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The role of the spatial dimension within the framework of sustainable landscapes and natural capital

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Abstract

A new paradigm of Natural Capital and Sustainable Landscapes has been suggested. It implies the integration of economic, environmental and social-cultural qualities in a physical setting while focusing on functions in terms of goods and services for people. Due to its anthropocentric perspective it pays less attention to landscape structure and spatial arrangement compared to the widely applied patch-matrix concept. The matrix of land use elements provides the key to understanding land use systems and land use changes and it can play an important role in understanding land use pattern and their dynamics. But one of the remaining constraints for a direct application of landscape ecological concepts in practice is the lack of agreed ways to combine environmental, socio-economic and societal/cultural views. This paper examines both paradigms, asking: does the spatial arrangement of land use types add specific qualities beyond statistical measures of their existence and quantity? For instance, can a landscape be sustainable, as long as 20% of the land use is extensive, 10% is protection area, etc., no matter where the respective patches are, which typical size and shape they have, how connected patches are and how often incompatible land use types are adjacent? This paper elucidates spatial concepts for sustainable landscapes with an emphasis on the role of GIS.

Keywords: Sustainable landscape; Landscape planning; Landscape metrics; GIS; Landscape scenario; Leitbild

1. Introduction

1.1. Spatial constructs to decomposing complexity

Landscape ecology as a discipline has, in part, focussed on the description of structure and pattern, especially through the use of Geographic Information Systems (GIS). These pattern descriptions are the basis for the exploration of ecological mosaics (Turner, 1990; Forman, 1995; Wiens, 1995; Turner et al., 2001) and the foundation of a spatially explicit consideration of space based on relatively homogeneous patches as basic spatial entities (Wiens, 1995; Gustafson, 1998). It is generally believed that an area dissected into patches can be analysed and modelled more efficiently than dealing with a complex system as a whole. Environmental management has predominantly focused on individual ecosystems but is increasingly confronted with the problem of managing and planning entire landscapes which often consist of complex, interacting mosaics of different habitat patches and ecosystems (Potschin and Haines-Young, 2001). Generally, a complex entity is composed of different elements that interact and combine in a way that may not be obvious at first.

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Planners and environmentalists break down complex problems into compartments (sectoral approach) or themes (object-oriented or theme-oriented) accounting for the fact that landscapes are spatially and functionally heterogeneous, more so than ecosystems. Recently, models have been applied to the observed patterns and processes and research gradually moves from description to statistical interference (Turner et al., 2001; Bissonette, 2003). Planners offer guidelines for future developments, informing the decision making process (Botequilha Leitao and Ahern, 2002).

Nature is complex. To partition complexity is defining a typical human behaviour (Simon, 1962; Koestler, 1967). For example, structural complexity may refer to the compositional diversity and configurational intricacy of a system; functional complexity emphasizes the heterogeneity and non-linearity in system dynamics; and self-organizing complexity hinges on the emergent properties of systems co-evolving with their environment primarily through local interactions and feedbacks at different spatiotemporal scales (Wu and Marceau, 2002). We can observe a shift of systems thinking in complex ecology (Peterson and Parker, 1998): research increasingly deals with emergent properties of non-linear adaptive landscape system, spatio-temporal complexity and chaos, scale (scale invariance and covariance), hierarchy, cross-scale dynamics, and non-linear physics based holistic landscape ecology. New methods such as coupled map lattice, non-linear thermodynamics-based Markovian model, multiscale entropy analysis, self-assembling of networks, and detecting noise-induced structures in spatiotemporal data have been introduced (Brown et al., 2002). The idea that the concept of complexity is inseparable from perception depends on the scale of observation under study. There is no scale for observing all phenomena as illustrated in Fig. 1.

In order to understand complex systems, it is often convenient to consider a simpler system that exhibits the type of behaviour of interest (Simon, 1962). In sustainable landscape management we are mainly concerned with the notion of long term stability/resilience and the fact that the domain of attraction of a stable equilibrium may depend upon slowly varying biophys-

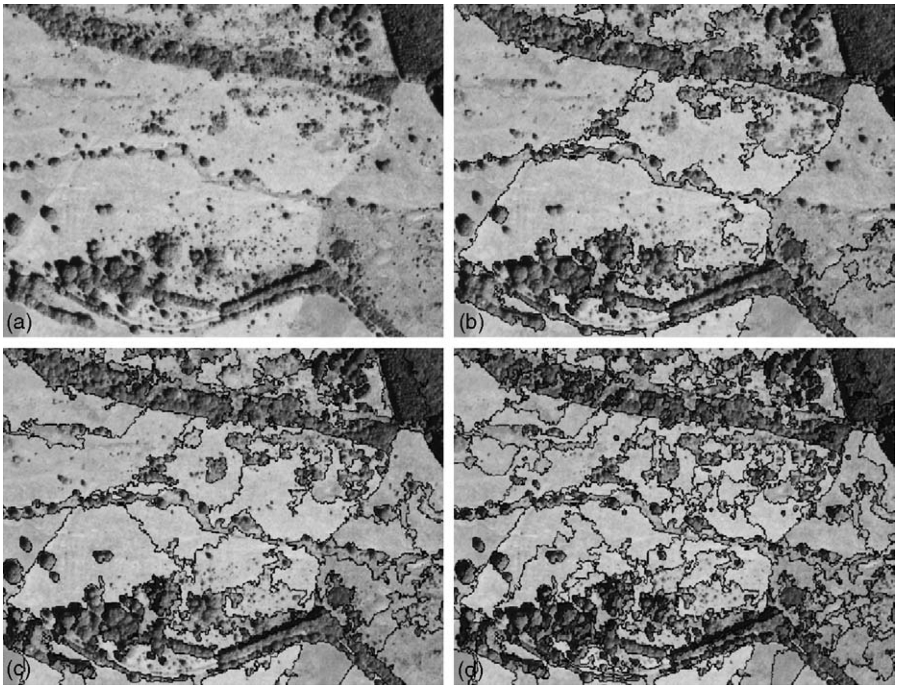


Fig. 1. Dissecting a remotely sensed image (a) into three different levels. The images (b) to (d) represent increasingly detailed segmentation levels and increasing numbers of segments. None of these levels is 'right' or 'wrong' but might be appropriate or inappropriate for a certain application and a given scale.

ical parameters and fast changing human-induced disturbance (Botequilha Leitao and Ahern, 2002; Antrop, 2003). This task is extremely difficult and we have to look for easier approximations of these processes. Ludwig et al. (1997) demonstrated the complexity of the task to clarify the concepts of sustainability and resilience even for the subset of natural systems. Brown et al. (2002) describe recent progress and future prospects for understanding the mechanisms of complex systems as power laws which express empirical scaling relationships that are emergent quantitative features of biodiversity. One of the major issues of this paper is the ability to take into account the multiplicity of (spatial) scales of study so that each phenomenon studied at its specific level can be integrated through hierarchically organised spatial concepts.

1.2. The landscape concept and spatial questions

Landscape refers to a common perceivable part of Earth's surface (Zonneveld, 1995). Land use is the most dynamic aspect. Crop rotation and changing land use year after year is a 'normal' change and is not considered as a disturbance that breaks continuity. Changing land use in this manner seldom changes the whole landscape and may even be a specific character of it. Landscape change does happen when gradually the land cover transforms to a new dominant type and also causes structural change (Antrop, 2003). Another landscape will be formed when the new forms of land use demand larger fields, special treatment of the soil, terrain levelling, removal of hedgerows and new enlarged roads. Change and continuity are related to speed and magnitude of the overall land use and land organization. The use of aerial photography after the Second World War stimulated the study of landscape in a broader multidisciplinary field (Forman and Godron, 1986; Zonneveld, 1995). Theories about changes were developed and the human impact on the natural environment is considered as the most important factor of change nowadays, acting more and more at a global scale (Goudie, 2000). It is widely agreed that people living in the landscape must be actively integrated in the landscape planning process (Forman and Collinge, 1997; Volker, 1997; Botequilha Leitao and Ahern, 2002). If the residents do not have satisfying opportunities to influence the development of their landscape they might no longer fulfil their needs and identify themselves with their everyday landscape (Buchecker et al., 2003). Social problems can be a direct consequence (and cause) of environmental problems (Saunders and Briggs, 2002). Concepts, therefore, for sustainable landscapes should not only focus on sustaining the physical landscape resources, but they should also ensure quality of life of the people living in the landscape.

For particular situations, examples exist to model consequences in the form of scenarios and impact maps for a particular scale. The impacts of economic forces and environmental policies are difficult to forecast spatially due to the highly variable ecologies across regions (Webster, 1997) but ample studies demonstrate the possibility to combine both a local and regional perspective using a spatial framework (Dramstad et al., 1996; Hermann and Osinski, 1999; Botequilha Leitao and Ahern, 2002). A good example is land abandonment in Europe which is due to severe changes in agricultural economics. This process is ongoing in large parts of Europe but it is expected to be escalating over the next years. The spatial patterns of these expected severe land use changes, affecting millions of hectares of land, are important. The combination of GIS and spatial modelling tools will support research questions like: "where will land abandonment happen if no policy actions are set" or "which areas will be more or less affected if subsidies are increased or decreased"? The identification of risk zones may help planners at local, regional and national levels, to focus their activities on problem areas and to differentiate strategies between low-potential and high-potential areas. The example of modelling land abandonment certainly includes both ecological and economic aspects. Risk and potential are typically envisaged through additive models which indicate areas of superimposition of factors such as recent and historic land use, areas of legal restriction, environmental parameters on topography, soil, vegetation, or land use. Usually, they do not take into account the quality of life of the residents.

1.3. Objectives of this paper

This paper puts an emphasis on spatial representations in the context of sustainable landscapes, namely maps and representations incorporating temporality and dynamic modelling which are important tools for the analysis of landscape ecological processes, and for the visualization of alternative land-use scenarios. Many social and economic data are only available for certain administrative levels. The availability of spatial data in digital form is a prerequisite in landscape analysis, to monitor landscape change and to evaluate landscape functions. GIS offer powerful tools for spatial analysis (Openshaw and Clark, 1996; Longley et al., 2001) but are not exclusive to model complex systems spatially. GIS is both a toolbox and methodology at the same time (Pickles, 1997). It needs methodologies to integrate qualitative and quantitative information across spatial and temporal scales. The sophistication and usefulness of GIS is not necessarily proportional to complexity but it is hypothesized that they are in principle codifying and empowering human understanding of nature.

Haines-Young (2000) suggested a new paradigm for landscape ecology based on the concept of natural capital: sustainable landscapes. It reflects the increased human influence on landscapes and the increasing demand to reveal the human population as part of the landscape. Haines-Young claims that current landscape models are mainly science-based and that these models cannot define in any complete sense an optimal or sustainable landscape. He argues that in order to deal with landscape sustainability we must recognise that in any situation there is a whole set of landscapes that are more or less sustainable, in terms of the outputs of goods and services that are important to people. In this paper, I critically discuss this concept, the underlying ideas of ecosystem functions and their valuation and I take up the challenge to juxtapose it to the spatially explicit patch-matrix concept (Forman and Godron, 1986).

This paper discusses several leading issues in landscape science. Namely the importance of understanding concepts, research and applied approaches and methods with respect to their informational characteristics and the potentials for development provided by the recently emerged arena of multi-scale segmentation/object-relationship modelling. It makes a case for this, demonstrating the significance of the spatial, and the limitations of statistical approaches based mainly around a patch-matrix model. Throughout I aim to link these issues to ones of landscape policy planning and sustainable management, in particular the assessment approach of natural capital applied to sustainable landscape planning.

2. The ‘landscape concept’ and some ecological concepts with relevance to ‘Sustainable Landscapes’

2.1. Landscape change: decoupling ‘sustainability’ and ‘development’

Humanity has influenced and dramatically changed at least 90% of Earth’s landscape (Naveh, 2000; Sanderson et al., 2002). The influence of human beings on the planet has become so pervasive that it is hard to find adults in any country who have not seen the environment around them reduced in natural values during their lifetimes. This includes woodlots converted to agriculture, agricultural lands converted to suburban development, suburban development converted to urban areas (Sanderson et al., 2002). The cumulative effect of these many local changes is the global phenomenon of human influence on nature, a new epoch some call the “anthropocene” (Steffen and Tyson, 2001). The patches created (see below) may already depend more on human actions than on natural ecological conditions. In cultural landscapes the ecological and socio-economic realms are intricately linked. We need concepts to predict and to manage future land use but we are just beginning to parameterize issues related to the new economy, changing lifestyles and different priorities in land consumption.

The concept of ‘Sustainable Development’ is relatively well known and often serves as a guideline in spatial planning (Volker, 1997; Webster, 1997; Grossmann, 2000; Botequilha Leitao and Ahern, 2002). It is not comprehensively discussed in this paper. Rather, I concentrate on spatial aspects of sustainability and (spatial) indicators for sustainable landscape management. First, I suggest decoupling sustainability and ‘development’. Decision makers have to decide what we must adapt to, what we must try to control, what we should alter, and what we should leave as is. Development is often implicit in the concept of sustainability, but the goal is no longer progress in the sense of more, further and higher, but a new and qualitatively different set of aims (Grossmann, 2000). Depending on the initial situation, development towards this goal can involve quantitative increase (as will be the case for most of the world’s less developed nations), or drastic cutbacks in material flows and resource consumption as in most industrialized countries. When ‘sustainability’ and 'development' are intrinsically coupled together this implies that there is 'change'. These changes leave footprints on Earth's surface and we will be able to detect and measure the resulting change within the environment. The whole human centred concept of sustainability is centred on the well being of humans, namely to enhance and maintain the well-being of future generations, individuals and communities. Land, and the uses to which it is put, are influenced by almost every area of policy and by every sector of the economy. Conversely, changes in the land have widespread impacts elsewhere, for example, on water quality, on amenity and on nature.

2.2. Spatial matters: natural resource accounting and indicators

Neither increasing nor decreasing resource consumption will automatically lead to a 'sustainable landscape'. For instance, decreasing land use intensity through land abandonment processes will not happen spatially randomly, it might increase or decrease existing disparities in resource consumption. In highly human-influenced landscapes this process must be planned and its methods must be explicit and replicable (Forman and Collinge, 1997). This is particularly relevant in the era of GIS-based spatial analysis, through which methods and procedures can be subjected to rigorous tests of accuracy and replication. A GIS-based planning process is regarded to be explicit and transparent if the assumptions, variables and goals are clearly presented (Ahern, 1999). This, in principle, allows interdisciplinary collaboration and non-expert participation as often realised in the generation of alternative planning scenarios (Forman and Gollinge, 1997; Botequilha Leitao and Ahern, 2002; Tress and Tress, 2003).

The concept of sustainable landscapes does not focus on the analysis of a status quo. It is about what landscapes of the future will look like and how will they function, based on our understanding of current environmental and human conditions. This paper highlights the 'spatiality' of the underlying questions of sustainable land use without neglecting the role of participatory approaches: what spatial patterns and processes will be evident, assuming expected global, regional and local conditions? Only a spatial framework offers foundation for providing answers to these questions: "The spatial solution is a pattern of ecosystems or land uses that will conserve the bulk of, and the most important attributes of, biodiversity and natural processes in any region, landscape or major portion thereof" (Forman and Collinge, 1997, p. 129). Consequently, evaluating sustainability cannot be performed 'a-spatially'. By this, I mean it cannot be performed on administrative units based on statistical reports and national statistics. What is needed is a regionalisation of statistics through an integration of various types of information (measurements, sample data, areal data).

This spatial balancing should combine the classical landscape planning (predominantly for the protection of environmental compartments and recreation properties as zones in landscapes) with a weighting of a distinct, but in terms of environmental protection broad set of environmental indicators. This combination leads to environmental performance indicators or spatially related eco-balances (Pauleit et al., 2005). Fig. 2 illustrates the ability to combine different data types and spatially reconfigure areas for planning. Some of the ecological planning units and procedures (e.g. regions, landscapes, watersheds, land consolidation acts) are of high importance of having state-of-the-art sustainability indicators. The acceptance of the end users has to be achieved by taking on board stakeholders (see Sec-

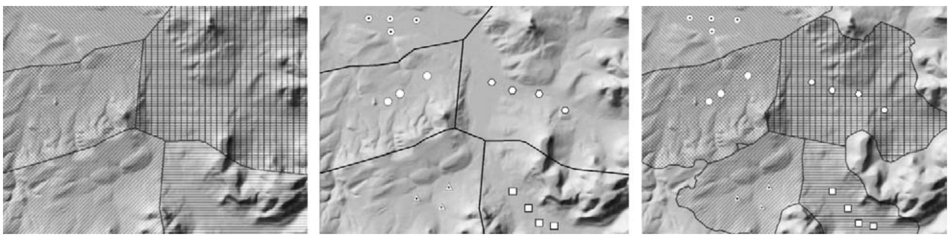


Fig. 2. Example of reconfiguration of socio-economic units (left) using empirical point data (centre) into new spatial units through spatial interpolation methods.

tion 5). The need to report regularly on the state of the environment at the local, national and international levels has become a preoccupation for a number of organizations (Haines-Young, 1999). But we lack spatial approaches to differentiate sustainability in practice in order to maximise natural economic effectiveness and efficiency and maintain a necessary balance among resource accessibility, requirements, and capacity to meet requirements.

There is an increasing need for indicators which capture the links between the economic, social and environmental dimensions. Haines-Young (1999) highlighted the merits of the 'indicator approach' compared to one based on natural resource accounting. He argued that, for 'State of the Environment' reporting to be effective, it must embody a model that describes the processes of land use/land cover change. The indicator approach, as currently employed by several European countries, is a relatively loose framework for achieving such a systematic view mainly based on land cover. Technically, the challenge lies in the integration of 'hard' (measured, mapped, interpolated) and soft data (e.g. well-being of people living in a landscape), of spatially explicit measurable data and vague but important concerns of people living in landscapes. Methodologically, different scientific areas and sub-models are to be incorporated, so that it is possible to predict and evaluate the environmental outcomes of alternative courses of action for policy. GIS provide the analytical tools and methodologies for spatial integration of the different scientific areas and sub-models. The spatial organization of the composing elements is a key to describe the functions and processes within a landscape. If the composition changes their connecting relationships will change too, since the functioning of a landscape and its structure are intimately related (Forman and Godron, 1986; Dramstad et al., 1996; Antrop, 2000; Turner et al., 2001; Nagendra et al., 2004).

2.3. *Spatial heterogeneity and establishing relatively homogeneous spatial units*

Ecosystem structure and function are essential to understanding in sustainable landscape planning (Forman, 1995; Ahern, 1999; Botequilha Leitao and Ahern, 2002). The understanding of the dynamic relationship of landscape pattern to process is therefore fundamental (Turner et al., 2001; Bissonette, 2003). A common proposition leading to the patch-matrix (-corridor) paradigm (Forman and Godron, 1986; Forman, 1995; Wiens, 1995), is that landscapes are considered to be mosaics of smaller entities, mostly called patches, which are relatively homogenous in a sense that the 'within-patch-heterogeneity' of some phenomena under consideration are less than the differences to its surroundings. The patch-mosaic paradigm evolved rapidly in North America (Krummel et al., 1987; O'Neill et al., 1988) and dominates landscape research (Bastian, 2002; Wu and Hobbs, 2002). European schools of landscape ecology developed different approaches, e.g. the large scale analysis methodology in the German Democratic Republic with its conception of natural landscape units, called geochores. The geochores as geographically defined units were regarded as associations or mosaics of basic topic elements. Tope or topic refers to a particular locality. One important feature of these geochores is their heterogeneous structure. The properties of choric spatial units result from the association of combinations of topic elements, as well as their arrangement in space. Finally, on a higher level of aggregation, geochores have new properties beyond the mere sum of the parts and are regarded as functional regions. Both approaches, the 'North American' and the 'Central European', focus on spatial heterogeneity and regard it as a measurable expression of the overall spatial complexity or variety of an area. Both approaches rely on mappable basic spatial units at a given scale. "Nanochores have a simple pattern of arrangement of the topes with a nearly homogeneous form of spatial relations and interlinkages between them" (Haase, 1989, p. 31).

Patchiness refers to a particular spatial pattern: bounded elements in a background matrix. Although the specific arrangement of patches may take a variety of forms, the basic structure is the same, and it is relatively well-defined. In contrast, any form of spatial variation, from an unbounded gradient to a collection of various patch types (and including a simple patch-matrix system) is heterogeneous. For sustainable landscapes it is concluded that we do not know the exact amount of heterogeneity necessary at a landscape level but we assume that a certain degree of heterogeneity is needed. The delineation of relatively homogeneous areas is a prerequisite for most approaches in landscape research. Two main approaches can be distinguished: to optimize homogeneity or to concentrate on discontinuities assuming that the areas between discontinuities are relatively homogeneous. Since the 1970s, dozens of techniques have been proposed for edge detection in remote sensing and GIS data sets (Pitas, 1993). Significant progress has been achieved in image analysis with a focus on edge enhancement (visually emphasizing the boundaries in a picture) and edge detection. In remote sensing, heterogeneity is often reduced to 'texture'. Texture can be characterized e.g. through a structural approach, an image is assumed to be composed of primitive elements (pixels) that can be characterized in groups by their shape and size as well as their pattern of repetition. However, because image processing encounters problems similar to those met in field ecology (i.e., misclassification or unclear repetitive patterns) statistical approaches are often preferred including autocorrelation functions, autoregressive models, spatial intensity co-occurrence probabilities, and structural element filtering in field data, for reducing noise and for the ability to detect small edges (Fortin et al., 2000). Alternative approaches to overcome the problem of scale-dependency include fractal geometry (Krummel et al., 1987; Milne, 1988) or lacunarity analysis (Plotnick et al., 1993; MacIntyre and Wiens, 2000).

2.4. Landscape, scale and hierarchy

Scale is a key issue in sustainable planning (Ahern, 1999). Conventionally, the hierarchy of scales (Allen and Starr, 1982) refers to organizational levels: cell, organism, population, ecosystem, landscape, biome, and biosphere. Due to the interdependencies of ecosystems, a planning approach is needed that examines a site in its broader context. The landscape provides an approximately useful context for sustainable planning (Forman, 1995). Scale and scaling became increasingly popular in ecology in recent years (Allen, 1998; Peterson and Parker, 1998; Marceau, 1999; Hay et al., 2002) as research has shifted from local to broader scales and many environmental and resource management problems can only be dealt with effectively at broad scales (Ahern, 1999; Botequilha Leitao and Ahern, 2002). In order to develop fuller understanding of landscape process we must understand broad-scale patterns and processes and relate them to those at fine scales with which we are most familiar. In both cases, transferring information between scales is essential (Wiens, 1995; Allen, 1998; Wu, 1999) but difficult. It requires upscaling and downscaling techniques (Hay et al., 2001; Wu, 2004). For example, contrast properties of heterogeneous habitat template in general compared to heterogeneity of a template defined at a particular scale are scale-independent, but heterogeneity of a specific template is scale-dependent and changes with altered scale (Kolasa and Rolo, 1991). Generally, extrapolation of experimental results from fine to broad spatial scales can be fraught with problems (Murphy, 1989). While fine-scale experiments may yield some useful information of relevance to broad-scale processes, its validity in broad-scale contexts must always be carefully assessed (Hobbs, 1999).

For several understandable reasons, landscape ecologists and planners highlight 'the landscape scale' which refers to a particular geographic extent (Lavers and Haines-Young, 1993). It is said to be appropriate for sustainable planning (Ahern, 1999) because it is sufficiently large enough to contain a heterogeneous matrix of landscape elements that provide a context for mosaic stability (Forman, 1995). Ecologists discuss more critically the term landscape scale, especially if it is used synonymously with the term 'landscape level' (Allen, 1998; King, 1999). King (1999) and other ecologists reject the hypothesis that there is a 'landscape scale' and that a particular scale is inherent to the concept of a landscape. But landscape planning needs a particular scale which corresponds to a 'window of perception' (Hay et al., 2001). Lavers and Haines-Young (1993, p. 65) provide a relative flexible definition: "The 'landscape scale' is simply that at which one considers the pattern and interaction between the various mosaic elements of patch, edge and corridor".

Holling (1992) demonstrated that landscapes are generally structured according to scaling regions with distinct dimensions connected by transition zones. Krummel et al. (1987) developed a method for detecting distinct scales of pattern for mosaics of irregular patches using fractal analysis (Mandelbrot, 1983). Most appropriate methods to detect discontinuities and characterize boundaries depend of the spatial resolution and the measurement type of the data. Only recently, methods were developed to detect spatial scales (Lindeberg, 1994; Fortin et al., 2000; Grossi et al., 2001; Hay et al., 2002) but these methods are not able to explain hierarchical relationships between the scales. Certain scales are related to specific research questions and vice versa (Table 1). Hierarchy theory

Table 1
Different levels of investigation, related research questions and example methods with an emphasis on the landscape level

Scale	Research question	Methods and indicators
Landscape level	Landscape diversity	Indices of landscape patterns
		Historic reference conditions
		Remote sensing and GIS
	Habitat availability and distribution	Indices of landscape patterns
		Historic reference conditions
		Remote sensing and GIS
	Changes in landscape elements	Indices of landscape patterns
		Historic reference conditions
		Remote sensing and GIS
Community or ecosystem level	Management actions or natural disturbance affects on species diversity	Species diversity indices
	Function of species in community or ecosystem	Functional group and guild analysis
	Level of protection in areas with high species richness	e.g. Gap analysis
Species/population level	Species-population trends	Abundance indices
		Population estimates
	Anthropogenic or natural disturbance	Abundance indices
		Population estimates
	Probability of species or population persistence	Population viability analysis
Genetic level	...	...
	...	...

(Koestler, 1967; Allen and Starr, 1982; O'Neill et al., 1986) is a way of ordering observational scales in a way that draws attention to the mechanisms and constraints that operate at a given level and how these change among levels. The theory does not maintain that every ecological system must necessarily be hierarchical. Rather, it points out that stable complex systems often take on such a structure (O'Neill et al., 1986). Hierarchies, however, are artificial constructs that we impose on nature: we categorize phenomena into levels that are logically related (Burnett and Blaschke, 2002). These categories cannot reflect the true complexity of interactions within natural systems. Hierarchical levels may be seemingly clearly defined when we are dealing with units such as cells, organisms or populations, but they are not intuitively apparent when we are dealing with spatial variation.

2.5. Landscape composition and ecosystem functions

The complexity of landscapes is largely determined by the number of ecosystem types, their characteristics (e.g. in terms of structure and functioning), their size and shape, and their connectivity (Forman, 1995). A large amount of evidence suggests that complexity at the landscape scale may have large consequences on regional to global scale processes. For instance, the presence and arrangement of keystone ecosystem types such as wetlands or riparian areas often determine total carbon and nitrogen balance of a region. Arising research questions include: (i) what effect does landscape complexity have on ecosystem functioning at large scales? (ii) how do these relationships between complexity and functioning vary with ecosystem processes of interest? and (iii) how will global change affect landscape complexity, and in turn ecosystem functioning?

Ecosystem functions are divided into four categories according to De Groot (1992): regulation, habitat, production and information functions. Marks et al. (1992) developed a conceptual framework and practical guidelines for the assessment of the fulfilment of specific functions by a particular landscape. A function of a given landscape unit is understood as the performances and tasks the landscape ecosystem is fulfilling. Their concept is based upon experiences in geoecological mapping. Complex ecosystem processes can be divided into the transfer of matter and energy across landscapes and between the land surface and the atmosphere. Examples of such processes include biogeochemical cycles, surface hydrology and water, and energy exchange. They deal with the interaction of landscape patterns and regional disturbance regimes. For example, landscape patterns influence the spread of fire and the probability of logging, which in turn govern large-scale ecosystem functioning. Although matter and energy exchange, movement of organisms, and disturbance are treated largely independently, there are interactions between them (e.g., migration across landscapes of new functional types which affect nutrient cycling and probability of fire). Changes in the structure of the landscape can have ecological effects such as modifying nutrient transport and transformation (Hobbs, 1993; Mander et al., 2000) and affecting species persistence and biodiversity (Dale et al., 1994; Opdam et al., 2003; Jentsch et al., 2003). While ecosystem processes are more and more understood, we have little knowledge about the material and energy flows between ecosystems and how the arrangement of ecosystems in space control the ecological patterns and processes that result.

3. Spatial analysis and the spatial dimension of sustainable landscapes

3.1. GIS

In the late 1980s and throughout the 1990s research frameworks and applications in various disciplines evolved which emphasize spatial relationships. Some of this growth has led to increased sophistication in the description of spatial patterns (McGarigal and Marks, 1994; Forman, 1995; Gustafson, 1998; Turner et al., 2001), aided by more powerful spatial statistics (Liebhold and Gurevitch, 2002), techniques for detecting patch boundaries (Fortin et al., 2000), Geographic Information Systems (Burrough and McDonnell, 1998; Longley et al., 2001) and remote sensing. These developments led to the foundation of Geographic Information Science (‘GIScience’, Goodchild, 1992; Longley et al., 2001). The power of a GIS is its ability to synthesize information about spatial phenomena by integrating geo-referenced data to show the original data and derived information in new ways and perspectives. As computational power has increased, modellers have developed simulation models that incorporate spatial variation by allowing individual cells in a spatial grid to undergo dynamics that are spatially linked in various ways. Cellular automata are one version of such models, and some individual-based models incorporate a spatial dimension as well (e.g. Takeyama and Couclelis, 1997). Much of the recent growth in spatial theory has involved elaboration and extension of patch-based population models (e.g. metapopulation theory Hanski and Simberloff, 1997). The development of certain new, specialized statistical metrics has been motivated by the emerging field of landscape ecology, which focuses on spatial processes operating over various spatial extents (Forman and Godron, 1986; Krummel et al., 1987; Turner, 1990).

Patch-based population theory has developed in several ways and a variety of patch arrangements and interactions have been incorporated into models (Turner et al., 2001) but also spatially explicit individual based models are readily available today (e.g. Topping and Jepsen, 2002). They are by no means restricted to ecological parameters: innovations in simulation have been supported by conceptual developments in articulating dynamics and complexity and technical developments in GIS and object-oriented programming, coupled with increases in the availability of high-resolution data sets. These models emphasize spatial, disaggregated, flexible, dynamic, and more realistic approaches for example to modelling urban systems (Batty and Torrens, 2001). With theoretical roots in artificial life, non-equilibrium physics and computational economics, cellular automata and multi-agent systems are being rapidly developed for simulating cities (Torrens and O’Sullivan, 2001). Cellular automata are generally well suited to the simulation of urban infrastructure (land-use transition, real estate development and redevelopment and urban growth) while multi-agent systems are more appropriately applied to population modelling (residential location, pedestrian movement and traffic simulation). These tools all enhance our ability to test spatially explicit theory and are powerful tools to analyse ecological events in a spatially explicit framework (Grimm, 1999). For a critical review of spatial statistics see Openshaw and Clark (1996) or Liebhold and Gurevitch (2002).

GIS have emerged over the last 20 years as an effective tool not only for analysing spatial data but also for evaluating resource management alternatives (Hermann and Osinski, 1999; Kangas et al., 2000;

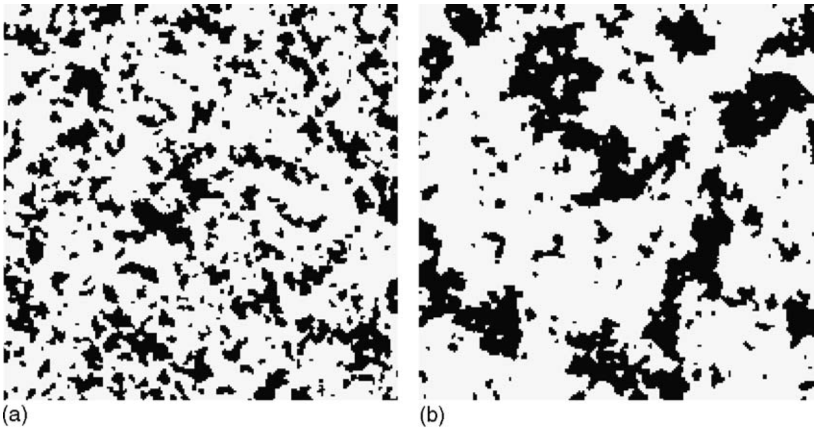
Appleton et al., 2002; Seppelt and Voinov, 2002). One of its strengths is that it allows us continually to reconfigure the data in ways that are most appropriate for our changing needs and points of view. Spatial statistics tend to perform the same sort of task, but in a more abstract way, allowing us to make generalizations about what we see in the data, to extract hypotheses from them, or, finally, to use them to test hypotheses. This is a positive view on the potential of GIS. In reality, in many cases data are simply stored and processed in a GIS centred around the patterns of land cover and land use, and of social, economic, and demographic characteristics. But these conditions constantly change, both because the spatial structures represented are themselves inherently unstable, and because they are typically exposed to external phenomena that also force change. Today, environmental data collection programmes are designed for periodic data collection and updating, typically in a manner that provides a regularly updated picture of current conditions. Planners and decision makers need to know not only the current state of affairs but also require some idea of future conditions. Ideally they would like to be able to see the possible consequences of the plans and policies they may have under consideration. This is often realised in a finite set of scenarios or through one of the many different predictive computational modelling techniques available (Seppelt and Voinov, 2002). The latter mainly use regular tessellations like regular grids or lattices and support the search for 'optimal' spatial decisions (Martinnez-Falero et al., 1998; Makowski et al., 2000; Seppelt and Voinov, 2002).

Remotely sensed images are essential data sources for landscape analysis. Aerial photography and satellite imagery are useful in characterizing landscape because variation in the image is usually highly correlated with variation in the landscape. No other survey technique can operationally provide a regularized survey of landscape with which to assess landscape level patterns and change. However, remotely sensed images, like all observations of reality, are an imperfect capturing of patterns, which are themselves an imperfect mirror of ecosystem and human-induced processes (Burnett and Blaschke, 2003). Until recently, most effort went into the exploitation of pixel characteristics rather than into the exploration of pattern and only for very specific purposes (e.g. automated road extraction from high-resolution imagery) applications of operational information extraction from raw data exist (Blaschke and Strobl, 2001). Forest inventory and agriculture routinely use information from satellites and these technologies are built into scenario development. But, for example, surveys of past processes remain largely outside the investigative capacity of recent spatial technologies.

3.2. Landscape analysis and spatial indicators

Data on land use, land use intensity and land use diversity are important indicators for sustainable land management. Many spatial indicators are only suited to describe specific aspects of spatial patterns. The common indicators are often called 'landscape metrics' (Gustafson, 1998; Blaschke, 2000; Botequilha Leitao and Ahern, 2002). A better way would be to distinguish between metrics (neutral values) and indicators (some meaning attached to it through definition and possible range). An initial and seemingly straightforward question is whether the patterns of two maps are similar (see Fig. 3). The palette of geostatistical tools to answer this is powerful today but, like classic statistics, is not automatically protected against pseudo correlations. Approaches include autocorrelation analysis or the use a polygon-by-polygon comparison, in which various characteristics of the polygons can be compared, starting with the extent to which polygons on one map coincide with those on the other, but also including other measures of polygon similarity such as size and compactness. An example for landscape metrics is adjacency. If we analyse in the context of sustainability how much neighbouring conflicts (e.g. between areas of high nature conservation value and intensive agricultural use) a certain area is opposed to, we have to define thresholds of potential impact. In other words, if we are able to set thresholds for these spatial conflicts, we have the tools at hand for determining a rational use of existing resources that anticipates the possible long-term effects on the environment of the decisions taken (Hermann and Osinski, 1999; Blaschke, 2000; Turner et al., 2001, Appleton et al., 2002; Botequilha Leitao and Ahern, 2002; Verburg et al., 2004).

Interspersion/Juxtaposition Index, Contagion Index or Cohesion Index and other indices compare adjacency frequencies between classes and describe connectivity (Schumaker, 1996; Wu et al., 1997; Gustafson, 1998). A variety of examples exist, many of



	# patches	Patch density	Mean patch size	Area weight. mean patch size	Edge length	Mean shape index
a	318	0.80	125.8	19955.4	9393	1.37
b	165	0.41	262.4	19907.2	5404	1.32

Fig. 3. Selected landscape indices describing two different subsets of a habitat map. The map is aggregated in a binary classification with black colour indicating suitable habitat. Both subsets comprise about 70% non-suitable area (white) and 30% black areas. Only few indices are sensitive to changes in pattern.

them exhibit a very simple method of counting edges and patches. The amount of edge between each land use is usually determined by summing the number of interfaces between adjacent cells of different land uses, then multiplying by the length of a cell or by using polygon data to represent the land uses and measuring adjacency directly. The amount of edge between all categories can be statistically analysed and may provide insights regarding the degree of convolution of edges and consequently about the potential for species dispersion from a patch since in general a straight boundary tends to have more species movement along it, whereas a convoluted boundary is more likely to have movement across it (Forman, 1995; Dramstad et al., 1996.). The complexity of patch perimeters is measured using fractal dimensions (Mandelbrot, 1983), which can be used to compare the geometry of landscape mosaics (Milne, 1988). Some problems with landscape metrics are illustrated in Fig. 3. The two graphics exhibit a totally different appearance even if both classes (for the sake of simplicity aggregated to black and white classes 'suitable habitat' and 'not-suitable habitat') cover about 30% suitable habitat (black) in both images but only few metrics react to these differences (Wu et al., 1997; Hargis et al., 1998; Tischendorf, 2001; Wu, 2004). Another often underestimated problem is the 'study area bias' (Blaschke and Petch, 1999): depending on the delineation of the study area boundaries significant differences may occur (Saura and Martinez-Milan, 2001).

What have been described so far are mainly descriptive, structural indicators which measure the physical composition or configuration of the patch mosaic without explicit reference to an ecological process. Resulting figures are of limited use in the context of sustainable development, but landscape indices have been applied to compare heterogeneity between different landscapes (O'Neill et al., 1988; Turner, 1990; Hulshoff, 1995; Pan et al., 1999), to predict response variables of ecological processes (Wiens et al., 1993; Schumaker, 1996; Mander et al., 2000) and specific aspects of the survival of populations in heterogeneous landscapes (Dale et al., 1994; Wiens et al., 1997; With et al., 1997; Fahrig, 1998).

Many studies revealed significant statistical relationships between landscape indices and dependent variables, suggesting the general potential of landscape indices to predict ecological processes (Tischendorf,

Table 2
Examples of widely used structural indicators and corresponding research questions

Criterion for structural assessment	Parameter/key issue	Corresponding question	Metrics employed
Area characterization	Area sensitivity of species	What is the area size of the respective habitat type?	Class area (CA)
		What is the average patch size and how are the values distributed?	Mean patch size (MPS), Patch size standard deviation (PSSD)
		How many patches comprise a respective habitat type?	Number of patches (NP)
Core area and ecotone analysis	Size and number of core areas/effective areas for edge-sensitive species	How large is the ecologically effective area for edge-sensitive species in the entire landscape?	Total core area (TCA)
		How large is the core area of all patches of a given habitat type?	Total class core area (TCCA)
		Of how many disjunct core areas are all patches of one class comprised?	Number of core areas (NCA)
		Which percentage of the patch is core area?	Core area index (CAI, landscape and classes)
		What is the degree of decimation? Does a core area exist and in how many parts is it split?	Cority
Structural ‘richness’/ fragmentation	Density of linear elements/structural richness vs. fragmentation (quality to be considered)	How much of a landscape or patch type is comprised of edges?	Total Edge (TE)
		What is the density of edges in a hectare?	Edge density (ED)
		What is the average length of edges within all patches of a patch type?	Mean patch edge (MPE)
Form description	Compactness (optimized interior)	How compact are the patches in average (in comparison to a circle)?	Mean shape index (MSI)
	Edge-interior-ratio	How large is the patch in relation to its edge?	Mean perimeter-area ratio (MPAR)
	Complexity of shape	How complex or irregular is the form of the patches?	Mean fractal dimension (MFRACT)
Connectivity/isolation	Distance (usually Euclidean)	How distant is the next patch of the same habitat type?	NNDIST, NNID
	Maintenance of metapopulation/functional connectivity, integrity/connectivity vs. connectedness	Ecological importance of neighbouring patch of the same habitat type within specific dispersal range.	PX92
		How well is a specific patch integrated in the arrangement of neighbouring patches?	PX94 PXFRAG
Landscape diversity	Proportion, distribution and dominance of habitat types	What is the percentage of a specific habitat type in the landscape?	Proportion
		How many classes are represented in the landscape?	Relative richness
		What is the amount of ‘information’ per patch within the landscape?	Shannon’s diversity
		How is this information distributed?	Shannons’s evenness
		Is a habitat type dominating?	Dominance

Table 2 (*Continued*)

Criterion for structural assessment	Parameter/key issue	Corresponding question	Metrics employed
Subdivision	Fragmentation/dissection/isolation of remnants	What is the remaining degree of coherence, i.e. how likely are two randomly chosen locations not part of the same un-dissected patch?	DIVISION
		How many patches remain at a given degree of division?	SPLIT
		What is the average size of those?	MESH

Index nomenclature according to McGarigal and Marks (1994) and Jaeger (2000).

2001). However, relationships between landscape indices and response variables of ecological processes may be non-linear (With and Crist, 1995; Wiens et al., 1997; Hargis et al., 1998; Blaschke and Petch, 1999), including thresholds at which ecological processes may change dramatically (Wu, 2004). For instance, the survival probability of a population may severely decrease after a certain proportion of habitat is removed from the landscape (Fahrig, 1998). The measures which Hargis et al. (1998) examined were relatively insensitive to variations in the spatial arrangement of patches on a landscape. Mean nearest neighbor distance and mean proximity index both quantify distances between patches in a cluster, but neither are designed to place the cluster in the context of the landscape window. Edge density, contagion, and perimeter-area fractal dimension are all metrics of landscape pattern caused by size and shape of patches and their proportional representation on a landscape, but none can differentiate the spatial relationship among patches (for an overview of structural metrics see Table 2). Recently, indices were introduced which are more sensitive to the spatial arrangement (Jaeger, 2000; Fjellstad, 2001; Ludwig et al., 2002; Verburg et al., 2004). For the discussion of sustainable landscapes it is concluded that landscape metrics provide mainly descriptive spatial values for a limited set of spatially dissected realities but are less suited for continuous processes through space and time.

3.3. The need for functional indicators

Applications of structural indices often focus on vegetation or habitat structure since vegetation is relatively easy to map. The measuring of edge can for instance be used to create the greatest amount of interface between upland vegetative community and the aquatic environment to facilitate the exchange of energy and nutrients between these two systems. Beyond the investigation of species presence, abundance or dispersal, the usefulness of landscape metrics for the understanding of interactions between landscape balance and land use at various levels of scale is increasingly investigated, namely addressing processes of soil erosion, ground water recharge, surface runoff of material into river and lakes, using remotely sensed data and complex GIS-based models (Thierfelder, 1998; Wrbka, 1998; Zhang et al., 1998).

Many examples demonstrate that spatial configurations can in principle have functional significance in spatial systems. The functional relevance of the computed value is left for interpretation during a subsequent step. What are needed are functional landscape metrics (Vos et al., 2001; Turner et al., 2001; Opdam et al., 2003) which explicitly measure landscape pattern in a manner that is functionally relevant to the organism or process under consideration (Vos et al., 2001; Bissonette, 2003). This requires additional parameterization prior to their calculation, such that the same metric can return multiple values depending on the user specifications. For instance, mean nearest neighbour distance is based on the distances between neighbouring patches of the same class. The mosaic is in essence treated as a binary landscape (focal class versus everything else) and a single value for this metric is returned (McGarigal and Marks, 1994; Wiens, 1995). This is a structural metric because the functional meaning of any particular computed value is left to subsequent interpretation. Conversely, connectivity metrics that consider the permeability of various patch types to movement of the organism or process of interest are functional metrics. There are an infinite number of values that can be returned from the same landscape, depending on the permeability coefficients assigned to each patch type. The computed metric may be functionally relevant only for a particular parameterization. In most empirical studies conducted, most species attributes are linked to landscape pattern using single-year distribution or turnover patterns (Reich and Grimm, 1996; Harrison and Taylor, 1997). These produce regression models that usually are hard to extrapolate to other landscape areas and to the long-term chance of persistence (Ter Braak et al., 1998; Vos et al., 2001). At a landscape level, Mander et al. (2000) presented some examples of ecological consequences due to the ongoing changes in land use and land cover in Estonian agricultural landscapes during the 1990s. MacIntyre and Wiens (2000) demonstrate a use of the lacunarity index (Plotnick et al., 1993) to quantify landscape function and describe a disparity between landscape pattern and landscape use. Zebisch et al. (2004) have proved response functions of biodiversity attributes built on spatial heterogeneity to be particularly useful to explaining the impact of land-use shifts on biodiversity. Ludwig et al. (2002) create a leakiness index to differentiate the landscape function to retain vital system resources such as rainwater and soil.

3.4. A multiscale object-based GIS framework

The patch-matrix paradigm is widespread but it has its limitations to model complex landscapes especially through time and across scales. The discussed landscape metrics focus on one scale or a few data sets at different scales and at given sets of spatial arrangements. The relationships between scales are difficult to model, although some examples for upscaling and downscaling were mentioned. As Koestler (1967) points out, when we turn from the universe in miniature (e.g. the cell) to the universe at large, we find hierarchic order. He concludes that whenever we find orderly, stable systems in nature, we find that they are hierarchically structured. Without structuring of complex systems into sub-assemblies, there could be no order and stability. Wu and Loucks (1995) suggest the integration between hierarchy theory and patch dynamics via the hierarchical patch dynamics paradigm and lay a theoretical framework for a theory-driven breaking down of ecological complexity through a hierarchical scaling strategy. Wu (1999), drawing on Koestler (1967) concepts of flux rates in hierarchy, suggests that ecological systems are nearly completely decomposable systems because of their loose vertical and horizontal coupling in structure and function. The term “loose” suggests decomposable, and the word “coupling” implies resistance to decomposition. Following these ideas we accept scale non-linearities as discussed before. Koestler (1967) introduced a methodology which translates hierarchy theory to landscape ecological analysis: holons (from the Greek word *holos*) are then used synonymous with patches: the ecological unit at a particular scale of consideration. It is at the same time a (sub)whole and consist of sub-wholes of a lower order.

Building on Koestler’s ideas, the concept of sustainability will not be satisfactorily supported by a ‘thematic layering’ of information. But how to define meaningful levels and how to dissect reality spatially at these levels? Burnett and Blaschke (2003) build in their approach on Koestler’s ideas of multi-levelled hierarchies. Like societies they treat landscapes as multi-levelled hierarchies of semi-autonomous sub-wholes branching into sub-wholes of a lower order. The term ‘holon’ (Koestler, 1967) refers to these intermediary entities which, relative to their sub-ordinates in the hierarchy, function as self-contained wholes, relative to their sub-ordinates as dependent parts.

The multiscale segmentation/object relationship modelling methodology suggested by Burnett and Blaschke segments information (usually remote sensing images plus any georeferenced information). Generally, an advantage of segmentation to classification of pixels is that the resulting division of space tends to involve fewer and more compact subregions. Alternatively, classification-based approaches within geostatistical frameworks exist (Atkinson, 2001). The multiscale segmentation based approach is designed to utilize information in the scales inherent in our spatial (image) data sets in addition to a range of auxiliary data sets, including for airborne and satellite data, but also to the scales of information inherent in single images. Technically, segmentation is not new (Haralick and Shapiro, 1985). Recently, many new segmentation algorithms and applications have been tested in GIScience applications, but few of them lead to qualitatively convincing results while still being robust and operational (Blaschke and Strobl, 2001). The multiscale segmentation based methodology uses

Table 3
The multi-scale segmentation/object relationship modelling methodology for landscape analysis of Burnett and Blaschke (2003)

Step (1) GIS building	The main prerequisite is the collating of geographic information into database of geo-referenced survey, sample and auxiliary data. Survey data includes any systematic and continuous assay of landscape, e.g. digital aerial photograph mosaics, airborne spectrometer swathes and satellite images. Sample data may include distribution and habitat data from bird and insect investigations, or the distribution and species of dominant trees. Auxiliary data include other data sets which could be considered to be part of either category, for instance derived vector data such as topographic contours, road network and cadastral information, and raster digital elevation models (DEM). All three types of spatial data are geo-referenced, stored and visualized using any GIS.
Step (2) Segmentation	The multi-scale segmentation, searches for the gradient of flux zones between and within holons (patches): areas where the varying strengths of interactions between holons produce surfaces. Multi-scale segmentation equates to searching for changes in image object heterogeneity/homogeneity.
Step (3) Object relationship model building	A model of the relationships between the segmented image objects is built. Some object relationships are automatically derived. For instance, the characteristics of level –1 objects (such as mean spectral values, spectral value heterogeneity, and sub-object density, shape and distribution) can be automatically calculated and stored in the description of each level 0 object. Other relationships are semantic, requiring the knowledge of the expert on the landscape in question. This relationship model information is stored in the system through a variety of mechanisms, for example as attributes in GIS vector objects or in a proprietary object-orientated database format.
Step (4) Visualization	The output of the object relationship mode is usually a map which emphasizes some objects and relationships over others depending on the research question. For instance, in an urban forest example, the visualization rules can be designed to hide sub-objects below certain super objects (e.g. houses, roads), while showing deeper levels of object hierarchy within ‘forest’ and ‘agriculture’ super-objects.
Step (5) Quality assessment	Quality assessment is essential, both at the final stage when a visualization (map) has been derived from the system, and at each of the preceding stages. Derived data sets, e.g. those generated by algorithms that search for dominant tree crown positions, must also be assessed for error.

the aforementioned hierarchical patch dynamics theory (Wu and Loucks, 1995) for guidance. The five components are described in Table 3. It can, on demand, produce candidate discretizations of space i.e. maps. The initial GIS database building stage can be considered as quasi-independent of specific research questions. With a modicum of change (in segmentation levels, relationship model and visualization rules), the same system can be tuned for a variety of different needs. This way, the multiscale segmentation based approach is very flexible and can embrace any kind of spatial information (Fig. 4). The methodology is also relatively reproducible, compared to human interpretation. The methodology provides some feedback on uncertainty in the classification, and through its ‘modelling nature’ provides for an examination of what aspect of the system, whether data or heuristic, is weakest. Burnett and Blaschke (2003) demonstrate the applicability of this methodology. More specifically, they produce visualizations of the landscape with discretization of roads, settlements, forest and pasture elements. Within the object relationship modelling step, the ‘within patch heterogeneity’ measure (mean spectral difference between all sub-objects) was successfully applied to characterize shrub encroachment on pastures of a European cultural heritage landscape in Germany. Burnett et al. (2003) extended the methodology to mire mapping and monitoring. Hay et al. (2003) applied three different multiscale analysis methodologies to characterize a forested landscape in Canada. It is concluded that this methodology is an alternative or extension to the patch-matrix model since it embraces multiscale analysis and modelling.

4. The natural capital/sustainable landscapes paradigm

4.1. Spatial planning for sustainable landscapes

There are multiple dimensions to sustainability including, economic, social, ethical and spatial. Ahern (1999, p. 175) points out that landscape planning is most fundamentally linked with the latter, the spatial dimension and “predominantly at the scale of the landscape”. Despite the concerns of ecologists (see Section

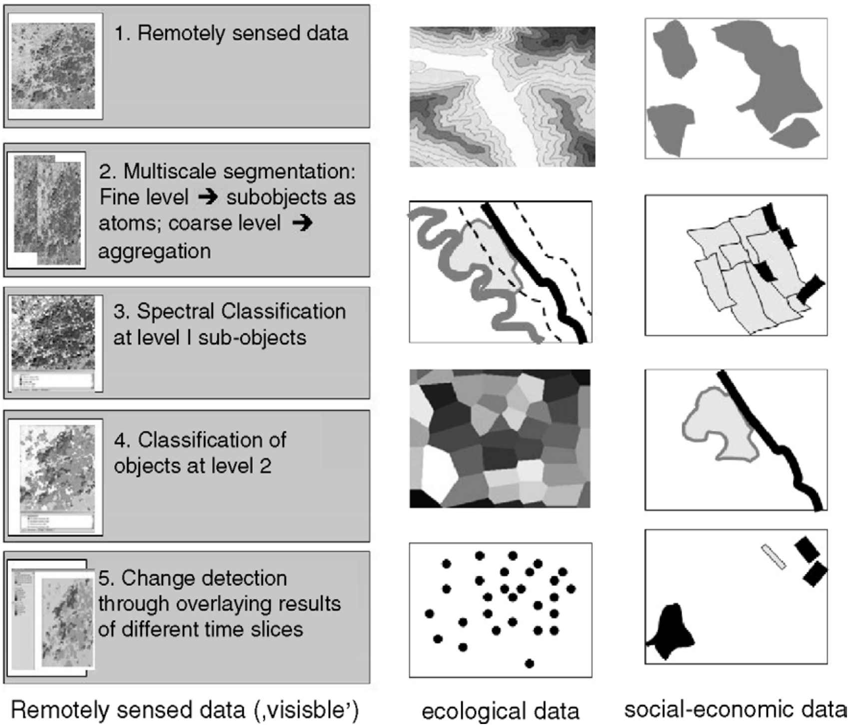


Fig. 4. Integrating socio-economic data within the multiscale segmentation methodology.

2.4) it will not only pragmatically but also legally be the era of landscape planners. Although defined functionally and more precisely the ecosystem is by definition vulnerable to irrevocable disturbance and is therefore, as Ahern (1999, p. 176) concludes, not an appropriate scale for landscape planning. The ecosystem is a useful spatial unit but it is spatially too limited to understand the "horizontal" or chorological patterns and processes (Zonneveld, 1995). Landscapes, more so than ecosystems, are spatially heterogeneous and are spatial arrangements of a number of different ecosystem types that may be linked through various energy and material flows (Potschin and Haines-Young, 2001). Although not measurable directly, these spatial parameters are to be included in land use planning through modelling and interpolation methods.

Natural resources and economic activities are interlinked to highly complex levels. The environmental characteristics of an area affect the amount of materials, effort and money required to yield a particular product or economic award. This suggests that there should be spatial patterns to the sustainability of the landscape to support different land uses at different times, and that these patterns should coincide to a large extent with spatial patterns in environmental characteristics. Individual plans for certain natural resources and regions are often prepared (e.g. river basin management plans, biodiversity management plans). However, landscapes are increasingly regarded as 'multi-functional' (Brandt and Vejre, 2003). In order to encompass some natural boundaries the scale of a plan may be such that local people find it difficult to relate to. This increases the need for a more anthropogenic perspective as discussed below and a multiscale perspective (Hay et al., 2002, 2003; Wu, 2004).

Landscape ecology has achieved significant progress in defining spatial elements and a framework based on them (Forman, 1995; Zonneveld, 1995; Turner et al., 2001). Although alternatives exist (e.g. gradients, Müller, 1998) it seems to be widely accepted that spatial planning needs tangible elements, both from a legal and a pragmatic point of view (Hermann and Osinski, 1999; Botequilha Leitao and Ahern, 2002). One of the remaining constraints for a direct application of landscape ecological concepts in practice is the lack of agreed ways to combine environmental, socio-economic and societal/cultural views. While environmental data are mostly measurable for points or areas (precipitation, temperature, altitude, exposition, soil organic carbon, ground and surface water, air pollution), the spatial dimension of social implications, cultural heritages or the well being of people is more difficult to parameterize. This integration is necessary. Landscape should be considered as holistic, relativistic and dynamic (Antrop, 2000; Naveh, 2000). The term 'landscape' is used as an abstract concept, but also to refer to a particular area and, implicitly, to a particular scale as discussed before. Politicians and large parts of the population refer largely to the aesthetic values attached to it. We frequently talk about specific cultural monuments or nature sites. The landscape scale seems to be of vital importance at a broader level for planning and managing processes of sustainable development. It is about what we see but also about the processes that have created what we can see. Thus, it is about the beauty of Tuscany or the Scottish Highlands and, inherently, it is about the ubiquity of pattern and the uniqueness of specific patchiness in some factor of interest at some scale (Forman, 1995; Turner et al., 2001; Bissonette, 2003).

Currently, we are lacking methodologies which incorporate processes of human settlement and agriculture and the natural forces that have created a landscape into spatial planning. Landscape is something that has great public appeal: people possibly find it hard to understand the more obscure aspects of resource protection or biodiversity but they feel very strongly about issues to do with landscape, mainly at the local level, where many communities have a strong sense of place. Current methodological frameworks based on the patch-matrix paradigm are limited in reflecting these needs.

4.2. The sustainable landscapes paradigm

For landscapes, more so than for ecosystems, it is widely acknowledged that social, economic and cultural factors are key to sustainable management. To some degree it is the people living there which make a difference (Haines-Young, 2000; Buchecker et al., 2003). Their interests have to be incorporated in planning but their interests have also to be limited if it leads for instance to an overconsumption of resources. Haines-Young (2000) addresses this need for a more flexible and more dynamic consideration of environmental and economic interference by suggesting the natural capital/sustainable landscape (subsequently: "sustainable landscapes") approach. He builds on the "Quality of Life Capital" approach mainly developed in the UK (Countryside Commission, 1997; Haines-Young, 1999) and an economic valuation of ecosystem functions and services. The economic value of ecosystems is generally acknowledged. Costanza et al. (1997) described a methodology and a worldwide valuation of ecosystem services. de Groot et al. (2002) suggested a typology and integrated framework for assessment and valuation of ecosystem functions. The proposed framework, in combination with a comprehensive data base of ecosystem services and values, can help identify information gaps in the literature and could serve as a launching point for research strategies in the field of ecosystem service valuation. Once operational, it would be an important tool for more integrated cost-benefit analysis and greatly enhance more balanced decision-making regarding the sustainable use and conservation of natural ecosystems and their many goods and services.

The term 'natural capital' has so far been used in various meanings and contexts. Haines-Young follows the ecological foundations of the approach laid out by Costanza et al. (1997) and used by the UK's Countryside Commission (Haines-Young, 2000). Initially the term 'environmental capital' was used. The term 'Quality of Life' was substituted later, to tie in with the quality of life aspects emphasized by the recent publication of the suite of sustainability indicators for the UK, called "Quality of Life Counts" (Potschin and Haines-Young, 2003). These authors apply the 'Quality of Life Capital' approach by using it to examine the golf course developments along the Swiss–German–French border. In order to explore what benefits the approach might offer the analysis of future developments they compare it with a typical Environmental Impact Assessment. They conclude that the Quality of Life Capital approach considers more of the environmental and social constraints on the development. It thus provides a framework in which judgements about the significance of the impacts and the acceptability of alternatives can be made.

This concept is expressed as a new paradigm although the usage of the term 'sustainable landscapes' is certainly not exclusive (Buttimer, 2001; Saunders and Briggs, 2002). But what distinguishes the paradigm from many other approaches is that it is relatively concrete and the suggested methodologies are tangible and can be applicable in practice. Furthermore, it contributes to the decoupling of 'sustainability' and 'development' which is important in the regional and local context of sustainability. Sustainability as understood by Haines-Young (2000) refers to the promotion of planning scenarios with carefully defined objectives that aim to achieve a sustainable flow of goods and services that ensure or enhance quality of life. This way, the concept aims to balance ecological conditions and the well being of the people living in the landscape. It is holistic and anthropocentric at the same time and in principle fulfils the demands of an integrative approach: "For successful achievement of sustainable landscapes, it will be necessary to manage landscapes as a whole rather than the piecemeal approach to management employed at present" (Saunders and Briggs, 2002, p. 76).

4.3. Critique of the sustainable landscapes paradigm

Some advantages of the suggested paradigm are obvious: (a) it seeks to integrate ecological, economic and cultural values in defining the goals and guidelines for the planning process, (b) it reflects the dynamic and open system character of a landscape and the permanent exchanges between diverse constituent elements, (c) it allows for a more flexible consideration of landscape change since it suggests concentrating on the character of change processes rather then on the land cover change as such, (d) it aims to integrate stakeholders and in general the people being part of the landscape into the planning process, (e) it aims to transfer the concept of ecosystems functions to the landscape level where a 'service' is not necessarily identical with a single ecosystem or patch, in some cases it might explicitly refer to a specific mosaic of ecosystems.

Some advantages may be less obvious: by not explicitly fixing a spatial arrangement of land use, a more diversified landscape and a higher variability in ecosystem complexity are provided. Such a diversified landscape is likely to have a wide range of ecological niches conducive to enhancing biodiversity and at the same time ensure sustainability of the landscape. It is sometimes argued that the maintenance of the overall sustainability of a system demands a loosely coupled management (Ehrenfeld, 1991). It bears the potential to integrate traditional landscape management knowledge. Traditionally, many societies have viewed their land use activity in a given landscape as part of an integrated land use management, wherein human managed ecosystems are closely linked to a variety of natural systems (Ramakrishnan, 1999; Ramakrishnan et al., 2000). For instance, the diversity of cropping and resource systems that form part of the landscape serves not only as a major means of protecting ecological integrity at the landscape level, but also acts as the knowledge and resource base that makes adaptivity possible. Traditional societies adapt their land use practices both in space and time to cope with uncertainties in the environment and/or to capture market opportunities (Ramakrishnan, 1999).

Some critical questions remain. One is addressed by Haines-Young (2000, p. 8): "how would we recognise a sustainable landscape if we saw one?" But he points this question at traditional approaches and implicitly at the patch-matrix paradigm. Potschin and Haines-Young (2003) address this problem and seek to demonstrate that the sustainable landscape paradigm can give us a sustainable measure where the 'classic' patch-matrix paradigm cannot. The integration of societal values and the general orientation of 'goods and services for people' seem to be feasible in particular situations. But it is not clear as yet how to integrate the societal values, especially reflecting the well being and the feelings of the people living in the landscape in a transferable and repeatable manner. In theory the orientation on functions of ecological systems provide clues about the underlying mechanisms that constrain ecological complexity and regulate biodiversity but practical examples are missing. On the one hand, the anthropogenic functions and their mapped or assumed distribution may represent outcomes similar to law-like processes of physics, chemistry and biology. In principle, the paradigm accounts for biodiversity beyond species and includes functional diversity and the diversity of landscapes. Still, it remains a fundamental challenge to assign economic attributes to particular species, communities or ecological events (Jentsch et al., 2003). It might be even more difficult to appraise aspects of functional diversity. While many of the ecological mechanisms, at least in small scales, are well understood (e.g. thermodynamics, conservation of mass and energy, atomic particles and chemical elements, erosion processes and evolution by natural selection), it is far from clear how the anthropogenic pattern and processes act and interact with the ecological processes to produce the emergent spatial patterns. Although stakeholder and public involvement in shaping landscapes is desirable there remains the challenge of elucidating how both types of processes give rise to landscapes that are simultaneously extremely variable and highly constrained.

The frameworks for the assessment of ecosystem functions, goods and services (de Groot et al., 2002) promises to make the comparative ecological economic analysis possible. These checklists are useful in that they have enabled workers to link ecosystem functions to the main ecological and socio-cultural valuation methods. But ecological relationships are complex, often vary regionally and may be different for areas where predominant land uses vary. This complicates the estimation of cut-off values of the different functions or thresholds for management. Observed changes for the spatial configuration under observation may be associated with several interacting factors, rather than a single causal agent. This way, the spatial entities serve as containers of functions and even relatively simple thresholds may be difficult to handle spatially.

The concept of de Groot et al. (2002) does not describe how spatial relations are taken into account in the valuation process. For the monitoring of planning scenarios, developing policies or agri-environmental programmes, indicators have to reflect site-specific features in order to be meaningful. Less site-specific indicators, which are more readily available, tell little about effects in local areas. Indeed, they may fail to disclose significant developments at a local or regional level. The proposed framework of de Groot et al. (2002) provides total values of the goods and services provided by an area but the concept has, so far, not provided spatially explicit expressions of the values in the form of maps. The impact of many polluting, depleting or beneficial processes will depend on site specific characteristics such as geology, topography or climate and they might interact in a specific manner if certain land use types border to specific objects.

Finally, a spatial approach enables us to look at ecosystems and landscapes in a holistic way and address systemic characteristics such as quality and vulnerability. A sound methodology is needed to integrate peoples' concerns and their integration in regional resource management and stakeholders future visions (Costanza, 2000; Fish et al., 2003). The need to integrate the people living in the landscape and the stakeholders or 'connoisseurs' (Arler, 2000) is widely acknowledged and it is assumed that this will enhance our understanding of landscape change, but the actual location of such actively integrated persons within the wider landscape matrix is important (MacFarlane, 2000).

4.4. Putting the sustainable landscapes paradigm in practice: the need for a Leitbild

Sustainability is a goal everybody agrees to, but no one knows how to achieve it. Since the act of sustainable planning is a heuristic process, where we learn by doing, observing and recording the changing conditions (Franklin, 1997) continuous monitoring is essential to determine whether or not society is approaching the goal of sustainability. Monitoring also requires thresholds for the various goods and services. In natural systems, an ecological threshold refers to a point at which relatively rapid change occurs from one ecological condition to another. In nature, few relationships show constant change in one attribute in response to another. Rather, they mostly show points or zones at which marked change in one attribute occurs in response to a small additional change in one or more influencing factors. Potschin and Haines-Young (in press) suggest a 'tongue model', which defines a 'multifunctional choice space', set by the combination of biophysical limits and economic and social values, in which landscape trajectories can be considered.

The sustainable landscapes paradigm provides notions of quantities of ecosystem goods and services from the perspective of people living in a particular landscape. But it does not provide concepts of spatial arrangement. Monitoring necessitates clear objectives, especially environmental quality targets and specific goals for policy action ('environmental action targets'). Environmental quality targets refer to a particular state of the environment towards which efforts are geared. They comprise elements derived from both the natural sciences and the domain of social ethics, and combine scientific knowledge with social valuation of the environmental assets to be protected and the level of protection to be afforded. Environmental quality targets are defined for the anthroposphere and/or the ecosphere in respect to objects or environmental media, and are oriented to the regeneration rate of key resources or to the ecological carrying capacity, to the safeguarding of human health and to the needs of present and future generations.

This requires the integration of a Leitbild concept which is widely debated in the German-speaking literature (Potschin and Haines-Young, 2003). The term Leitbild is used to refer to a statement of some future desired state or situation. Potschin and Haines-Young suggest this term also for the English-speaking literature since there is no direct translation for both the 'future desired conditions' and the 'vision'. Environmental action targets may refer to the quantity and quality of material throughput within an economic system and be formulated as volume reduction targets. Such targets are aimed at quantitative reductions in the consumption or throughput of material and energy resources. But to put them in practice, they have to be translated into a plan and/or a *Leitbild*. This poses the challenge to 'value' resource loss (e.g. contingent value) or quantify alternative tradeoffs for locations and different regions. Official practice until now has shown that, for a variety of reasons, environmental interests tend to have less chance of being asserted in the assessment of interests than opposing interests (e.g. business, transport or regional policy interests). It may be dangerous to enforce ecological thresholds in a people- or stakeholder-based planning process. Clearly some of the stakeholders will advocate ecological considerations. But, do we define at which point ecological thresholds have to limit individual interests? For instance, how should we achieve sustainable use of certain species in a manner considered legitimate by the stakeholders?

5. Resulting research questions

5.1. Sustainable landscapes: problems to be solved

More research is needed for a scientifically based framework to evaluate avoidance and compensation rules. Compensation should only be on option if degradation or harm cannot be prevented or compensated for to the required extent. According to the common understanding of sustainability, an intervention is balanced or equalized if, when it has ended, no major or persistent disruption of the balance of nature remains and the visual appearance of the landscape has been restored or reshaped in an appropriate way (Forman, 1995; MacFarlane, 2000; Appleton et al., 2002). Research must focus on parameterizations of this 'balance of nature' and disruptions of it. A specific aspect is again the spatial dimension: the nature conservation rules on interference can only be utilized for achieving sustainable development if the spatial arrangement of habitat types and land use classes is embedded in an evaluation scheme. Landscape management demands a variety of responses that are location-specific, in terms of land use activities linked with natural resource management such as, hydrology regime, sustainable soil fertility, biodiversity and biomass production.

The suggested paradigm is not mature as yet, although first applications exist (Potschin and Haines-Young, 2003). The proposed shift away from ecological orientation to a more anthropocentric focus, in which landscapes and the ecosystems associated with them are viewed as a resource, requires a number of measures, e.g. transparent evaluation methods. The desired measures, including potential positive and negative externalities, shall support the sustainability impact assessment, the assessment of the inter-relations of environmental, economic and social impacts of policies and measures in qualitative and quantitative terms. The precautionary principle and the regional aspects to sustainable development will be key elements to be taken into account. It is not yet clear how to quantify accumulating or compensating effects of adjacent or near developments. Future research has to focus on the estimation of thresholds of sustainability and externalities, and on the development of tools for integrated, yet spatial sustainability assessment. The definition and estimation of scientifically based thresholds of sustainability and points of no return as a tool for the sustainable management and characterization of the state of the environment will have to be addressed. This implies an equal balance between the necessary ecological, social and economic dimensions. But most of the social and economic dimensions have so far been treated statistically for administrative units only. The challenging task, therefore, is to estimate cumulative, interactive effects over time caused by current and foreseeable actions and the coupling of data with policy judgements reflecting costs. In order to calculate cumulative effects based on spatial concepts, we need spatial tools such as are provided within GIS.

5.2. *Changing landscape patterns: monitoring needs*

Not all landscape pattern and not all spatial relationships are relevant. There are effective and non-effective properties associated with patterns and patterns may not have any ecological relevance for the question at hand (Hargis et al., 1998; Bissonette, 2003). Research efforts have considerably advanced our understanding of the interrelationship between landscape structure and ecological functions and have led to multiscale landscape analysis (Hay et al., 2002; Wu and David, 2002). But, as stated earlier, we know less about the relationship between human behaviour and landscape functions but we hypothesize that changes in economy and society are directly reflected in the character of landscapes, both in terms of their form and function and the landscape planning and management challenges which they present (Wood and Handley, 2001; Botequilha Leitao and Ahern, 2002). Wood and Handley (2001) seek to explore the phenomenon of landscape dynamics using a systems perspective to help define drivers of landscape change, subsequently using these as the basis for a framework of landscape planning, design and management to anticipate and resolve the adverse impacts of change. Two drivers of landscape change are identified: 'obsolescence' and 'dysfunction'. The former summarises the effect of changes in land use, and the latter addresses conflicts between competing uses. They claim that this concept helps to describe the particular pressures of post-industrial society on landscape integrity, but no particular practical examples are provided.

As discussed, landscape change is a normal process, since it is just a physical reflection of changes in land use and history teaches that it is impossible and undesirable to freeze land use change. From another perspective, it confronts us with a problem, since, in recent decades, the rapid land use changes that have taken place dramatically affected the qualities and internal coherence of many landscapes all over the world. This is reflected in the changing spatial patterns of landscapes. Broadly speaking, landscapes with small spatial and historic patterns are changing into landscapes with larger spatial patterns, while losing the historic qualities and spatial heterogeneity. As biological diversity is lost, ecosystems become less complex (Saunders et al., 1991). This sets in train a sequence of events that in many cases leads to ecosystem degradation. These changes can be dramatic and have important and long-lasting consequences (Saunders and Briggs, 2002). Simplified ecosystems become less resilient and there are fewer components to buffer the blows inflicted by drought, fire, exotic species and climate change (Saunders et al., 1991). Spatial heterogeneity is an important asset for the long-term resilience of ecosystems and landscapes and it is necessary to characterize spatial heterogeneity quantitatively over a range of scales. Because today's spatial pattern results from yesterday's dynamic processes, pattern analysis may reveal critical information on properties of underlying processes (Wu et al., 1997). Multiscale and multitemporal analyses are mandated. Both dimensions are reflected in state-of-the-art GIScience research (Longley et al., 2001) but the sustainable landscapes paradigm does currently not provide a conceptual framework for the integration of the reciprocal relationship between spatial pattern and ecological processes.

The environmental impacts of landscape changes request a sound theoretical and methodological foundation of land-use decisions including landscape ecology concepts and spatial analysis. We need ways to monitor and analyse these changes to inform politics (Botequilha Leitao and Ahern, 2002) and a commonly agreed methodology of what landscape is, what landscape structure is, and we need transferable measures for certain qualities. The patch-matrix concept is an understandable and operational framework for the spatial-quantitative analysis and many examples exist to quantify changing patterns spatially since human activities generally introduce rectangularity and rectilinearity to landscapes producing regular shapes with straight borders (Krummel et al., 1987; O'Neill et al., 1988; Turner, 1990; Forman, 1995; Saunders and Briggs, 2002). Landscape metrics provide measures for the degree of complexity of spatial units and mosaics (Fig. 3 and Table 2) and there are many studies that relate the complexity to ecosystem stability or species survival (e.g. Dramstad et al., 1996; Goudie, 2000; Moser et al., 2002).

5.3. Which patterns are needed?

As discussed earlier, the world is patchy (Wiens, 1995; Dale, 1999; Bissonette, 2003). When we seek spatial pattern and evidence of spatial non-randomness, we find it is the rule rather than the exception. Landscape ecologists study spatial pattern to infer the existence of underlying processes, such as movement or responses to environmental heterogeneity (Kolasa and Pickett, 1991). Spatial structure indicates intraspecific and interspecific interactions such as competition, predation, and reproduction but can also be shaped by abiotic processes. We have many terms to describe various aspects of non-randomness in spatial data. The terms 'aggregated', 'patchy', 'contagious', 'clustered' and 'clumped' all refer to positive, or 'attractive', associations between individuals or events in point-referenced data. The terms 'autocorrelated', 'structured' and 'spatial dependence' indicate the tendency of nearby samples to have attribute values more similar than those farther apart (Liebhold and Gurevitch, 2002; Perry et al., 2002). Observed heterogeneity may also be driven by resource availability. Clearly, care is required in inferring causation, since many different processes may generate the same spatial pattern. Spatial pattern has implications for applied problems such as the management of biodiversity (Jentsch et al., 2003). Some effects are illustrated in Fig. 5. As we are starting to understand some of these complex relationships between spatial structure and ecological parameters, we currently have very limited knowledge about integration of these relationships in planning scenarios. How much spatial diversity is needed? How much connectedness is needed? How little spatial diversity can we accept?

5.4. Spatial entities versus services for people

Recent research emphasizes that both the way we are dissecting our reality and the scale of investigation influence the results significantly (Wu et al.,

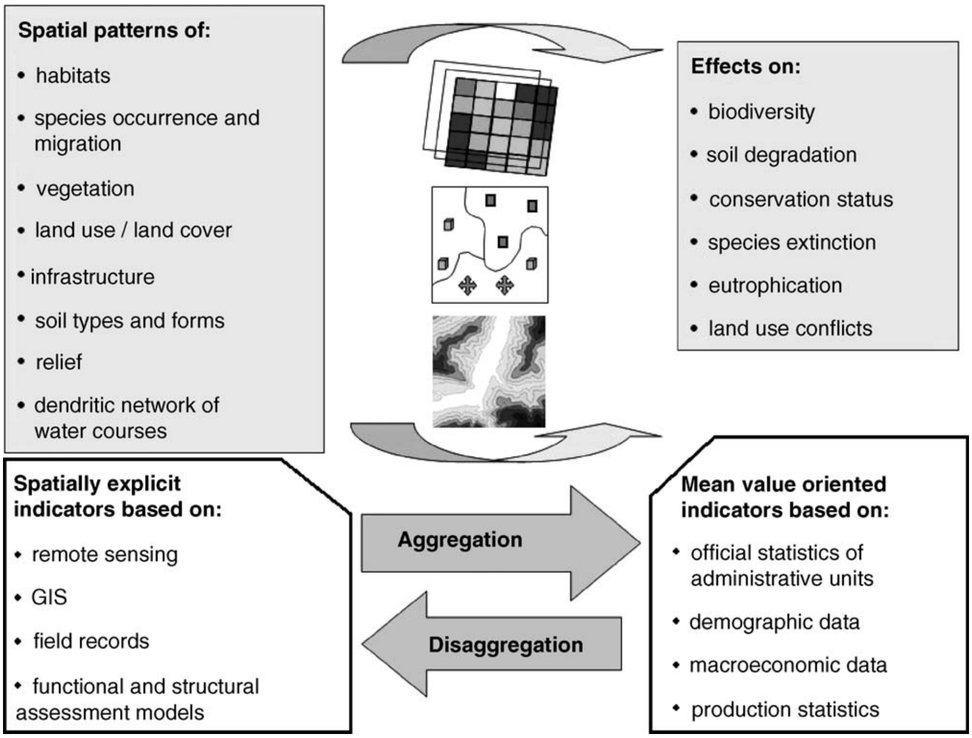


Fig. 5. The relevance of spatial reference for environmental indicators.

1997; Hay et al., 2002, 2003; Wu, 2004). The methodological framework of Burnett and Blaschke (2003) tries to overcome the problems of 'discreteness' of analysis units and mono-scaled analysis by explicitly incorporating information from below the target level as a mechanistic level. It is still based on discrete spatial entities representing a continuous surface. In an attempt to represent complexity and a multi-scale view better, an object-based hierarchical structure was introduced to improve the integration of both raster and vector structures, and support facilities to handle fuzziness and uncertainty at the boundaries of homogeneous regions. Alternatives include fuzzy spatial objects (Cheng, 1999) or geographic fields (Cova and Goodchild, 2002). All these approaches address landscapes in terms of the physical patterns that we (or remote sensing) perceive. In landscapes, these patterns are usually determined by vegetation and topography. This perception may be convenient, but it may not reflect the spatial patterns of factors that influence other organisms. The sustainable landscapes paradigm might view some aspects of landscape mosaics better in terms of various costs and benefits. How to deal with distance and topological relationships of various existing and planned configurations and how to embrace concepts such as spatial gradients (Müller, 1998) and ecotones (Hansen and di Castri, 1992; Fortin et al., 2000) have to be laid out explicitly. The sustainable landscapes paradigm is flexible concerning the temporal dimension of sustainability compared to the patch-matrix concept but no applications demonstrate this as yet. Sustainable landscapes clearly reference the past, present and the future in terms of the relationships between the ecosystem and the economic and social dimensions of life today and the desire not to compromise future needs.

5.5. How to reflect the human dimension without overriding ecological functions?

Beyond the general intention to reflect the needs of the people and the mandate to seek for participatory planning in order to ensure the long-term acceptance of the planning directives, there is only fragmented knowledge about how human influence affects ecosystem functions and processes. We do not even know relatively simple relations like the consequences of interactions between human population density and the environment. The latter depends on the nature of the interaction and the particular species, ecosystems, or processes in question and is inherently multiscale in nature. The sustainable landscapes paradigm is by definition ('goods and services for people') linked with one level of scale: the human scale. This is reasonable since a sustainable landscape is defined as one in which the output of goods and services is maintained, and the capacity of those systems to deliver benefits for future generations is not undermined (Haines-Young, 2000). But how would we know? Certainly, the predictions are based on existing data and models associated with certain scales. While the patch-matrix paradigm has recently been developed further and applied to multi-scale analysis (Turner et al., 2001; Hay et al., 2003) the sustainable landscapes paradigm must develop multi-scale methodologies in order to be applicable.

A challenge for both the sustainable landscapes paradigm and the patch-matrix paradigm is the integration of aesthetic values. Landscape planning is concerned with balancing ecological and anthropogenic values such as aesthetics or the visual landscape. But how do we balance the sometimes incompatible goals in the context of sustainable landscapes? Parsons (1995) demonstrated this conflict using the example of wood patches and hedges in an open landscape. Densely vegetated wood patches support the diversity of wildlife habitats, while evidence from literature on the aesthetic quality of natural environments has repeatedly established that people tend to prefer more open grassy areas punctuated by occasional groupings of trees and shrubs. But, in landscape planning, too often form and aesthetics take prevalence over functional aspects (Hobbs, 1999).

5.6. How to include topology and hierarchy?

Many ecological processes of interest in global change studies, such as productivity, biogeochemical cycling, and water and energy exchange, operate at a number of scales. While most process level research is conducted at the patch scale, the valuation of ecosystems (Costanza et al., 1997) and the methodology of de Groot et al. (2002) respectively extend the analysis of ecosystem processes to the global scale. But it remains unclear how to incorporate hierarchy. In some cases, the ecosystem function or the 'service' might depend on the association of different ecosystems or habitat in space. For any use or activity, the area being planned is embedded within a larger ecological system, and will affect other parts of the systems non-linearly since functional relationships exist to and from that larger system (Forman, 1995). Some argue even that context is usually more important than content (Dramstad et al., 1996). The critical question is when can ecosystem processes simply be aggregated as an area-weighted sum of patches, and when does distribution and patchiness, adjacency and remoteness, fragmentation and connectedness, rather than just abundance of landscape elements affect these processes? Further, when must material and energy exchange among landscape units be considered to develop adequate estimations at large scales?

6. GIS integration

The premise of landscape ecology, that spatial context makes a difference in ecological patterns and processes does not hold for any case, but there are many studies and empirically validated facts underpinning the importance of the spatial (Wiens, 1995; Wrbka, 1998; Fjellstad, 2001; Turner et al., 2001; Bissonette, 2003). GIS is not a solution to environmental problems but a powerful analysis, integration and visualization tool for sustainable environmental management. Although the choice of scale may be strongly influenced by existing data, the selection is important for establishing a common system for data integration, for addressing data-quality issues, and for specifying detection limits. An understanding of the precision and accuracy of the data within a GIS is also important. Even though a GIS can mechanically reformat and transform data from different sources into a common system, it is the responsibility of the GIS user to determine the consequences of integrating data that have been collected at different scales, represented by different topological structures, digitized with varying degrees of precision, or containing other sources of errors. Unfortunately, elegantly drawn GIS maps usually do not convey the uncertainty associated with boundaries or contour lines. However, advanced cartographic techniques provide improved interpretability of the spatial analysis results.

A spatial concept enables us to understand better the specific characteristics of sites and the nature of the interaction between ongoing or periodic human actions such as agricultural practices, permanent or long lasting land use changes and the environment. Together with territorially differentiated information on driving forces and the state of the environment, a spatially explicit landscape approach can form the basis for describing in a relatively simple way the balance between agricultural activity and the ecosystem of which it is part. Mapping and, more generally, providing basic information for planning does not automatically lead to any betterment or sustainability, but there is potential for an integrated and scientific planning through a meaningful use of relatively mature computer-based tools and methods dealing with spatial information. Analysis at large spatial scales can never replace the need to understand structure and function at the ecosystem level of organization.

First attempts have been made to develop a theoretical base and associated spatial techniques to identify resource management regions integrating the citizens' views by means of GIS (Brunckhorst et al., this volume). It was laid out in this paper that day-to-day planning and decision making depend on the ability to associate different sorts of land-related information with its physical location. Today, GIS alleviates many of the underlying problems and provides much more than record management and maintaining spatial information. Geographic databases interfaced to predictive models provides a resource for professionals and decision makers with the capability to undertake more 'what if' analysis and comparison of alternatives. Thus, they can support envisioning the future, although most effort in 'futures modelling' has so far focused on extrapolating past trends rather than envisioning alternative futures (Costanza, 2000).

Natural science research focuses on an assessment and understanding of ecological patterns and processes. Social science usually calculates values of the goods and services by assessing the costs and benefits and the social acceptability of changing the environment. Integrated approaches shall take into account how and why societal conditions that positively or negatively affect the environment are made. We are technically equipped to serve for these integrated needs and appropriate methodologies are currently being developed.

The sustainable landscapes paradigm is new ground for landscape ecology and landscape planning. Perhaps it is currently immature in some respects. Despite the discussed remaining problems, this paper emphasizes the potential of the paradigm. It advocates a ‘spatialisation’ of the framework suggested by Haines-Young (2000). The spatial dimension of sustainable landscapes engages processes and relationships between different land use types, ecosystems and landscapes at different scales, and over time and requests a conceptual framework for sustainable landscape planning based on landscape ecological concepts and spatial concepts. Although the author believes that the development of a combined methodological framework is of central importance to a meaningful use of GIS for landscape analysis and landscape planning, the development is slow.

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