are calculated using the actual rate of adjacency of each occurring class type with all other class types. The resulting values express the probability of adjacency of different class types. Herewith, contagion can give an idea about the extent of aggregation or clumping of patches. High values indicate big continuous areas, while small values represent many small, dissected areas. On the other hand, juxtaposition and interspersion metrics indicate how 'well mixed' the patches in a landscape or the pixels in an image of different types are – for example, the version implemented by McGarigal and Marks (1994, p. 58) in the Fragstats software is based on "patch" adjacencies, each patch is evaluated for adjacency with all other patch types. This means that, while the values of the juxtaposition metric in the example in Figure 2.3, on page 48, will increase from left to right, while values of the contagion metric will decrease. Various modified versions of the contagion metric have been proposed for use in description and quantification of forest fragmentation (O'Neill *et al* 1988, Li and Reynolds 1993). For a discussion of the usefulness of this and similar 'advanced' metrics, see Frohn (1998).

Amongst the shape metrics are indices of 'fractality' of the patches, following the definition by Mandelbrot (1967), and thus the assumption of self-similarity, i.e. that pattern observed at one level are repeated at higher and lower levels or larger and smaller spatial scales. It is generally believed that high fractal values reflect natural conditions (de Cola 1989, Hargis and Bissonette 1998) while if the values are lower, the pattern and thus the landscape must be assumed to be artificial/un-natural. The self-similarity if "real" fractal patterns should make them insensitive to scaling effects, but Frohn (1998) found that there is an intimate relation between scaling properties and fractal properties of land cover classes and patches.

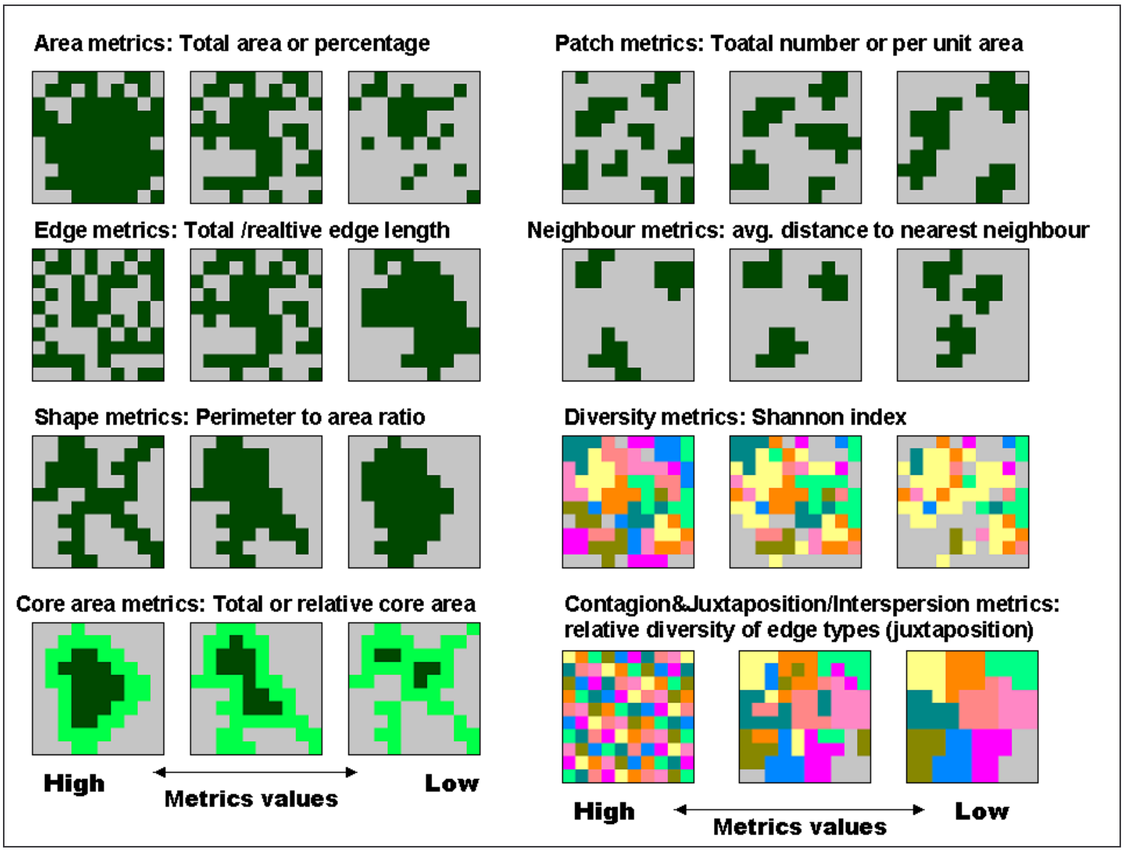

Figure 2.3 Examples of the eight main types of spatial metrics defined by McGarigal and Marks (1994), partly after Häusler *et al* (2000), fig. 8.

2.3.1.4 Perspectives for the use of spatial/geo-referenced data for environmental analysis

Spatial metrics can function as indicators that can be compared between landscapes or watersheds – preferably using "natural" instead of administrative units. It is assumed that differences in the values of these metrics reflect real differences in landscape quality/naturalness/usefulness as habitat. Additional information could possibly be gained by calculating spatial indices at different resolutions (with grain sizes equivalent to the different sensors) and displaying them together - or using them as a N-dimensional data set, as demonstrated by Riitters *et al* (1995).

When ‘returned’ into a GIS, in geo-referenced format, spatial metrics can serve as indicators of structural diversity and are therefore potentially a landscape management tool. Geographical Information Systems in themselves are useful tools for linking and visualising geo-referenced data, area-covering maps and statistical data computed for administrative units. O’Neill *et al* (1999) state that: *“The combination of remote imagery data, geographic information system software and landscape ecology theory provides a unique basis for monitoring and assessing large-scale ecological systems.”* This claim can be justified by be results from projects such as the national GAP project in USA (Scott and Jennings 1998, Stoms 2000) and the Britsh ‘countryside survey’ (Bunce *et al* 1996, Brandt *et al* 2002), with assessments of the usefulness at pan-European scales in EU-DG AGRI *et al* (2000) and at global scale by Riitters *et al* (2000). An example of application of selected spatial metrics for provision of base-line information on the structure of forests within a natural reserve area is provided by Luque (2000), who chose metrics of Diversity, Dominance and Contagion to represent the diversity of forests and Fractal Dimension to represent spatial pattern (complexity) at two thematic levels: separate forest classes and forest-non-forest. Temporal analyses of changes in forest structure were then performed, based on a series of satellite images covering a period from 1972 to 1991. Griffiths *et al* (2000) however warn that the land cover map for the countryside survey, produced from Landsat TM data and including 25 target classes is *not* a map of biotopes - as it can be shown that the level of detail is much lower that in the Corine biotopes classification (see end of section 2.3.2.2).

After the listing of potential spatial metrics, it is possibly worth recalling Forman’s (1995) demands for an ideal shape index, that should :

- be easy to calculate,
- work over whole domain of interest,
- unambiguously and quantitatively differentiate between different shapes, and finally
- permit the shape to be drawn based on knowledge of the index number alone.

Unfortunately he had to conclude that such an index could not exist. Which means that the actual challenge is to find a combination of spatial metrics and possibly other geographical information, that together provide a useful description of forested landscapes.

2.3.2 Uses of Earth Observation techniques in landscape analysis

The term Earth Observation (EO) is used here, because it is typically directed more towards environmental management applications, than Remote Sensing (RS), a term which can be used to describe the use of satellite imagery anywhere (for instance on other planets). At the same time the EO concept includes airborne photography and scanner data, but the use of satellite data has two main advantages over airborne data. First of all it makes possible a regional approach, where the area of investigation only depends on the extent of the areas from where data are available. Secondly it makes it possible to directly assess changes over large areas over time, such as monitoring of deforestation or afforestation, although it must be kept in mind that also repeated airborne data acquisition can be a relevant tool.

2.3.2.1 Potential uses of Earth Observation data for landscape analysis

It is generally agreed that effective mapping and monitoring can be carried out using optical satellite data of high to medium spatial resolution (around 20 to 200m ground resolution cells), such as for Landsat TM and SPOT data as described by Cohen and Spies (1992), Häme *et al* (1999 and 2000), Häusler *et al* (2000), McCormick and Folving (1998), Pitt *et al* (1997) and many others. The use of Landsat MSS data is described by Hall *et al* (1991), Ripple (1994) (in combination with NOAA AVHRR data), Mayaux and Lambin (1997), and of WiFS data by Häme *et al* (1999) and Häusler *et al* (2000). A typical approach is to 'calibrate' or train large area classifiers on low or medium resolution data using the high resolution data as a sort of 'ground truth' (Mayaux and Lambin 1995, Häme *et al* 1999).

The use of radar (microwave) sensors still have some way to go for operational classification purposes (Kasischke *et al* 1997), but the use of multi-polarised channels seems promising

(Saatchi and Moghaddam 2000, Corr *et al* 2003). It should be noted that data from the Shuttle Radar Topography Mission (SRTM) that was accomplished in February 2000 is currently becoming available in the form of high resolution (1 to 3 second resolution, corresponding to 30-100m cell size) topographic/elevation models and probably also useful land cover information[5] (Rabus *et al* 2003).

It is widely recognised that maps of habitat diversity as derived from remote sensing data can potentially provide powerful indirect indicators of species diversity (Noss 1990, Bell *et al*. 1991, ref. in Stork *et al*. 1997). The review in the previous section (2.3.1) shows that it makes some sense to assign structural and ecological meaning to selected spatial indices, as derived for landscape ecological applications in RS and GIS, and to calculate these for subsets of large land-cover maps. Furthermore, the resulting thematic layers can be applied directly as map information in management of structural and (thus) biological diversity – thereby ensuring multiple uses of the image data, which can otherwise be expensive.

An obvious potential pitfall in the application of spatial data for assessment of biological diversity is that no standardised way exists in which to map and analyse land cover. Using terms from a more "physical" approach to remote sensing, a list of factors influencing the results of spatial analysis of land cover maps include (Duggin and Robinove 1990):

- Thematic resolution (i.e. the number of vegetation or land cover classes) The thematic resolution is of great importance for first of all the edge-and diversity metrics, a scheme with more classes automatically will produce maps with more edges (borders between patch types) and a higher number of different classes within a (sub-) landscape.
- Spatial resolution (i.e. precision of vector data and grain size of raster data), closely linked with scaling issues, as discussed in section 2.3.3.3).

[5] Information on the mission and the results at http://www.jpl.nasa.gov/srtm/ Data are located at ftp://edcsgs9.cr.usgs.gov/pub/data/srtm/ (February 2004 3-sec data were available only for the Americas and Eurasia)

- Temporal resolution (i.e. how much land cover changes from season to season and year to year). It is of importance to know whether a change in land cover as appearing on EO derived maps reflect real changes or e.g. sensor degradation or different weather conditions at the time of acquisition. Even more, large areas land cover maps like the CORINE are mosaics of classifications done on imagery from different years, the current CORINE database having differences of up to ten years between neighbouring countries.
- Sensor system and image processing (more or less refined) influence, there can be many causes such as point spread function of the radiometer, robustness of classification algorithms, pre- or post processing filtering of image data.

Other factors than spatial pattern are of great importance for real and potential bio diversity, such as available energy for photosynthesis and actual evapo-transpiration, as described in section 2.2.2 on Habitat quality. Values of these can be derived from remote sensing data, typically from low-resolution sensors such as the AVHRR instrument on the NOAA satellites (Cihlar *et al* 1997), or the MODIS instrument on the Terra satellite (Moody and Woodcock 1994, Running *et al* 1994). The applications of both instruments for description of ecosystems using remote sensing are discussed in Justice and Townshend (1994) and Waring and Running (1998, chapter 7). See (Goward 1989, Wulder 1998) for reviews of ‘bioclimatological’ i.e. vegetation applications, and Roughgarden *et al* (1991) for a general discussion of the (potential) role of Remote Sensing in ecology.

2.3.2.2 Use of remote sensing for forest and land cover mapping

The process of getting from “raw” satellite data to land cover “maps” is by now well established in applied Remote Sensing (Cihlar and Jansen 2001), and includes such steps as geo-referencing, calculation of spectral indices, supervised or unsupervised classification (Campbell 1996 chapters 10 and 11), clean-up operations such as low-pass filtering or merging of classes (McCormick 1996, Banko and Kusche 2000) and export to GIS data formats for further analysis (Wilkinson 1996).

For some years ecologists and foresters have recognised, that remote sensing techniques can deliver useful data for land cover mapping and forest inventories. Blackburn and Milton (1996), McCormick *et al* (1995), Ekstrand (1994), Cohen and Spies (1992) provide specific examples of how parameters relevant to forest management and ecology are derived, and Pitt *et al* (1997) and Innes and Koch (1998) review the state-of-the-art within the field of "forestry remote sensing" with special focus on ecological applications. Wulder (1998) discusses the 'trade off' that must be made between cost and detail (see table 3, p. 455) when choosing between air photography and satellite images. He also compares spectral vs. spatial techniques (such as textural metrics) and finds the former more mature and better tested. Païvinen and Köhl (1997) provide an assessment of feasibility of remote sensing in forest applications for harmonisation of forest data. A similar approach was taken by the BEAR-project (Larsson *et al* 2000, see also section 2.1.4.4), in defining key factors of forest diversity, although with less focus on satellite data. Some research has focused on whether and how natural forest can be distinguished from managed forest using remote sensing techniques (Franklin and McDermind 1993, Nel *et al* 1994). Häme *et al* (2000) present a new method for the estimation of forest variables at sub-pixel level. In this study, problems associated with using conventional image classification techniques when pixels do not belong exclusively to one distinct ground class are addressed, and an approach presented for overcoming this by applying a probability based classification method. Of special interest for forest monitoring is the completion of a 'Forest Probability Map' covering the entire European continent, based on a mosaic of NOAA AVHRR images, with original pixel size 1*1 km. This employed an approach similar to the one used by Foody *et al* (1999), although at a very different spatial scale, as in that study, airborne scanner data with a resolution around 4 m were used to identify and characterise forest gaps originating from wind throw.

An interesting approach to solving the problem of what spatial entities to use as the basal mapping units for assessing diversity is to use catchment areas, also referred to as watersheds. These have the advantage of being functional natural units, that can be delineated from digital terrain models or existing maps and analysed using a hierarchical approach, ranking the watersheds from headwaters (highest altitude, often forest covered) to the uplands of large rivers. For an approach examining landscape patterns at catchment level see Hunsaker *et al* (1996), and Tinker *et al* (1998). In the latter study, a number of different spatial metrics were calculated from Fragstats software (McGarigal and Marks 1994). They were subsequently reclassified into uncorrelated components, using principal components analysis (PCA), in an attempt to find few significant parameters describing the environmental state of the watersheds, in this case especially the process of forest fragmentation.

The European Environment Agency (EEA) is carrying out a continental level land-cover mapping project, through the Topic Centre for Land Cover (ETC/LC)[6]. This 'Co-ordination of Information on the Environment' (CORINE) land cover database has been created mainly through manual interpretation of satellite imagery, mostly from the Landsat TM and SPOT XS sensors. The CORINE land cover (CLC) dataset has a nomenclature of 44 land cover classes, organised hierarchically at three levels. The first, highest level has 5 classes and corresponds to main categories of LUC: artificial areas, agricultural land, forest and semi-natural areas, wetlands, water surfaces (EU-DG AGRI *et al* 2000, table 1, p.4); the second level has 15 classes that cover physical and physiognomic entities in more detail (urban zones, forest, lakes etc.); the third level is composed of all 44 classes, including only three forest classes: coniferous, deciduous and mixed, but other classes such as "agro-forest areas" and "woodland-shrub" might be included in analyses of forest structure at landscape level, depending on the objectives. CLC data are available in vector and raster format, the raster data as 100*100 or 250*250 meter cells (note that these data are 'created' by sampling the vector data). CLC data have the potential to become powerful tools for monitoring the sustainability of land use in Europe, especially in combination with the CORINE biotopes database, that is being assembled by EEA as part of the NATure/LANd Cover information package

[6] The status of the project can be followed at: http://terrestrial.eionet.eu.int/CLC2000 (accessed 22/2 2004)

(NATLAN). With these, it should be possible to compare landscape metric over large areas (Jongman 1994, Gallego *et al* 2000), however the accuracy still has to be evaluated – as it seems to vary form country to country (Dubs 1999).

2.3.2.3 Approaches to use of remote sensing for forest monitoring

Forestry applications have hardly been considered so far in the design of remote-sensing projects and sensor-configurations. This complicates the process from data acquisition or changes in the management of spatial data (typically substitution of traditional forest maps with GIS systems) to changes in land use practices (Blaschke 1999). The same situation exists for conservation management and monitoring of biological diversity – no dedicated spaceborne missions exist (Innes and Koch 1998). Thus it is up to the scientific community working with forest applications to find the best ways of applying this technology and the data streaming from it. In doing so, it should be kept in mind that the 'raw' outputs from airborne and satellite sensors are nothing but measurements of emitted and reflected radiation, and estimates of e.g. biomass, are inferred based on statistical relations. This obviously puts some limitations on the types of information, that can be expected to be derived from remote sensing. In each case the analyst or organisation monitoring a forest environment must make clear what kind of information is required and check whether remote sensing can really deliver that, or if other data sources have to be drawn upon. Table 2.4, is an attempt to link some forest ecology and –management concepts with terms used in and parameters available from remote sensing data sources. The role of Remote Sensing for Gap Analysis or similar large area monitoring and planning applications, is thus to provide information on the location, extent and shape of potential habitats for the objects in question which need protection/monitoring, be it plant or animal species or habitats or ecosystems.

Monitoring of ..	**Elements of Diversity**		
Using..	Composition	Structure	Development
Ecological concept	Identity Species composition	Spatial pattern Network	Change
Entities that must be measured	Stand type Stand age Stand density	Number, size and shape of patches Distance between patches of same type	Clearance Gap creation Growth
Relevant Image Processing Technique(s)	Classification	Spatial and textural analysis	Change detection

Table 2.4 Working concept for forest diversity assessment, modified from McCormick and Folving (1998).

The processes within forests that control structure and thereby forest diversity and the suitability of the forest as habitat is illustrated in Figure 2.4, on page 60, which is a modified 'Strommel diagram', together with the approximate spatio-temporal domain of different ecological processes, the domains where the different types of biological diversity are observed and the domain covered by operational remote sensing. Blackburn and Milton (1996) discuss gap creation mechanisms (the function/development component) process and regeneration dynamics, natural successional processes in deciduous woodland at landscape level (landscape-community according to Figure 2.2, and how these can be monitored using remote sensing techniques, with the New Forest in England used as test area.

It is important to recognise that Remote Sensing offers some approaches that are different from, but possibly complementary to the use of landscape level spatial metrics. With these approaches, other types of information can be extracted from remotely sensed data, and used for classification purposes and determination of surface parameters. Analysis of spectral properties of the surface, as derived from RS data have long been used for assessment of vegetation health and forest damage (Häusler and Akgöz 1997, Kenneweg *et al* 1997) and chemical composition of the foliage (Martin *et al* 1998, Blackburn 1998), and a variety of 'spectral indices' have been developed to describe vegetation properties (Leblon *et al* 1993, Blackburn 1998, McDonald *et al* 1998,). Other methods include texture analysis (Cohen and

Spies 1992, Nel *et al* 1994, Coops and Culvenor 2000), spectral un-mixing (Cross *et al* 1991, Garcia-Haro *et al* 1996, Peddle *et al* 1999, Brown *et al* 2000), use of geometrical-optical models (Albers *et al* 1990, Jasinski 1990, St-Onge and Cavayas 1995) and time series analysis (Cihlar *et al* 1997, Waring and Running 1998, chapter 5).

2.3.2.4 Applicability of EO data for assessment of forest and landscape diversity

Not only has the direct use of RS data for the assessment of biological diversity been disputed (Roughgarden *et al*. 1991, Mann and Plummer 1993 and 1995, Roe 1996), in general the practical applications of satellite RS in forestry remain unclear (Blaschke 1999). Nevertheless, remote sensing data are beginning to be used for assessment of structural diversity, especially within the field of Landscape Ecology (Eiden *et al* 2000). Furthermore there are some examples of use in Gap Analysis projects, mostly derived from USA and Australia, where the landscape units are generally of larger extent and simpler composition compared to those found in Europe (Scott *et al* 1993, Scott and Jennings 1998, Loomis and Echohawk 1999).

Hunter (1990, in Koch (2000)) describes seven 'criteria' for the classification of forest diversity: species composition, age structure, horizontal spatial heterogeneity, edges, islands, vertical structures and (the presence of) dead trees. These correspond quite well to a short list of indicators of forest naturalness (Riitters *et al* 1992, Davies and Moss 2002), from the structural and compositional domains of diversity. In Table 2.5 on page 58, these are compared to the remote sensing data sources that are available today – in the form of images at the visible and near-infrared wavelengths, thus excluding RADAR and Light Detection and Ranging (LIDAR) techniques. For a discussion of these techniques and their applicability, see Innes and Koch (1998).

	Data source	**High resolution Aerial photogr.**	**Low res. (high alt.) Aerial photogr.**	**Very high resolution satellite imagery**	**High resolution satellite imagery**	**Medium resolution satellite imagery**	**Low res. satellite imagery**
Feasibility/	Spatial Resolution	<1 m	1-5m	~1m	5-50m	50-500m	>500m
Forest feature	species composition	+	+	+	+	+	+
	age structure	+	+	+	?	-	-
	horizontal spatial heterogeneity	+	+	+	+	?	?
	Edges	+	+	+	+	+	+
	Islands	+	+	+	+	+	+
	vertical structures	?	?	?	-	-	-
	dead trees	+	?	?	-	-	-

Table 2.5 Features considered relevant to forest diversity and the potential of different sensor types to monitor them. Based on Hunter(1990) in Koch (1999), and Wulder (1998, table 3 p. 455).

+ : detection/mapping is possible, ? : dubious/not verified, - : not possible.

The possibility of assessing species composition at even low resolutions, are based on the results from large area mapping projects, applying the AVHRR instrument of the NOAA satellites (Cihlar *et al* 1997, Häme *et al* 2000, Riitters *et al* 2000). The first European forest map reported to be made was based on NOAA-AVHRR data with 1 km spatial resolution (Häusler *et al* 1993), in which it proved possible to map forest over large areas – even under very different terrain and climatic conditions.

Errors, noise and potentially bias (on reflectance values and thus land cover proportions) are added to the satellite imagery by atmospheric properties and terrain effects, and the establishment of time series for environmental monitoring is sensitive to degradation or change of sensor response to the upwelling reflected radiation. The sensor models used for corrections of reflectance/temperature values, may simply not be valid (for instance due to degradation of the instruments on board the satellite), or they may be used in inappropriate ways (Duggin and Robinove 1990, McGwire *et al* 1993). Finally strong bias can be added from the whole suite of methods/software for image processing and handling of geographic (vector) data. These processing steps include reflectance correction, geo-referencing, segmentation, classification and spatial (clean up) filtering procedures (Moody and Woodcock 1994, Duggin and Robinove 1990). Mapping of structural factors such as edges and 'islands' (typically equal to number of patches) is obviously highly scale dependent, as the edge-length and the number of patches/island will decrease with increasing grain size, in a non-linear way (Benson and MacKenzie 1995, Riitters *et al* 1997).

So why use high-resolution satellite data at all? The first reason is that the (spatial) information that we get from them is closer to or more directly related to the "processes" taking place in the landscape than cadastral maps or statistical information (Blackburn and Milton 1996, Pitt *et al* 1997, Pitkänen 1998, Lucas and Curran 1999). Figure 2.4, on page 60, is intended to illustrate how remote sensing techniques, as available at the moment, fit the spatio-temporal dynamics of forest ecosystems. The lower and the left side of the RS box represent the best obtainable resolution/grain size, the right and upper sides represent the maximal possible extent or coverage using single images (with mosaicing, Word-Wide coverage is possible, as already demonstrated in various land-cover-mapping exercises). The second reason for using satellite image data, is that they allow us to check how well connected the indices calculated from medium resolution data are to the information that can be extracted from low-altitude, aerial photographs – a data source considered too expensive for mapping and monitoring of larger areas (Wulder 1998).

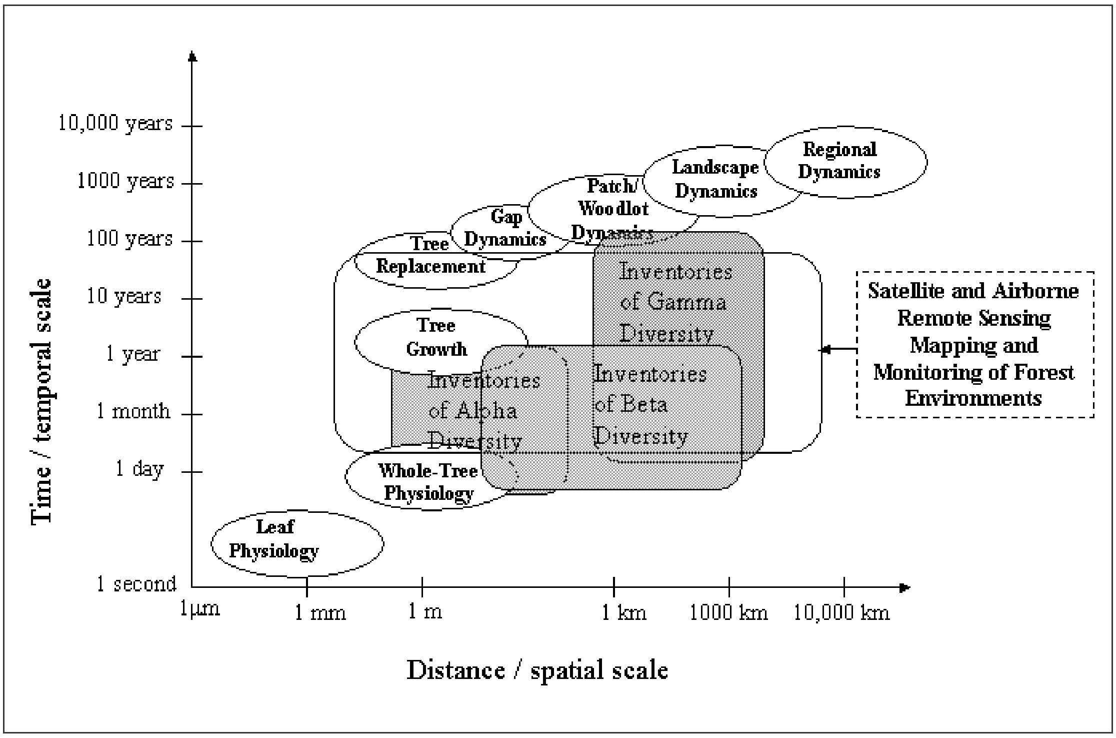

Figure 2.4 A hierarchical representation of forest dynamics and the role for Remote Sensing in monitoring of forest environment. Adapted from King (1990).

In spite of the scepticism expressed above, there has been little doubt that in Europe there is a strong potential for mapping the diversity of land cover types (in this case more or less equivalent to vegetation types), situated within wholly or partly forested landscapes, by use of Remote Sensing techniques (Blaschke 1994, McCormick *et al* 1995, Häme *et al* 1999, Häusler *et al* 2000). Furthermore, some improvements appear to be possible, based on expected developments in conceptual and mathematical models, software and sensors (higher spatial resolution as well as multi-spectral sensors). This should make possible operationalisation of RS data in the following fields:

- Detection of areas threatened or in need of special management techniques/consideration, e.g. fire or erosion risk.
- A better understanding of relations between spatial/textural measures/information from high resolution to medium scale spectral and/or spatial information

Potential advantages from the use of Remote Sensing in large-area environmental monitoring include:

- Satellite image data offer synoptic views, also over larger areas.
- Satellite image data makes it possible to repeatedly update Land Cover maps. Compared with LUC maps from other sources, such as 'normal' topographical maps. This will be an advantage when analysing habitat structure, as land cover maps/images show what type of (and/or how much) vegetation is actually present in the landscape.
- Remote Sensing techniques have been developed for monitoring vegetation health, and can be used for detecting sudden changes such as wind throw, clear-cutting and burned areas.
- Comparison will be possible across borders and administrative levels, independent of ownership of the areas of interest.
- Satellite image data can provide unbiased historical datasets, in the best case more than 30 years back (launch of first Landsat satellite in 1972), though the results must be tested for sensitivity to different kinds of changes (sensor type, resolution etc.).
- As different types of information are available from RS/EO data with inherently different spatial resolution/grain size, additional information can be gained from combinations of these, as they reflect processes taking place at different hierarchical levels of the ecosystems.

2.3.3 Scaling issues related to raster GIS and EO derived image data

Contrary to traditional disciplines as biology and geography, work with digital EO data restricts the user to certain levels of observation – and the spatial resolution of the data automatically becomes the scale on which the data will be analysed. In the Remote Sensing community the term "scale" is often used synonymously with resolution, i.e. pixel size, and thereby becomes yet another sensor dependent parameter. However, natural phenomena occur in widely varying temporal and spatial domains, and ideally data sources should be selected from (monitoring) task to task and from (research) project to project, depending on the level of occurrence of the phenomenon under investigation, ref. Section 2.2.4.

A fundamental question is then, how to define the scale at which the processes of interest are taking place, and how, with that information available, to choose the correct solution of image data that will be used to map and monitor the objects of interest (Davis *et al* 1990, Stoms and Estes 1993).

2.3.3.1 Concepts of scenes, models and scale in Remote Sensing

Almost from the beginning of Remote Sensing as a discipline, it has been characterised by two very different approaches. On one side, much theoretical and practical work related to RS has been about acquisition of data and derivation of their physical meaning, typically reflectance (directional) and temperature. Practitioners of this approach have often worked with radiation and sensors, and calibration of these. Not surprisingly, many engineers and (geo) physicists have taken this direction – but it has also found applications in forestry, through analyses of leaf reflectance properties and light interception models for canopies (Jasinski 1990, Kuusk 1991, Blackburn 1998). On the other side, users of EO data from many various subjects such as geography, botany, agronomy – and certainly forestry, have expressed strong interest in immediate use of whatever data available for studies of phenomena on the Earth surface, preferably in a handy (GIS) format. This has called for application and development of statistical methods for image classification, feature extraction and change detection (Koch 2000, Banko and Kusche 2000).

Strahler *et al* (1986) provided a review of the developments outlined above, at a time when (what was then known as) high-resolution satellite data started to become available from the Landsat satellite's TM sensor and the SPOT satellites HRV and Panchromatic sensors. They recognised the need for a common ground or starting point and clear, common concepts to be understood and used by people working with RS data. Their proposed definitions are briefly reviews below.

A scene is defined as the spatial distribution of matter and energy-fluxes, on which a given sensor is measuring (Strahler at al 1986). An image is then a set of (distance) measurements over the scene, typically arranged systematically in rows and columns, so they can be treated as a matrix. The term “resolution cell” describes the area over which the measurements of the sensor are integrated or averaged, and that normally corresponds to a “pixel” in the images that are subsequently displayed and analysed.

Two sub-types of scene models are defined

a) Discrete models, where it is assumed that the scene consists of separate elements, ideally distributed on a homogeneous background. If the elements have different reflectance properties, they can be identified in time and space.
b) Continuous models assume that changes in matter and energy fluxes are continuous in time and space. It is possible to determine these properties as precise as the instruments allow – and to broaden these properties to cover larger scenes using averaged values from RS sensors. More and better measurements will provide a better description of the field of (the values of) properties such as crown cover or Leaf Area.

The elements in a discrete model are abstractions of real objects in the scene, for which it can be assumed that they have similar properties or parameters. Simple discrete models have only one (type of) element apart from the background, while complex discrete models have more, or even several types of background. The elements can be unique, or belong to one or more classes, it is then assumed that all elements in a class are characterised by the same set of properties/parameters. Thus, forest-non-forest maps belong to the simple discrete models, while land cover maps of CLC type, also known as categorical maps, belong to the complex discrete type.

Also the concept of nested models is useful. In these, the basic elements and their properties and parameters are used to infer properties of larger elements that are aggregated by smaller ones – such as the element forest may be composed of coniferous trees, deciduous tree and (litter covered) ground. Often the shadows from any of the basic elements constitute a separate class, such as in most approaches to spectral unmixing (Garcia-Haro *et al* 1996, Peddle *et al* 1999). Clearly, the theoretical/physical approach described above relate to deterministic models, using known properties of the scene elements to extract parameters of interest. In contrast, empirical models will associate sensor observations (pixel values) with certain elements, normally using statistical methods – as in standard Minimum Distance and Maximum Likelihood classifications, where the user supervises the selection of training areas, i.e. selected groups of pixels that are known to belong to a certain element type or class.

Types of RS scenes can also be categorised based on the relation between the size of certain (selected) elements and the sensor resolution, i.e. pixel size. Strahler *et al* (1986) introduced the concept of H- and L-resolution. It should be noted that the concept of resolution here is relative, thus not similar to what is elsewhere called high- and low-resolution (imagery, as e.g. used in Table 2.5 on page 58).

H-resolution denotes a situation when the elements are notably larger than the resolution/pixel size, while on the contrary, at L-resolution they are notably smaller than the resolution. This implies, that in H-resolution imagery, the elements can be directly seen, identified, labelled, measured and counted, while at L-resolution a parameterisation of the spatial distribution is necessary if anything is to be known of their size and proportion, this leading directly to sub-pixel analysis (Woodcock *et al* 1994, Peddle *et al* 1997). Furthermore, L-resolution imagery should have two-dimensional stationarity to allow mapping of the scene properties over several pixels – meaning that the same spatial pattern and/or texture should be present all over the scene, or at least the segment being investigated or characterised. This obviously calls for a working segmentation of the images before sub-pixel properties are assessed on L-resolution imagery – such as is for instance the case when assessing forest composition and internal structure using Landsat TM and SPOT HRV imagery (Wulder 1998, McCormick and Folving 1998).

Undertaking environmental analyses with use of RS imagery forces the analyst to use data acquired at certain levels of observation, making their spatial resolution the scale at which analysis is carried out – still knowledge of the scale of the objects in the imagery will be important in order to know whether the data type and methods used are feasible, and if the characteristic scale of the imagery corresponds to the size of the 'real world' objects of interest. In an influential paper Woodcock and Stahler (1987) describe a simple method to show local variance in images as function of their spatial resolution. The variance in an image is described through gradual degradation to lower resolutions and calculation of the average variance in 3*3 pixel windows. Analyses of high-resolution aerial photographs from a forest area showed the local variance to be highest at a resolution equal to or slightly smaller that the diameter of the dominating objects of the images: the trees. Less clear results are achieved with images from urban and agricultural areas. A theoretic analysis with simulated images of dark disks placed randomly on a light background shows a curve that peaks at cell-sizes between ½ and ¾ of the objects' size. Irons *et al* (1985), studying actual and degraded Landsat TM data from a complex agricultural and urban landscape in Maryland, USA, point out two consequences of altering spatial resolution: that spectral variability often increases when spatial resolution is increased - and that statistical separability decreases as pixels become less homogeneous.

Raffy (1994), in a special issue of the International Journal of Remote Sensing on scaling, in the introduction paper titled "Change of scale: a capital challenge for space observation of earth", provides some good examples of how bad things can turn when e.g. merging data to change pixel size, and arguments that a 'spatialisation' method is needed if RS data are to be combined with computer simulation (of ecological processes).

2.3.3.2 Texture and scale in image processing

Similar to the difference between the statistical and the physical approach to analysis of multi-spectral data a difference exists between spectral and textural image analysis, or per-pixel statistics versus contextual or per-object statistics. In broad image processing terms, texture refers to the pattern of brightness variations within an image or a region of the image (Musick and Grover 1991, p. 79). When aerial photographs are used as the basis for manual/visual delineation and labelling of spatial entities (such as forest stands), the analyst is using the textural properties of the image, as well as the average colour or grey level values of the segment of interest. Obviously, this has been done as long as aerial photography has been available for mapping and landscape analysis – in the process giving birth to the discipline of landscape ecology (see e.g. Forman 1995, ch. 1). On the other hand, much of the scientific progress related to and spurred by the development of new satellites and sensors, has been directed towards achieving a better understanding of the spectral properties of land surfaces and vegetation (Woodcock and Strahler 1987). However, with increased availability of panchromatic image data from the SPOT, IRS-C and IKONOS satellites, attention has again been drawn to the possibilities of gaining extra information from images through analysis of the relation between pixel values at different positions in the matrix – since it has already for some time been known that textural features can reduce the classification error rate and improve the analysis (Haralick *et al* 1973). A distinction can be made between a <u>structural</u> approaches which resemble the way humans perceive visual impressions, and <u>statistical</u> approaches, where certain, pre-defined parameters are calculated from sub-images or windows (Sali and Wolfson 1992). The structural approach assumes that the images consist of primitive elements or objects (in this case patches of a certain shape and size), repeated in a certain pattern, and that differences in texture result from differences in the elements, the pattern of their repetition or both (Musick and Gover 1991). In the statistical approach, texture is modelled as a grey-level function, with more or less continuous values over the land surface – depending on the value of interest and of the size of the window uses in the calculation.

The link between variograms and geo-referenced image data is provided by the key concept of geostatistics: the regionalised variable, which is defined as any variable of which the geographic position is known (Vogt 1992). Within EO based mapping for forestry, semivariongrams have been used for analysis of canopy structure (Cohen and Spies 1990, Levesque and King 1996), tree growth in grasslands (Hudak and Wessman 1998) and various stand parameters (Franklin and McDermid 1993). These studies conclude that costumised texture windows (for which the semivariograms are calculated) are most useful for estimating canopy coverage.

2.3.3.3 The influence of scale changes on land cover classification and spatial metrics values

Ideally, it should be possible to predict the values of spatial metrics at one resolution from the same or other metrics at higher or lower resolution, in the latter case it would help extrapolation of structural properties over large areas using low-resolution RS data. Such an approach was attempted by Mayaux and Lambin (1997). They found that integration of spatial information into a correction model to retrieve fine resolution cover-type proportions from coarse resolution data improved the reliability of the estimates by up to 35%. The Matheron index calculated from NOAA AVHRR images was used as estimator, and correlated to cover proportions derived from Landsat TM images.

The Modifiable Areal Unit Problem (MAUP) was first identified by S. Openshaw, who defined it as a form of ecological fallacy associated with the aggregation of data into areal units for geographical analysis – where aggregated data are treated as individuals in analysis (Openshaw 1977 and 1984, Marceau and Hay 1999). The concept is also widely used in social sciences, e.g. it is recognised that census layout in form of size and shape and (demographic) composition of statistical units will strongly influence the results (Green and Flowerdew 1996). Hay *et al* (1997) propose ‘object specific upscaling’, a procedure in which the spectral ‘influence’ of image-objects are spatially modelled and integrated within a user defined upscaled representation (which could be a land cover map at lower spatial resolution). Marceau and Hay (1999) describe Remote Sensing as a particular case of the MAUP, and propose this as an explanation to many of the inconsistencies observed in studies where EO data were used to produce thematic maps or as inputs to physical models – without the scale taken explicitly into account.

Aggregation A common problem in aggregation of LUC data is the variability in results obtained through variations in the shape of areas, an example of this is the dependence of forestry statistics on how the basic spatial units, the stands or forest management units are delineated. Cao and Lam (1997) point to the similarity between trials with different aggregation levels and mechanical ‘zooming’ in and out to find the best ‘focus’ of an image of a certain area. All methods that involve modifying the units of measurement and reporting will lead to loss of details. Some methods however better retain statistical characteristics of the original data, while others are better for revealing spatial patterns at another resolution. Within a particular aggregation level, some classes are better classified at fine spatial resolutions, while others require coarser spatial resolutions (Marceau *et al* 1994b). All aggregation methods lose details, but some better retain statistical characteristics of the original data, while other methods are better for revealing spatial patterns at another resolution (Bian and Butler 1999). The same authors find that the averaging method for aggregation produces data and errors with the most predictable behaviour. Using simulated images gives better control of statistical and spatial characteristics of the data, and is suitable for *systematic evaluation of aggregation effects*. If research is focused only on the effect of aggregation on model output, it will not be possible to separate inherent flaws of the methods from operational errors. Skov-Petersen (1999) describe the aggregation of point data on buildings types and uses as well as floor space to a grid covering the entire surface of Denmark, and summarise the considerations that must be made during the aggregation process: *Fidelity,* *Reality, Objectivity, Accessibility, Data-handling, Sensitivity to lack of accuracy of single in-going points, Handling of 'noise'.*

Coarsening or degradation of images results in images with a larger pixel or grain size, where each pixel holds information representative of several pixels in the original imagery. Theoretically, application of this operation will mean that the larger or more common land cover classes will tend to become more dominant, while smaller or less common classes will have even smaller proportions or completely disappear (Gustafson and Parker 1992). The magnitude of this effect although depends on how clumped, spread or fragmented these land cover classes are (or the elements/objects belonging to them). Moody and Woodcock (1994) performed a simulation from 30 m resolution, Landsat TM based maps to test the use of MODIS based land cover maps, and found that while class proportions change in a regular way, the proportional errors differ between classes. Degradation of image data from higher to lower resolutions should ideally simulate sensor response (Townshend and Justice 1988), so that degraded images from for instance Landsat TM would resemble images form the IRS-WiFS sensor. When spatial degradation or thematic and spatial aggregation is performed, it must be considered whether to apply methods/algorithms that account for the (relative) importance of different land cover/vegetation types, typically through application of a weighting function, rather that ‘brute force’ methods that treat all pixel values or land cover classes equally.

Wickham and Riitters (1995) analysed the behaviour of metrics of diversity and structure (contagion) for a data set derived from aerial photographs at 4, 12, 28 and 80m, and found that metrics values were not ‘dramatically’ affected by this scale change. Wu *et al* (2000) demonstrated use of scale variance analysis and landscape metrics as methods for testing as well as describing multi-scale or hierarchical structures in landscapes. Response curves of metrics values as function of grain size were found to characterise different metrics types and to differentiate between landscapes better than variograms and scale variance curves. Wu *et al*

(2002) and Wu (2003) further developed the use of these response curves, now termed scalograms, for characterisation of metrics at class- as well as at landscape level and also investigated the response of metrics values to extent, i.e. the size of the image or window (in terms of number of pixels) for which the metrics are calculated.

In summary, finding an appropriate scale of measurement for geographical entities remains a fundamental, still unresolved problem (Marceau *et al* 1994a, Wu 1999), thus it shouldn't be expected that this project will provide a final solution, it rather aims at providing recommendations on methods to better overcome the problems of using and extracting spatial metrics from multi-resolution data.

2.3.3.4 Evaluation of spatial metrics using neutral models

When spatial metrics are calculated from images with different grain sizes and extents, and differences in metrics values observed, one can ask whether these are due to real differences in the two landscapes represented in the images or to scale effects. Thus methods are desirable that isolate the effects on observed landscape or habitat structure, which are induced by changed point of view. Also methods that evaluate the usefulness of fine-scale detail in examining broad-scale patterns[7] are important steps in development of useful and reliable models (Gardner and O'Neill 1991). One such approach to assess the influence of scale of observation is neutral models, which in the two-dimensional form are more or less realistic artificial maps showing the distribution of a number of 'classes'. Neutral models do NOT model landscape processes, they are rather used to produce data for comparison with maps of real landscapes, in order to identify non-random patterns, resulting from processes that can hopefully better be described/quantified (With and King 1997).

Early uses of neutral models were based on percolation theory (Gustafson and Parker 1992, O'Neill *et al* 1988), and the outputs had the form of random maps. Later spatial contagion was

[7] When for instance aerial photographs and satellite imagery are combined.

introduced into maps by adjustment of the correlation among sites (Gardner and O'Neill 1991). A new 'generation' of models applied fractal algorithms and hierarchical random landscapes (With and King 1997, fig. 1), and since then still more advanced algorithms have been applied, in order to create models that produce realistic images and thus metrics values. For a review of landscape simulation methods see Saura and Martinez-Millan (2000). The general purpose of using neutral landscape models is twofold (With and King 1997):

- Determine the extent to which structural properties of landscapes (such as patch size and shape, connectivity) deviate from theoretical spatial distributions, random or structured.
- Predict how ecological processes will be affected by known spatial structures.

The first approach uses models to find out how processes affect landscape patterns, the second uses models to investigate how ecological processes are affected by known spatial structures.

The performance of spatial metrics on neutral landscapes has been used for interpretation of the significance of these metrics when they are calculated on real landscapes, by separation of the effects of topography, natural disturbances and human activities from the expected behaviour of the metrics if such effects were absent (Gardner and O'Neill 1991).

Johnson *et al* (1999) extended the concept of neutral landscape models to provide a general Markovian model of landscape structure (based on assumptions of landscape development processes). A stochastic transition matrix was used to create patterns, which were compared with maps of real landscapes, watersheds in Pennsylvania, USA. Saura and Martinez-Millan (2000) present a new simulation method: Modified Random Clusters (MRC), that provides more general and realistic results than commonly used landscape models and describe the development of the Simmap software[8] where it is implemented. Saura and Martinez-Millan (2001) apply simulated landscapes based on MRC to asses the sensitivity of map extent (corresponding to window size) and Saura (2002) uses simulated thematic (land cover)

[8] Description, instructions for use and contact details for the programmer at: http://www.udl.es/usuaris/saura/simmap.htm (accessed 23/2 2004).

patterns to assess the influence of minimum mapping unit (MMU) on the values of a number of spatial metrics.

2.3.3.5 Perspectives for scaling of calculated spatial metrics

Textural measures can be of great value and improve classification, although it must be stated that when textural measures are used in image processing and reporting in this thesis, the approach is implicitly *statistical* (as opposed to modelling). The MAUP provides a relevant approach to the problem of how robust spatial metrics are to changes in spatial resolution and change of reporting unit (e.g. level of watershed or administration). Also the strategy described by Woodcock and Strahler (1987), see section 2.3.3.1, could potentially be applied on landscape or land cover maps, for calculation of spatial metrics in windows of increasing size. When the index values cease to change, the texture values or contrast between neighbouring cells decrease, or both, it must be assumed that the typical or characteristic size or distance for the actual landscape has been passed. Such an approach will build on and investigate the hypothesis that texture at one level (coarse) corresponds to spatial structure at another (finer). This might provide an approach to the regionalisation and the MAUP problems, that is relevant with forest maps and CLC- or National Vegetation Classification type (as used by the United States' GAP project) land cover maps in raster format. The variance approach described here could be supplemented by scalograms as described by Wu *et al* (2002).

In the context of this thesis and the challenges it poses, it is important to test whether the effects are the same when degrading land cover maps from finer to coarser resolution as they are when classifying EO imagery that has corresponding fine and coarse resolutions – as this is crucial for approaches that *extrapolate* from localised, well described plot or test areas to larger regions. This might allow extrapolation beyond the landscape level in subsequent experiments with large land cover datasets, in order to test whether spatial metrics are comparable at the continental scale.

2.3.4 An example of quantification of spatial structure using EO data: description and measurement of fragmentation

This section will provide an example of how the fragmentation concept can be assessed operationally with EO data through the application of spatial metrics. Fragmentation was chosen partly because interest in the concept has been expressed from environmental managers, and partly because the literature is rich with examples of how spatial metrics of forest structure in general and fragmentation in particular are defined and applied, in very different ways.

Forman (1995, p.39) defines fragmentation as the *breaking up* of a habitat, ecosystem, or land-use type into smaller parcels (considered to be one of several spatial processes in land transformation) and later on states that the concept includes perforation and shrinkage (ibid p. 408). Frohn (1998, p. 9-10) adapts this definition to an EO context and sees fragmentation as *the opposite of contagion*, which he defines as the tendency of land covers to clump into a few large patches. The term fragmentation can also be used to describe a landscape where areas of forest have been removed in such a way that the remaining forest exists as islands of trees in a cutover environment (Natural Resources Canada 1995, in Dobbertin 1998). The major concern with fragmentation is in this case the effect of the loss of contiguous forest cover on species movement and dispersal, making relevant (and possible) the application of Island Biogeography models to the 'fragmented' landscape, while Fry (1996) argues that habitat loss in general have more serious effects that changes or differences in spatial distribution with constant or equal habitat area, see Figure 2.5 on page 74. According to Kouki and Lofman (1998), the concept of fragmentation has been widely applied in recent years to denote *landscape transformation* from uniform to more patchy and heterogeneous types, although the usage of the word has not been consistent. According to Delbaere and Gulinck (1994) the term fragmentation refers to the broader term *connectivity* (and can thus be defined as lack or loss of connectivity).

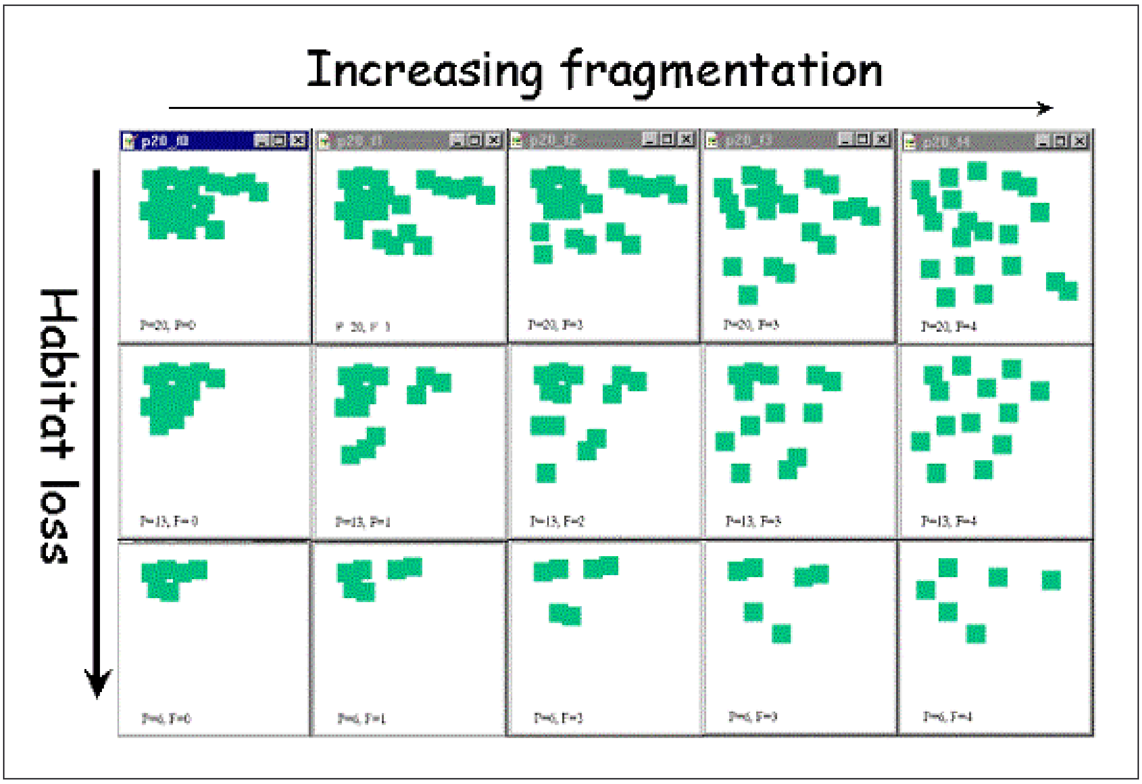

Figure 2.5 Conceptual model of how fragmentation is related to habitat loss. The size of the arrows indicates the respective importance of the processes. After Fry (1996).

In a continental-level study Skole and Tucker (1993) defined fragmented forest as isolated patches < 100 km^2 area, while Mayaux *et al* (1998) simply define an area to contain fragmented forest if within an AVHRR pixel, approx. 1 km^2, the forest cover is between 10 and 70 % of the surface. These two definitions seem to be more *ad hoc* for specific and mapping scale purposes, and defined more from knowledge about the properties of the remotely sensed data that happen to be available than from knowledge about the processes to be mapped and monitored. Riitters *et al* (2000) performed an analysis of forest fragmentation based on 1-km resolution land-cover maps for the entire globe, the Global Land Cover Characteristics database (GLCC). The measurements used a 'moving windows' approach with window sizes ranging from 81 km^2 (9*9 pixels, "small" scale) to 59,049 km^2 (243*243 pixels, "large" scale). The value calculated for the window was then used to characterize the fragmentation around the central pixel – if it was forested, otherwise it would be left blank -

with metrics of fragmentation based on the occurrence of adjacent forest pixels. The types of forest structure used was: Interior, Edge, Perforated, Transitional, and Patch – based on the relation between total forest cover (percentage) and the proportion of pixel-pairs including at least one forest pixel where both pixels are forested. Using this method it was possible to characterise fragmentation patterns of different forest types and to examine differences between continents.

Several spatial metrics have been proposed and tested for the description and quantification of the level of fragmentation of landscapes as recorded in maps and images, and only a minor selection will be applied here.

The Matheron index is one of the more commonly used indices in studies on landscape and forest fragmentation, especially those who focus on forest / non-forest interfaces (European community, 1995) - as it is basically a normalised edge length measure, defined as:

$$M = 10 * \frac{\textit{number of runs between forest and other LC type pixels}}{\sqrt{(\textit{number of forest pixels})} * \sqrt{(\textit{total number of pixels})}} \qquad [1]$$

The index has been used as a tool (Mayaux and Lambin, 1995 and 1997) to describe the fragmentation of forest cover as observed in Landsat TM images as well as in NOAA AVHRR images and to derive a correction function for use of the latter for the creation of tropical forest maps. Mertens and Lambin (1997) used the index to describe the spatial fragmentation of forest cover - as one amongst several spatial variables, some derived with GIS analysis. In European Commission (1995), the index is calculated for representative forest / non-forest interfaces on land cover data derived from NOAA AVHRR data in 34 'sites' that are found to be typical for forested, tropical regions.

Also more sophisticated measures have been used of which some are mentioned here, for a more in-depth review, see McGarigal and Marks (1994) and Riiters *et al* (1995). Amongst the indices to have drawn most attention are those which attempt to measure fractal dimension, which can be seen as a measure of as well the self-similarity as of the complexity of patch shape/borders (Mandelbrot 1967). Already Ramstein and Raffy (1989) link this measure with the structure of variograms derived from image data. De Cola (1989) found that forests have high fractal dimensions - and that for agricultural regions the fractal dimension is inversely related to land-use intensity. Hargis *et al* (1998) developed a new measure termed "mass fractal dimension", and compared it with other commonly used landscape measures, but found that no measure could differentiate between landscape patterns with dispersed vs. aggregated patches. The contagion metric has been used, revised and subjected to some criticism, as there are diverging opinions about what it actually measures (Frohn 1998, Hargis *et al* 1998). The same thing can, to some degree, be said about the different measures of fractal dimension (Olsen *et al* 1995, Frohn 1998) and some warnings are found in literature on the subject, that the use of complex quantitative descriptors for overall general concepts should be done with great care (McGarigal and Marks 1994, Brandt and Holmes 1995, Frohn 1998). In a study modelling the dynamics of butterfly populations in a real landscape, Moilanen and Hanski (1998) found that the impact of landscape structure only is influential on species persistence within a certain interval along a gradient of fragmentation. This could by to the use of fragmentation metrics for stratification of larger areas before regional analyses are performed.

Frohn (1998) proposed two mathematically simple indices for quantification of fragmentation, as alternatives to the more complicated indices of contagion and fractal dimension. They are defined as follows. The number of Patches Per Unit area (PPU):

$$\mathrm{PPU} = \frac{m}{(n * \lambda)} \qquad [2]$$

where m is the total number of patches (in the window), n is the total number of pixels in the area of interest (window) and λ is the scaling constant equal to the area of a pixel. Dependent on the extent of the area of interest the unit of λ can be m^2, ha or km^2. The advantage of the PPU index is that it reflects the number of patches, normally thought of as something describing much of the information on the structure of a classified image.

The Squareness of Patches (SqP) index is defined as:

$$\text{SqP} = 1 - \frac{4*\sqrt{A}}{P} \qquad [3]$$

where A is the total area of all pixels and P is the total perimeter of all pixels belonging to the land cover class of interest in the area (window). The theoretical value for this index is between 0 and 1; 0 is for the case of the landscape mask element (the forest) consisting of one large square; if it is made up of more patches, the values will be > 0; the value will approach 1 when the cover type becomes more scattered over the landscape.

Once these spatial measures of (forest) fragmentation have been defined and selected, scaling issues must be considered with special focus on fragmentation. The central problem is whether is it the same processes that are observed when landscapes are imaged at different resolutions. Thus scaling effects should be quantified, in order to determine if efforts can be concentrated on assembling land cover datasets at one specific (standard) resolution, or if it will be possible to recommend a series of spatial metrics that allow comparison between data derived from images with different resolution or summarised over different spatial units. These issues are addressed in the following chapters of this thesis.

2.4 Conclusions on the use of spatial and Earth Observation data for monitoring of sustainable land use and biological diversity

In this section, the findings and considerations in the literature review are summarised and evaluated, with applications for environmental monitoring and management in mind.

2.4.1 Forest mapping and monitoring

At the local or Forest Management Unit (FMU) level, maps constitue an integrated management tool and foresters are normally familiar with use of aerial photography. This should be seen as an advantage and taken into consideration when EO data are introduced in management practices. GIS is increasingly being installed and used for forest management at the lowest administrative levels, aided by the developments in surveying techniques through cheap and easy-to-use GPS equipment. Thus remote sensing data have the potential to become increasingly integrated in GIS applications for improved land cover classification, better assessment of (production related) stand parameters and change detection, i.e. updating of forest inventory maps. The real challenge is to make use of the EO data for ecologically oriented purposes as well, either by the actual agents, the forest mangers themselves or public or private ecologists/environmental experts cooperating with the forest administrations. In order to make this happen, the role of the scientific community is to provide methods for utilizing EO data in combination with forest inventory data as well as with data for conservation planning and monitoring and ecological/biodiversity surveys.

At regional and national levels, where EO data currently have few forest applications – at least in Europe - EO data is expected to be used for broader overviews of landscape structure, such as in Gap Analysis and for regular updates of forest statistics. High resolution EO data could also be used for monitoring the environmental conditions around protected areas, e.g. by assessing edge effects due to land use changes. In general EO data can supplement statistical data such as results of national forest inventories that are without spatial aspects, in the sense that values are reported for administrative units.

For assessment of sustainability and potential biological diversity, large amounts of information with potential use are available, recently also through Internet-applications, at low or no cost for researchers. Such data (sets) include national and regional forest inventories and maps, forestry statistics, data on forest ownership, protection status etc., national monitoring programs for monitoring of biodiversity that the countries have committed themselves to according to the CBD, and statistics about e.g. forest products, tourism and agriculture. At European level data are available through EUROSTAT, EEA, and potentially EFIS, and forest fragmentation can be quantified through analysis of existing EO based forest maps.

2.4.2 Land cover mapping and Landscape monitoring

Landscape diversity can be quantified through analysis of existing LUC maps, as demonstrated in EU-DG AGRI (2000) and Gallego (ed. 2002). The existing CLC database and national LUC mapping initiatives provide useful input data for landscape level analyses. Within the EU, national land evaluation, survey or mapping initiatives are often modified or at least it is made sure that the outcome can form part of CLC, meanwhile providing information at higher thematic and spatial resolutions, see Brandt *et al* (2002), Weiers *et al* (2002), Büttner *et al* (2002). Landscape level metrics can be calculated from CLC data and used to establish comparisons between regions and countries. Such metrics however, should not stand alone, but rather be used along with (other) agri-environmental indicators (see European Environment Agency 2001, Gallego 2002).

LUC data can serve as contextual information for assessment of habitat quality, at correctly chosen spatial scales, and will be indispensable for applications of (calculations based on) island biogeography, meta-population theory or the patch-matrix-corridor model. Hemeroby levels or index values could be calculated from LUC data, preferably in combination with information on land use history and on point 'sources' of human activity, pollution etc. Spatially explicit models of ecological processes, including animal movements and species colonisation and extinction could help establish statistical relations between values of spatial metrics and either habitat quality or species richness of landscapes. These relations might differ with the size (extent) of the landscapes investigated. Thus neutral models could be used to assess the effects of extent before moving-windows methods are applied for calculation of metrics (map) over large areas. In addition such models (outputs) may help separate scale influence of metrics values from differences due to real-world differences in spatial structure.

Figure 2.6 is intended to provide a conceptual overview of the factors involved in a system for assessing landscape structure (of which forest structure is a special case), integrating remote sensing and 'ancillary' data, probably using a Geographic Information System for the data management.

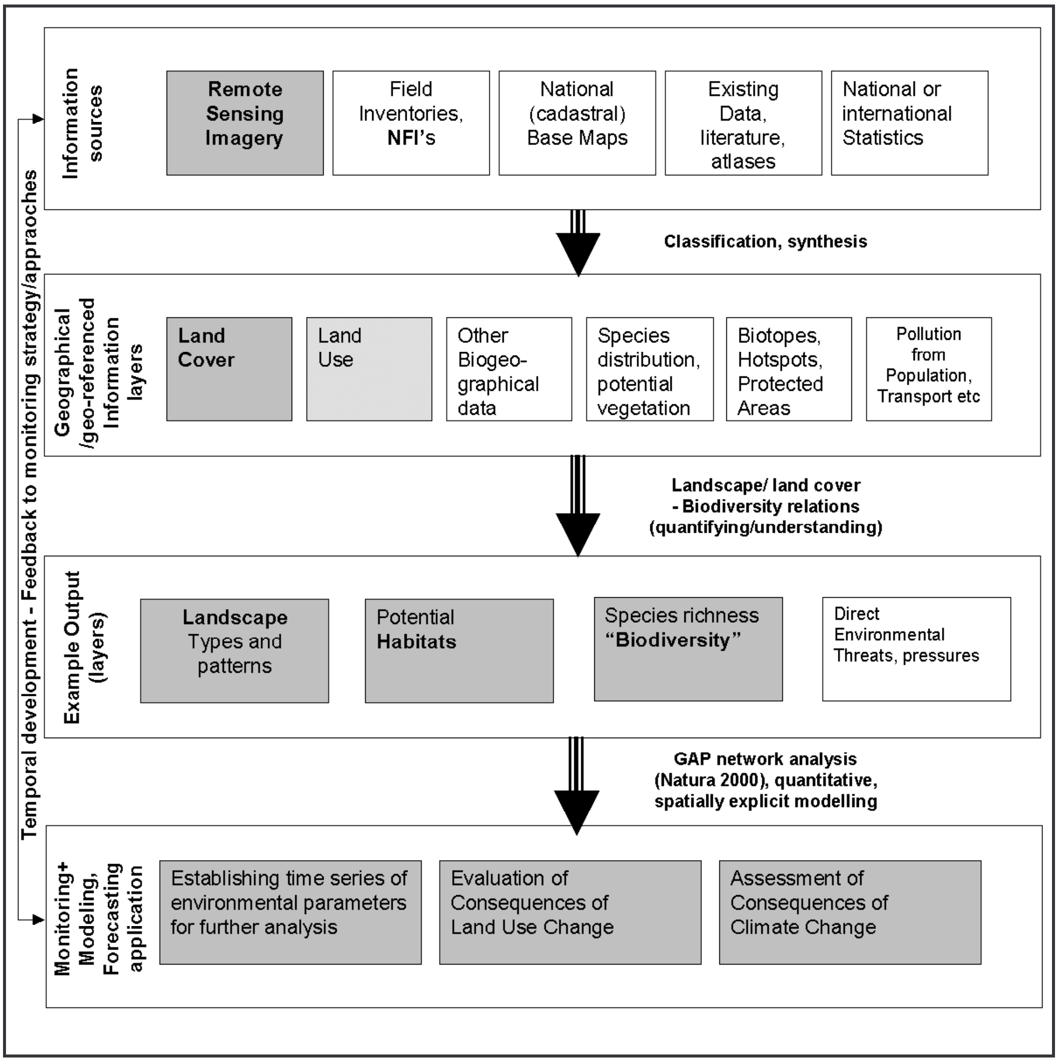

Figure 2.6 Conceptual model for integration Earth Observation data with other information sources for environmental monitoring in a habitat based monitoring approach (from Estreguil *et al* 2001).

2.4.3 Applications of spatial metrics in an EO-GIS framework

Relative to traditional land use maps, land cover/vegetation maps derived from EO data provide more relevant input data for calculation of metrics such as diversity – on the other hand there are several potential error sources in the processing chain from spectral bands of satellite data to classified images. The choice of which geographical data to use for specific management/monitoring tasks will however depend on the information need as well as availability and price of data and not least the on potential to combine data and metrics for description (and prediction) of biological properties of the landscape or forest of interest – and on the ability to detect changes when data from different times are compared. This potential can be clarified through scaling and sensitivity analyses.

There is a certain pedagogic value of calculating spatial metrics from LUC, in the sense that it makes the user think in landscape ecological terms (patches, corridors, edges etc.). Furthermore implementation of moving-windows methods, in line with those envisioned by O'Neill *et al* (1997) will be useful for illustration purposes, as demonstrated by Häusler *et al* (2000). However, when the outputs from such calculation are used as raster-GIS layers there may be particular scaling problems associated with the window size(s) used – especially if the 'maps' are made from input data with different pixel/grain size.

Through this literature survey, relationships between on one hand biological diversity and naturalness (state) of landscapes and on the other hand spatial metrics derived from EO data of the same areas have been identified, some simple and some rather complex, based on intricate numeric models. It follows from the discussion above that is relevant to focus further studies on development of methods to derive indicators from EO data, which meet the information needs of potential users. Such metrics must contain information about processes or 'state variables' that is of concern (ref. Table 2.5) or central in reporting according to e.g. the Helsinki process or for the EU member states in relation to designation and monitoring of Natura 2000 habitat areas. Examples are forest fragmentation, landscape diversity, connectivity and disturbance. Also temporal metrics like change rate would be relevant as indicators. It is however still important to keep in mind what purpose the spatial metrics are being calculated for, and who will in the end be using them.

A possible flow of information and decisions in the application of software for calculation of spatial metrics aimed at forest or landscape management is outlined in Figure 2.7, which is partly based on the recommendations in McCormick and Folving (1998) and Häusler *et al*

(2000). This figure will be used for discussion of the actual implementation of spatial metrics calculation and image and landscape analyses performed during the studies for this thesis.

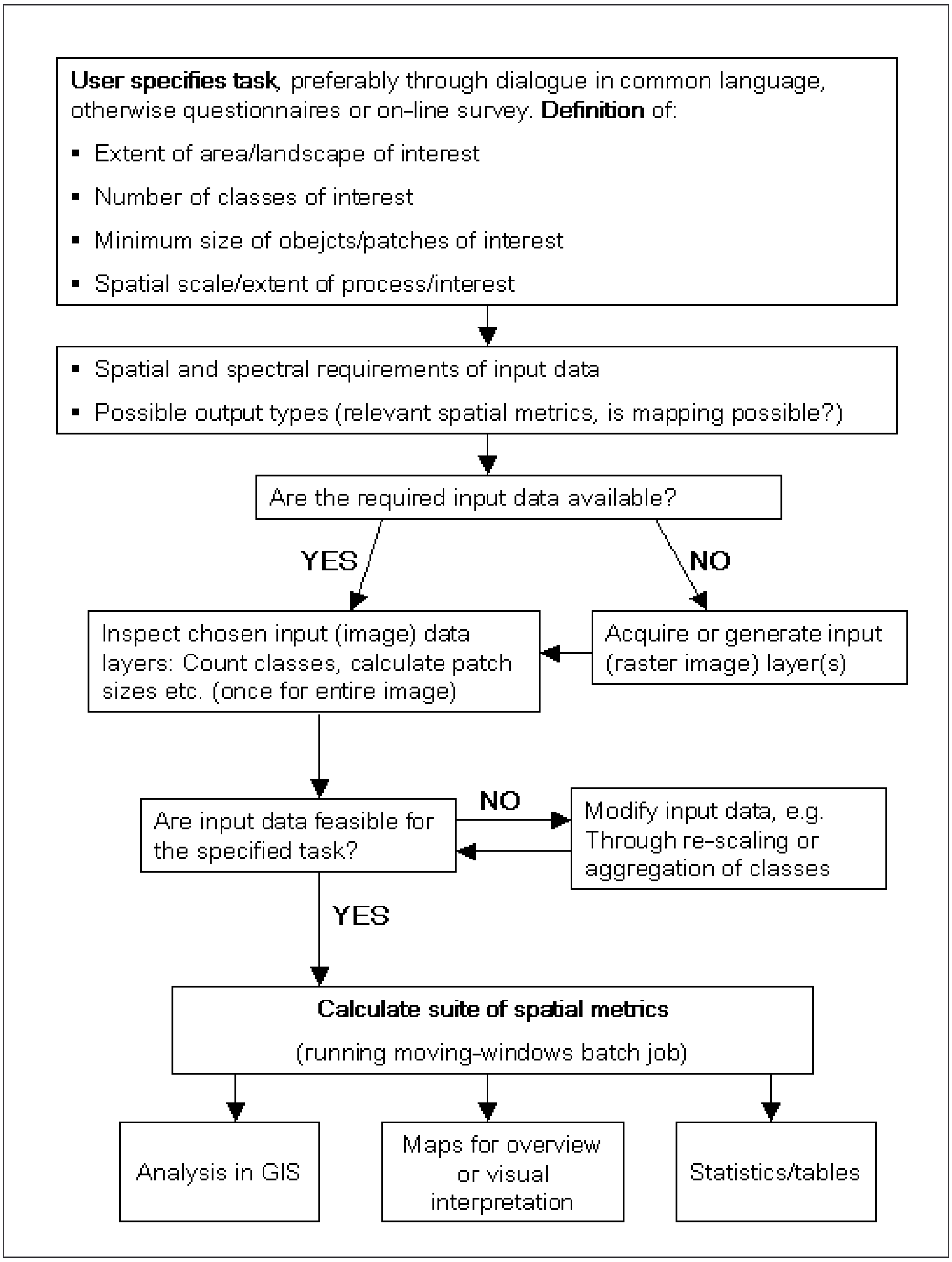

Figure 2.7 Proposed schedule for landscape ecological analysis using EO data and spatial metrics.
