Mapping and Modelling
Multifunctional Landscapes

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This research was conducted under the auspices of the Graduate School of Socio-Economic and Natural Sciences of the Environment (SENSE)

Mapping and Modelling Multifunctional Landscapes

Louise Willemen

Thesis
submitted in fulfilment of the requirements for the degree of doctor
at Wageningen University
by the authority of the Rector Magnificus
Prof. dr. M.J. Kropff,
in the presence of the
Thesis Committee appointed by the Academic Board
to be defended in public
on Wednesday 12 May 2010
at 4 p.m. in the Aula.

Louise Willemen
Mapping and Modelling Multifunctional Landscapes,
152 pages.

Thesis, Wageningen University, Wageningen, The Netherlands (2010)
With references, with summaries in Dutch and English

ISBN 978-90-8585-626-9

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Chapter 1

Introduction: mapping and modelling multifunctional landscapes

Based on: L. Willemen, P.H. Verburg, K.P. Overmars, M.M. Bakker
In: New perspectives on agri-environmental policies; a multidisciplinary and transatlantic approach (2010), Eds. S.J. Goetz, and F. Brouwer

Background

People use and modify landscapes. Within landscapes humans try to improve their livelihood by converting land cover, extracting resources and redirecting water flows. These actions indirectly influence underlying biophysical processes of the landscape (Vitousek et al., 1997). Intended or unintended changes of the landscape alter landscape functions (DeFries et al., 2004; Palmer et al., 2004; Kareiva et al., 2007). *Landscape functions* describe the ability of a landscape to provide goods and services to society. Such goods and services include, amongst others, food and timber production, fresh water supply, climate regulation, landscape aesthetics and recreational opportunities. These are all benefits of the landscape that contribute to human well-being. People thus depend on landscape functions and therefore good management of landscapes is essential for sustainable human development (MA, 2003).

In this chapter background information is provided on the concept of landscape functions and it is explained why and how this concept needs to be studied more in depth. Then the objective and resulting research questions are presented and an overview of this thesis is given.

Functions of landscapes

The concept of landscape functions originates from Ecology. In the 1970s ecologists started to identify the benefits of natural ecosystems for society in order to promote nature conservation and to support spatial planning actions (Van der Maarel and Dauvellier, 1978; Van der Ploeg and Vlijm, 1978). Only in the 1990s the concept of ecosystem functions gained momentum in the scientific literature (e.g. De Groot, 1992; Costanza et al., 1997; Daily, 1997). Although currently many definitions are available, ecosystem functions are generally seen as characteristic of an ecosystem that provides goods and services to satisfy human needs.

The term *landscape function* in this thesis is used in analogy with the concept of ecosystem functions: it indicates the capacity of the landscape to provide goods and services to society. The reason for specifically addressing landscapes and not ecosystems is because landscapes consist of different systems, arranged in specific spatial patterns. This thesis addresses land systems that are strongly modified by humans, such as agricultural and peri-urban areas. Landscapes are considered holistic spatial systems in which humans interact with their environment (Naveh, 2001 ; Bastian, 2004), while ecosystems are often perceived as merely natural and semi-natural systems (e.g. Daily, 1997; Egoh et al., 2007; Cowling et al., 2008). As a product of landscape functions, *landscape services* are defined as the flow of goods and services provided by the landscape to society. These landscape services (short for landscape goods and services) are the connection between the landscape and human benefits, i.e. the actual contributions to well-being (De Groot et al., 2010). Besides landscape services, also other terminologies have been introduced in the scientific literature to address services that are provided in natural and cultivated systems. These include, land-use functions (e.g. Pérez-Soba et al., 2008), land functions (e.g. Bakker and Veldkamp, 2008; Verburg et al., 2009) and environmental services (e.g. Barton et al., 2009; Turner II, 2010). However, as the term landscape explicitly includes the interplay between humans and their environment, we consider *landscape* functions and services in this thesis the most appropriate terms (as for example in Bastian et al., 2006; Gimona and Van der Horst, 2007; Lovell and Johnston, 2009; Termorshuizen and Opdam, 2009).

Landscape service supply is not equally distributed over the landscape. The amount of service supply depends on location-specific and temporal landscape characteristics (Wiggering et al., 2006; Egoh et al., 2008). Landscape service supply of a location can be quantified by the actual service supply (e.g. quantity of food produced) or by its value. An assessment of the amount of supplied services always proceeds the valuation the service supplied by a location (Hein et al., 2006). Basically, three aspects drive the value of landscape services: ecological, socio-cultural, and economic (MA, 2003). The ecological aspects encompass the health status of a system, measured by ecological indicators such as diversity and integrity (De Groot et al., 2010). Socio-cultural measures relate to the importance people give to, for example, the cultural identity of landscape or recreational possibilities (e.g. Alessa et al., 2008). And last, the economic measures which relate to the goods and services consumed, or used as input in an economic production process (e.g. MA, 2003; TEEB, 2009). In order to calculate the overall value of landscape services a number of methods have been developed to also translate ecological and socio-cultural measures of landscapes into monetary terms (e.g. Costanza et al., 1997; Costanza and Farber, 2002 ; Hall et al., 2004; Hein et al.). The advantage of using a single value-unit (e.g. money) is that it can not only represent all different value-domains in one measure, but it can also be used to assess the overall value of multiple landscape services in a multifunctional landscape.

Multifunctionality

Landscapes provide often more than one service at the same time, resulting in multifunctional landscapes. A landscape could, for instance, be used for agricultural production, facilitate recreational activities and provide habitats for wildlife at the same time. The concept of multifunctional landscapes first appeared in scientific publications in the 1980s (e.g. Niemann, 1986). Nowadays, the scientific literature describes the concept of multifunctionality from different disciplinary backgrounds. Besides multifunctional landscapes (e.g. Brandt and Vejre, 2004), also multifunctional agriculture (e.g. Hall et al., 2004; Bills and Gross, 2005; Van Huylenbroeck et al., 2007), multiple services from natural areas (e.g. MA, 2003; Tallis et al., 2008a), and multifunctionality in relation to regional development (e.g. Heilig, 2003; Wiggering et al., 2003) are frequently studied. Although approached from different perspectives, all these studies relate to describing interactions between landscape functions.

Over the last two decades, several international organisations included the concept of multifunctionality into their agricultural development strategies. In 1992, for example, the role of multifunctional agriculture in relation to rural development was addressed in Agenda 21 of the United Nations' Rio Earth Summit (UNCED, 1992). Some years later, the Food and Agriculture Organisation of the United Nations (FAO) highlighted important multiple functions of agricultural areas for rural livelihood (FAO, 1999). The Organization for Economic Co-operation and Development (OECD) and European Union introduced the concept into their new conceptual approach for agricultural policy, recognising that 'agriculture' does not solely include agricultural production but embraces a whole range of functions (OECD, 2001). And finally, the latest European Common Agricultural Policy reforms were also based on the concept of multifunctionality. These reforms gave policy makers the opportunity to shift the focus of subsidy programmes from a primarily production focus to a stronger attention to the social and environmental functions of agriculture (EC, 2004). Due to these reforms payment schemes and subsidies of farmers in the European Union relate now also to the non-commodity services they supply.

The recognition of multifunctional landscapes also appeared in national political arenas. In regions with high pressure on land, the concept of multifunctional landscapes is expected to play a role in reducing conflicting claims on land while complying with societal needs for landscape services (Brandt and Vejre, 2004). In this way, the notion of landscape multifunctionality became a part of several comprehensive spatial planning strategies (see e.g. Dijst et al., 2005; VROM, 2006; Cairol et al., 2009; Vejre et al., 2009). One of the major reasons for policy makers to focus on multifunctionality is that the total service supply of multifunctional areas is assumed to exceed the service supply of mono-functional locations (Brandt and Vejre, 2004; De Groot, 2006). However, not all landscape functions can be combined without influencing the overall provision of landscape services because of trade-off effects or conflicts between different stakeholder groups.

Relevance of mapping and modelling landscape functions

The concepts of landscape functions and multifunctionality are currently thus included in many different policy strategies. Additionally, efforts to include management of landscape services into planning practices have increased strongly (e.g. Daily and Matson, 2008; Tallis et al., 2008a). Especially for areas with high pressure on land resources, good management of interacting functions within a multifunctional landscape seems crucial. Landscapes are spatially diverse and therefore landscape functions are unequally distributed over an area. In order to adequately manage landscape functions knowledge is needed on where and how much landscape services are being provided (Egoh et al., 2008). Also, to be able to make decisions regarding trade-offs at multifunctional sites we need to understand where and how much landscape functions interact with each other (De Groot et al., 2010). And finally, to evaluate management strategies information is needed on the location and quantity of landscape function dynamics within a changing landscape. However, to date, there appear to be no examples of complete spatial assessments of the quantity and value of service supply in multifunctional landscapes under different land management strategies (ICSU et al., 2008). Therefore, we assume that quantitative maps of landscape functions can support decision makers to design spatial policies, while spatially explicit models can be used to evaluate the effect of land management strategies.

Why maps?

Limited information is available on the spatial distribution of landscape functions. Current land-use maps relate primarily to the classic spatial policy focus on agricultural and urban development. These land-use maps are based on the directly observable land cover. However, it is hypothesised that many landscape functions cannot be directly linked to land cover. This limits the use of current maps in landscape function studies. This limitation is illustrated by an example in the central region of The Netherlands. The land-cover maps in Figure 1.1 indicate that mainly urban expansion has taken place between the years 2000 and 2006. With an increase in urban area, changes in use of the surrounding rural landscape likely takes place. For example, more land in peri-urban areas will be used for outdoor recreation. The land-cover maps in Figure 1.1 do not show any change in landscape functions in agricultural areas, represented by the land-cover classes 'arable land' and 'pastures'. However, according to Dutch farm census data, an average growth of 8.3% in the number of farms that incorporated recreational activities into their operations occurred between 1999 and 2005. In the same period, an additional 11.3% of farms per year were participating in nature and cultural heritage conservation programs. So, even though the land cover does not show any change, the functions of the agricultural landscape did change.

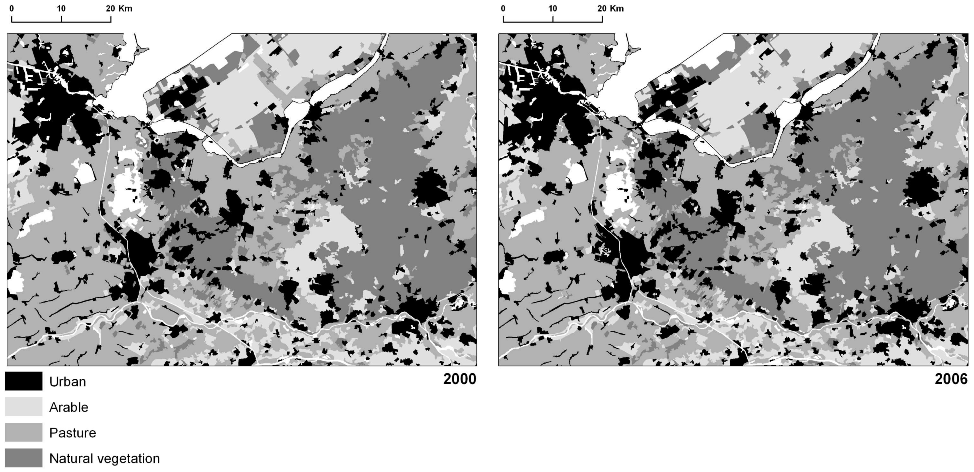

Figure 1.1 Land cover in the central part of The Netherlands in 2000 and 2006, reclassified from the CORINE dataset (Hazeu et al., 2008)

When the distribution of landscape functions is made spatially explicit, potential conflicts between landscape functions can be identified and minimised. For example, in the last decade many modern windmills have been placed in the Dutch rural areas. This new function of the landscape to provide 'green' energy soon appeared to be in conflict with the bird habitat function, i.e. many birds died in a collision with the blades (Winkelman et al., 2008). To better deal with these conflicting functions in planning strategies, the two landscape functions were quantified and mapped to indicate areas where wind energy production would least affect bird habitats (Aarts and Bruinzeel, 2009). This example shows that spatially explicit information on landscape functions can mitigate function conflicts.

Why spatial models?

Landscapes are continuously changing and therefore the provision of landscape services is subject to permanent change. Most of these dynamics in the landscape are induced by people and influence landscape functions directly (Vitousek et al., 1997; DeFries et al., 2004; Palmer et al., 2004). Scientists have intensively studied causes and impacts of human-induced landscape dynamics. This resulted in a wide variety of spatial modelling approaches to describe, monitor and explore landscapes and their future changes (see overviews by Parker et al., 2003; Gutman et al., 2004; Verburg et al., 2004; Lambin and Geist, 2006). These research efforts however focus on land-use changes without explicitly including landscape functions or demand for such functions. For policy makers and planners the effect of management strategies on landscape functions is of great interest

(OECD, 2001; EC, 2003; VROM, 2006). Spatially explicit land-use models have proven to be valuable tools to explore future scenarios and to assess impacts of change in policy making (Uran and Janssen, 2003; Geertman and Stillwell, 2004; Kok et al., 2007; McIntosh et al., 2007). Therefore, including landscape functions into spatial models is expected to further the understanding on their feedbacks and interactions which could improve decision making processes (Cowling et al., 2008; Carpenter et al., 2009; Paracchini et al.). Both the construction of such models and the interpretation of their results can enhance understanding of the landscape function dynamics (Parker et al., 2008; Claessens et al., 2009).

There are several reasons why landscape functions are mostly not included in current land-use change models. First, the land-use modelling approaches are mainly based on land-cover maps (e.g. Figure 1.1), while, as shown above, landscape functioning extends beyond land cover. We assume that landscape functions can be defined by a range of biophysical and socioeconomic characteristics, of which land cover is only one aspect (Figure 1.2). The quantitative relationships between these landscape characteristics and landscape functions however need to be defined (ICSU et al., 2008; Renting et al., 2009; De Groot et al., 2010). Second, few land-use models actually quantify the service supply or land-use outputs per area (Lambin et al., 2000). In order to account for the spatial variation of landscape functions within a landscape, the actual amount of service supply to society needs to be quantified. Third, human-induced changes in land-use models are usually driven by a demand for commodity goods like agricultural products and urban areas. Current land-use models do not take into account demands for non-commodity landscape functions such as cultural value of a region, recreational opportunities, and biodiversity support (Heilig, 2003). Demand for services is assumed to be a driver of land management decisions and spatial policies (Figure 1.2). The resulting societal actions adapt the landscape in such a way to ensure the continued flow of services (DeFries et al., 2004; Bastian et al., 2006; Nelson, 2006). By explicitly including landscape functions into a spatial model, societal actions can be described and evaluated. Finally, many locations in the landscape are multifunctional. Because of possible interactions between landscape functions, these multifunctional locations need special attention when exploring dynamics of landscape functions (Figure 1.2). Present land-use modelling approaches cannot take into account the interactions at multifunctional locations as these approaches typically assign a single land-use type to a specific location.

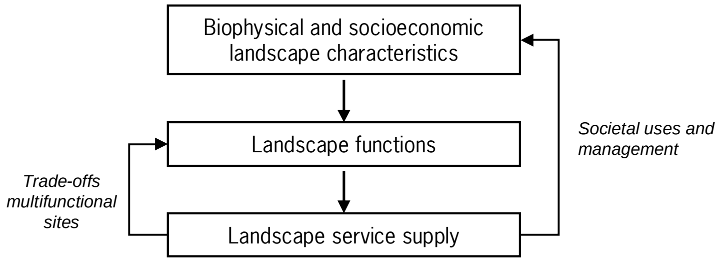

Figure 1.2 Hypothesised relations and feedbacks in space and time of landscape processes influencing landscape functions.

Contents of the thesis

Objectives

As people depend on landscape functions, effective management of landscapes is essential to safeguard the flow of landscape services. However, information on the spatial distribution and dynamics of landscape functions to support such management is limited. Current landscape research lacks methodologies to quantify and map landscape functions with the aim to explore the dynamics of multifunctional landscapes. The overall objective of this thesis is therefore to analyse and quantify *spatial aspects* of both *landscape functions* and *multifunctionality* and to develop a methodology in which landscape function *dynamics* are modelled. The methodological outcomes of this thesis should have the potential to support decision-making on future landscape management.

To achieve this objective the following research questions are defined:

1. How can landscape functions be described, quantified and mapped?
2. How can interactions between landscape functions at multifunctional locations be identified and quantified?
3. How can changes in landscape service supply and value be quantified to evaluate landscape management strategies?
4. How can dynamics of multifunctional landscapes be modelled in space and time?

The general focus of this thesis lies on the development of methodological approaches, rather than on presenting clear guidelines for landscape management. The resulting methodologies should have to potential to be applicable in other studies. An application of these methodologies is given based on data of a case study area, the Dutch Gelderse Vallei region. Additionally, this thesis addresses landscape functions as a result of spatial patterns of the landscape and regional socioeconomic characteristics. Individual decision making processes (as in Pfeifer et al., 2009; Valbuena et al., 2010) or economic processes driving changes in landscape services supply are considered beyond the scope of this thesis.

Making landscape functions spatially explicit adds an innovative component to research conducted in the field of quantification of multiple landscape services. While most other quantification methods lack a spatial component, this thesis aims at developing a methodology to quantify spatial variability of landscape functions. Furthermore, a methodological framework that explicitly includes quantitative and spatial information on landscape functions and their interactions, should lead to novel spatial modelling approaches to describe the dynamics of multifunctional landscapes.

Study area

All analyses presented in this thesis are based on data of the Gelderse Vallei region in The Netherlands (Figure 1.3). The Gelderse Vallei is a prominent agricultural region within the densely populated Netherlands. Because of the diverse biophysical characteristics and pressure of land resourses, multifunctionality is a key aspect in the current land-use planning for this region.

The Gelderse Vallei is a shallow valley formed by a glacier that covered a part of The Netherlands in the Saale period (approximately 150 ka BP). In this period push moraines were formed which now border the valley. The difference in elevation in the study area causes a gradient in many biophysical conditions like in the hydrology and soils. This diversity in biophysical conditions forms a basis for diverse landscape functions. Total size of the study area is about 750 km^2 of which currently 70% is under agricultural use, 17% of the land is covered by urban areas and the remainder of the area is composed of natural areas, infrastructure and water. Because of current spatial policy, ecological corridors are being created to connect two national parks located on both sides of the study area. These national parks enhanced the development of a large tourism sector in the region (some 300 000 overnight stays per year: Provinces of Gelderland and Utrecht, 2005). Additionally, the region contains approximately 20% of the intensive livestock production (pork, poultry and eggs) of The Netherlands (CBS, 2008a). Through an increase in population and built-up areas, the Gelderse Vallei region is gradually transforming from rural to peri-urban. Based on the current trends, the population in the region of almost six hundred thousand inhabitants in the year 2000 is expected to increase with four percent by the year 2015 (CBS, 2008a). Peri-urban developments can be observed from, for example, the increase of rural estates and hobby horses and stables, which are becoming a common aspect in this region (Van der Windt et al., 2007).

At the end of the 20th century conflicts between different landscape functions led to several problems in the Gelderse Vallei. The region suffered from pollution and eutrophication of the natural environment because of the intensive livestock production, losses of cultural-historical landscapes because of strong urban development and significant economic losses due to livestock diseases. With the intention of specifically solving these problems, a new spatial planning strategy was introduced in 2004. The Reconstruction Act focuses specifically on the multifunctionality of the Gelderse Vallei, aiming at separating conflicting functions and joining compatible functions as much as possible (Provinces of Gelderland and Utrecht, 2005). Therefore, policy makers in the study area could profit from a methodology to make landscape functions and multifunctionality spatially explicit and to explore future changes in landscape functions. This thesis explores how this need can be addressed.

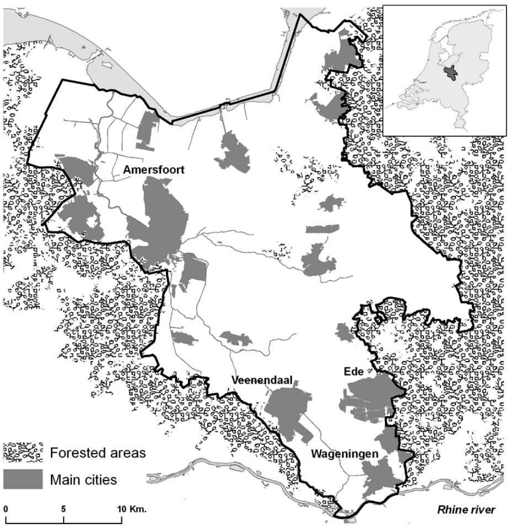

Figure 1.3 Study area of the Gelderse Vallei; within the inset the location of the study area in The Netherlands.

Outline of the thesis

The structure of this thesis follows successive steps to address the overall objective and research questions (Figure 1.4). *Chapter 2* presents a methodological framework to quantify landscape functions and to make their spatial variability explicit. In this chapter three methods are presented to map and quantify landscape functions depending on the availability of spatial information. The results are subsequently used in *Chapter 3* to define multifunctional areas and to identify and quantify interactions between landscape functions. Different aspects of the landscape function interactions are addressed including landscape characteristics that influence landscape function interactions, interrelations between landscape function capacities and the effect of multifunctionality on landscape service supply. In *Chapter 4* the change in landscape service supply and value under influence of policy measures are evaluated. Changes in service supply quantities are explored using a unit-less index related to the level of service provision and an estimation of the value of these services in monetary terms. In *Chapter 5*, a multi-scale modelling approach is proposed to analyse the spatial and temporal dynamics in landscape service supply based on the insights gained in the previous chapters. In this modelling approach we explicitly address, the multifunctional character of the landscape, the different spatial levels at which interactions between landscape service supply, demand and land management occur, and the trade-offs in service supply levels as a result of land management actions. To conclude, in *Chapter 6* the presented methodologies and findings are discussed, together with the possible implications of landscape function mapping and modelling for sustainable land management.

Chapters 2 to 5 are written as independent papers for scientific journals and can therefore also be read separately.

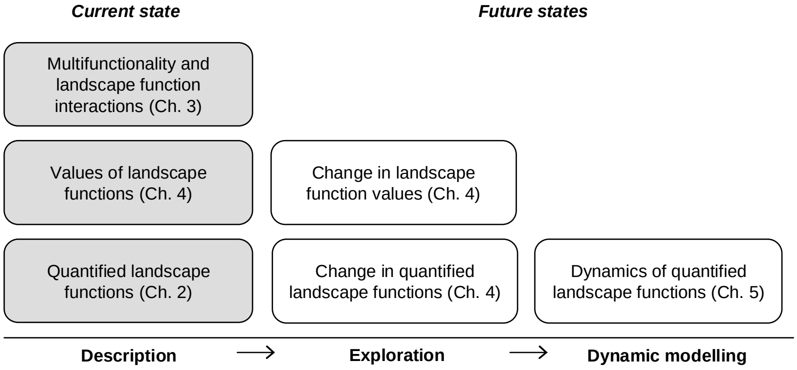

Figure 1.4 Overview of successive methodological steps in the different chapters of this thesis; from describing landscape functions to a dynamic modelling approach.

Chapter 2

Spatial characterisation of landscape functions

Limited information is available on the spatial variation of landscape functions. We developed a methodological framework to map and quantify landscape functions depending on the availability of spatial information. In this framework three different methods were proposed (i) linking landscape functions to land cover or policy defined areas, (ii) assessing landscape functions with empirical models using spatial indicators and (iii) assessing landscape functions using decision rules based on literature reviews. The framework was applied to the Gelderse Vallei, a transitional rural area in The Netherlands. We successfully mapped and quantified the capacity to provide services of eight landscape functions (residential, intensive livestock, drinking water, cultural heritage, tourism, plant habitat, arable production, and leisure cycling function) for this region. These landscape function maps provide policy makers valuable information on regional qualities in terms of landscape functionality. Making landscape functions spatial explicit, adds an important component to research conducted in the field of quantification of landscape services.

Based on: L. Willemen, P.H. Verburg, L. Hein, M.E.F. van Mensvoort
Landscape and Urban Planning, 88 (2008), 34-43

Introduction

Landscapes are able to fulfil many different functions. Based on the definitions of de Groot (1992) and the Millennium Ecosystem Assessment (2003) we define 'landscape function' as the capacity of a landscape to provide services to society. These include, for example, the provision of goods like harvested crops or timber, and services like landscape aesthetics, provision of habitat or regulation of water systems. Landscape functions are not evenly distributed over a region because of the socioeconomic and biophysical variation of the landscape and the spatial and temporal interactions between the different components of the landscape (De Groot, 1992; Wiggering et al., 2006; Syrbe et al., 2007).

From the 1990s onwards, landscape functions and multifunctionality have become important concepts in policy making, in particular within the European Union (FAO, 1999; OECD, 2001; Hollander, 2004; Wilson, 2004; Bills and Gross, 2005). For example, the European Union's Common Agricultural Policy (CAP) reforms of 2003 were strongly based on the concept of multifunctionality (EC, 2004). Additionally policy makers nowadays have to deal with an explicit demand for landscape services from local and national stakeholders (Hein et al., 2006). However, information on landscape functions is often lacking for policy making (Pinto-Correia et al., 2006; Vejre et al., 2007). Existing landscape models to support policy making mostly either deal with land-cover patterns (Geertman and Stillwell, 2004; Verburg et al., 2004) or are strongly sector-oriented (Heilig, 2003; Meyer and Grabaum, 2008).

In the last decades considerable progress has been made in analysing and quantifying a multitude of landscape functions. A large number of studies have focused on various aspects of landscape functions and its multifunctionality (Costanza et al., 1997; Costanza and Farber, 2002 ; Dijst et al., 2005; Potschin and Haines-Young, 2006b). However, an issue that is not yet sufficiently resolved is how the spatial heterogeneity of landscape functions can be accounted for (Troy and Wilson, 2006; Meyer and Grabaum, 2008).

Spatial information of landscape functions is scarce as only some landscape functions directly relate to observable landscape features (e.g. built-up area and residential function, or forest and timber production). Spatial information of other landscape functions depends on additional intensive field observations or cartographic work.

The objective of this chapter is to present a methodological framework to quantify landscape functions and to make their spatial variability explicit. We present three methods to map and quantify landscape functions depending on the availability of spatial information (i) linking landscape functions to land cover or spatial policy data, (ii) empirical predictions using spatial indicators and (iii) decision rules based on literature reviews. An application of the methodology is illustrated for the Gelderse Vallei region of The Netherlands

Data and methods

Landscape functions

In this study eight landscape functions were analysed, namely, the capacity of the landscape to provide, (1) areas for residential use, (2) locations for intensive livestock husbandry, (3) information on cultural heritage, (4) zones for drinking water extraction, (5) an attractive landscape for overnight tourism, (6) habitat for rare, endemic and indicator plant species, (7) arable agriculture production fields, and (8) an attractive landscape for leisure cycling. This selection of landscape functions was based on their different levels of spatial information availability and the current spatial planning policy focus of the case study region (Provinces of Gelderland and Utrecht, 2005).

All eight functions were assigned a so-called function proxy variable which could be quantified. Where possible, these proxy variables presented the function capacity measured in units relating to the anthropogenic use, or services of the landscape (Table 2.1.). In this chapter when 'landscape functions' are mentioned, we actually refer to the measurable proxy variable for that specific function.

Table 2.1. Overview of the selected landscape functions with their proxy variable and available delineation data.

Landscape function	**Function definition** *The capacity of the landscape to provide:*	**Function proxy for capacity measure**	**Delineation level and data source**
Residential	Areas for residential use	Population per residential neighbourhood	Complete, land-cover data
Intensive livestock	Locations for intensive livestock production	Economic farm size (Dutch Standard Unit)	Complete, land-cover data
Cultural heritage	Information on cultural heritage	Unchanged land-use in policy defined historical landscapes (%)	Complete, policy documents
Drinking water	Zones for drinking water extraction	Drinking water pumping license (m^3/yr)	Complete, policy documents
Tourism	An attractive landscape for overnight tourism	Tourist accommodation suitability	Partial, accommodation sites
Plant habitat	Habitat for rare, endemic and indicator plant species	Conservation Value index	Partial, observation sites
Arable production	Crop production fields	Yield (ton/ha)	Partial, observation sites
Leisure cycling	An attractive landscape for leisure cycling	Potential leisure cycling population	Not available

Overall methodology

The overall methodological framework is based on the available data on the location of the selected landscape functions. Driven by the link between landscape functions and observable landscape features or policy delineation, three different levels of landscape function delineation, in terms of location and extent, can be distinguished (Figure 2.1.):

1. Complete delineation: Landscape functions are directly observable from the land cover or are defined by policy regulations.
2. Partial delineation: Non-directly observable landscape functions whose delineations are non-comprehensive or based on sample point data. Function data originated mainly from field observations.
3. No delineation: Not-directly observable landscape functions lacking any direct spatial referenced information on their location.

These three levels of landscape function delineation form the basis of our different landscape function mapping approaches. In this framework, functions are quantified based on the actual or potential services they are providing.

The first group consists of landscape functions with complete delineation data, so location and extent of each of these functions is exactly known. This spatial information is based either on directly observable cover data or on through policy delineated areas. Spatially referenced data were used to quantify the capacity of the function at that location.

The second group consists of landscape functions with incomplete delineation data, so location and extent of these functions is only partly known. The lack of delineation data is related to the fact that these landscape functions can not directly be observed from the landscape. It is assumed that land cover, biophysical and socioeconomic landscape components can be used to describe the location and capacity of landscape functions. These different landscape components were translated into spatial indicators. Multivariate regression techniques were used to empirically quantify the influence of these spatial indicators on function variability. In the next section these techniques will be discussed in detail. Using the empirically derived relations, the partially delineated landscape function was extrapolated to a quantitative landscape function map covering the whole study area. After defining the function capacity a threshold was introduced to delineate the assumed presence of the landscape function for human use or policy making.

The third group consists of landscape functions lacking any delineation data, so no data on function location and extent are available. In this case spatial indicators and literature based decision rules were used to come to a quantitative landscape function map. Here again a threshold value was determined to delineate the area in which the function was considered present for human use or policy making.

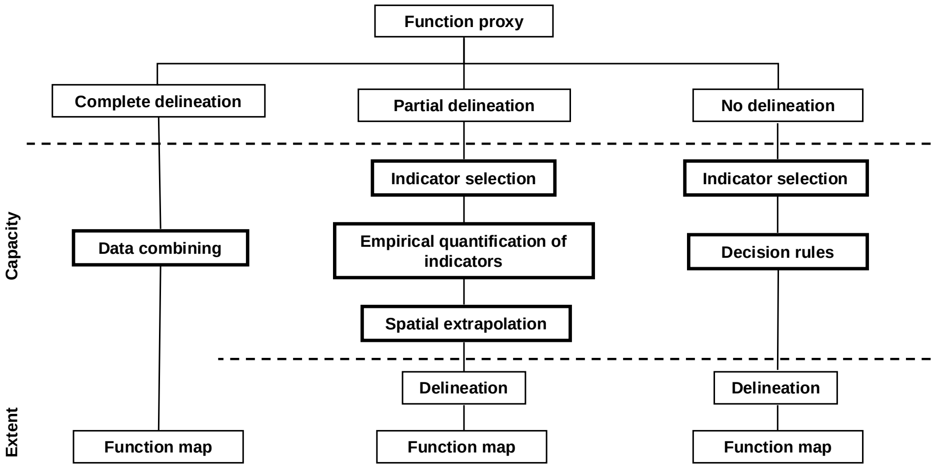

Figure 2.1. Overview of the presented methodology; from landscape function proxy to quantitative and delineated landscape function map.

Analyses in this chapter are based on data for the year 2000, unless mentioned otherwise. All topographic data in this study were derived from the topographical 1: 10 000 map (TDK, 2005) and land-use data originated from the Soil Statistics survey (CBS, 2002). Data sources were converted to a raster format with a spatial resolution of 100 meter, at this resolution we conducted all spatial calculations and presented all maps. Spatial data processing was done using ArcGIS 9.2. All statistical calculations were carried out using the statistical package R 2.2.

Quantifying and delineating landscape functions

Delineated functions

Four landscape functions were completely delineated for our study area. Using land-cover information the *residential* function was delineated by the location of residential neighbourhoods and quantified by the population per neighbourhood (CBS, 2000). The *intensive livestock* husbandry function was delineated by the location of intensive livestock farms and quantified by the economic farm size in Dutch Standard Unit, DSU (Alterra, 2000b). Only farms larger than 20 DSU (gross production larger than €28 000 Euro per year) were taken into account, as agricultural production of smaller farms is too low to sustain a minimal income.

The landscape function providing information on *cultural heritage* was delineated using the location of high value historical landscapes as defined by the province. As authenticity of the landscape is considered an important aspect of this landscape function (Daugstad et al., 2006), the percentage of unchanged land-use between the year 1900 and 2000 within 300 meter of each raster cell was used to quantify the function. The fourth complete delineated function in our study area was the *drinking water* extraction zone function. Here we used policy defined groundwater protection zones for function delineation. Within these protected areas water remains in the underground aquifer for approximately 25 years before being extracted. The permitted quantity of drinking of water that companies may extract (in m^3/yr) at these locations was used to quantify this function.

Partial delineated functions

Tourism

The capacity to provide an attractive landscape for overnight *tourism* was quantified by means of tourism suitability. Function data on tourism suitability were available through the current locations of rural accommodation sites. Delineation of this function was considered partial as the suitable landscape for tourism goes beyond the location of tourism accommodations. In this analysis accommodation types included camp sites, chalets and group accommodation sites (KvK et al., 2005). Hotels were not included as they are often located in urban areas and do not solely host tourists.

In the study area 397 raster cells contained one or more tourist accommodations. The selection and quantification of the tourism function was based on a logistic regression. A logistic regression estimates the probability (0 to 1) of the occurrence of an event based on a set of independent variables. This regression type requires a binary dependent variable (in this case 'presence' and 'absence' of accommodation sites). Therefore, an equal number of absence cells (397) were randomly sampled from the study area. A mask of 500 m by 500 m around all presence locations was introduced to avoid 'absence sampling' in the direct neighbourhood of observed accommodation locations.

The selection of potential indicators for suitable tourism locations was based on European studies carried out to identify attractive rural areas for tourism (Goossen et al., 1997; Walford, 2001; EC, 2002; Roos-Klein Lankhorst et al., 2005). The most important landscape characteristics for tourism in the Dutch context were: land cover, level of disturbance, recreation possibilities and accessibility.

Land-cover indicators, primarily related to landscape aesthetics, were included by taking the percentage of agriculture, built-up and natural areas surrounding the tourist locations. These three land-cover classes were chosen as they contributed most to landscape perception (Van den Berg et al., 1998; Roos-Klein Lankhorst et al., 2005). The built-up land-cover class included overall built-up area and land cover related to industrial activities. To take into account different scale levels at which land cover might influence tourism suitability a radius of 500 m and 5 km was used to describe the surrounding land cover. Besides the specific land-cover classes also the line of sight indicating the openness of the landscape was included (Weitkamp et al., 2007). The level of disturbance was expressed by the distance to a highway (the major source of noise in the study area) and distance to intensive livestock farms indicating the level of smell disturbance. Recreation possibilities were indicated by the distance to natural areas large enough for recreation (> 1 km^2), density of trails in natural areas, distance to swimming locations, presence of cultural historical elements in the neighbourhood and local road network for cycling recreation. Accessibility was calculated measuring distance to main roads and highways.

A stepwise logistic regression in both directions (following Vernables and Ripley, 2002) was used to make a selection of predictive variables based on the Akaike's information criterion (AIC) scores. A lower AIC indicated a better fit with a greater parsimony. To ensure independence among the variables, the variance inflation index (VIF) was calculated. The VIF indicates the effect of each other independent variable on the standard error of the regression coefficient (Hair et al., 1998). The performance of the final model was assessed by the area under the curve (AUC) of the relative operating characteristic, indicating the ratio of true positive and false positive predictions for an infinite number of cut-off values (Swets, 1988). The AUC values can vary between 0.5 (completely random prediction) and 1 (perfect discrimination).

We cannot assume that all locations without tourism accommodation are simply not suitable. To account for this uncertainty in the tourism accommodation absence data, we repeated the random sampling of tourism accommodation absence points 100 times. The tourism suitability model was therefore calculated 100 times and the average regression results (beta estimates and AUC) are presented in this chapter.

To validate the accuracy of the tourism model, regression models were fitted using only 75% of the data. The remaining 25% was considered independent and used to test the prediction accuracy. This procedure was repeated based on the 100 different datasets. For each model the AUC, based on the 25% of the data, was calculated. After obtaining information on model behaviour by this split-sample validation the regression model (with average betas estimates) based on the full dataset was used to extrapolate the tourism function suitability for the whole study area. The probability, or suitability, value of 0.50 was used as threshold to define the function delineation.

Plant habitat

The landscape function providing *habitat* for rare, endemic and indicator plant species was quantified using a nature value index. Delineation data came from a nature value inventory carried out and made available by the Province of Gelderland (Rijken, 2000). This inventory included point locations spread over the study area at which occurrence of plant species was recorded. Hertog et al. (1996) used these plant species occurrence data as input for the calculation of the biodiversity conservation value (CV). For each observation point this conservation value index was calculated taking into account characteristics of all plant species at that specific location. These species characteristics are national and international rareness, trend in occurrence, vulnerability and importance of the species for a specific vegetation type. Based on these characteristics the conservation value was determined, ranging from a value from 0 to 10, with 10 being the locations with the highest plant nature value. To avoid over-representation and to reduce spatial autocorrelation, the conservation value of raster cells containing more than one observation was averaged, resulting in 738 raster cells containing plant habitat data (from the period 1998 to 2001). Contrary to the binary tourism accommodation data, these plant habitat function data consisted of continuous sample data. Therefore, we used for the empirical analysis of this function a regression type for continuous metric dependent data: a multiple linear regression.

The most important characteristics of landscape functionality for plant habitats were included in the plant habitat function assessment. These were soil type, groundwater level, nitrogen availability, and land cover (Noss, 1990; Van Ek et al., 2000; Wamelink et al., 2003). The biophysical conditions were derived from soil (De Vries et al., 2003), modelled groundwater (Finke et al., 2004) and assessed excess nitrogen (Gies et al., 2002) maps. Land-cover indicators included the main land-cover classes (forest, open nature, arable and grass lands, urban area, and infrastructure) and their size and log distances.

Variables were selected by a stepwise linear regression based on the AIC and tested for independence using the VIF. Performance of the final model was indicated by R-squared. The final regression model was used to extrapolate the conservation values for the whole study area, excluding all built-up areas. All areas with a conservation value higher than 5 were considered areas where landscape has the capacity to provide good habitat for rare, endemic and indicator plant species (Hertog and Rijken, 1996) and was therefore used as a threshold for the function delineation.

To test how well the plant habitat regression model was able to estimate conservation values, model accuracy was determined by a 10-fold cross-validation (Fielding and Bell, 1997; Hair et al., 1998). Conservation value data were randomly split into ten approximately equal-sized groups. Each group was considered an independent validation data. The validation dataset was used to validate the model which was calibrated using the other 9/10 of the data. The R-squared was computed for each of the ten validation groups. Additionally, the standard deviation of the beta estimates of the ten different calibration models was computed to derive information on the model's stability.

Arable production

The third landscape function having a partial delineation dataset was the arable production function. Function delineation data were based on the location of arable production fields. Arable agriculture was considered not fully delineated by land cover because of rotation practices. Arable fields are not at the same location every year and therefore land-cover maps generally may not correctly display the spatial delineation of the arable production function. The arable production function was quantified based on the crop yield (ton/ha) reported per postcode area (23 in total). Maize is the only commonly grown arable crop in the study area, therefore only maize production data were considered. Yield data came from a survey carried out by the Dutch Agricultural Economics Research Institute in 2005. Each maize field was assigned the value of the average maize production of the postcode area in which it was located. The final dataset contained 588 maize fields with production data.

Important landscape characteristics to explain the spatial variation in arable production in the Netherlands are soil type, groundwater level (Wijk et al., 1988) and farm characteristics. In our study area maize is mostly cultivated by dairy farms as it serves as fodder crop for their cows. Farm characteristics of dairy farms, including average farm size in hectares and number of farms per postcode area, were derived from farm census data (Alterra, 2000b). Soil types (De Vries et al., 2003) and the modelled groundwater levels (Finke et al., 2004) were aggregated to field level.

A multiple linear regression was used to analyse the relations between the arable production function and landscape data. Using a stepwise approach in both directions based on the AIC a selection of predictive landscape variables was made. To decrease the spatial autocorrelation effect in our analysis we applied regressions on 100 randomly sampled fields and repeated this 100 times. The average beta coefficients of the 100 regression models were used to extrapolate the estimated crop yields to all areas under agriculture use in the study area. Afterwards a minimum yield threshold of 35 ton/ha was introduced to define the function delineation. This is the minimum estimated maize yield within the 95% interval for the case study region (CBS, 2000). Model accuracy and stability was determined by a cross-validation using the left-out data points of the repetitive random sampling procedure. Within each model run the R-squared of the validation data was calculated and averaged over the 100 model runs.

Not-delineated function

Leisure cycling

To assess the leisure cycling function the following landscape characteristics were included: residential locations, population, average cycling distance, cycling facilities, and visual and noise disturbance elements like industry, business parks and highways (Goossen and Langers, 2000; Gimona and Van der Horst). The majority of leisure cycling primarily takes place in the direct neighbourhood of residential areas (Goossen et al., 1997; CBS, 2000). As leisure cycling requires cycling facilities, all areas with small local roads within a distance of 5 km around each residential neighbourhood were included as leisure areas. All locations with highways, industry, business parks and waste dumps were excluded from the suitable leisure cycling areas. Based on the population that could reach the suitable cycling area, the leisure cycling function was quantified. The leisure cycling area was delineated by excluding all areas with a potential leisure population of smaller than 10 000.

Function map results

Delineated functions

The quantitative landscape functions maps of *residential* areas, *intensive livestock* farm locations, *and drinking water* extraction zones are presented in Figure 2.2a, b and d. To improve visibility of the *intensive livestock* point locations, we mapped the so-called 'odour circles' of 400 m around each farm location (VROM en LNV, 1985) with summed Dutch Standard Units. The assessed quantitative *cultural heritage* map is presented in Figure 2.2c. Based on secondary data we tried to validate the plausibility of this assessment. In The Netherlands several cultural landscapes are protected on national level by the so called Belvedere Act (OCW et al., 1999). One of these nationally protected landscapes is located in the study area. The location of this protected landscape was compared to the location of the highest values on our cultural heritage function map. Both showed a clear spatial overlap indicating that our assessment was reasonable, although more areas at our assessed map scored as high as the nationally protected cultural landscape.

Partial delineated functions

Tourism

Resulting from the averaged regression outcomes of the 100 model runs ten variables significantly explained tourism accommodation locations (see Table 2.2). The variables distance to highway, high density of small local roads, a high percentage of accessible surrounding natural areas and a high percentage of clustered natural areas showed a positive relation with tourist accommodation locations. Areas further away from a highway and in a neighbourhood with many local roads that could facilitate recreational cycling together with accessible natural areas with a high amount of clustered natural areas led to a higher probability for tourism locations. A high percentage of natural areas became significant on a coarser spatial scale, i.e. natural areas in a radius of 5 km showed a positive correlation with suitable tourist locations. The variables openness, distance to natural areas larger than 1 km^2, high percentage of industrial elements and homogeneous natural, agricultural and surroundings showed a negative relation with tourist accommodation sites. The negative influence of both a high percentage of natural and agricultural land cover in the direct surroundings (500 m) indicated that most tourist accommodations are located in heterogeneous land cover areas. The sign of the estimated beta coefficients of the predictive tourism suitability variables in our study coincided with earlier publications on favourable landscape characteristics in The Netherlands (Goossen et al., 1997; EC, 2002; Roos-Klein Lankhorst et al., 2005).

The average AUC of the 100 logistic regression models was 0.84 (Table 2.2) which in land-use studies is interpreted as "very good" (Hosmer and Lemeshow, 2000; Lesschen et al., 2005). The standard deviation in Table 2.2 indicates the stability of the beta estimates over the 100 runs. The VIF of all variables remained under 10, so all variables could be considered independent (Hair et al., 1998). From the validation datasets containing 25% of the data we obtained an average AUC of 0.85. This indicated that our models which based on only 75% were very well able to predict the location of tourist sites.

Using the average betas of the 100 runs, the probability for a suitable landscape for tourist accommodations was estimated. All areas with a probability higher than 0.5 were considered areas where landscape has the capacity to provide an attractive landscape for tourist accommodations (Figure 2.2e). Interpreting the predicted tourism suitability map, tourism areas are mainly located on the border of our study area where a mix of natural and agricultural areas is found.

Table 2.2 Multiple logistic regression results for the tourism suitability function (n=794). Means of the beta coefficients, AUC and standard deviations (S.D.) are based on 100 runs.

Variable	Mean beta estimate	S.D.
Intercept	1.3576	0.3103
Agricultural land cover within 500m (%)	-0.0195	0.0031
Natural land cover within 500m (%)	-0.0578	0.0039
Clustered natural area within 5km (%)	0.0247	0.0087
Openness (m)	-0.0004	0.0000
Distance to highway (m)	0.0001	0.0000
Industrial elements, within 500m (%)	-0.0343	0.0037
Distance to natural area >1 km^2 (m)	-0.0002	0.0001
Distance to swimming location (m)	-0.0001	0.0000
Accessible nature, within 500m (%)	0.0242	0.0000
Local roads within 500 m (%)	0.0388	0.0048
AUC	0.84	0.01

Plant habitat

Following the regression model, six variables could explain the variability in important plant habitat suitability (see Table 2.3) in our study area. Wetter areas in winter time, when the highest groundwater level occurs, further away from forest or open nature showed a lower conservation value. Sandy, sandy clay and peat soils had higher plant conservation value than other soil types (peaty sand, loam, heavy sandy clay, clay and heavy clay). So, variables related only to groundwater level, soil type and land cover. Two other variables (distance to highway and excess nitrogen) were found significant but were removed from the model as no processes could be linked to these. Their influence was contrary to what was expected. Removal of these variables did not change the sign of the beta coefficients of any of the explanatory variables.

The regression model showed an R-squared of 0.47, and independent explanatory variables (maximum VIF 2.7). A residual analysis did not reveal any high leverage data points. The 10-fold cross-validation resulted in an average model accuracy of R-squared 0.46 and as the betas estimates did not show any large fluctuations, we considered our model stable.

Using the regression model, conservation values were estimated and all areas with a conservation value higher than 5 were mapped (Figure 2.2f). Interpreting the function map, high conservation values were only present in natural areas. In The Netherlands cultural landscapes are perceived as important habitats for rare plant species (Kleijn et al., 2001), but this was not supported by our plant habitat map. To account for possible different habitat requirements and therefore spatial indicators for plant species in natural and agricultural areas two extra regression analyses were carried out. One analysis was based on conservation value observations in natural areas, and one on observations in agricultural areas. The R-squared was calculated for the complete study area based on the two land-cover specific models and the overall model as presented in Table 2.3. The land-cover specific models performed less than the overall model (R^2 0.26 vs. R^2 0.47). This difference can partly be attributed to the high influence of the distances to natural land-cover variables. In the natural land-cover model the distance to natural areas was logically not found significant as all locations had the same value there, 0 m. Additionally, our landscape variables could not explain well the variation in nature values within the agricultural land cover. This could be due to variation in agricultural management practices which were not included in our analyses.

To validate the plausibility of our plant habitat model, the predicted high nature value areas (conservation value less than 5) were compared with the location of the State Nature Monuments (LNV, 1998). These State Nature Monuments have a strict protective status because of their exceptionally high nature value. The spatial comparison showed that only two out of five State Nature Monuments appeared in the predicted high nature value function map. This discrepancy between predicted and observed values could be a result of the generalisation of landscape characteristics related to nature value. Different plant communities with different habitat requirements could have similar conservation values. Therefore our plant habitat model is very likely to be biased towards the most abundant plant community habitat requirements.

Table 2.3. Multiple linear regression results ($p<0.001$) for the plant habitat value function.

Variable	Beta estimate
Intercept	8.5482
Highest groundwater level (cm below surface)	-0.0100
Sandy soil (yes/no)	1.0465
Sandy clay soil (yes/no)	1.1458
Peat soil (yes/no)	0.8474
Log distance to forest (m)	-0.1984
Log distance to open nature (m)	-0.5378
Residual standard error	1.506
R^2	0.47

Arable production

The production of arable crops could be explained by seven variables (Table 2.4). Areas with low groundwater levels in summer showed a negative relation with the yield versus areas with a low groundwater level in winter time showing higher yields. Sandy, sandy clay and peaty sand soils have a positive relation with maize yield, compared to peat, loam, heavy sandy clay, clay and heavy clay soils. Also two farm characteristics of the postcode areas showed a relation with the arable crop yield: postcode areas with more and larger sized farmed had higher yields. So, although agriculture in The Netherlands strongly relies on management, maize yields could still be partly predicted by spatial indicators related to land with expected most favourable characteristics.

The 100 repetitions of the regression model based on 100 sampled fields showed an average R-squared of 0.40 and a maximum mean VIF of 3.4 indicating that all explanatory variables in the model can be considered independent. Additionally, no high leverage data points were detected in the regression models. As a result of the limited number of yield data (23 postcode zones), some variables (soil types) showed strong fluctuations in the beta estimates within the model runs (Table 2.4). However, comparing the mean beta coefficients resulting from the random sample models (n=100) with the betas of the regression model of the full data set (n=588), the betas did not show any large differences. The cross-validation of the 100 model runs resulted in an average model accuracy of R-squared 0.36.

Figure 2.2g shows the expected locations where the landscape provides suitable arable production fields.

Table 2.4. Multiple linear regression results for the arable production function. Means of the beta coefficients, R-squared and the standard deviations (S.D.) are based on 100 runs (n=100).

Variable	**Mean beta estimate**	**S.D.**
Intercept	31.1606	4.9134
Lowest groundwater level (cm below surface)	-0.0101	0.0119
Highest groundwater level (cm below surface)	0.0100	0.0091
Sandy soil (yes/no)	1.4325	1.2719
Sandy clay soil (yes/no)	1.8912	2.8915
Peaty sand soil (yes/no)	1.1766	3.1978
Average farm size per postcode area (ha)	0.2649	0.0977
Number of farms per postcode per km^2	46149	11727
R^2	0.40	0.01

Not-delineated function

The not-delineated function - provision of an attractive landscape for *leisure cycling* activities - is presented in Figure 2.2h. Interpreting the delineated *leisure cycling* map almost the whole study area contains an attractive landscape for leisure cycling activities. However the potential leisure cycling population is especially concentrated around and between main residential areas.

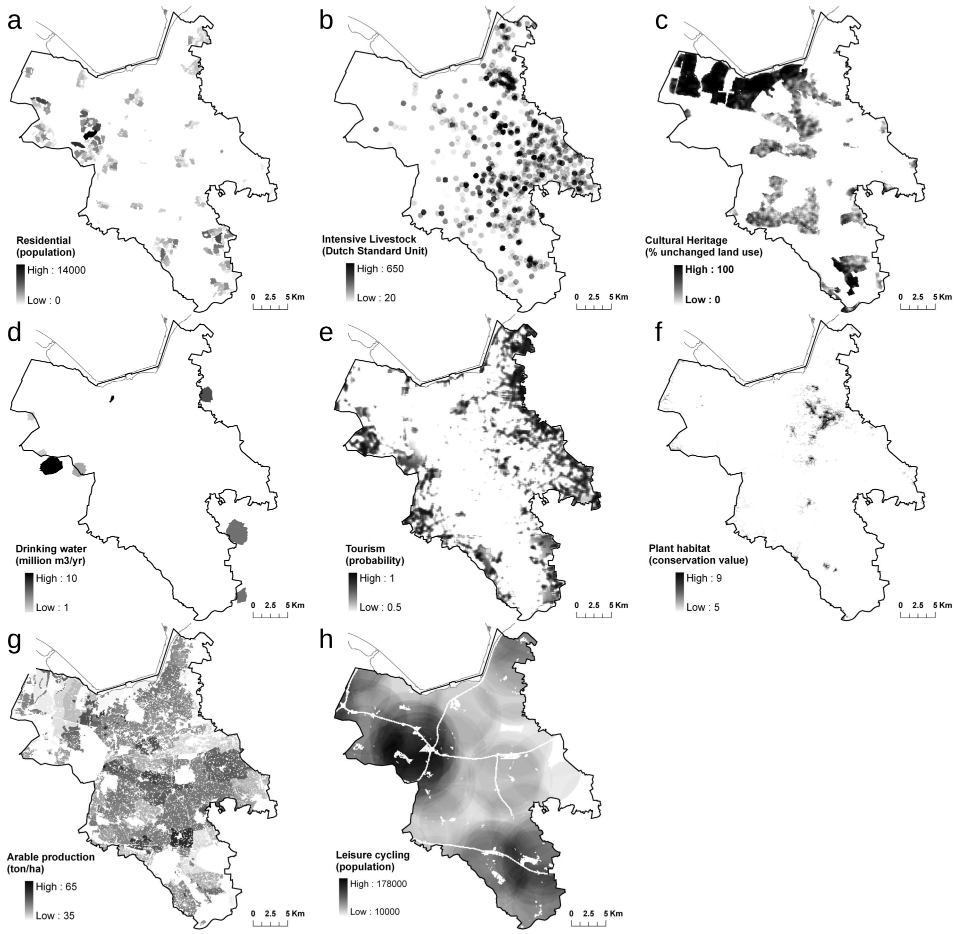

Figure 2.2 Landscape function maps. a) Residential function, b) Intensive livestock function, c) Cultural heritage function, d) Drinking water function , e) Tourism suitability function, f) Plant habitat function, g) Arable production function, h) Leisure cycling function.

Discussion and conclusions

Evaluation of the methodology

The presented mapping methodology accounts for varying availability of information and different properties of landscape functions. The diverse choice of methods seems to be inherently linked to delineating and quantifying landscape functions.

A first observation regarding the general validity of the methodology is that the subdivision based on available information could in some cases be argued: when using delineations of functions based on policy assignment, such as the case of cultural heritage areas in this study, it is not sure that the policy designated areas correspond with the areas that have the highest capacity in providing this function. Such policy function delineation can easily change due to changes in policy focus.

Furthermore, in this chapter, functions described by partial delineation data were extrapolated using empirically quantified relations with spatial indicators. Using empirical models gives the researchers the possibility to identify and quantify site-specific relations between landscape functions and the environment. Empirical techniques are per definition data-driven. This implies that the scale of analysis is primarily defined by the scale of the input data. In regional scale studies, like ours, explanatory input data are generally available at a coarse scale. At this scale only overall patterns and relations between phenomena can be identified (Verburg and Chen, 2000). In our study, aggregated landscape data such as soil type, land-use and topographic features could already explain a large part of the spatial variability of the landscape functions.

Several earlier studies have used spatial indicators together with decision rules to map a range of landscape functions or their supplied services (e.g. Haines-Young et al., 2006; Gimona and Van der Horst, ; Meyer and Grabaum, 2008). Like the leisure cycling function in this study, these authors reviewed the literature to define spatial requirements of landscape functions. Decision rules based on literature make best use of available knowledge and underlying theories. A drawback of this approach is that these decision rules are based on general assumptions not on site-specific quantified relations.

By weighing spatial indicators within the decision rules, a gradient in landscape function suitability could have been obtained like, e.g. the recreation quality map in Haines-Young (2006). We decided not to use a weighing system because we lacked information to justify such a quantification of our spatial indicators. Instead we made a binary leisure cycling suitability map in which the gradient was determined by the potential usage.

All functions in our study were quantified based on the provision of the actual or assessed services. Other, so-called valuation studies have been carried out trying to quantify the value of the supplied services (Costanza et al., 1997; Turner et al., 2003; De Groot, 2006; Hein et al., 2006). The valuation of services very much depends on demand and appreciation of these benefits to humans (Turner et al., 2003). We decided not to focus on this human valuation but rather on the capacity of the landscape to provide these services. The quantitative landscape function maps resulting from our methodology could serve as an input for future valuation exercises.

As in many landscape studies, proper validation data were lacking to validate the landscape function mapping exercise thoroughly. Only two maps could be validated for its plausibility of location using secondary data sources (plant habitat and cultural heritage function). Additionally three functions maps which were based on empirical extrapolation techniques could be cross-validated for model accuracy (tourism, plant habitat and arable production function). Due to this lack of validation data, uncertainty in the function maps could not be quantified. Therefore we consider a clear communication of data choice and all assumptions to end-users as an important aspect of the presentation of the results.

General applicability

Although the proposed methodology has been specified in detail for our case study area, the general approach should be applicable in other case studies as well. Undoubtedly, different areas will have different data availability, different function definitions and thresholds apply. But we believe that by following our framework best use can be made of available data and the inherent characteristics of different landscapes, also in other regions in the world.

The quantitative landscape function maps resulting from this methodology could support policy makers and spatial planners by providing insight into the functional capacities of the landscape. Regional qualities in terms of functionality can easily be interpreted from the landscape function maps. Additionally, the selection and quantification of spatial indicators, which in this study were used to extrapolate functions, can give insight to important landscape components and underlying processes explaining landscape functionality, providing that these are based on causal relations. By showing where functions depend on and through what factors they could be enhanced, the identified landscape indicators can support land-use management, for instance when assessing the potential impact of policy implementations on landscape functionality (Groot et al., 2007; Meyer and Grabaum, 2008). Especially in regions where specific landscape function data are lacking or incomplete, but where spatial biophysical, socioeconomic and land-cover data are already available, the proposed methodological framework helps generating more information on the landscape without intensive new data gathering is necessary.

In our study area many locations with multiple landscape functions were present. Possible interactions between landscape functions could be analysed by overlaying function maps and comparing spatial indicators. In this way, areas can be identified in which function interactions lead to possible synergies or conflicts. Policy makers can use this information to design spatial policies and (ex-ante) evaluate the effect of their land-use strategies on the capacity of the landscape to provide services. Especially for areas with high pressure on land resources, good management of interacting functions within a multifunctional landscape could promote sustainable land-use. Within such a sustainable land-use, societal demands should be satisfied in the most optimal way (Wiggering et al., 2006).

Making landscape functions spatial explicit, adds an important component to research conducted in the field of quantification of services. While most other quantification methods lack a spatial component, we presented a first step in the methodological development to quantify spatial variability of landscape functions. Accounting for this spatial variation has large potentials to improve further quantification efforts. Furthermore, as the presented method explicitly considers the spatial heterogeneity and complexity of landscape characteristics, it contributes to an integrated policy support aiming at strengthening sustainable management of multifunctional areas.

Chapter 3

Quantifying interactions between multiple landscape functions

Rural landscapes are often multifunctional, meaning that at one single location different services are being provided. Multifunctionality is spatially heterogeneous as not all areas are equally suitable to supply multiple services. This suitability depends on favourable biophysical and socioeconomic conditions and interactions between landscape functions. The objective of this chapter is to identify and quantify interactions between landscape functions in a diverse and dynamic rural region, the Gelderse Vallei in The Netherlands. First, multifunctionality in the study region is identified and quantified. The results of these analyses are used to study three aspects of landscape function interactions (i) influence of landscape characteristics on function interactions, (ii) interrelations between landscape functions and (iii) effect of multifunctionality on the different landscape functions. Landscape functions do not equally interact with each other, some landscape functions are affected negatively by the presence of other functions while other landscape functions benefit from multifunctionality. At multifunctionality hot-spots different landscape functions are present that are enhancing each other. Additionally, in our study area it appears that mainly locations with landscape functions that sub-optimally provide services are strongly multifunctional. Quantification and an improved understanding of landscape interactions will help to design and evaluate spatial policies related to the provision of multiple services by the landscape.

Based on: L. Willemen, L. Hein, M.E.F. van Mensvoort, P.H. Verburg
Ecological Indicators, 10 (2010), 62-73

Introduction

In Europe, rural areas take up approximately 80% of land areas (OECD, 1994; EC, 2003). As rural landscapes contain different landscape functions, such as agricultural production and cultural functions, their space can be used for more purposes than just agricultural production alone (OECD, 2001). Based on the definitions of De Groot (1992) and the Millennium Ecosystem Assessment (2003) we use the term 'landscape function' to indicate the capacity of a landscape to provide services to society. These services are all benefits people obtain from landscape, such as food, fresh water and recreational benefits (Millennium Ecosystem Assessment, 2003). In multifunctional areas, more than one landscape function to provide these services is present at a single location. Barkman et al. (2004) define multifunctionality of the landscape as "the phenomenon that the landscape actually or potentially provides multiple material and immaterial 'goods' to satisfy social needs or meet social demands". By including the landscape's potential in this description, a landscape is by definition multifunctional. However, the capacity of a landscape to provide services might be below an acceptable quantity, leading to a negligible value for human use or policy making. In this study we only consider locations at which more than one landscape function has the capacity to provide sufficient services to be of interest for human use, as multifunctional landscapes.

Not all areas and all combinations of landscape function are equally suitable for a multifunctional land-use. This spatial heterogeneity is caused by differences in biophysical and socioeconomic conditions supporting different landscape functions (De Groot, 1992; Wiggering et al., 2006; Syrbe et al., 2007; Metzger et al., 2008). Additionally, multifunctionality is influenced by interactions between landscape functions (Sattler et al., 2006; Groot et al., 2007; Van Huylenbroeck et al., 2007). A landscape function interaction can be described as the effect of one landscape function on another landscape function. Such interaction can influence the capacity of the landscape to provide services. Landscape interactions can be ordered into three classes: (1) conflicts, the combination of landscape functions reduces a landscape function in its provision of services to society, (2) synergies, the combination of landscape functions enhances a landscape function, or (3) compatibility, landscape functions co-exist without reducing or enhancing one other. For example, intense residential use and provision of plant habitat are conflicting landscape functions as the presence of the residential function likely decreases or excludes the plant habitat function. In another case a landscape function directly provides favourable conditions to another landscape function and synergy takes place. For instance in areas with a nature and tourism function. Natural areas directly and positively influence the tourism function, as the tourism suitability of an area will increase by the presence of nature (Van den Berg et al., 1998; Roos-Klein Lankhorst et al., 2005). In case of compatibility landscape functions do not reinforce nor reduce one another, even though these functions are present at the same location.

Because of the current increase in societal demands for recreational space and conservation of the natural biodiversity along with agricultural production, the demand for services provided by rural areas has increased (Baudry et al., 2003; Hall et al., 2004; Buijs et al., 2006), while the total amount of rural areas in the European Union is decreasing due to urban sprawl (Reginster and Rounsevell, 2006). Therefore, recent spatial policies pay much attention to the multifunctional character of landscapes (OECD, 2001). The identification and subsequent understanding of landscape function interactions depends on the influence of landscape characteristics and the role of other landscape functions. Such understanding will help to design spatial policies and assess the effect of their land-use strategies on the capacity of the landscape to provide services (Sattler et al., 2006). Especially for areas with a high pressure on land resources, good management of interacting landscape functions could contribute to sustainable land-use.

Interactions between landscape functions in multifunctional areas have been studied before (e.g. Gomez-Sal et al., 2003; O'Rourke, 2005; De Groot, 2006; Sattler et al., 2006). These studies focused on either a qualitative description of landscape function interactions or on quantitative landscape function trade-off analyses. All lacked, however, a strong quantitative spatial component. Only Chan et al. (2006) and Egoh et al. (2008) quantified spatial associations between ecosystem services but did not specifically focus on function interactions. The objective of this chapter is to identify and quantify interactions between landscape functions in a rural region, the Gelderse Vallei region in The Netherlands. This region has a high diversity of landscape functions and strong land-use dynamics and therefore provides and appropriate case study. We analyse three different aspects of the landscape function interactions: (1) What landscape characteristics influence landscape function interactions? (2) How do landscape function capacities interrelate? And (3) How does multifunctionality affect different landscape functions?

Data and Methods

Research approach

The overall approach to study landscape interactions at multifunctional location consists of three steps. First, the variation of the current landscape functions is made spatially explicit by mapping all landscape functions using landscape indicators. Second, multifunctionality is quantified and mapped by combining the separate landscape function maps. Third, by using landscape indicators and the quantified multifunctionality maps, we analyse three different aspects of landscape functions interactions. The analyses of interactions includes a) identification of landscape characteristics influencing landscape function interactions, b) calculation of correlations between landscape function and c) determination of the effect of multifunctionality on a single landscape function. Based on the analysis of multifunctionality effects, multifunctional hot-spots are identified. In these analyses we assume that the current spatial patterns of landscape functions result (partly) from the function interactions. Figure 3.1 schematically summarises our approach.

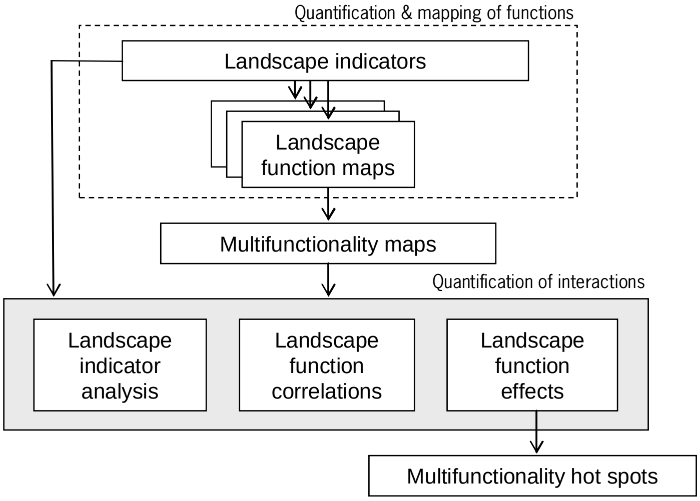

Figure 3.1. Overview of the research approach; from landscape indicators to multifunctional hot-spots.

Following the current focus of spatial policies of the study region seven landscape functions are included in our analyses; the capacity of the landscape to provide, (1) residential use, (2) intensive livestock husbandry, (3) cultural heritage, (4) overnight tourism, (5) habitat for rare, endemic and indicator plant species, (6) arable agriculture production, and (7) leisure cycling (Provinces of Gelderland and Utrecht, 2005).

All analyses in this chapter are based on data for the year 2000. Topographic and land-use data for our study area are derived from the Dutch topographical map (TDK, 2005) and Soil Statistics Survey (CBS, 2002) which were both created at a scale of 1:10 000. All spatial data sources are converted to a raster format with a resolution of 100 by 100 meter, which is used to conduct all calculations, indicator definitions and map presentations. To avoid border-effects in our results we carried out our spatial analysis for the study area plus a buffer of approximately 20 kilometres. Spatial data processing is done using ArcGIS 9.2. All quantitative analyses are carried out using the statistical package R 2.2 (R Development Core Team, 2008).

Quantifying and mapping landscape functions

Common techniques to observe the landscape focus on land cover and therefore do not provide enough information on the spatial variation of all landscape functions. For this reason alternative methods are needed when studying and mapping landscape functions (Brown, 2006; Haines-Young et al., 2006). In Chapter 2 we developed a methodological framework to spatially describe landscape functions using landscape indicators, representing a variety of biophysical and socioeconomic landscape elements and processes. Apart from land-use/cover data these landscape indicators are also based on alternative data sources including, among others, census data, spatial policy documents, and biophysical data. Key in this methodological framework is the availability of spatial information on the location and capacity of each landscape function. For some landscape functions the exact location is known because they can be directly observed (e.g. built-up area for residential function). While for other landscape functions the exact location needs to be assessed as direct observation from the landscape is not possible (e.g. the location of the tourism function can not be directly observed as this function relates to many different landscape characteristics). So based on the available landscape function delineation data an adequate mapping method needs to be chosen. In this chapter, when 'landscape functions' are mentioned, we refer to a measurable variable in units relating to the anthropogenic use or policy focus (e.g. yield in ton per ha) for that specific function. Table 3.1 lists the landscape function names and their quantified variables as used throughout this chapter.

Following the methodology described in Chapter 2, three of the seven selected landscape functions (residential use, intensive livestock husbandry and cultural heritage) can be directly mapped based on indicators derived from land cover and policy documents describing their exact location. These three landscape functions are subsequently quantified using indicators reflecting the actual provision of services. The residential function is mapped using topographical data on residential areas together with demographic information on the population per postal code area. The intensive livestock husbandry function is delineated by the known location of intensive livestock farms and quantified by the farm size in standardised livestock units (Alterra, 2000b). The landscape function providing information on cultural heritage was delineated using the location of high value historical landscapes as defined by the province government. As authenticity of the landscape is considered an important aspect of this landscape function (Daugstad et al., 2006), the percentage of unchanged land-use between the year 1900 and 2000 within 300 meter of each raster cell was used to quantify the function.

For the four other selected landscape functions no complete delineation data describing their exact location in our study area is present. For three of these landscape functions; overnight tourism, habitat for important plant species, and arable agriculture production, point observations on the location of service provision is available. This allows us to empirically quantify and map these landscape functions based on regression analyses. Using a regression analysis, significant landscape indicators are selected that relate to locations at which services are provided by the landscape functions. These indicators are hereafter used to extrapolate these landscape functions to the complete study area. For example, the tourism function is assessed by regressing landscape characteristics to the locations of the current tourism accommodations. In our study area about 400 raster cells contain one or more tourist accommodations, which include camp sites, chalets/cottages and group accommodation sites. Hotels are not included in the analysis as they are often located in urban areas and do not solely host tourists. The selection of landscape indicators and quantification of the tourism function is based on a logistic regression. This regression type requires a binary dependent variable (in this case 'presence' and 'absence' of accommodation sites). Therefore, an equal number of absence cells are randomly sampled from the study area. A mask of 500 by 500 meter around all presence locations is introduced to avoid 'absence sampling' in the direct neighbourhood of observed accommodation locations. The selection of potential indicators for suitable tourism locations is based on European studies carried out to identify suitable rural areas for tourism (Goossen et al., 1997; Walford, 2001; EC, 2002; Roos-Klein Lankhorst et al., 2005). The most important landscape characteristics for tourism in the Dutch context are: land cover, level of disturbance, recreation possibilities and accessibility. All significant independent variables are included in the regression model and used to map the tourism probabilities for the study area. A similar approach was followed for the plant habitat and arable agriculture production functions. For the plant habitat analysis data are used from a large scale nature value inventory (Rijken, 2000). This inventory includes point locations spread over the study area at which occurrence of all plant species was recorded. Hertog et al. (1996) used these plant species occurrence data to calculate the suitability of a location to provide habitat to rare, endemic and indicator plant species. For our study area these species mostly relate to vegetation growing under mesotrophic wet conditions. The landscape characteristics to estimate suitable plant habitats included soil type, groundwater level, nitrogen availability, and land cover (Noss, 1990; Van Ek et al., 2000; Wamelink et al., 2003). The plant suitability figures together with variables describing landscape characteristic are included in a linear regression analysis to explain and estimate the plant habitat function. The arable production function is assessed using land-use data combined with yield data from a survey carried out by the Dutch Agricultural Economics Research Institute. Landscape indicators describing soil type, groundwater level and farm characteristics are used to estimate the location and quantity of the arable production function using a linear regression model.

One landscape function lacking any spatial information on the location and quantity of provided services, the leisure cycling function, is assessed and mapped based on landscape indicators and decision rules derived from the literature. Using spatial information on residential locations, average cycling distance, cycling facilities, and visual and noise disturbance elements like industry, business parks and highways, the areas suitable for leisure cycling function are mapped (Goossen et al., 1997; Goossen and Langers, 2000; Gimona and Van der Horst). The leisure cycling function is quantified by the population that is within reach (i.e. closer than 5 km away) of the suitable cycling area (CBS, 2000).

For all landscape functions maps thresholds indicating the minimum for occurrence are introduced to only present the locations at which landscape functions provide enough services to be of interest for human use. We refer to Chapter 2 for detailed information on the complete landscape function mapping methodology and internal validation analyses.

Quantifying and mapping multifunctionality

To quantify multifunctionality, the number of landscape functions at each location (represented by a raster cell) and the summed capacity of service provision of the overlapping landscape functions are calculated. This results in two quantified maps indicating the level of multifunctionality. In order to determine between which overlapping landscape functions interactions can take place, all landscape function combinations in our study area are listed and assigned a unique ID number. To quantify the total potential provision of services at multifunctional locations, the capacities of all landscape function are normalised and summed. A min-max normalisation is used to normalise landscape function capacities to a 0 to 1 scale, similar to the procedure used by Gomez-Sal et al. (2003) and Gimona and Van der Horst (2007). This normalisation technique is, however, very sensitive for minimum and maximum values. To avoid erroneous transformation due to outliers, all landscape function maps are first *winsonised* based on the 5-95 percentile of the assessed capacity range, i.e. all values outside the 5-95 percentile are respectively assigned the 5th or 95th value (Venables and Ripley, 2002).

Quantifying landscape function interactions

Following Figure 3.1 three methods are applied to describe the different aspects of interactions between landscape functions at multifunctional locations.

Landscape indicator analysis

Landscape indicators are the basis of our spatial definition of landscape functions. In this analysis we indentify what landscape characteristics influence landscape function interactions. A landscape function synergy can be found when a favourable landscape function indicator is directly linked to another landscape function. Like when, for instance, a landscape function contributes to landscape characteristic (e.g. plant habitat function contributes to the creation of natural area) which positively relates to another function (e.g. the tourism suitability increases when it is in close proximity to natural areas). When the opposite is the case, a landscape function is negatively influenced by a landscape characteristic generated by another landscape function, these landscape functions could be conflicting. Finally, locations containing landscape characteristics that are beneficial for multiple landscape functions can support landscape function compatibility.

The same landscape indicators that are used to spatially define landscape functions are included in this interaction analysis. In this analysis we list all indicators showing relations with more than one landscape function together with their possible effect on landscape function interactions. Landscape functions having similar landscape requirements could be joined in a multifunctional landscape. In case of conflicting landscape requirements, multifunctionality is not supported and might lead to a decreased provision of the total services.

Landscape function correlations

The second analysis quantifies the relations between landscape functions across the study area directly, in comparison with the first analysis that focuses on landscape function indicators. To test and quantify relations between each landscape function pair, Spearman's rank correlations are calculated. We based the correlation analyses on a random sample representing approximately 10% of landscape function data to decrease spatial-autocorrelation effects between observations.

Likely, two landscape functions will not equally interrelate. For example, the relation plant habitat to tourism can be different from the relation tourism to plant habitat (positive versus negative). To account for these possible asymmetrical correlations, we select per landscape function all observations at which the landscape function is present. From this selection, correlation coefficients between all landscape functions are calculated. In case another landscape function is not present at that location, a capacity of 0 is assigned to that function observation. For example, to quantify the correlation between areas with a tourism function and arable production function, first, all locations with a tourism function are selected and correlated with the capacity of the arable production function, secondly, all locations with an arable production functions are selected and correlated with the capacity values of the tourism function. This approach results in an asymmetrical correlation matrix.

The correlation figures indicate a trend in capacity between the landscape functions, from which landscape function interactions can be derived. For example, in case of a negative correlation a conflict between landscape functions can be assumed; the capacity of a landscape function decreases as the other landscape function gets stronger.

Multifunctionality effect analysis

To quantify multiple function interactions we study the effect of the presence of different landscape function combinations on the capacity of a specific landscape function to provide services. Knowing this effect of multifunctionality on the capacity of a landscape function, combinations of landscape functions leading to high service provision can be identified. We consider all multifunctional locations at which combination of landscape functions results in an increased capacity for two or more landscape functions, a multifunctionality hot-spot (versus two or more conflicting landscape functions resulting in cold-spots).

For this analysis, the effect of each combination of landscape functions on the different landscape functions is first tested using an ANOVA. In case the capacity of a landscape function differs among the landscape combinations, we define for which combination of landscape functions their capacity significantly deviates from the expected capacity. The expected capacity is calculated from the average capacity among all function combinations. The differences in capacity are tested using a Welch t-test, a type of t-test that does not assume equal variances in the two samples (Venables et al., 2003). These statistical analyses are carried out on a 10% random sample of our functions maps. Landscape function combinations that only occasionally appear in our data (falling in the > 97.5 percentile of the cumulative area distribution) are removed from our data set. To decrease the effect of large landscape function groups on the overall mean, the average values of the function capacity are calculated from the complete sample data set, that is to say, all values before rare landscape function combinations data are removed.

Results

Quantifying and mapping landscape functions

Landscape indicators that are used to quantify and spatially describe landscape functions are listed in Table 3.1. The table also indicates the relative weight of each indicator to describe the capacity of landscape functions. For the three landscape functions that are quantified using regression analyses (tourism, plant habitat, and arable production), the standardised beta coefficients indicate the indicator weighting (Bring, 1994; Menard, 2004). In Table 3.1 also the landscape indicators that are used to define a boundary condition (not the capacity) of a landscape function are listed, these lack a quantified weighting. Figure 3.2 shows all quantified and maps landscape functions for our study area.

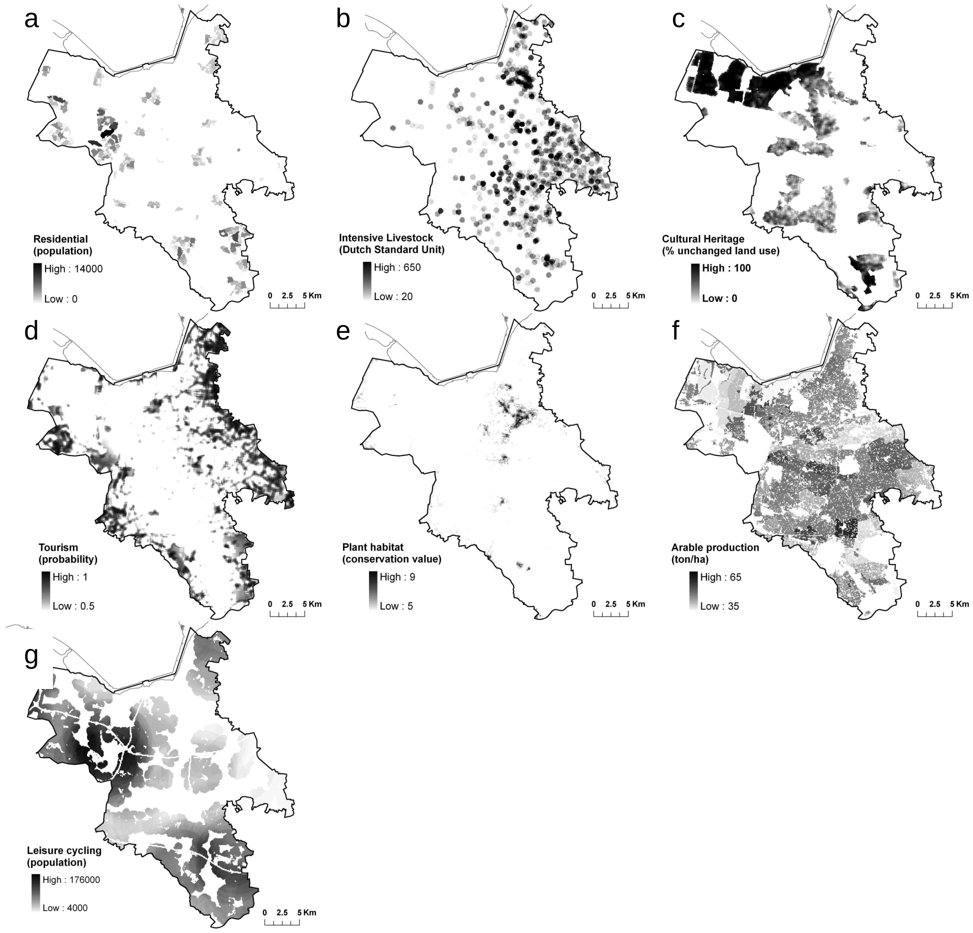

Figure 3.2 Landscape function maps. a) Residential function, b) Intensive livestock function, c) Cultural heritage function, d) Tourism suitability function, e) Plant habitat function, f) Arable production function, g) Leisure cycling function.

Table 3.1 Landscape functions together with their weighted indictors for function quantification. Landscape indictors only defining function delineation are indicated with a weight '–'

Landscape function	Landscape indicator	Weight
Residential	Residential area	-
	Number of residents per ha	1
Intensive livestock	Livestock farm zone	-
	Livestock production and farm size (Economic Units)	1
Cultural heritage	Policy indicated area	-
	Unchanged land-use/cover (% within 250 m radius)	1
Tourism *	Homogeneous agricultural land-use (% within 500m radius)	-1.13
	Homogeneous natural area (% within 500m radius)	-3.08
	Clustered natural areas > 1km^2 (% within 5 km radius)	0.86
	Openness of landscape (line of sight m)[1]	-0.84
	Distance to highway (m)	0.36
	Presence of business park / Industry (% within 500m radius)	-0.40
	Proximity to natural areas (m)	0.49
	Proximity to accessible natural area (m)	1.04
	Presence of small roads (% within 500m radius)	1.03
	Proximity to recreation facilities (m)	0.53
Plant habitat *	Winter groundwater level (cm below surface)	-0.12
	Sandy soil (no= 0 yes=1)	0.23
	Sandy clay soil (no= 0 yes=1)	0.13
	Peat soil (no= 0 yes=1)	0.14
	Proximity to open natural area (m)	0.56
	Proximity to forested natural area (m)	0.26
Arable production *	Summer groundwater level (cm below surface)	-0.10
	Winter groundwater level (cm below surface)	0.12
	Sandy soil (no= 0 yes=1)	0.15
	Sandy clay soil (no= 0 yes=1)	0.09
	Peaty sand soil (no= 0 yes=1)	0.08
	Average farm size (ha)	0.40
	Number of neighbouring farms (farms per km^2)	0.69
Leisure cycling	Distance to residential areas < 5km	-
	Absence of highways	-
	Presence of small roads	-
	Absence of business parks/ Industry	-
	Potential number of visitors	1

* Landscape functions of which the weight indicates the standardised beta coefficients.

[1] See Weitkamp et al (2007) for a full description of the indicator calculations

Quantifying and mapping multifunctionality

The multifunctionality map based on the number of overlapping functions shows a strong spatial variation in number of landscape functions in our study area (Figure 3.3a). The seven landscape functions make a total of 76 different function combinations (including both mono and multifunctionality) in our study area. The most abundant landscape function combinations, in terms of percentages of the total study area, are *leisure cycling & arable production* (8.1%), *arable production & cultural heritage* (6.9%) and *leisure cycling & tourism & residential* (6.4%).

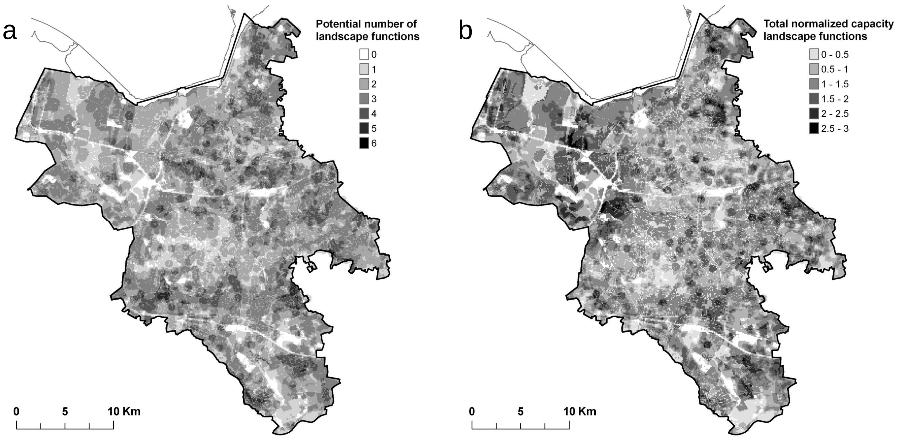

Figure 3.3 Multifunctionality in the study area based on a) assessed number of landscape functions, b) summed normalised capacity of the seven landscape functions.

Figure 3.3b shows the summed capacity to provide services of all landscape functions, which varies in our study area from 0 to 3, out the maximal attainable capacity score of 7. Visual observation reveals a clear difference in pattern between the amount of landscape functions present (Figure 3.3a) and the summed function capacities (Figure 3.3b). The landscape function combination *leisure cycling & arable production & intensive livestock & cultural heritage* (covering 2.0% of the total area) showed with 1.52 the highest average summed capacity, followed by *leisure cycling & arable production & tourism & intensive livestock* (2.6% of the total area), with a capacity of 1.46, and *leisure cycling & arable production & tourism & intensive livestock & cultural heritage* (0.4% of the total area), which had a summed capacity of 1.45. Meaning that within our study area at these multifunctional locations the highest total capacity to provide services is present.

Quantifying landscape function interactions

Landscape indicator analysis

Landscape characteristics are identified that positively or negatively influence multiple landscape functions and therefore multifunctionality (Table 3.2). Four landscape indicators (proximity to natural areas, population, residential area and farm size) are directly related to landscape functions (plant habitat, residential, insensitive livestock and arable production functions, respectively) which provide favourable landscape characteristics for another landscape function. So at these locations a landscape function synergy is supported. For example, the plant habitat function generates natural areas, which are enhancing the tourism function (see Table 3.1). The opposite applies for the landscape indicator representing homogeneous agricultural land-use and large field sizes as in this case a landscape function generates unfavourable conditions for another landscape function. These indicators are related to large scale arable production but this characteristic will decrease tourism suitability (see Table 3.1) and therefore lower the multifunctional potential. All areas containing a landscape characteristic that has a similar effect on multiple landscape functions (all are either positively or negatively effected), support compatibility and therefore multifunctionality. For instance, areas far away from highways positively affect both the tourism and cycling leisure functions, therefore these locations could be suitable location for both landscape functions. Landscape indicators from Table 3.1 that are not listed in Table 3.2 indicate unique landscape function requirements and are therefore assumed to not directly influence landscape function interactions and multifunctionality.

Table 3.2 Landscape indicators affecting landscape functions together with their possible landscape function interactions and effect on multifunctionality. (Table continues on the next page)

Landscape indicator	Positive effect	Negative effect	Interaction	Multifunctionality effect
Proximity to natural area	Plant habitat, tourism		Synergy	Positive
Residential area	Leisure cycling, residential		Synergy	Positive
Population	Residential, leisure cycling		Synergy	Positive
Farm size	Arable production, intensive livestock		Synergy	Positive
Field size/homogenous agricultural land-use	Arable production	Tourism	Conflict	Negative
Presence sandy & sandy clay soils	Arable production plant habitat		Compatible	Positive
Amount of small roads	Tourism, leisure cycling		Compatible	Positive

Distance to highway		Tourism, Leisure cycling	Compatible	Negative
Presence of Business park/Industry		Tourism, Leisure cycling	Compatible	Negative
Winter groundwater level	Arable production	Plant habitat		Negative
Summer groundwater level	Plant habitat	Arable production		Negative

(Table 3.2 continued)

Landscape function correlations

All significant Spearman correlations coefficients between landscape function pairs are given in Table 3.3. Between many pairs of landscape functions a significant correlation is found, however the small coefficients indicate that most of these correlations are not very strong. The strongest correlation is found between *cultural heritage & tourism* (-0.54), meaning that at locations with a high cultural heritage capacity a low tourism suitability is found (which can also be visually interpreted from Figure 3.2). This negative correlation is a result of the predominantly open grasslands far away from forested areas that characterise the highly valued cultural landscapes in our study area. These landscape characteristics do not coincide with the landscape requirements for the tourism function (Table 3.1). Table 3.3 shows many negative correlations which are the result of the absence (i.e. zero values) of other landscape functions (5-18 % of landscape function observations are located in mono-functional areas). For example, the negative correlation between valuable cultural landscapes and leisure cycling is related to absence of leisure cycling function in these cultural heritage areas. In our study area the cultural heritage landscapes are in general too far away from urban centres to have leisure cycling. The intensive livestock function, one of the known conflicting landscape functions of our case study area, shows (surprisingly) positive correlations with areas plant habitat and tourism. So, large intensive livestock farms are located in areas with high capacities for tourism and for plant habitat. This example indicates that the direction of the correlation coefficients can not in all cases be interpreted as a current function interaction.

That two landscape functions do not have mirrored correlations is explicitly shown from the function pairs *residential & tourism*, and *leisure cycling & tourism*, as these landscape function pairs show opposite correlations. Locations with tourism show a positive correlation with leisure cycling capacity, while areas with leisure cycling show a negative relation with the landscape's tourism capacity. So, tourism areas contain very suitable leisure cycling areas, but cycling areas are often not suitable for overnight tourism. The leisure cycling function has fewer landscape requirements compared to the tourism function.

Table 3.3 Spearman correlation coefficients between landscape functions (p< 0.01), between brackets the number of observations in the sample set. 'NA' indicates complete absence of a landscape function pair, '-' indicates not significant correlations.

	Landscape functions including absence						
Landscape function presence (number of observations)	**Cultural heritage**	**Intensive livestock**	**Residential**	**Tourism**	**Plant habitat**	**Arable prod.**	**Leisure cycling**
Cultural heritage (2588)		-0.32	-0.07	-0.54	-0.15	-0.10	-0.14
Intensive livestock (3196)	-0.08		-0.09	0.11	0.06	-	-0.14
Residential (1123)	-	-0.18		-0.31	-	NA	-
Tourism (3387)	-0.13	-0.06	0.12		-0.09	-0.27	0.07
Plant habitat (538)	-	-	NA	-0.22		-0.29	-0.22
Arable production (5665)	-0.19	0.14	NA	-	-0.17		-0.04
Leisure cycling (5221)	-0.08	-0.19	0.14	-0.07	-0.16	-0.15	

Multifunctionality effect analysis

After leaving out all rare landscape function combinations, 41 combinations were included in the multifunctionality effects analysis. The ANOVA results ($p < 0.01$) indicate that all seven landscape functions show a difference in capacity among landscape function combinations. So for each landscape function in our study area, the provision of services differs when other sets of landscape functions are present. Based on the t-tests results we list combinations of landscape functions that significantly deviate from the overall capacity mean of a landscape function. To simplify the presentation of the results, we only show the results for function combinations covering more than 1% of the total area in Table 3.4. As an illustration we plotted the results for the plant habitat function in Figure 3.4. The plant habitat function has a significant lower capacity when it is located in multifunctional areas with *cultural heritage & leisure cycling*, *leisure cycling & tourism*, *arable production* and *arable production & cultural heritage*. These function combinations, however, cover less than 1% of the total study area and are therefore not presented in Table 3.4. Locations only containing the plant habitat function (combination ID 29, this 'combination' includes one landscape function), clearly show a higher plant habitat capacity than all other function combinations. So, at mono-functional locations the best conditions are present to find important plant species. Actually, although not presented in the simplified Table 3.4, for almost all landscape functions (except for the intensive livestock function) mono-functionality leads to a significantly higher capacity to provide a service.

Table 3.4 Landscape function combinations with a positive and negative deviation of the mean function capacity (Welch t-test, $p < 0.01$). Only function combinations covering more than 1% of the total area are displayed.

Function (mean)	**Positive deviation**	**Mean**	**Negative deviation**	**Mean**
Cult. heritage (0.50)	▪ Cult. heritage & arable prod. ▪ Cult. heritage ▪ Cult. heritage & arable prod. & leisure cycling	0.69 0.67 0.54	▪ Cult. heritage & intensive livestock & arable prod. ▪ Cult. heritage & leisure cycling & intensive livestock & arable prod.	0.34 0.30
Int. livestock (0.30)	▪ Intensive livestock & leisure cycling & Tourism & arable prod. ▪ Int. livestock & tourism & arable prod. ▪ Int. livestock & arable prod.	0.37 0.36 0.33	▪ Int. livestock & leisure cycling & arable prod. ▪ Int. livestock & cult. heritage & arable prod.	0.26 0.23
Residential (0.34)	▪ Residential & Tourism	0.48	▪ Residential & leisure cycling & tourism	0.29
Tourism (0.45)	▪ Tourism & leisure cycling & residential ▪ Tourism ▪ Tourism & int. livestock ▪ Tourism & leisure cycling	0.57 0.55 0.51 0.51	▪ Tourism & arable prod. & int. livestock ▪ Tourism & arable prod. ▪ Tourism & leisure cycling & arable prod. & int. livestock ▪ Tourism & residential ▪ Tourism & leisure cycling & arable prod.	0.41 0.38 0.36 0.34 0.32
Plant habitat (0.22)	▪ Plant habitat	0.39		
Arable prod. (0.56)	▪ Arable prod. & int. livestock ▪ Arable prod. & int. livestock & leisure cycling ▪ Arable prod.	0.68 0.66 0.59	▪ Leisure cycling & cult. heritage & Arable prod. ▪ Arable prod. & leisure cycling & tourism ▪ Arable prod. & tourism ▪ Arable prod. & cult. heritage	0.50 0.49 0.45 0.45
Leisure cycling (0.39)	▪ Leisure cycling & residential ▪ Leisure cycling ▪ Leisure cycling & tourism & residential ▪ Leisure cycling & tourism ▪ Leisure cycling & arable prod.	0.58 0.52 0.47 0.45 0.45	▪ Leisure cycling & cult. heritage & arable prod. ▪ Leisure cycling & int. livestock & arable prod. ▪ Leisure cycling & tourism & arable prod. ▪ Leisure cycling & tourism & Int. livestock ▪ Leisure cycling & tourism & arable prod. & int. livestock	0.34 0.34 0.32 0.29 0.19

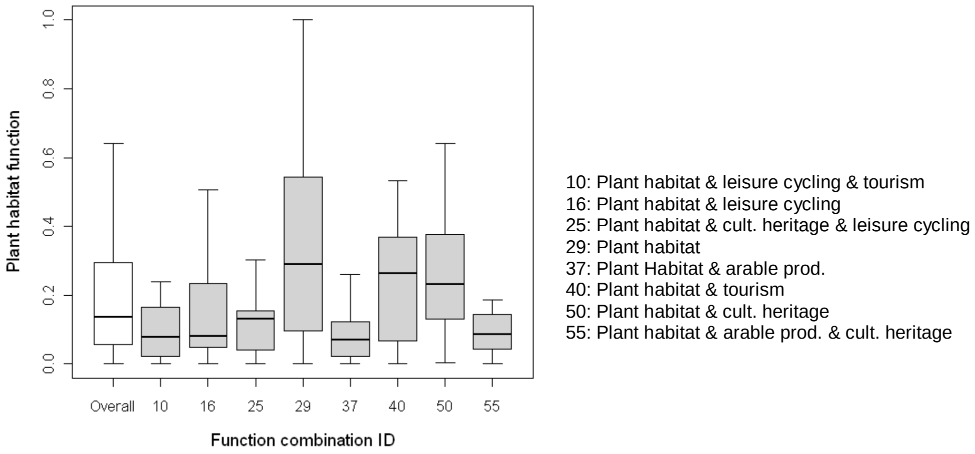

Figure 3.4 Standardised plant habitat function per landscape function combination.

To identify multifunctionality hot-spots, we selected from the landscape function combinations covering 97.5% of the total area those combinations that showed a positive deviance from the mean for two or more landscape functions. All these combinations of landscape functions, together with their assumed interactions, are listed in Table 3.5. For example, *leisure cycling* and *tourism* are affected positively by the presence of each other. Therefore we can conclude that at these locations a synergy between two landscape functions is taking place, resulting in a multifunctional hot-spot. This result differs from the correlation analyses in which we find a negative correlation between leisure cycling and tourism, as we here limit our analyses to locations at which both landscape functions are present. The landscape function combination *leisure cycling & tourism & residential* shows significantly higher capacities for both *tourism* and *leisure cycling* but a significantly lower residential function capacity. However, as two landscape functions show a function synergy all locations with this function combination are labelled as ‘multifunctional hot-spot’.

Table 3.5 Multifunctional hot-spots and cold-spots landscape and the assumed interactions between these landscape function combinations.

Landscape function combinations	**Interaction**
Hot-spots	
▪ Arable prod. & intensive livestock	Synergy
▪ Leisure cycling & tourism	Synergy
▪ Leisure cycling & tourism & residential	Synergy & conflict
Cold-spots	
▪ Leisure cycling & tourism & residential & intensive livestock	Synergy & conflict
▪ Leisure cycling & tourism & arable prod. & intensive livestock	Synergy & conflict
▪ Leisure cycling & cultural heritage & arable prod.	Synergy & conflict
▪ Leisure cycling & intensive livestock & arable prod.	Synergy & conflict
▪ Plant habitat & cultural heritage & arable prod.	Conflict
▪ Leisure cycling & plant habitat & cultural heritage	Conflict
▪ Leisure cycling & cultural heritage & arable prod. & tourism	Conflict
▪ Cultural heritage & intensive livestock & arable prod.	Conflict
▪ Cultural heritage & leisure cycling & intensive livestock & arable prod. & tourism	Conflict
▪ Tourism & intensive livestock & cultural heritage & arable prod.	Conflict
▪ Tourism & arable prod.	Conflict
▪ Cultural heritage & leisure cycling & intensive livestock	Conflict
▪ Leisure cycling & tourism & arable prod.	Conflict
▪ Tourism & cultural heritage & arable prod.	Conflict

The multifunctionality hot-spots of our study area are presented in Figure 3.5a. Figure 3.5b shows all locations at which multifunctionality leads to a lower capacity of at least two landscape functions, i.e. multifunctional cold-spots.

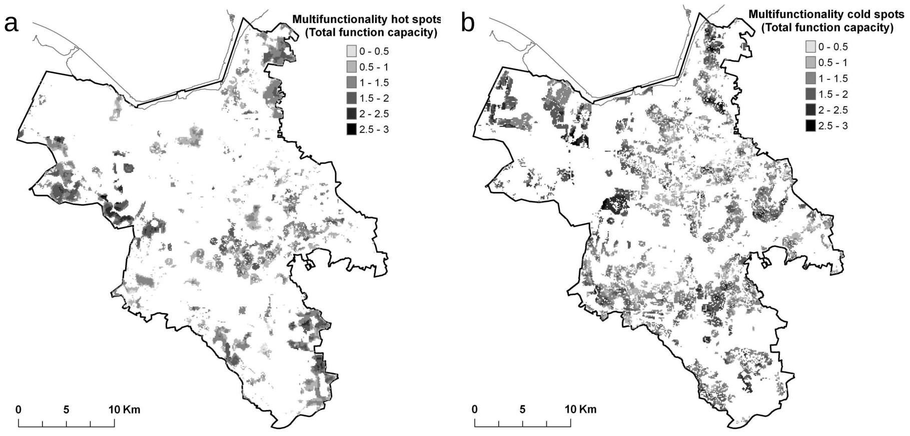

Figure 3.5 Multifunctionality a) hot-spots and b) cold-spots.

Discussion

Methodology for quantifying interactions

The three methods to quantify landscape interactions can be seen as complementary to describe different aspects of multifunctionality. First, to obtain detailed information on the landscape requirements of each landscape function, landscape indicators are analysed. Knowledge on what landscape characteristics enhance or hamper landscape functions can be used to identify locations with suitable conditions for multiple functions. Secondly, to understand landscape function dynamics, we calculated correlations between all landscape function capacities. This method can be used to foresee trends resulting from landscape function dynamics; the direction of the correlation coefficient indicates the expected effect of an increase or decrease of a landscape function on another landscape function. Thirdly, the effect of multifunctionality on the capacity of individual functions is calculated from which multifunctional hot and cold-spots are identified. This improved understanding of landscape function interactions could be used to design and evaluate spatial policies aiming at enhancing multifunctionality or one specific landscape function within a multifunctional area. Stimulation of multifunctionality is likely to be successful at locations with favourable conditions for multiple functions in combination with synergising landscape functions. Spatial policies could therefore focus on changing landscape conditions or by promoting specific combinations of landscape functions. Knowledge on trends between function capacities can be used in policy questions regarding landscape function dynamics. The presented overall methodology is generic and could therefore be applied in other study areas or to study interactions with potential functions currently not provided in the area.

A number of methodological issues need to be mentioned in order to ensure correct interpretation of the results. First of all, we assume that the current capacity of landscape functions is properly reflecting functions interactions. However, we cannot prove causality in any of the three methods used to define function interactions. Especially in the correlation and t-test methods the underlying relations are mainly unclear and therefore difficult to evaluate for assumed causality. Our results indicate that the intensive livestock function is positively correlated with plant habitat and tourism, which is very likely not a result of causality but rather a description of the current situation where both potentially conflicting functions are present in the same area. The effect of this conflict could become apparent in due course. The temporal scale at which changes in landscape functions and their related interactions show its effect is highly variable, some interactions take place immediately whilst others could take years (Lindborg and Eriksson, 2004). Time series data are needed to study these possible temporal effects, however due to data restrictions we could not carry out a time lag effects study.

Another issue relates to the landscape function interactions analyses which are all based on linear methods, indicating overall trends. A landscape function interaction could however become effective or even change when a certain threshold or optimum point is reached (Daugstad et al., 2006; Hein, 2006; Groot et al., 2007). However, too little information on such thresholds for our landscape functions is available to develop non-linear models.

Additionally, in this landscape function study we are dealing with data containing potentially high levels of uncertainty. The landscape function maps, on which all our analyses are based, are generated using different methods, all subject to different levels (location and nature) of uncertainty (Walker et al., 2003). Internal validation exercises on landscape function mapping were carried out (Chapter 2) but uncertainty throughout the modelling exercise has not been quantified. Before making a next step to communicate results from our approach to planners and policy makers the different dimensions uncertainty need to be better determined and understood.

To aggregate the landscape functions maps to a map indicating multifunctionality, all landscape functions in this study are given an equal weight. By straightforwardly assigning equal weights to the different landscape function we did not include any preferences or level of importance. However, for policy purposes landscape functions could get different weights, related to policy objectives or economic values of landscape functions (Turner et al., 2003; Hein et al., 2006; Gimona and Van der Horst, 2007; Meyer and Grabaum, 2008). An economic valuation of landscape functions could then be used to study at which locations a multifunctional landscape provides the highest monetary benefits to society.

A last methodological issue relates to our assumption that landscape functions only interact at overlapping function locations. Our landscape functions are delineated based on thresholds related to the minimum capacity of the landscape function at interest for policy making. Areas outside these landscape function boundaries are excluded in our analyses. Also the effect of scale or the distance to landscape function in our interaction analyses is not analysed in this chapter. This scale effect could however play in important role describing multifunctional areas (Hein et al., 2006; De Groot and Hein, 2007).

Monofunctional versus multifunctional

Multifunctional areas have often been stated to provide in total more services to society than mono-functional locations (FAO, 1999; OECD, 2001) Our data show indeed an increase in total landscape functionality with an increase in the number of landscape functions. However, our data show also a clear negative trend between the average capacity of a landscape function and the number of landscape functions (Figure 3.6). So, with multifunctionality the total function capacity increases while the average capacity of a landscape function decreases. Looking at the average maximum capacity values in Figure

3.6, multifunctional areas seem to contain multiple low providing functions rather than one dominant function with some very small functions. This could either indicate that mainly at marginal areas multifunctionality emerges or that multifunctionality goes at the expense of single functions. The landscape function correlations presented in Table 3.3 and landscape function effects in Table 3.4 also showed that most landscape functions provide more services at mono-functional than at multifunctional locations.

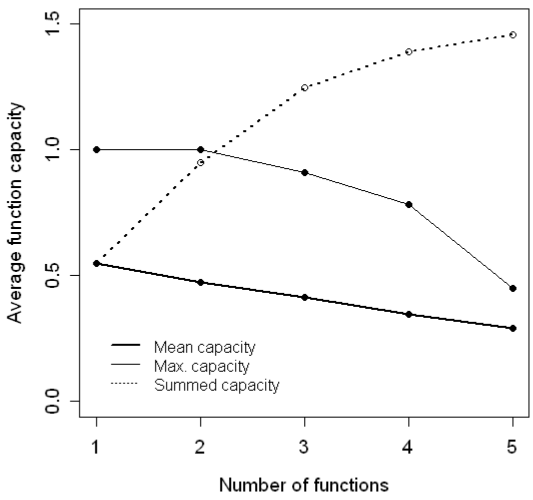

Figure 3.6 Relation between the number of landscape functions and the average mean, maximum and summed capacity.

Conclusions

Our study presents a further step in exploring the complex system of interacting landscape functions in relation to spatially heterogeneous multifunctional landscapes. In this chapter we describe different aspects of landscape function interactions using quantitative and spatial explicit information on landscape characteristics and multifunctionality. Landscape functions interact with each other in different ways, some landscape functions are affected negatively by the presence of other functions (e.g. plant habitat function) while some other landscape functions seem to benefit from multifunctionality (e.g. leisure cycling and tourism functions).

Our research also supports the need for regional approaches rather than a sectoral focus in order to study benefits of the landscape for society. We show a trend that at multifunctional locations the total provided services by the landscape is higher than at mono-functional sites. On the other hand, single landscape functions at multifunctional locations seem to provide fewer services than at mono-functional locations, only at multifunctional hot-spots multifunctionality does not lead to a decrease of the expected capacity to provide landscape services. The presented quantitative and spatially explicit approach highlights interactions between landscape functions which hamper or stimulate the landscape to provide these multiple services.

Chapter 4

Evaluating the impact of spatial policy on future landscape services

In this chapter we analyse the potential impact of an integrated policy package for the Gelderse Vallei region in the Netherlands on seven landscape services (residential use, intensive livestock husbandry, drinking water supply, attractiveness for overnight tourism, habitat provision for rare, endemic and indicator plant species, arable agricultural production, and attractiveness for leisure cycling). The spatially explicit methodology focuses on the changes in landscape properties resulting from the implementation of these policies and its effects on the supply of landscape services and economic values of the landscape services. After the policy implementation the strongest increase in services supply is expected in rural areas while the strongest increase in value is expected to occur in (peri-) urban areas of the study area. Additionally, we conclude that the policy package leads to an increase in multifunctional areas. This study presents one of the first comprehensive methodologies to quantify and analyse spatial variation in economic value of landscape services in time, and therefore can contribute to well-informed management of landscapes.

Based on: L. Willemen, L. Hein, P.H. Verburg
Submitted

Introduction

Besides producing agricultural commodities, rural regions provide a multitude of services that benefit people. Based on the definitions of 'ecosystem functions' (De Groot et al., 2002; MA, 2003), we use the term 'landscape function' to indicate the capacity of a landscape to provide services to society. These services include, amongst others, benefits such as food and timber production, fresh water supply, and recreational opportunities. The potential to provide such services depends on the spatial configuration and components of the landscape (Wiggering et al., 2006; Syrbe et al., 2007; Egoh et al., 2008). By changing landscape properties, human activities can directly or indirectly affect the supply of landscape services (Bastian et al., 2006; Nelson, 2006). Spatial policies are designed to influence the landscape in such a way that the provision of one or more landscape services is improved (Daily and Matson, 2008). For example, creating buffer zones around natural areas may improve wildlife habitats, land consolidation of arable land will enhance agricultural production, and creating access to natural areas can boost recreational activities. However, such changes in the landscape may affect each landscape function in a different manner leading to trade-off between different functions (Chan et al., 2006; Nelson et al., 2009). Such trade-offs challenge the design and implementation of regional spatial policies.

An ex-ante evaluation of the consequences of spatial planning and policy on the supply of landscape services can support decision making (Bockstael et al., 1995; Verburg et al., 2009). Landscapes are spatially diverse leading to unequal distribution of landscape services over an area. An evaluation of policy effects should therefore be spatially explicit as policies are likely to have a location-specific effect on the provision of landscape services. Additionally, changes in service supply need to be quantified to support decisions regarding possible trade-offs between landscape functions. To be able to compare service provision across different landscape functions, service supply should be standardised to the same units of value. Increasingly, economic valuation techniques are used to quantify landscape functions and their value for society (MA, 2005; Fisher et al., 2008; Schaeffer, 2008; Carpenter et al., 2009). Different approaches have been developed to value landscape services in monetary units (see for an overview e.g. MA, 2003; Turner et al., 2003; Zandersen and Tol, 2009). Monetary valuation can, for example be of use for analysing trade-offs in landscape service supply (Turner et al., 2003; Naidoo and Ricketts, 2006). Several studies have included economic values in spatial policy evaluations (Bateman et al., 2005; Troy and Wilson, 2006; Grêt-Regamey et al., 2008). Alternatively, landscape functions can be valued in non-monetary measures representing standardised service supply (E.g. in Chapter 3 or in Gimona and Van der Horst, 2007; Nelson et al., 2009). Using these standardised measures, each landscape function can be evaluated by quantifying the relative change in service provisioning, without explicitly considering the economic value of a landscape function. To date, studies tend to focus either on a detailed spatial description of service supply or on a valuation of landscape services without explicitly taking into account the spatial variation in service supply. To our knowledge, no earlier study has presented a spatially explicit economic valuation based on the quantified service supply for a broad range of landscape functions.

The objective of this chapter is thus to analyse the change in landscape service supply and value under influence of policy measures in the rural Gelderse Vallei region of the Netherlands. In this chapter we analyse the impact of implementation of policy plans on seven landscape functions: residential use, intensive livestock husbandry, drinking water supply, attractiveness for overnight tourism, habitat provision for rare, endemic and indicator plant species, arable agricultural production, and attractiveness for leisure cycling. We quantitatively and spatially explore the changes in service supply using two measures; (i) a unit-less index related to the level of service provision and (ii) an estimation of the value of these services in monetary terms. We present a methodology that addresses three questions; (i) How will landscape properties change after policy implementation?; (ii) How will changed landscape properties affect the supply of landscape services? and (iii) How will a change in service supply translate into a change in economic value of the study area? We will show that the quantification, valuation and mapping of landscape services can support management and planning activities of multifunctional landscapes.

Data and methods

Study area

In the Gelderse Vallei study area, tension exists between different land-uses as a result of simultaneous claims for space. In 2002, the Dutch national government introduced a new law to tackle competing land demands in the Dutch rural areas in an integrated manner: the Reconstruction Act. The integrated spatial policy plans evaluated in this chapter are based on the Reconstruction Act of which the implementation for the study area is defined by the provincial administrations. The policy plans envision regional development through improving the conditions for supplying multiple landscape services to fulfil the increasing demands of society (Provinces of Gelderland and Utrecht, 2005). Living conditions and economic development should benefit from the new spatial planning. The first policy measures based on the Reconstruction Act were implemented in 2005, and in 2015 all measures are foreseen to be put in place.

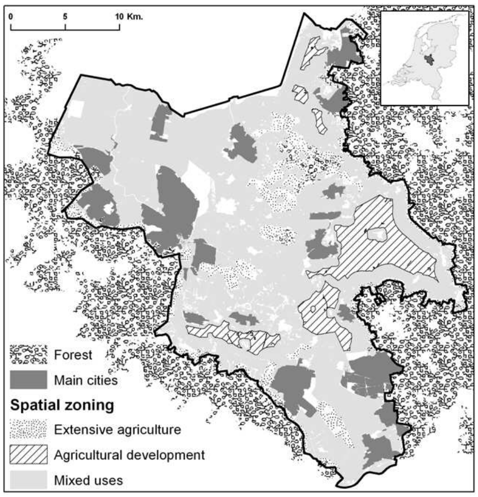

Figure 4.1 Location of the study area within the Netherlands and the intensive livestock regulation zones as defined by policy.

The implementation and regulation of the Reconstruction Act in the study region is strongly based on spatial zones to separate conflicting landscape functions. As especially intensive livestock husbandry causes conflicts with other functions in the study area, three zones regulating the degree of development in intensive livestock sector have been defined; (i) Agricultural development zones, in which priority is given to growth and establishment of intensive livestock farms; (ii) Extensive agriculture zones, in which priority is given to nature development, residential use and recreation. In this zone the growth of intensive livestock farms is strongly regulated, but large profitable farms are eligible for a full financial compensation from the Dutch government to be reallocated to agricultural development zones; and (iii) Mixed zones, where residential and recreational uses, nature and agriculture should develop side by side. All urban areas are excluded from intensive livestock development zones (Figure 4.1).

Methodological approaches

To analyse the change in landscape service supply and value under influence of a set of spatial policies, three methodological steps were taken. The first step describes the effect of changes in land management on landscape properties as a result the implemented policies. Landscape properties include biophysical properties (e.g. soil type or groundwater level) socioeconomic properties (e.g. land-use or population pressure), and spatial characteristics (e.g. clustering or proximities measures). Landscape properties determine the presence of landscape functions and the spatial variability of their supply of landscape services (Chapter

2 and Diaz et al., 2007; Egoh et al., 2008; Tallis and Polasky, 2009). In the second step landscape properties are used to quantify and map the level of service provision of the different landscape functions, before and after policy implementation. In the third step, service supply is valued in monetary units, accounting for spatial variation of landscape service supply. Different valuation approaches are used to estimate the economic value of market and non-market landscape functions. As the focus of this chapter is on demonstrating an interdisciplinary approach to evaluate spatial policies, simplified methods are used to estimate economic values of landscape services.

To quantify the changes in landscape service supply we compare the situation in 2000 to the expected situation in 2015, when the Reconstruction Act is planned to be completely implemented. In this study all spatial data sources are converted to a raster format with a spatial resolution of 100 by 100 meter. To facilitate the visual interpretation of the results, all landscape function maps are aggregated to 186 postal code zones and administrative neighbourhoods of the study area.

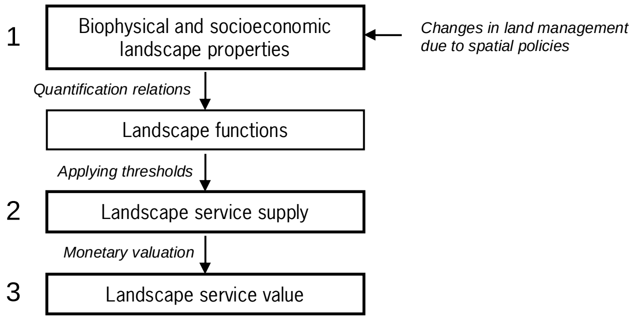

Figure 4.2 Three steps in the spatially explicit methodological approach; from landscape properties to landscape service value maps.

Assessment of landscape properties

The first step in our overall methodology is to assess the changes in landscape properties. In addition to the integrated policy plans a range of development trends are also evaluated for their spatial alterations of the landscape, both in socioeconomic and in biophysical terms. Including development trends is needed to adequately describe the future situation in which policy plans are put into effect. Therefore, we base the assessment of the changes in landscape properties on demographic and economic prognoses and trends, construction licenses, and environmental regulations. Translating spatial policy plans (e.g. construction sites for residential areas) to location-specific landscape properties can be done straightforwardly. On the other hand, (policy induced) development trends often lack detailed information on spatial variability, as these trends are mostly given for administrative units (e.g. population growth per province). Therefore, translating these trends into landscape properties requires that these trends are made spatially explicit. In this section we briefly describe the translation of non-spatially defined trends and spatial policies into future landscape properties for the study area.

Development trends

Demographic trends on population growth are for the study area available at municipality level (CBS, 2008a). We included an increased spatial variation in these trends by distributing the expected population per hectare over all residential areas within the municipality in 2015, using the size of the residential area as weight. This results in the estimated population distribution map for 2015. Additionally, agricultural development trends are used to assess the spatial variation of the change in intensive livestock farms. Since a number of decades, there has been a negative trend in number of farms in the Netherlands. The Dutch Farm Accountancy Data Network includes development trends regarding the number and size of farms in the study area (Provinces of Gelderland and Utrecht, 2005). The prognosis for 2015 is an overall decrease in the number of intensive livestock farms of approximately 40% in the study area. However, among farm size classes this percentage differs. Based on the farm size and growth restrictions as defined by the intensive livestock development zones, we estimated the location and size of livestock and arable farms by 2015 (see Appendix).

Spatial policies

A spatially defined landscape change, as described in the regional Reconstruction Act, is the planned conversion of 3800 hectares of agricultural land into natural areas to create ecological corridors (VROM, 2006). Additionally, 560 hectares of new residential area are envisioned in housing construction plans for the study area (Nirov, 2007). Changes in infrastructure include the construction of a highway in the central part of the study area. In most cases these planned land-cover conversions take place at locations that are currently used as agricultural land. Combining the land-cover map of 2000 with these foreseen landscape changes, a new land-cover map for the year 2015 is constructed for the study area. Additionally, to reduce the chance of water stress for natural vegetation the Reconstruction Act states that the amount of extracted groundwater needs to decrease. Therefore the overall drinking water extraction in 2015 is foreseen to decrease to 94% of the overall amount of water extracted in 2000. Drinking water companies foresee that this reduction goes alongside with a similar expected decrease in water losses due to leakages (VEWIN, 2005).

Quantification of landscape service supply

In the second step of the methodology, landscape service supply and its spatial distribution are assessed based on the spatially explicit landscape properties resulting from the first step (Figure 4.2). In Chapter 2 we quantified the relation between landscape functions and landscape properties. In this section we give a short description of the quantification method of service supply and mapping procedure for the years 2000 and 2015 per landscape function. Based on the situation in the reference year 2000 and the expected landscape properties in the year 2015, landscape function maps are created and subsequently compared. To allow for comparison between different landscape functions and years, all aggregated functions maps are normalised between 0 and 1 based on their minimum and maximum values for 2015.

Residential

Service supply of the residential function is quantified by means of the number of people living in a residential area. Based on the population statistics for the year 2000 and the growth prognoses for the year 2015, the population per residential area was estimated for both years. The population per hectare of the study area is mapped for the years 2000 and 2015.

Intensive livestock husbandry

The provided services of this function are mapped based on intensive livestock production, measured in the Dutch Standard Unit (DSU). The location and size of each farm in the study area is derived from farm census data for 2000. Combining the assessed livestock farm distribution map for the year 2015 with the agricultural growth data, results in a map representing the DSU per livestock farm in 2015. On average a livestock farm in the study area possesses four hectares of land surrounding the farm. Therefore the supplied services of each farm are distributed over the four surrounding hectares.

Drinking water

Service supply of drinking water function is described by the amount of water extracted by drinking water companies per year. In the study area, all drinking water is extracted from protected groundwater extraction zones. Annual extraction volumes are used to quantify the function. According to reports of the extraction companies these volumes varied from 3000 m^3 per hectare to 14 000 m^3 per hectare in 2000. This function is mapped based on the extraction volumes in cubic meter per extraction zone in 2000 and the expected volumes in 2015.

Tourism

The provided services for tourism are expressed as the suitability for overnight tourism. In Chapter 2 we described the location of tourism accommodations by ten landscape properties using a logistic regression model. These ten properties include land cover, level of disturbance, recreation possibilities and accessibility measures. The variables representing these landscape properties and their standardised beta coefficients are presented in Table 4.1. Using the logistic regression model the suitability for tourism in 2000 was estimated. Some of the landscape properties describing the tourism function are expected to change by 2015. Using the expected land-cover changes, related properties (e.g. percentage of agriculture land) are redefined for the year 2015. This new set of landscape properties is put into the regression model to estimate the tourism suitability for 2015. All locations with a probability of 0.5 or higher are mapped as suitable locations for tourism.

Plant habitat

Service supply of the plant habitat function is quantified by an 1 to 10 index referring to the suitability of a location to provide habitat to rare, endemic and indicator plant species (Hertog and Rijken, 1996; Rijken, 2000). For our study area these species mostly relate to vegetation growing under mesotrophic and wet conditions. A linear regression model describes the relation between observations on plant habitat suitability and groundwater level, soil type and natural land cover, see Table 4.1 and Chapter 2. Based on the variables representing the landscape properties and the regression model, the plant habitat function is estimated for the year 2000. Only locations with a good plant habitat suitability (index value larger than 5) are included in the plant habitat map. By changing the independent variables of regression model according to the expected land-cover changes the plant habitat function is estimated and mapped for 2015.

Arable production

Service supply of the arable production function is mapped based on the production of the only common arable crop in the study region, fodder maize. By a linear regression on a set of selected maize plots, landscape properties explaining the yield per hectare were identified in Chapter 2, see also Table 4.1. Using this regression model the arable production for 2000 is assessed. Only the locations with an estimated yield higher than 35 ton per hectare are included in the arable production map. Two of the seven landscape properties included in the model, the average farm size and farm density, are expected to change by 2015. The new set of independent variables and the regression model are used to assess the yield of maize per hectare for 2015.

Table 4.1 Landscape properties and their quantitative relation with service supply (based on Chapter 2). The landscape properties that are assumed stable over time are presented in *Italic*.

Landscape function	Landscape properties	Quantified relation
Residential	Residential area	
	Number of residents per ha	1
Intensive livestock	Livestock farm location	
	Livestock production and farm size (Economic Units)	1
Drinking water	Extraction zones	
	Water extraction (m^3)	1
Tourism *	Homogeneous agricultural area (% within 500m radius)	-1.13
	Homogeneous natural area (% within 500m radius)	-3.08
	Clustered natural areas (% within 5 km radius)	0.86
	Openness of landscape (line of sight in m)	-0.84
	Distance to highway (m)	0.36
	Presence of business park / Industry (% within 500m radius)	-0.40
	Proximity to natural areas (m)	0.49
	Proximity to accessible natural area (m)	1.04
	Presence of small roads (% within 500m radius)	1.03
	Proximity to recreation facilities (m)	0.53
Plant habitat *	*Winter groundwater level (cm below surface)*	-0.12
	Sandy soil (no= 0 yes=1)	0.23
	Sandy clay soil (no= 0 yes=1)	0.13
	Peat soil (no= 0 yes=1)	0.14
	Proximity to open natural area (m)	0.56
	Proximity to forested natural area (m)	0.26
Arable production *	*Summer groundwater level (cm below surface)*	-0.10
	Winter groundwater level (cm below surface)	0.12
	Sandy soil (no= 0 yes=1)	0.15
	Sandy clay soil (no= 0 yes=1)	0.09
	Peaty sand soil (no= 0 yes=1)	0.08
	Average farm size (ha)	0.40
	Number of neighbouring farms (farms per km^2)	0.69
Leisure cycling	Distance to residential areas <5 km	
	Absence of highways	
	Presence of small roads	
	Absence of business parks/ Industry	
	Potential number of visitors	1

* Relations are quantified based on a regression analysis

Leisure cycling

The service supply for leisure cycling is defined by the potential number of people visiting the location for cycling recreation. Of the Dutch population approximately 70 % participates in cycling recreation, making on average four trips in 2000 and an expected three trips per year in 2015 (CBS, 2008a). An average cycling tour starts at home and covers approximately 15 km in a 5 km radius, following small local roads through areas with little disturbance from highways and industry (see Chapter 2). Based on the suitability of the landscape and the potential leisure population, the leisure cycling function is quantified and mapped for 2000. To map leisure cycling in 2015, estimations of the change in the potential number of people recreating per year and infrastructure changes are included.

Valuation of landscape services

The economic valuation of landscape services for the years 2000 and 2015 is based on a selection of economic indicators that are linked to the landscape function maps. All landscape functions are valued in euro (€) per hectare per year. For each landscape service a value is estimated for both evaluation years based on (i) the amount of service supply per hectare and (ii) the price levels. Our valuation approach does not include market models to estimate supply and demand curves. Instead we use an approach based on market prices. For services with a market value our valuation is based on establishing the net value generated for each service, i.e. the gross value (price times quantify) minus the costs of producing the services (including cost of all inputs and depreciation of capital goods). Labour costs are however included in the net value. We are aware that these prices only reflect part of the value of a provided landscape service. Therefore the values assigned to landscape services can only be seen as an illustrative measure. For the year 2000, the prices are derived from different sources, including national statistics and company financial reports. For the year 2015, price levels are not available or price forecasts contain a high uncertainty. Therefore, we decided to base the 2015 price levels on the prices that occurred in the year 2007, i.e. the latest year for which the relevant national statistics are available. Hence, it is assumed that all prices in 2015 equal the prices in 2007. We are aware that this creates a bias, but further estimation of price developments is beyond the scope of this chapter. Landscape services of which a significant part of the values are not reflected in market transactions (e.g. recreation and tourism) are additionally valued by simplified monetary measures indicating the consumer surplus based on literature. All prices and consumer values are expressed in 2007 euros. Hence, the prices that were recorded in 2000 are converted to 2007 euros, using a 2.2% annual inflation rate, the average inflation rate in the period 2000 to 2007 (CBS, 2008a).

Residential use

The residential service is valued based on the price of land under residential use. It is assumed that this value can be assessed based on the average house price and the number of houses per hectare of residential area. The house price is estimated based on the average real estate tax values per municipality of 2007, and the growth in house prices between 2000 and 2007 (CBS, 2008a). For the study area the average value per house increased from

€211 500 in 2000 (€182 300 in 2000 euros) to €259 000 in 2007. The prices of 2007 are used to assess the house prices of 2015. To translate these house prices into a value per year, the by the Dutch government proposed maximum annual rent of 5.4% of the house value is taken (VROM, 2005). Because this study aims to estimate the value of land for residential purposes, the construction costs of the house are subtracted from the annual value. The construction costs normally are about 70 percent of the real estate value (Bouwfonds, 2006). For 2000 the land value per house is multiplied by the number of houses per residential area. For 2015 housing development plans are included to define the residential areas and number of houses (Nirov, 2007). The value of the residential service is mapped per hectare under residential use for both years.

Intensive livestock husbandry

The service supply of the intensive livestock function is valued based on the net value added (NVA) generated by a livestock farm. The Dutch farm size index DSU is used as starting point in these calculations. The DSU is included in Dutch farm census data and indicates agricultural production units expressed in a monetary term. The monetary term is obtained by taking gross farm revenues minus the costs for variable production inputs such as raw materials, fertilisers and pesticides (one DSU refers to €1390 in 2000 and is, based on 2007 data, estimated to be €1400 in 2015 (LEI, 2008c)). To come to the net value added (NVA) of an intensive livestock farm, all depreciation and interest costs are subtracted from the DSU. Since the NVA of intensive livestock farms strongly fluctuates per year, the average annual NVA between 2001 and 2006 is used (LEI, 2008b). Based on the above, the NVA is estimated at 29% of the gross revenues for 2000 and 2015. For each (predicted) farm size and location the NVA is calculated and mapped for the year 2000 and 2015.

Drinking water

The value of the service supply of the drinking water function is estimated using the NVA of the drinking water companies and expressed per cubic meter water. Hence, the water extraction volumes for 2000 and 2015 are linked to the consumer prices of drinking water per cubic meter. These prices were €1.40 in 2000 and are estimated at €1.35 in 2015, based on 2007 prices (Hydron, 2004; Vitens, 2008). This decrease in real value of water relates to a lower price increase as compared to the inflation rate. The NVA as a percentage of the gross revenues is 20% for 2000 and is expected to be 23% in 2015 (Hydron, 2004; Vitens, 2008). Per drinking water extraction zone, the (expected) NVA generated by water extraction per hectare is mapped for the year 2000 and 2015.

Tourism

The total value of the service supply of the tourism function is estimated by accounting for both the NVA generated by the tourism sector (producers) and the net benefits accrued to tourists (consumers). To estimate the value of the tourism service related to the tourism industry, the NVA of the economic turnover of the tourism sector is calculated for 2000 and estimated for 2015 based on 2007 values. In 2000 a total 4.5 million overnight stays in tourist accommodations (camping, holiday homes, group accommodations) were registered for our study area. On average, a tourist spent €23 per day during a stay (CBS, 2008a; NBTC, 2008). In 2015 a slight increase (to 4.6 million) in overnight stays and a small decrease in expenses (to €22.50) is expected (CBS, 2008a; NBTC, 2008). The NVA of a tourism accommodation equals on average 32% of the total turnover in 2000 and is expected to be 34% in 2015 (CBS, 2008b). To estimate the value for consumers of the tourism service, we consider a range of studies to assess the consumers' surplus accruing to visitors of natural areas in The Netherlands. In particular, we considered the results of a travel cost method for wetlands (Hein et al., 2006) and a national park (Van der Heide, 2005) and a contingent value method for a lake area (Van der Veeren, 2002). Based on the results of these studies we assume that visitors accrue a surplus of €2 per tourist per day in both evaluation years. For both evaluation years the sum of the two values is distributed over the area suitable for tourism (i.e. the area falling within a 5 km circle from a tourism accommodation) and weighed by the number of overnight stays per accommodation site.

Plant habitat

The service supply plant habitat function has no direct market value and is therefore valued based on an assessment of the purchase and management costs related to conserving plant habitat in the Netherlands. This valuation approach differs from the approaches used for other landscape functions in this chapter, as this valuation is based on costs instead of the net benefits (i.e. benefits minus costs). It is assumed that these costs indicate a minimum willingness to pay of the Dutch society for one hectare of natural land. In 2000 on average €38 250 per hectare (€33 000 in 2000 euros) was spent to purchase land for nature development (M&NC, 2005). In 2015 this value is estimated at €36 000 per ha, based on 2007 data. To convert these figures to annual values, the maximum leasehold price for agricultural land is used, which set at 2% of the total value of the land (LEI, 2008a). To the land values the costs for nature development and maintenance are added. The Dutch government has defined a subsidies scheme per nature type to stimulate nature development in rural areas (LNV, 2008). These subsidy values vary between €26 and €1300 per hectare natural land per year with a median at €130. The subsidy scheme is linked to the map representing an inventory of the nature types in the study area (LNV, 2003). These nature types include, amongst others, nutrient poor grasslands, peat lands, heather and managed forests. Using the delineation of the plant habitat function maps for 2000 and 2015, the costs of nature conservation are calculated and mapped for each nature type.

Arable production

The service supply of the arable production function is valued by linking the NVA of the crop market prices to the yield (ton per hectare) estimations of the function maps of 2000 and 2015. For the year 2000 the average market price for one ton of fodder maize was €24.35 with a NVA of 30.5%. In 2015 the prices are estimated to be €27.60 with a NVA of 28.8 %, based on 2007 values (LEI, 2008a).

Leisure cycling

The economic value of the service supply of the leisure cycling function consists of a value accruing to the recreational sector offering various services to cyclers as well as a value to the cyclers themselves. To estimate the value of the leisure cycling service related to the recreation sector, the potential leisure population is linked to the NVA of the turnovers of recreational expenses. In 2000 at national level these expenses were approximately €4.65 per person per biking trip (mainly on drink/food consumptions) and in 2015 on average €6.18 is expected to be spent, based on 2007 statistics (CBS, 2008a). The NVA of total turnover of cafes and restaurants in the Netherlands in 2000 was 34% and estimated 36% in 2015 (CBS, 2008a). The total value for the recreation sector (indicating the producer surplus) is therefore assumed to be the number of people times the expenditure per person times the NVA (as percentage). The surplus accrued to visitors by leisure cycling is assumed to be €2 per person per day based on earlier recreation studies in a Dutch context (Van der Veeren, 2002; Van der Heide, 2005; Hein et al., 2006). The sum of the NVA and WTP of the leisure cycling function is distributed over the area suitable for leisure cycling and weighed by the potential number of people recreating at that location.

Results

Assessment of landscape properties

Based on policy plans and development trends we mapped the expected changes in land cover, drinking water extraction, intensive livestock and arable farms, and population for the year 2015. From these maps we subsequently derived the landscape properties to map and quantify landscape functions and their service supply.

Quantification of landscape service supply

For the years 2000 and 2015 the provided services of the seven main landscape functions of our study region were mapped for each postal code area. In Figure 4.3 we present the landscape function maps for the year 2000 and the change in service supply between the years 2000 and 2015. Only postal code areas with a change in service supply of more than five percent of the average postal code service supply in 2000 are mapped as change. Smaller differences in change in service supply are indicated as ‘no change’.

Several observations from the change maps as presented in Figure 4.3 can be made. First, all landscape functions, except for the drinking water extraction and arable production function, show an overall increase in service supply by 2015, with the largest gains for the plant habitat and tourism functions. Nevertheless, within the study area different patterns of change can be observed. Second, the foreseen effect of policy has a wide spread impact on some functions (plant habitat, tourism, arable production), while other functions are more locally affected (residential, cycling leisure). The planned policies to stimulate nature development (i.e., implementation of ecological corridors), clearly increases in spatial extent and quantity of the plant habitat function by the year 2015. Some postal code areas are foreseen to lose small patches of nature, which leads to a decrease in plant habitat. The tourism function is for a large part defined by the presence of natural areas and is therefore also expected to increase strongly in most areas. Third, around the residential centres, the leisure cycling function is foreseen to increase in service supply. This increase is not a result of an improved spatial configuration but is caused by an increase of population in nearby residential areas i.e., more people that can reach the location for cycling leisure. Fourth, the largest reduction in service supply is seen in the arable production function, as agricultural land is being converted to other land-uses. This land-use conversion however did enhance the service supply of other functions (plant habitat, tourism, residential use). Other trade-offs can be observed near cities, where urban expansion is at the cost of the cycling leisure function. However these trade-offs are also spatially variable, as can be observed from Figure 4.3. For example, an increase of residential use leads not at all locations to a decrease in cycling leisure.

The seven normalised function maps are summed to visualise the total service supply in 2000 and the change by 2015 (Figure 4.3). As all landscape function maps are normalised to a 0 to1 scale, the maximal achievable index would be 7. However the service provision in the study area varies between 0 to 2.5. In 2000 the strongest providing multifunctional areas are located the east of the study area. In this part no changes in service supply are foreseen by 2015, while the overall picture of the study area shows an increase in overall service provision. A few postal code zones show a reduction in service supply. These areas are mainly the locations where agricultural production (arable and livestock) is foreseen to decrease.

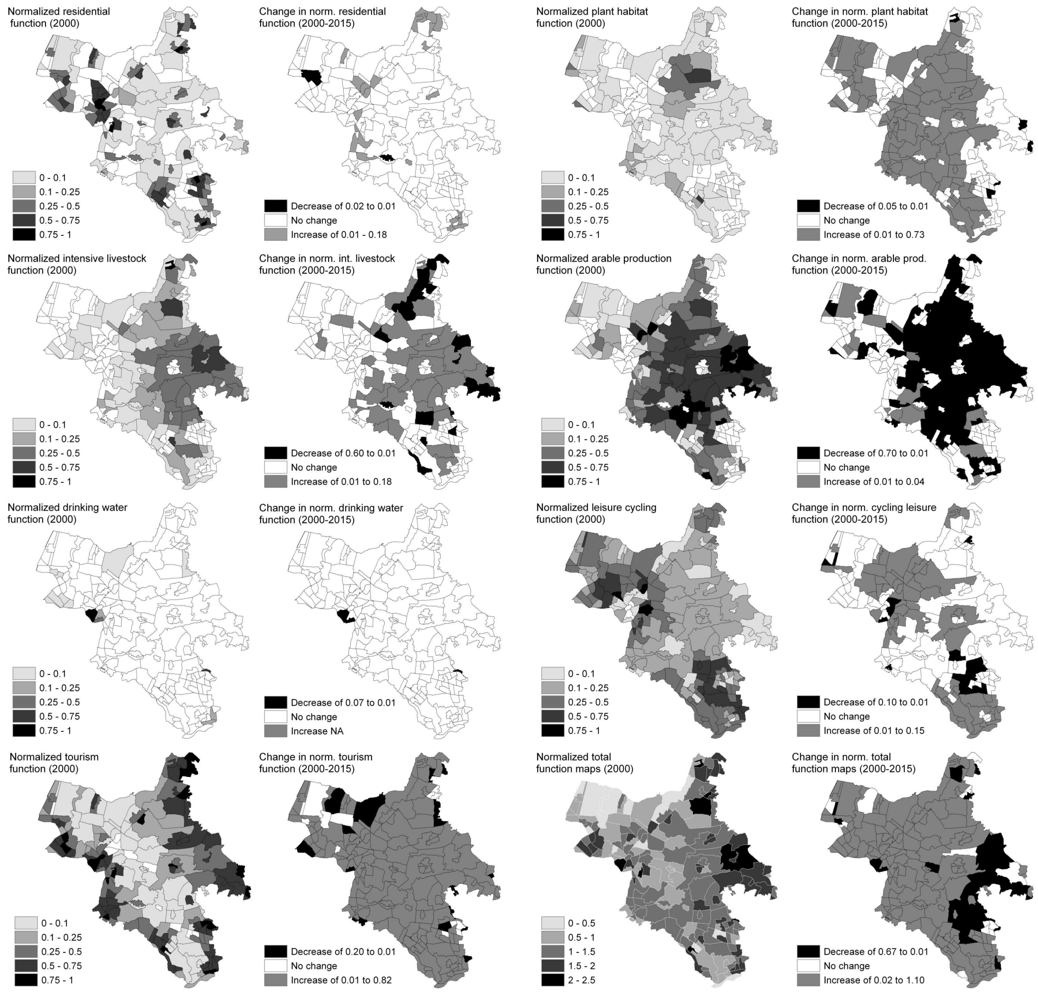

Figure 4.3 Normalised service supply in 2000 and the normalised changes between 2000 and 2015 for each landscape function and the total service supply in the study area. Only areas with a difference >5% of the average service supply per postal code in 2000 are mapped as 'change'.

Valuation of landscape services

The results of the valuation of the landscape functions are presented in Figure 4.4. Per function we show the value of the landscape service per hectare per year and its change between 2000 and 2015. Here again, we only classify the difference in service supply between the two evaluation years as 'change' when the service provision increased or decreased with more than five percent of the average service supply per postal code in 2000. In Table 4.2 we present the average value per hectare of a landscape function before aggregation to postal code level, and the total value per landscape function for the study area as a whole.

For the year 2000 monetary valuation maps show similar patterns as seen in service supply maps, because the valuation of services is directly linked to service supply. The economic value of all landscape functions is foreseen to increase by 2015, except for the arable production function (Table 4.2). However, at some locations an increase instead of a decrease in monetary value of arable production is found. This increase is a result of an increase in net value added per hectare, resulting from increased production efficiency and market prices, which outweighs the decrease due to loss in agricultural land. Next, the impact of the spatial zones regulating intensive livestock development is visible in the map indicating change in economic value of the intensive livestock function (Figure 4.4). This function shows a decrease in monetary value in the postal code zones located in the 'extensive agriculture zones', in which the growth of intensive livestock farms is strictly regulated. In other zones, the overall economic value of the intensive livestock sector is foreseen to grow despite a decrease in intensive livestock farms, so the remaining farms become more intensive. Another observation from Figure 4.4 is that the tourism function shows a decrease in economic value by 2015 in many areas which had the highest economic value in 2000. This is a result of the expansion of the tourism function towards the central part of the study area, leading to lower average value per hectare, even though the overall value of tourism of the study area is expected to increase.

Table 4.2 The average value per hectare per year before aggregation to postal code area, and the total value per landscape function of the study area.

Landscape function	**Valuation indicator**	**2000**		**2015***	
		Mean value €/ha	**Total value millions €**	**Mean value €/ha**	**Total value millions €**
Residential	Price residential land	76 278	700.5	108 366	995.2
Livestock	NVA per livestock farm	8 442	27.4	13 460	29.8
Drinking water	NVA per m^3 water	4 663	1.9	4 848	2.0
Tourism	NVA per accommodation + WTP per tourist	1 295	34.3	1 017	35.3
Plant habitat	Price of land + nature subsidy	922	3.3	1 046	10.3
Arable	NVA per arable farm	341	2.6	352	2.5
Cycling leisure	NVA leisure sector + WTP per tourist	127	5.9	182	8.4

NVA: Net Value Added, WTP: Willingness To Pay

* Based on price levels of 2007

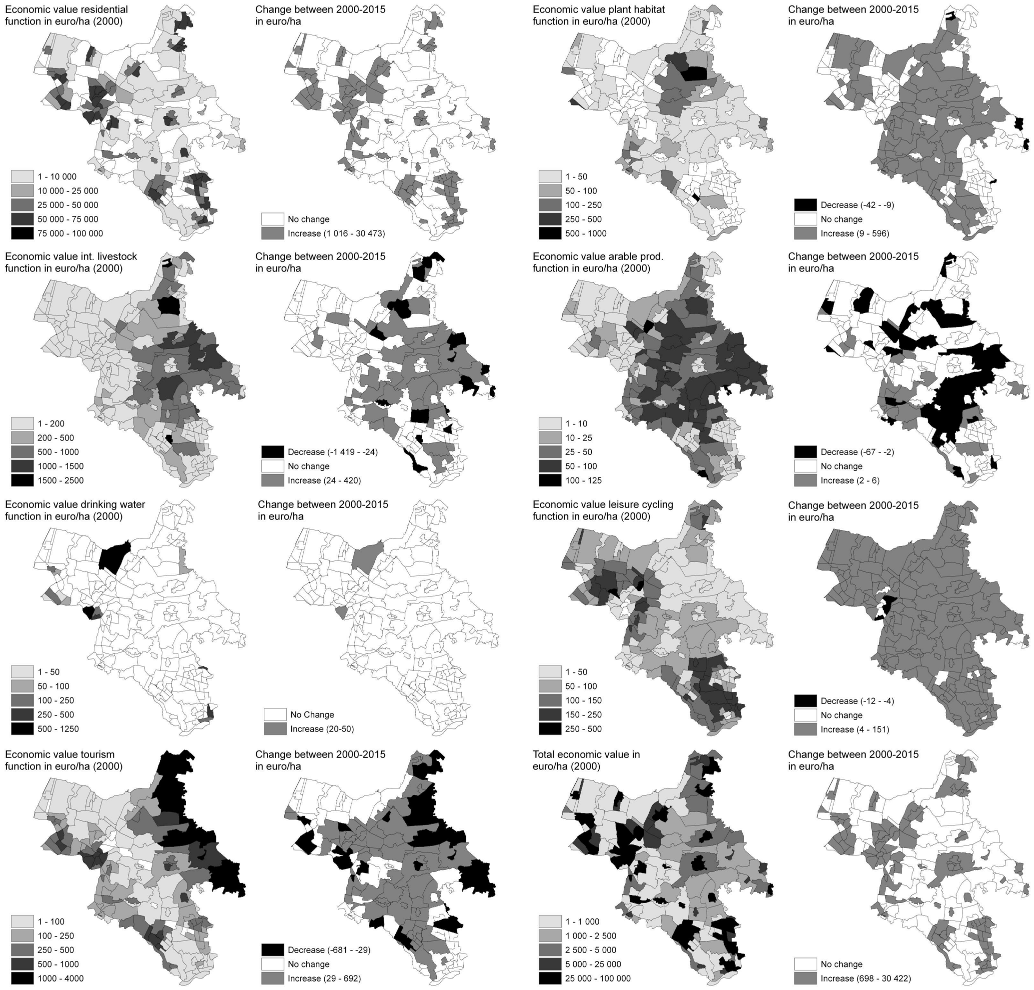

Figure 4.4 Economic values in euro per hectare in 2000 and the changes in value between 2000 and 2015 for each landscape function and the total economic value in the study area. Only areas with a difference >5% of the average economic value per postal code in 2000 are mapped as 'change'. Note that the total value maps do not include plant habitat values.

We summed all landscape function maps in order to visualise the spatial distribution of the total value. The plant habitat values are excluded from this summation as the different valuation measure of this function does not allow for a direct comparison with the other service values. Looking at the spatial distribution of the aggregated total value of the year 2000 we clearly recognise the high value of urban areas (Figure 4.4). Therefore, mainly in areas with urban expansion a strong increase in the total value of landscape services is expected in 2015. For most other areas no large changes in the economic value are observed. This means that at most locations the decreases in economic value are compensated by increases of other landscape functions. In Table 4.2 the total value per landscape function in euro per year for the study area and the average value per hectare is given for the year 2000 and 2015. In both years the capital intensive functions; residential use, tourism and intensive livestock contribute most to the total value of the region.

Discussion

Policy impact

The integrated policy package that is addressed in this study aims to enhance regional development through an improved spatial structure for agriculture, nature, forests, landscape, recreation, water, environment and infrastructure (Provinces of Gelderland and Utrecht, 2005). In this section, we discuss the major policy impacts on the spatial pattern of service supply and value for our study area.

From our results we can observe that by 2015 the strongest increase in services supply is found in rural areas while the strongest increase in value is foreseen to take place in and around urban centres. Also, we observe a general trend that the value of all studied landscape functions increases by 2015, except for the arable production function. So for our selection of landscape functions, the policy package will likely be successful in achieving its objectives. In Figure 4.5 the foreseen changes between 2000 and 2015 in normalised service provision and economic value (with 2015 values based on the price levels of 2007) are presented per landscape function. The percentage of change between the landscape functions in the study area clearly differs between the two methods for landscape service quantification. So the choice of quantification method likely influences the interpretation of the policy impact. For example, when looking at the cycling leisure function the change in service supply only increases a few percentages but the economic value is foreseen to increase by almost 50%. This strong increase in value is a result of the increasing demand and therefore economic value (prices) of the cycling leisure function. A similar effect of price development on the service value is seen for the residential function and to a smaller extent for drinking water supply and arable production functions. For the interpretation of our results it is important to realise that differences between the change in normalised service provision and the change in economic values are caused by a price effect, as a consequence of economic development. Hence, changes in economic value are not solely an effect of the evaluated policy package; the strong increases in the economic value of some services (residential, cycling leisure) would also have occurred without policy intervention.

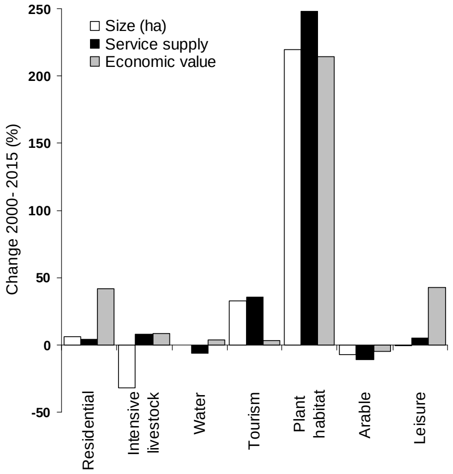

Figure 4.5 Overall change in landscape function extent, service supply and economic value of each landscape function for the study area between 2000 and 2015.

Quantifying change in service supply and value

This study presents one of the first attempts to make the economic value of landscape functions spatially explicit by linking economic values to supplied landscape services on a detailed spatial resolution. This step is perceived crucial in land management and for planning purposes (De Groot, 2006; Daily et al., 2009; De Groot et al., 2010). Even though quantified services give a direct view on the relation between the landscape and the supplied benefits, for decision making purposes it is important to translate these benefits into a unit that allows for trade-off analyses and cost-benefit comparisons. An additional advantage of evaluating changes in landscape functions based on changes in economic value is that an economic valuation takes societal preferences, which are reflected in the price (and value) of the supplied services, into account. One should, however, be aware that an economic valuation based on service supply measures leads to an increased uncertainty in the outcomes, as a result of the strong data requirements and assumptions to make calculations possible. Additionally, carrying out an economic valuation can be challenging for landscape services without a use value or market value, it might therefore be more meaningful to express these functions in service supply in addition to their assessed economic value (Schaeffer, 2008; Daily et al., 2009). Obviously the economic valuation carried out in this study could be improved by more detailed valuation data and analyses (e.g. like in Bateman et al., 2005; Maler et al., 2008). By describing the economic value as a function of service supply and demand, for example, more accurate trade-offs calculations between the total benefits for society (increase in services) and the losses in other landscape functions can be made (Fisher et al., 2008).

To evaluate changes in service supply it is needed to account for factors on different spatial scales (Hein et al., 2006; Carpenter et al., 2009). In this study we included national and provincial policies, regional characteristics and location properties to describe the future location and quantity of landscape services. To reduce the complexity of our analysis we did not explicitly include global processes and individual choices in our assessment, even though processes on these scales are known to play an important role (O'Rourke, 2005; Nelson, 2006; Yadav et al., 2008). Different spatial scales also play a role in linking economic values to landscape services (Bateman et al., 2005). By using market prices to quantify the economic value of a landscape service, often point observations need to be linked to the complete spatial extent of a landscape function. For example, in our study the tourism service is partly valued based on tourist accommodation revenues. We showed that not only the location of the camp site generates the service value, the surrounding landscape also contributes to this value. To adequately distribute the economic value to the total function area, the spatial extent of each landscape function needs to be known (i.e. the area of the landscape that contributes to the tourism function). The spatial extent of landscape functions is also reflected in the value per hectare. The value per hectare will decrease with the total needed land per function in the study area. Depending on the evaluation level, a hectare or the complete study area, a different ranking in values can appear. In our study the value of one hectare with intensive livestock is much higher than one hectare land having a tourism function, while the total value of the tourism service for the study area overtakes the value of livestock. However, one should be aware that the actual spatial extent of some functions goes beyond the limits of the study area; the intensive livestock function uses land for fodder production outside the study area. Additionally, the negative externalities of livestock production are not accounted for in this study.

Conclusions

This study presents one of the first attempts to make the economic value of landscape functions spatially explicit by linking economic values to supplied landscape services on a detailed spatial resolution. In this chapter we assessed the changes in landscape service supply as a result of the implementation of an integrated policy package and regional development trends. We used a methodology to translate policy implementation into changes in landscape properties and subsequently assessed how these changes affect the location and supply of landscape services and their value in the study area. The resulting maps visualise trade-offs between different landscape functions and we were able to show side-benefits as a result of policy implementation. Additionally, by taking into account the spatial scale of landscape functions we were able to show how non-overlapping functions can influence each other. These complex interactions often prove land management difficult. Also, as an addition to most policy evaluation studies so far, we included both the intensity and the extent of policy impact on the landscape to quantify the provisioning of landscape services.

The uncertainties in extrapolation methods, model and value assumptions, render our study as a methodological scientific contribution rather than a complete ex-ante evaluation or feasibility study to be used by policy makers. Therefore the current outcomes need to be considered indicative, and a stepping stone for future research. By linking service supply to economic values an important step is taken in institutionalising landscape services and guiding decision making (Cowling et al., 2008; Fisher et al., 2008; Daily et al., 2009). As landscape services link to both natural and agricultural systems, an evaluation of policy based on landscape services can make an important contribution to regional integrated assessments.

Appendix chapter 4

Changes in farm size and farm distribution

Farm size is in the Netherlands commonly expressed in Dutch Standard Units (DSU) representing, amongst others, the size of a farm in number of livestock units (e.g. in 2000 one DSU relates to three adult pigs or 385 chickens). In 2015 the number of livestock farms smaller than 50 DSU is expected to decrease by 55%, farms of size 50 to 70 DSU are foreseen to decrease by 45%, the class of farms between 70 and 100 DSU are expected to decrease by 19% while the number of farms larger than 100 DSU is expected to increase by 2% (Provinces of Gelderland and Utrecht, 2005). The intensive livestock development zones as described in the Reconstruction Act are used to explore the future situation of the intensive livestock sector in the study area. To create a map with possible locations of intensive livestock farms in 2015, current farms within the 'agricultural development zone', 'mixed zone' and outside the zones, are selected in an automated procedure. In this procedure farms are randomly selected based on their probability to continue farming given their current size and location. Subsequently, according to plans of the Reconstruction Act, all farms larger than 70 DSU within the 'extensive agriculture zone' are relocated to abandoned farms in the 'agriculture development zone'. The final map of intensive farm locations in 2015 is based on 100 repetitions of this procedure. The growth in terms of DSU of intensive livestock farms is expected to follow a different trend. Almost all intensive livestock farms are envisioned to grow between 2000 and 2015. The prognosis on the amount of growth depends on the location of the farm within the different spatial zones (RPB, 2007). In the 'agricultural development zone' a farm is expected to grow with 51%, in the 'mixed zones' a growth in DSU of 33% is expected, within the 'extensive agriculture zone' farms are not foreseen to grow, and outside the spatial zoning areas a decrease of 3.3% in DSU is expected. Combining the estimated locations and growth of intensive livestock farms, a map of the intensive livestock sector of the study area is made for 2015. Based on similar calculations the change in arable production farms between 2000 and 2015 is estimated and mapped. Here again, the repeated random selection procedure based on an overall farm abandonment chance (of 43%) and growth prognoses derived from the Dutch Farm Accountancy Data Network of arable farms are used to assess the situation in 2015.

Chapter 5

A multi-scale approach for analysing landscape service dynamics

Landscapes and the provision of landscape services are continuously changing as a consequence of shifts in societal needs. This chapter presents a modelling approach to analyse the spatial and temporal dynamics in landscape service supply as a result of a changing landscape and societal demand. In this approach we explicitly address (i) the multifunctional character of the landscape, (ii) the different spatial levels at which interactions between landscape service supply, demand and land management occur and (iii) the trade-offs in service supply as a result of land management actions. We first conceptualise the system in which landscape service dynamics take place. Next, we implement the resulting conceptualisation into a comprehensive modelling framework. Finally, this framework is applied to successfully simulate changes in several landscape functions in a rural region of The Netherlands. The outcomes of the application illustrate the relevance of modelling landscape service dynamics for environmental management and decision making. The presented modelling framework is a first step towards a dynamic modelling approach to better identify, map and quantify the dynamics of landscape functions and their provided services in time.

Based on: L. Willemen, A. Veldkamp, P. H. Verburg, L. Hein, R. Leemans
Submitted

Introduction

Landscapes provide a multitude of services that benefit people. Landscape functions indicate the capacity of a landscape to provide these services to society. Such services include, amongst others, food and timber production, fresh water supply, and recreational opportunities. Globally, the demand and use of landscape services per capita has increased exponentially since the 1960s and landscapes providing these services are changing rapidly (MA, 2005). These strongly related trends have led to changes in landscape service supply and have increased the complexity of interactions between society and their surroundings (Schröter et al., 2005; Diaz et al., 2007; Carpenter et al., 2009). Changes in landscapes together with the demand for landscape services are driven by a range of demographic, economic, political, cultural, and biophysical processes (MA, 2005; Nelson, 2006). Through land management and spatial policies, society can adapt the landscape in such a way that the provision of one or more landscape services is improved (DeFries et al., 2004; Bastian et al., 2006; Nelson, 2006). However, most landscapes provide multiple services and adaptations of the landscape often affect individual landscape functions in different manners. This can lead to unintended trade-offs in service provision (DeFries et al., 2004; Chan et al., 2006; De Groot, 2006). By effectively governing and guiding land management actions, undesired trade-offs between landscape functions could be minimised.

Landscapes are spatially diverse and therefore landscape service supply is unequally distributed over an area. Land management actions lead to location-specific effects and trade-offs, which makes it necessary to study changes in landscape service supply in a spatially explicit way (Chapter 3, Nelson et al., 2009). Additionally, policy and management actions often take place at other spatial levels than the local service supply and societal service demand, which increases the likelihood of conflicts in service supply and use (Evans and Kelley, 2004; Cash et al., 2006; De Groot, 2006). Therefore, processes and feedbacks between society and the environment at these different spatial levels should be understood and incorporated into decision making to support effective management actions (Kremen and Ostfeld, 2005; Carpenter et al., 2009).

Spatial models play an important role to systematically describe and understand feedbacks and interactions between society and the environment (Parker et al., 2008). Spatial models which are currently available to support planning activities, relate primarily to land-cover patterns with the classic policy focus on provisioning services, or focus solely on single landscape services (Verburg et al., 2004; Tallis and Polasky, 2009). Models focusing on land-use changes are developed to describe complex feedbacks between society and environment (Verburg, 2006; Parker et al., 2008) but are less suited to explicitly deal with the multifunctional character of a landscape and the different consequences of management decisions on the service supply of landscape functions (Pinto-Correia et al., 2006; Daily et al., 2009; Verburg et al., 2009). More recently, spatially explicit (ecosystem) service provision modelling tools have become available that describe multiple service supplies and different function interactions (Boumans et al., 2002; Gund Institute, 2009; Tallis and Polasky, 2009; Villa, 2009). These modelling initiatives are able to assess the impact of human activities on the provision and value of multiple services in space and time. However, these models do not explicitly simulate spatial and temporal feedbacks in landscape service supply as a result of dynamics in land management in relation to service demand.

The objective of this chapter is to present an innovative modelling approach that allows analysing the multi-scale dynamics in landscape service supply as a result of a changing landscape and societal demand. Drawing on the insights from land-use system and ecosystem modelling efforts, we explicitly address in this modelling approach (i) the multifunctional character of the landscape, (ii) the different spatial levels at which interactions between landscape service supply, demand and land management occur and, (iii) the trade-offs in service supply levels as a result of land management actions. In this chapter we focus on the exploration of possible spatial and temporal dynamics of landscape functions. Therefore our approach does not aim at finding an optimal configuration of landscape functions to maximise service supply for a region (e.g. Meyer and Grabaum, 2008). In this chapter we first describe our conceptualisation of the system in which landscape function dynamics take place. Next, we implement the resulting conceptualisation into a modelling framework. We finally apply the modelling framework to simulate a number of landscape services of the rural Gelderse Vallei region in The Netherlands to illustrate its potential for analysing the dynamics in multiple service supply.

A multi-scale system description

Figure 5.1 conceptualises the main processes and feedbacks that form the basis of the modelling approach. Here, all processes and interactions take place at three interconnected spatial levels; local level, management unit level, and regional level. These spatial levels represent respectively the landscape service supply, land management institutions, and societal service demand. Interactions between the spatial levels each have a temporal dimension. Higher level trends and processes (e.g. climate change, or market dynamics) also affect local service supply (Schröter et al., 2005; Nelson, 2006). In this system description, effects of these processes are represented in local and region system levels. Following the processes and components of Figure 5.1, a detailed description is given per spatial level in this section.

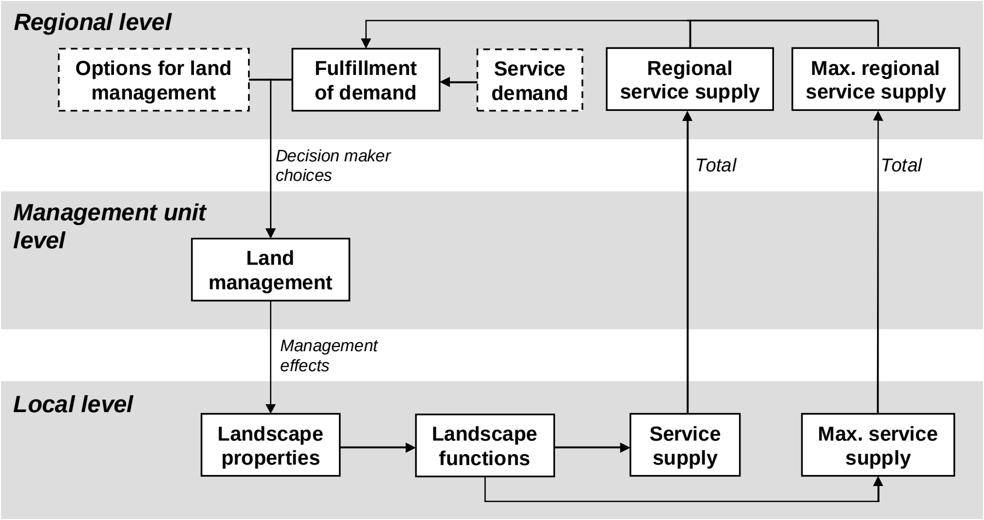

Figure 5.1 Overview of the conceptual spatial and temporal interactions in landscape service supply. Dashed boxes indicate scenario specific components.

Local level

The local level relates the level at which landscape services are supplied. The supply of landscape services is not evenly distributed within a landscape as they relate to spatially variable location and surroundings characteristics (Chapter 2 and Diaz et al., 2007; Egoh et al., 2008; Tallis and Polasky, 2009). These characteristics can be summarised as the *landscape properties* of a location, and include biophysical properties (e.g. soil type or groundwater level), socioeconomic properties (e.g. land-use or population pressure), and spatial characteristics (e.g. clustering or proximities measures). Every *landscape function* relates to a different set of landscape properties. Spatially variable landscape properties can determine the boundary constraints of a function or influence the level of *service supply*. For example, accessibility of natural areas is a boundary constraint for recreation, while the number of visitors who can reach the area indicates the actual service supply. Landscape functions are in this research directly defined and quantified by their service supply. Landscapes are often multifunctional, meaning that at a location more than one landscape function is present. At multifunctional locations landscape properties are present that support the provision of a bundle of landscape services. At some locations service supply is too low to be beneficial or relevant for society (e.g. only a few people that can reach a specific recreation area). Therefore minimal levels of service supply are defined as thresholds to spatially delineate landscape functions (like in Chapter 2 and Bastian et al., 2007), as shown in Figure 5.2.

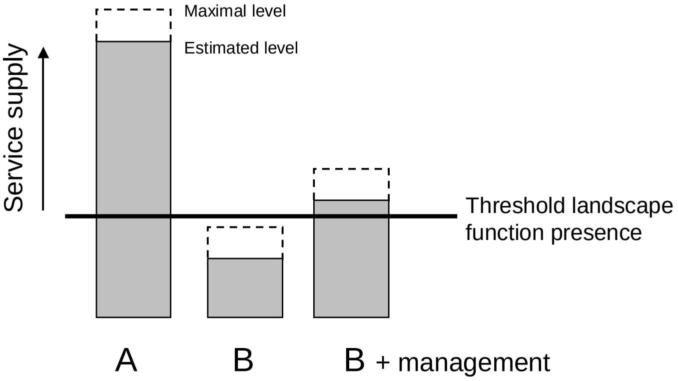

Figure 5.2 Schematic overview of the concepts of landscape function thresholds and service supply limits. At location A, the landscape function is considered present as service supply is above the minimal service supply threshold. Service supply at location A can vary between the threshold level and dashed line. At location B service supply is below a minimal threshold, therefore the landscape function is considered absent. Landscape properties at location B can be changed is such a way that the service supply passes the threshold and the landscape function is assumed present (B + management).

We consider landscape systems not static but flexible in terms of service provisioning. Therefore we do not describe service supply of a location by a fixed number but by a range of service provision. This range indicates an unused potential of landscape functions on top of their actual service supply. Due to these different levels of service usage, the level of landscape service supply can increase without a need of changing landscape properties. In this study we focus on the potential, *maximal service supply*. This upper limit of service supply differs per landscape function. When this limit is reached, the service supply is assumed maximal within the current spatial configuration of the landscape (Figure 5.2). The maximal level of service supply can represent the biophysical limits of a location, sustainable harvest levels (e.g. maximal amount of drinking water extraction without damaging the ecosystem), or societal preferences (e.g. maximal number of visitors for recreation areas without negatively changing the recreation function) (see e.g. Potschin and Haines-Young, 2006a).

Regional level

Services that are being provided at local level respond to the societal demand for services at regional level (Figure 5.1). Society is in this study defined as the total group of stakeholders within a region and has a certain (indirect) *demand for landscapes services*. This demand can change over time, for example due to increase in population. It is assumed that the *total supply* of landscape services of a region should meet this demand. The difference between regional demand and service supply, taking into account the maximal total service supply, is indicated by the *fulfilment of service demand*. For example, an increased demand for recreation could be accommodated within a current landscape configuration. As long as the service demand remains under the *maximal service supply* (Figure 5.2), it is presumed that the system has enough buffer capacity to deal with an increase in landscape service demand.

Through a range of *options for land management* society has the capacity to adjust landscape properties to ensure sufficient service supply. We assume that society will change landscape properties when service demand exceeds the maximal supply of services of a system (Termorshuizen and Opdam, 2009). Land management decisions that aim at improving service supply, take place based on a evaluation of management options, cost, and preferences and priorities (Groves et al., 2002; Tallis and Polasky, 2009). Management options include possible interventions to modify specific properties of the landscape (e.g. groundwater level or accessibility of natural areas). The selection of a management options is a result of societal willingness to pay for the conversion costs and preference for management locations. The preferred land management location differs per landscape function. For example, criteria to select locations to enhance habitat supply differ from the criteria to select the preferred sites to improve recreation possibilities. In case multiple functions have a shortage in service supply, landscape service with the highest societal priority are accommodated first. Priorities in landscape functions are based on, for example, the current policy focus, economic opportunities, or the highest shortage in service supply. Depending on the demand and valuation of goods and services, society will handle these priority and trade-off choices differently over time (MA, 2005; Nelson, 2006). Because of this large variety in choices we have summarised all land management options in as a ‘scenario specific component’ (Figure 5.1).

Management unit level

Human interventions as a result of a changing regional demand take place through *land management* at management unit level (Figure 5.1). We only address interventions that have a management institution within the system boundaries. The unit and therefore spatial extent at which land management actions take place, differs per landscape property that is considered to be changed. For example, groundwater is regulated by means of water management units, cattle density is controlled at farm level, and accessibility of nature is managed within park boundaries. This adaptive land management ideally takes place at locations where landscape service benefits or suitability can be maximised (Chan et al., 2006). However, management actions often do not occur at the same spatial extent as the smaller preferred management locations. For example, to be able to change the groundwater level at a location, the groundwater level of the complete management unit in which that specific location falls need to be adjusted. Because of these differences in spatial extent, land management can lead to trade-offs between functions not only at multifunctional locations, but also at locations within a similar management unit. In time, these trade-offs in service supply might influence the selection of most suitable land management locations or alter the ranking of function priorities. Therefore, land management can change all landscape functions and their service supply and might influence the complete feedback loop between local service supply and societal demand in a region (Figure 5.1).

Towards a dynamic modelling framework

To capture the system concepts and processes as presented in Figure 5.1, a spatially explicit modelling approach is needed in which the processes and interactions of the multi-level structure are described and quantified in time. Additionally, the modelling approach should deal with the multifunctional character of the landscape by quantifying possible trade-offs in service supply. In this section a step-wise description is given of the implementation of these system concepts and processes into a modelling framework. The modelling framework uses rasterised spatial data sources as input data. In the implementation of the system concept one raster represents the local level as described in Figure 5.1.

Step 1. Quantifying service supply at local and regional level

To quantify and map landscape service supply, spatial data representing landscape properties and their quantified relation with landscape services are used. Quantification of these relations is based on empirical techniques (logistic and linear regressions) and on a literature-based multi-criteria method as reported in Chapter 2. For example, linear regression models are used when field observations on service supply are present. In this case, service supply ($S_{L,t}$) of a location (L) at time (t), i.e. a raster cell, is estimated by a set of quantified relations (β) with landscape properties of a location (x_L), assuming zero error; see Equation 1:

$$S_{L,t} = \beta_0 + \beta_1 x_{1L,t} + \ldots + \beta_n x_{nL,t} + 0 \qquad (1)$$

where n is the index for number of landscape properties, β_0 is the service supply at reference locations, and S_L is the service supply above the minimal threshold that defines the presence of a landscape function (Figure 5.2).

In the case that no adequate field observations on service supply are available, decision rules are defined for a multi-criteria analysis. The actual service supply per location ($S_{L,t}$) is then described by spatial data representing a number (n) of binary landscape properties giving the boundary constraints (x_c) and a number (m) of local landscape properties which define the amount of service supply (x_L) (Equation 2).

$$S_{L,t} = x_{c1} \cdot \ldots \cdot x_{cn} \left(x_{1L} + \ldots + x_{mL} \right) \tag{2}$$

Based on the calibrated formulas and spatially explicit data on landscape properties, the estimated service supply (S_L) is mapped per landscape function for the complete region. Additionally, multifunctionality is quantified and mapped based on the number of functions present and the total service supply per location. To be able to aggregate service supply of landscape services measured in different units, the service supply is normalised to a 0 to 1 range, based on the minimum and maximum supply values.

To quantify the maximal level of service supply for landscape functions which are estimated based on regression analyses (Equation 1), the deviations of field observations from the predicted value (S_L) are calculated. We consider all values around the predicted service supply which fall within the 95% confidence interval of the regression model as likely supply values. So, the average upper value of the confidence interval (CI_{MAX}) indicates the maximum service supply of a location ($S_{L\ MAX,t}$) under the conditions set by the current (t) landscape properties (Equation 3).

$$S_{L_{MAX},t} = \beta_0 + \beta_1 x_{1L,t} + \ldots + \beta_n x_{nL,t} + CI_{MAX} \tag{3}$$

For landscape functions of which the service supply is not quantified based on regression analyses (e.g. Equation 2) a different method is needed. Depending on the landscape function, the maximal supply limit $S_{L\ MAX}$ can in this case be defined based on literature reviews, field experiences, or questionnaires. Therefore, for these landscape functions we cannot provide a generic equation to calculate the maximum service supply.

Service supply at the ith locations in the region is summed to define the aggregated regional service supply (S_R) for each landscape function (Equation 4). Likewise, the maximal service supply within the current set of landscape properties ($S_{L\ MAX}$) value is aggregated to the maximal service supply at regional level ($S_{R\ MAX}$) for each landscape function (Equation 5).

$$S_{R,t} = \sum_i S_{L,t,i} \tag{4}$$

$$S_{R_{MAX},t} = \sum_i S_{L_{MAX},t,i} \tag{5}$$

Step 2. Defining service demand at regional level

The demand for landscape services is defined by a scenario indicating the regional demand (D_R) at time (t). This demand includes both the demand from local actors as well as the demand coming from outside the region. An important aspect of ‘demand’ is that it relates to the provided services (e.g. ton/ha) rather than a certain amount of hectares. A yearly regional demand ($D_{R,t}$) can be derived from a long term policy target (e.g. an increase in the tourism services of 12% in ten years time). When dealing with such a pre-defined long term demand, the yearly demand for services ($D_{R,t}$) is adjusted for trade-off effects in service supply resulting from conflicting land management actions. So, when service supply decreases because of trade-off effects, these supply losses (S_{RTO}) are added to the regional demand in the next time step to assure a matching service supply over multiple years. Regional demand derived from a multiple year target is therefore determined as follows:

$$D_{R,t} = D_{R,t} + S_{RTO,t-1} \qquad (6)$$

Similarly, gains from management synergies are subtracted from the regional demand (D_R) in Equation 6. So, these adjustments are not made because of a change in the overall demand for a landscape service, but rather by a decrease of fulfilled of the demand by the landscape.

Step 3. Allocating land management per management unit

When the regional demand for a service (D_R) exceeds the buffer defined by the maximal regional supply ($S_{R\ MAX}$), landscape properties at management unit level are changed to enhance service supply. Service supply is increased by extending the areal of a landscape function, so new functions are added to locations. In this step, we decide on the location and magnitude of land management actions. For each landscape function land management options are indicated in scenarios. In an iterative simulation loop, landscape properties (x) are changed at preferred management units until service supply (S_R) matches service demand (D_R). The preferred management units are defined by rules relating to conversion cost (e.g. residential areas cannot be converted into nature), current policy and spatial plans, and locations with the highest suitability to extend a landscape function. Locations with the highest suitability are in our approach the locations with a service supply which is just under the threshold of minimal service supply (Figure 5.2). In case multiple functions have a shortage in service supply, the ranking in function priority defines the order in which changes are made. These function priorities are set for the complete region. After each time step in which management has taken place, the modelling framework recalculates all landscape functions (Equation 1 and 2, $t+1$). From the supply maps of t+1, the new service supply range is defined and it is monitored if land management led to an increased or decreased service supply and spatial extent for each landscape function.

Applying the modelling framework

The application of the modelling framework is based on data from the Gelderse Vallei region in The Netherlands. This rural but highly populated region encounters multiple claims on land due to an increased demand for landscape services (Provinces of Gelderland and Utrecht, 2005). Three landscape functions; plant habitat provision, arable production, and information on cultural heritage, are included in the modelling framework to demonstrate a simulation of the dynamics in landscape functions and service supply over time. These three landscape functions are selected because of conflicting landscape property requirements (plant habitat versus arable production function) and the negative feedback on service supply due to changes in land-use (cultural heritage function). The landscape functions are characterised using data at a raster cell resolution of 100 x 100 meter for the year 2000 and the model simulations run for 15 years (in steps of one year).

Plant Habitat
Service supply of the plant habitat function is quantified by using an index referring to the suitability of a location to provide habitat to rare, endemic and indicator plant species (Hertog and Rijken, 1996; Rijken, 2000). In our study area these species mostly relate to vegetation growing under mesotrophic and wet conditions. As described in the first step of the modelling framework, field observations on service supply are used to build a linear regression model. The regression model describes the relation between plant habitat suitability and groundwater level, soil type and natural land cover (Chapter 2). In Table 5.1 the quantified relations with landscape properties, the function threshold and the mean confidence interval limits of the plant habitat function are presented.

For the second step, change in service demand for the plant habitat function is defined by a spatial policy to increase the natural area. This policy aims at an increase of natural areas area of 90% by 2015 in the study (Provinces of Gelderland and Utrecht, 2005). We translated this demand to an annual increase for plant habitat services of 6% of the habitat supply of 2000.

Deep winter groundwater levels show a negative relation with plant habitat in our study area (Table 5.1).The preferred management option to enhance nature is to increase the winter groundwater with 10 cm per time step. Management actions (Step 3) are only allocated in non-built-up areas which are designated for nature development (VROM, 2006). The spatial units at which groundwater is managed are defined within agricultural areas, the bordering parcels with similar hydrological soil properties. Outside agricultural areas the hydrological soil classes determine the management units (Alterra, 2000a, 2006). The plant habitat function is assigned the highest priority because of the strongest increase in demand of the three example functions.

Arable production

Service supply of the arable production function is mapped based on the production of the only commonly grown arable crop in the study region: fodder maize. By carrying out a linear regression on a set of selected maize plots in the first step, the landscape properties explaining the yield per hectare are identified in Chapter 2. Details on the arable production function are given in Table 5.1.

For the second step the service demand is arbitrarily set on an extrapolation of the trend in maize production between 1995 and 2005. This trend is translated into the model as growth in annual demand of 3% of initial service supply. When the regional demand exceeds the total supply, the management option to enhance arable production is the expansion of arable fields. The preferred management site to converted land into new production fields are the locations that are already under agricultural use which are bordering current arable fields (Step 3). Opposite to the plant habitat function, a high winter groundwater has a negative impact of the arable production function (see Table 5.1).

Cultural heritage

In the first step of the modelling framework, cultural heritage is mapped based on a multi-criteria analysis. The cultural heritage function is delineated using the locations of high-value historical landscapes as defined by the Dutch government. The percentage of unchanged land-use between the year 1900 and 2000 within 300 meter of each raster cell is used to quantify the service supply, as authenticity of the landscape is considered a key aspect of the cultural heritage function (Provinces of Gelderland and Utrecht, 2005; Daugstad et al., 2006).

Because of the focus on preservation, service supply of this landscape function cannot be improved. Therefore, the demand for cultural heritage services is assumed not to increase over time and the function is not actively managed (Step 2 and 3). The cultural heritage function is negatively affected by changes in land-use (Table 5.1). So changes in arable expansion conflict with this landscape function. The cultural heritage function is assigned the lowest priority, implying that demand for the plant habitat and arable production services is accommodated without considering the effect on cultural heritage.

Table 5.1 Landscape properties, quantitative relation with service supply, and the mean upper confidence interval limit (CI Max). Quantified relations showing ‘–’, indicate a boundary constraint of the landscape function location. Landscape properties that relate to management actions are underlined.

Landscape function	Landscape properties	Quantified relation	Threshold	CI Max
Plant habitat *	Winter groundwater level (cm below surface)	-0.12	>=5	0.27
(0-9 index)	Sandy soil (no=0 yes=1)	0.23		
	Sandy clay soil (no=0 yes=1)	0.13		
	Peat soil (no=0 yes=1)	0.14		
	Proximity to open natural area (m)	0.56		
	Proximity to forested natural area (m)	0.26		
	Absence of built-up area	-		
Arable production*	Summer groundwater level (cm below surface)	-0.10	>=35	1.54
(0-60 ton/ha)	Winter groundwater level (cm below surface)	0.12		
	Sandy soil (no= 0 yes=1)	0.15		
	Sandy clay soil (no= 0 yes=1)	0.09		
	Peaty sand soil (no= 0 yes=1)	0.08		
	Average farm size (ha)	0.40		
	Number of neighbouring farms (farms per km^2)	0.69		
	Land currently under agricultural use	-		
Cultural heritage	Policy assigned cultural landscapes	-	>0	NA
(0-100%)	Percentage of unchanged land cover within a 300 meter radius between 1900-2000	1		

* Relations are quantified based on a regression analysis

Results

Figure 5.3 shows the regional changes in service supply for 15 years using the amount of service supply at *t*=0 as reference value. Trade-offs between nature development and arable production, and arable production and cultural heritage can be observed from the model results. On an aggregated regional level, these trade-offs differ over time, depending on the selection of management sites in a time step. An example of these temporal-spatial effects can be observed from the impact of arable expansion on the cultural heritage function. Arable expansion leads to changes in land-use. These changes decrease the authenticity of the area; the indicator used to quantify the cultural heritage function (Table 5.1). The first expansion of arable production fields (*t*=2) shows little effect on cultural heritage while from the second expansion (*t*=4) onwards, expansions are simulated to take place in or near cultural landscapes; this negative effect can clearly be observed from Figure 5.3.

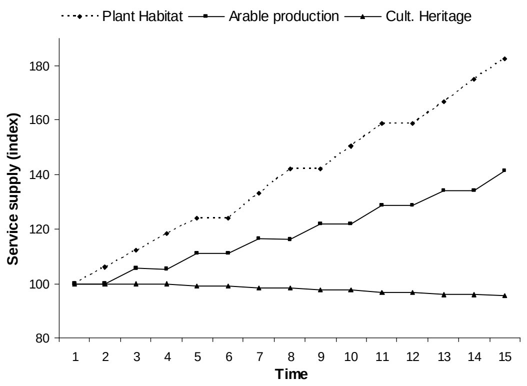

Figure 5.3 Relative changes in service supply at regional level as a result of groundwater management and agricultural expansion. Service supply at t=0 is used as reference value (100).

Figure 5.3 presents the net changes in service supply. In Figure 5.4 we show the trade-off relations between landscape functions a as result of the simulated land management actions, excluding the growth scenarios of service demand. Both relations are non-linear, with increasing trade-off effects with a higher regional service supply. So, trade-offs between landscape functions increase when more widespread management actions are needed to meet the demand for landscape services.

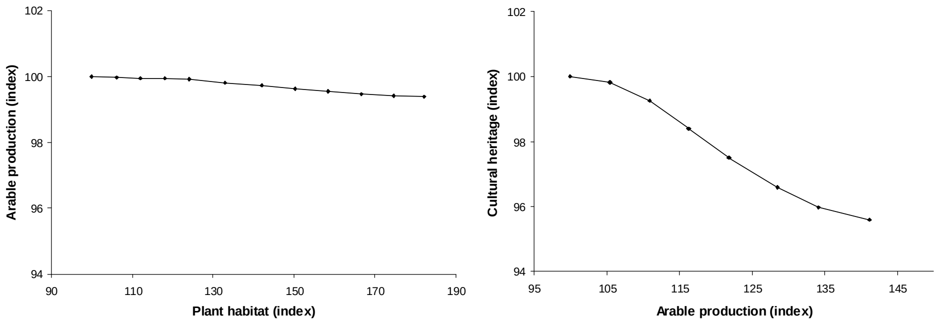

Figure 5.4 Trade-off relations in service supply at regional level, as a result of groundwater management and agricultural expansion, using the service supply at t=0 as reference value (100).

Figure 5.5 illustrates the effects of the simulated management actions on the spatial distribution of the landscape functions and service supply. Several effects of multi-level spatial interactions can be observed. The first relates to management actions that also positively or negatively influence service provision at other locations. For example, arable production decreases not only at locations where the plant habitat function is increased. This also happens when arable production fields are present in the same groundwater management unit as the selected site to improve plant habitat function. The grey areas on the plant habitat map in Figure 5.5 indicate the locations where service supply shows an unintended increase. Only locations where the plant habitat function is lacking can be selected to enhance nature. However, in Figure 5.5 also locations that already contained a plant habitat function show an increased service supply due to groundwater changes at management unit level. The second off-site effect can be observed from the cultural heritage functions maps The cultural heritage function is located in the policy assigned 'cultural heritage landscapes' and is affected by changes in land-use within a radius of 300 meter. In the southern part of the study area arable expansion is simulated, but the resulting changes in land-use leads to a decrease of in service supply of the surrounding cultural heritage function. So, changes in arable production lead to changes in cultural heritage up to 300 meters away (Figure 5.5).

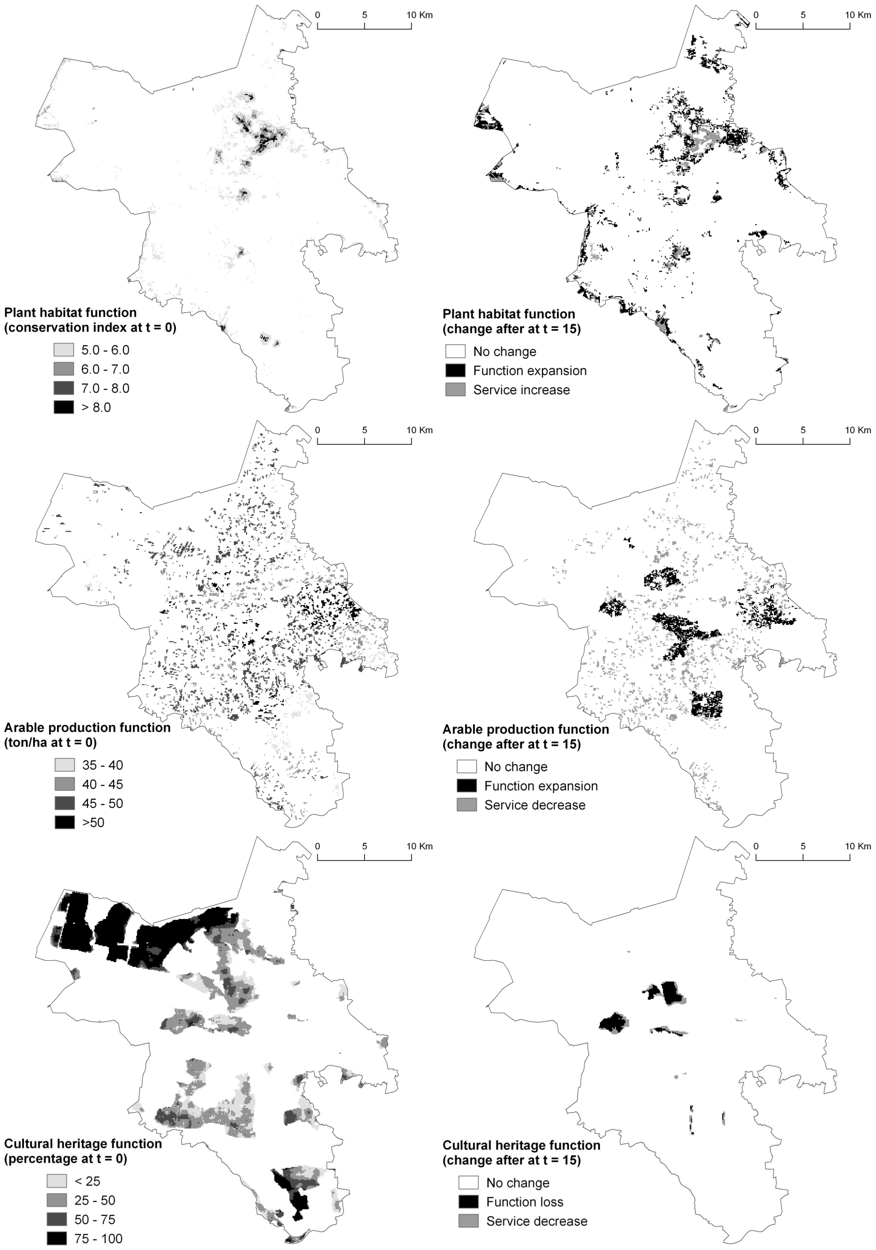

Figure 5.5 The spatial distribution of landscape services at the start of the simulations and the change in landscape function presence and service supply after 15 time steps

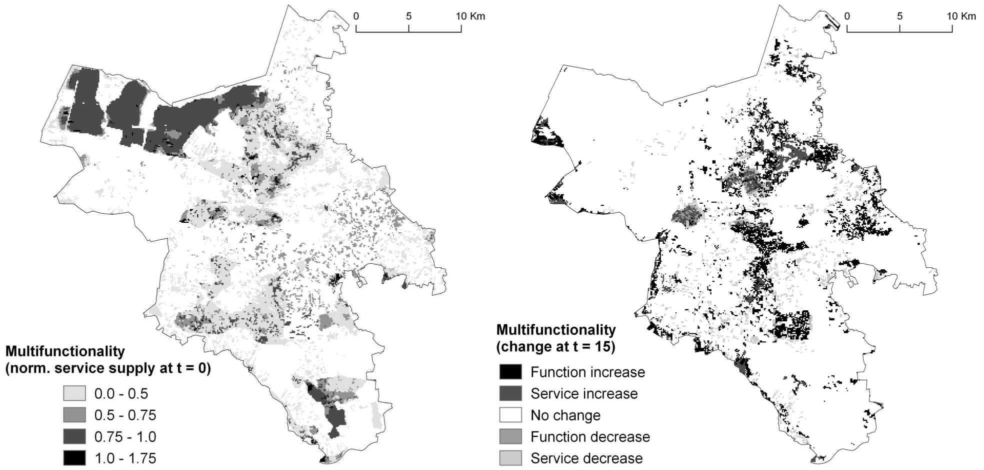

Figure 5.6 Multifunctionality maps expressed as the normalised and summed service supply at the start of the simulations, and the changes in number of landscape functions and service supply after 15 time steps.

In Figure 5.6 changes in multifunctionality as a result of simulated land management actions are mapped. Multifunctionality at *t*=0 is quantified by the normalised and summed service supply of the three landscape functions. The change in multifunctionality after 15 time steps is expressed as an increase or decrease in the number of functions and the total service supply. A decrease of the number of landscape functions or service supply indicates that at these locations management actions only lead to a decrease of land functionality. These areas (in light greys in Figure 5.6), can become hot-spots of conflicts between landscape functions. Comparing the Figure 5.6 with Figure 5.5 it can be observed that these hot-spot are mainly located in areas with a decreased agricultural production.

Discussion

Application of the modelling framework

The application of the modelling framework for a Dutch multifunctional landscape illustrates the relevance of analysing service dynamics in a quantified and spatially explicit way. At the aggregated regional level, the model simulates an overall increase in service supply as a consequence of land management actions. However, the mapped effects of the land management actions on the spatial distribution of service supply show that some locations strongly increase in total service supply, while other locations lose service providing capacity. The output maps visualise locations at which trade-offs take place in service supply as a result of land management actions (i.e. locations where the improvements in service supply lead to decreases in service supply of another landscape function). So, spatial patterns of trade-offs are made explicit for the study region using a spatial modelling framework.

To illustrate the capabilities of the modelling framework, we only simulated simple and straightforward management options and societal demands in a scenario. Due to the obvious oversimplification of reality in the scenario assumptions, the presented results cannot be used as plausible futures of the study area. However, the modelling framework is flexible, allowing more complex scenarios to be implemented. For example, more landscape functions could be added to better reflect reality. In the current model application only three landscape functions are included. These do not capture all possible trade-offs (or synergies) in the case study region. Additionally, policy plans and management constraints could be defined more explicitly in the model scenarios. For example, multiple claims on land may make it impossible to fulfil the service demand from mono-functional locations only. Real-life management objectives therefore focus on stimulating service provision of a limited areal extent, or by optimising the multifunctional potential of the land (Provinces of Gelderland and Utrecht, 2005; Meyer and Grabaum, 2008; Parra-Lopez et al., 2008). This objective can be implemented in the modelling framework by selecting only locations where other landscape functions will not be negatively affected by interventions. Related to possible landscape limitations, management scenarios could also aim at improving service supply at locations that are already providing the landscape service. Currently, increased service supply is obtained by management actions that lead to an expansion of the service supply areas. By not constraining the model to look for these solutions, we explored a scenario in which potential negative effects of land management are neglected in decision making.

Conceptual basis of the modelling approach

Our modelling approach is based on the concept that landscape systems are not static but flexible in terms of service supply. Different levels of services can be provided and, after adjusting the spatial configuration of the landscape, the total supply of services can even be enhanced. This flexibility provides us information on the need to adapt the landscape in response to service demand. By explicitly incorporating maximal levels of service supply, insight is gained when societal demands cannot be met. This leads to, for example, unsustainable use of the landscape or needs for importing services from outside the system.

Another conceptual characteristic of our modelling approach is that many interactions and feedbacks take place along different spatial and temporal scales. Temporal interactions are included in the path-dependency of interventions which affect service supply and the selection of sites for land management in the future. Spatial interactions take place at the three interconnected levels, generally representing the supply, management and demand of landscape services (Figure 5.1). In this multi-level system description, service demand and decision making on adaptive land management, are defined at a regional level. However, demand for services is spatially variable within a region. Defining different stakeholders groups, and specifying their spatial distribution, would give a better representation of service demand and the related land management actions. Areas with the highest demand for landscape services are often not the areas having the highest capacity to provide this service. This discrepancy can lead to tension in planning management activities, especially when access to services is at stake (Chan et al., 2006; Tallis and Polasky, 2009). So, the selection of locations to manage service supply should ideally incorporate the location of the beneficiaries. The need to explicitly define the spatial distribution of the supply and demand for landscape service depends on the landscape function. An overall regional demand can be sufficient for landscape services that can relatively efficient be replaced or transported (i.e. agricultural goods or drinking water). However, land-based services, like recreation possibilities, should be provided at accessible locations for stakeholders (i.e. recreation opportunities need to be provided near residential areas).

In our conceptual system description, management decisions are defined to take place homogeneously over the region. But management actions to regulate landscape services strongly depend on individual decision making processes (Pfeifer et al., 2009; Valbuena et al.). This means that land management actions are likely be to spatially variable; decision makers at management unit level can opt for different management activities. Including spatial variability in service demand and land management choices would increase the complexity of the system description. Additionally, the linear relations within the multi-scale human-environment system are often more complex and include slow and fast variables, non-linear relations and abrupt or irreversible changes (Carpenter et al., 2009; Koch et al., 2009). Adding such complexity will better represent reality, but lowers the feasibility to translate the conceptual system into an operational modelling framework.

The combination of concepts of flexible service supply, multi-level interactions, temporal feedbacks, and multifunctional location, distinguishes the presented approach from the current land-use and ecosystem models (see overviews Kienast et al., ; Verburg et al., 2004; MA, 2005; Matthews et al., 2007; Renting et al., 2009).

Relevance for landscape service governance

Good governance of landscape service is challenging because of the mismatch between human and environmental scales (Cash et al., 2006). Many scientists therefore called for internalising and explicitly incorporating service supply needs into policies and practices (Kienast et al., ; MA, 2005; Cowling et al., 2008; Daily et al., 2009; Fisher et al., 2009; Turner II, 2010). However, so far only few studies have investigated the impact of land management actions and policy plans on the supply of multiple landscape services (Cowling et al., 2008; Carpenter et al., 2009; Paracchini et al.). The presented modelling approach contributes to the development of policy support models to explore management and spatial planning options. Such approaches could assess the effect of different scenarios of landscape development on future landscape service supply. Policy makers need to deal with a demand for a broad range of landscape services. Modelling tools can be used to explore the fulfilment of the future societal demand for landscape services of a region. Trade-offs in service supply can lead to social, economic or ecological conflicts between stakeholder groups and people at different locations (De Groot, 2006). By identifying, quantifying and visualising undesired trade-offs, policy support tools could help adjusting management plans.

Currently, the modelling framework is based on service supply quantities and is not explicitly based on the value of landscape services as in other studies (see for an overview e.g. MA, 2003; Turner et al., 2003; Zandersen and Tol, 2009). Monetary values are important for cost-benefit analyses. Even though economic data could be added to the current modelling framework, the economic assumptions on management costs and economic effects of dynamics of service supply and value need to be improved in order to make the modelling framework suitable for supporting policy decisions on an environmental, societal and economic basis.

Conclusions

The presented modelling approach is a first step to better identify, map and quantify dynamic patterns of multiple landscape functions and their service supply. So far, land-use models could capture decision making, feedbacks and spatial and temporal dynamics, while ecosystem–based spatial models addressed complex processes relating to multiple service supply. We combined both approaches by explicitly addressing the interactions and feedbacks between landscape service supply, demand, and land management actions of the multifunctional landscape. The presented modelling framework is an example of new innovative landscape modelling approaches, which include multiple uses of the land, and which have a potential for quantitative assessments of ecosystem services provisioning for policy discussions on landscape management.

Chapter 6

Discussion and conclusions

Mapping and modelling multifunctional landscapes

The overarching methodological challenge addressed in this thesis is to develop new methods to describe the current and future spatial variability of multifunctional landscapes (Figure 1.4). The methodologies presented in previous chapters can be subdivided into two parts. The first part includes methodologies to describe, quantify, value and map the current state of landscape functions and multifunctionality. These methodologies help answering research questions with respect to the location and 'productivity' of landscape functions and assist in determining interactions between landscape functions at multifunctional locations. The second part includes methodologies to assess the future changes in multifunctional landscapes based on quantitative and spatially explicit information on landscape functions. These methodologies address questions in relation to monitoring landscape services as a result of new landscape management strategies and the dynamics of multifunctional landscapes in space and time

The presented methodologies are discussed in this chapter based on this subdivision. The associated research questions are answered by pointing out the methodological contributions of this thesis in relation to other approaches that are described in the scientific literature. Then, it is discussed how these methodologies and studies on multifunctionality can contribute to sustainable landscape management. Finally, some perspectives for future research on mapping and modelling multifunctional landscapes are presented.

Quantifying and mapping landscape functions

Several studies have shown that landscape functions and multifunctionality are unequally distributed over the landscape (Naidoo and Ricketts, 2006; Troy and Wilson, 2006; Egoh et al., 2008; Naidoo et al., 2008). However, so far a general methodological framework to describe this spatial variation was still lacking. This thesis presents such a methodological framework and the presented applications can provide guidance in future research efforts to further quantify and map landscape functions and multifunctionality. The methodologies presented in this thesis address the quantitative relations between landscape characteristics, functions, services and values (Figure 6.1). This sequence embraces the most important relations for studying landscape functions in a spatially explicit manner (Hein, 2006; Tallis and Polasky, 2009; De Groot et al., 2010).

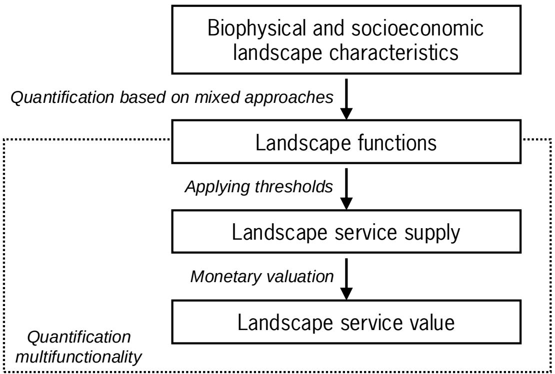

Figure 6.1 Methodological sequence for spatially explicit quantification of landscape functions, services and multifunctionality.

Overview of quantification and mapping approaches

Several studies have dealt with the question on how to quantify and map landscape functions and multifunctionality. The resulting maps of these studies are abstractions of reality. However, the way in which this reality is presented is subject to the choice of the methodological approach (Soini, 2001). In general, methodologies to (spatially) quantify the multifunctional character of the landscape focus on a single methodological approach. The quantification of landscape functions can be based on (i) ecological processes, (ii) empirical relations or (iii) expert knowledge, while the valuation of landscape functions can be based on a (i) economic valuation or (ii) social valuation of the provided services. In this section the strengths and weaknesses of these different methodological approaches are discussed for their suitability to quantify and map multiple landscape functions.

Methods that describe ecological processes of a landscape describe ecological indicators that represent landscape functions. Process-based relations with biodiversity, biophysical structures and processes of a landscape are used to quantify landscape functions (Kremen, 2005; Chan et al., 2006; Bartholomeus et al., 2008; Egoh et al., 2008). In this way, landscape functions are described based on value-free measures, such as tons of carbon sequestrated or cubic meters of water stored. Landscape characteristics that feed the process-based models have a spatial dimension. This spatial dimension allows for presenting the quantified landscape functions in maps. Maps that are based on process-based quantification methods, normally cover a small spatial extent, but few examples of large scale studies exist (e.g. Metzger et al., 2006; Metzger et al., 2008; Naidoo et al., 2008). As landscape functions are expressed in different service related units, the landscape function maps are standardised to be able to quantify the multifunctionality of sites. The spatial dimension of the underlying processes and process-based relations with the landscape make that quantifying landscape functions based on ecological models are very strong in quantifying and mapping biophysical landscape functions and services. A weakness of the ecological process-based methods is that they encounter difficulties when quantifying socio-cultural services like recreation and tourism (Kremen, 2005). Additionally, process-based methods require a high amount of input-data. This makes it challenging to use this quantification and mapping approach in large regional or continental studies. Also, additional factors that influence service supply will start playing a role when up-scaling process-based relations (Mander et al., 2005).

Landscape functions can also be quantified and mapped based on empirically derived relations with landscape components and processes. Empirical methods use statistical techniques to select and quantify landscape characteristics that explain observations on services supply of landscape functions (Diaz et al., 2007). So, contrary to process-based methods, empirical methods need observations on (proxies of) landscape functions to quantify the relation with landscape characteristics. The measurement unit of the observations of the landscape function determines the unit in which the quantification takes places (e.g. number of visitors, ton of crop produced). The found relations with these observations are spatially extrapolated to a larger region using the spatially continuous data on landscape characteristics. Here again, the units in landscape functions maps are expressed with different measures and are therefore standardised to quantify the multifunctionality of sites. The strength of empirical methods is that they can describe generalised complex relations without the need of including a precise understanding of underlying causal processes. Drawbacks, however, are that the observed relations do not necessarily indicate causality and empirical methods can only adequately describe what has been observed. The quantified relations are study area dependent, so extrapolation of the found relations introduces uncertainty.

Another approach of to spatially quantify landscape functions is the usage of expert knowledge and literature. Such information is used to develop general rules on linking landscape characteristics and landscape functions. Landscape functions are in this way normally quantified using a unit-less relative index, which is subsequently mapped (e.g. Kienast et al., ; Haines-Young et al., 2006; Pérez-Soba et al., 2008; Reyers et al., 2009). As landscape functions are quantified in indices, multifunctional sites can be quantified based on a direct aggregation of the different landscape functions. The strength of quantifying and mapping landscape functions based on such expert knowledge is that it is based on an easy to implement and parsimonious methodology. Because of the low-data requirements, landscape function maps based on expert defined spatial relations are mainly seen in large scale studies. However, this quantification method is based on generic relations and can therefore not take into account area-specific relationships. Additionally the defined relations can be prejudiced and expert dependent. These weaknesses introduce uncertainty in the final outcomes.

The quantification and mapping of landscape functions is normally based on one of these three methods. All methods have clear advantages and disadvantages that relate mostly to data availability, existing knowledge on the spatial distribution of landscape functions and the spatial scale of the study. In studies aiming at describing multifunctional landscapes, combining the three methods can improve mapping efforts. Because of the diverse character of the different landscape functions, the selection of quantification methodology should be based on the strengths of each approach and targeted at a specific landscape function. For example, biophysical landscape functions are best quantified using process-based methods, while expert knowledge can be best used to quantify and map socio-cultural functions that are determined by societal preferences. Empirical methods can be used to improve the expert based quantification and mapping efforts, when observations are present or as a second-best option when insufficient data are available for process-based methods. A combined use of these methods does require standardisation of the different landscape function units.

Besides quantifying landscape functions in terms of service supply (or expert based indices), quantified maps can be created based of the values of landscape functions. Valuation of landscape functions is generally speaking the domain of social and economic quantification studies. Most economic approaches value landscape functions, i.e. services, in monetary terms. The valuation can be based on the use and the non-use values of landscape services. Direct consumptive values, such as the current or future value of timber, fish or other resources, are included as use-values. The valuation can be extended by including non-use values that relate to the importance given to an aspect of the environment in addition to or irrespective of the use value, i.e. existence value (MA, 2003). Economic values need to be linked to landscape function maps in order to make them spatially explicit (Troy and Wilson, 2006; Nelson et al., 2009). These maps can directly be combined to quantify values of multifunctional locations, as all landscape functions are expressed in a single measure unit: money. Economic approaches are also often used in studies on the multifunctionality of agriculture (Groot et al., 2007; Van Huylenbroeck et al., 2007; Renting et al., 2009; Wilson, 2009). In this perspective multifunctionality refers to the fact that one economic activity (e.g. food production) can have different social, cultural and natural capital outputs (Van Huylenbroeck et al., 2007). The most important strength of quantifying landscape functions based on economic values is that information becomes available to directly support cost-benefit analysis. Additionally, economic valuations help in developing mechanisms that can compensate landowners for the services that their lands provide. However, there are limitations to the assessment of economic values. A drawback of quantifying landscape functions in economic terms, for example, is the difficulty to quantify the complex of supply-demand processes that determine economic values (Farber et al., 2002; Turner et al., 2003). These processes determine that as fewer services are being supplied or as the demand grows the value of the landscape service changes. Another drawback is that the economic valuation does not explicitly have a spatial dimension. This limits the mapping possibilities because of the difficulties of linking values to landscape characteristics that do have a spatial extent (Naidoo and Ricketts, 2006; Troy and Wilson, 2006; Grêt-Regamey et al., 2008). Sufficient contextual variation in the valuation studies is often lacking to link landscape value data to high resolution data on landscape services (Troy and Wilson, 2006). A third shortcoming is the challenge to unambiguously translate non-market values, such as aesthetics values, into monetary values. Besides these conversion difficulties, non-market values are also strongly based on individual preferences (Naidoo and Ricketts, 2006).

Valuation studies can also be based on qualitative socio-cultural measures. In these studies the description of landscape functions is normally based on stakeholder consultation. Information is gathered on people's motivations, perspectives, preferences and values (Soini, 2001; Brown, 2006; Alessa et al., 2008). Different people attach different values to services. From this viewpoint the mapping of landscape services is done by taking stakeholder perceptions and views as starting point (Parra-Lopez et al., 2008; Snep et al., 2009). Standardisation of social measures is needed to indicate the multifunctionality of the landscape (Alessa et al., 2008; Raymond et al., 2009). The strength of valuations based on social measures is the central focus on people. Social valuations support the research on motivations of socio-institutional entities, like land managers, in relation to decision making and management of multiple landscape functions (Renting et al., 2009). A drawback is the difficulty to make the measures spatially explicit. Participatory mapping is possible (Alessa et al., 2008; Raymond et al., 2009) but these mapping efforts cannot take into account the for the stakeholder invisible functions and function extents. Social valuations also centre on the socio-cultural functions like recreation and tourism, cultural diversity and identity (e.g. heritage value). Less obvious or indirect landscape services like biophysical landscape functions, frequently score low in social valuations (e.g. Raymond et al., 2009). Finally, the collection of data can be very time consuming and therefore costly (Cowling et al., 2008).

Economic and social valuations both make the importance of landscape functions for society explicit. Such an economic and social valuation adds extra information to the value-free measures of landscape function quantification in terms of service supply. As valuations typically are stakeholder dependent, clear information on the spatial distribution of values is often lacking. By linking the spatially explicit information on landscape service supply to service values, maps indicating landscape function values can be created. However, because of complex socio-economic processes that underlie valuations, a direct linkage might not be appropriate (Troy and Wilson, 2006; Spash, 2008). Assuming that landscape values are correctly represented in economic or social terms, ignoring this complexity can be a source of errors in the final landscape function maps. Unfortunately, an assessment of landscape service values including the complex socio-economic processes was beyond the scope of this thesis. Therefore, landscape function maps expressed in service supply in this thesis likely have a lower level of error propagation, while landscape function maps expressed in values illustrate the relative importance of landscape functions. However the quantification and valuation methods are complementary as they describe different dimensions and variations in space and time of landscape services.

So, the selection of quantification and valuation method determines the resulting descriptions, quantifications and mapping of landscape functions and multifunctionality. In this context, two additional methodological issues remain to be discussed, (i) the selection of landscape functions in relation to multifunctionality and (ii) the representation of the multi-level processes in quantifying and mapping of landscape service supply.

Landscape function selection in relation to multifunctionality

The selection and definition of landscape functions influences the identification of multifunctional locations. Ecological approaches have identified around twenty different functions of the landscape grouped in four categories: production, regulating supporting/habitat and cultural functions (MA, 2003; De Groot et al., 2010). Studies that focus on the multifunctionality of agriculture include in their definition of multifunctionality different economic activities on farms, like farm shops, on-farm care facilities and recreational facilities (Van Huylenbroeck et al., 2007). In this thesis the selection of landscape functions mostly relates to the functions as defined in the ecological approach but also includes more 'anthropogenic' functions (e.g. residential function) like in studies on the multifunctionality of agriculture. By including these latter functions the total service supply and number of landscape functions in peri-urban areas increases as compared with maps based on just ecological classifications, which are mainly based on the functions of natural areas. In this thesis, only eight landscape functions were selected that were mentioned in the regional spatial planning strategy (Reconstruction Act). If all defined landscape functions of the ecological classification would have been used, the multifunctionality of natural locations would inherently have increased.

The choice of unit in which landscape functions are defined is also subject to discussion. Some studies only measure direct benefits or the services relating to these benefits (e.g. Boyd and Banzhaf, 2007; Wallace, 2007; Wallace, 2008), while others also include intermediate and final services (e.g. Costanza, 2008; Fisher and Turner, 2008). This difference will affect the calculated total supply of services at multifunctional locations. In this thesis only the direct benefits expressed in services are quantified and mapped.

Multi-level processes

In the first part of this section, three methods to quantify and map service supply are presented. These methods relate service supply to spatial patterns of landscape components, which result from multi-scale processes. The spatial distribution and diversity of individual decision-making processes is not explicitly included in these mapping approaches. Information on land management, however, can additionally be considered to effectively capture multi-level processes that define landscape functions, as individual land managers influence landscape service supply (Yadav et al., 2008; Pfeifer et al., 2009; Piorr et al., 2009; Valbuena et al., 2010). For example, farmers manage a large collection of landscape elements (fields, margins, hedgerows, ponds etc.), which influence processes in the landscape at different spatial and temporal scales. Figure 6.2 illustrates landscape functions that are quantified and mapped based on a bottom-up land management and top-down landscape approaches. These maps relate individual management actions to landscape service supply. Figure 6.2a shows the percentage of farmers that participate in nature and landscape protection programmes (Pfeifer et al., 2009). This percentage is calculated per raster cell by aggregating the ten nearest farms. Aggregation of farm data gives an approximate of the ownership of land and was needed to protect the privacy of farmers. Figure 6.2b presents the top-down maps of the standardised plant habitat and cultural heritage functions (based on Chapter 2). Similar patterns between the maps can clearly be observed. Even though not all farmers had the option to enrol in protection programmes and not all land is owned by farmers so actions of other landholders are ignored, a relation between farm management and landscape functions seems present. Adequate data on individual land management are difficult to obtain, but when these data are available they can be used in empirical quantification and mapping efforts. Such efforts provide essential insights in the spatial patterns of the landscape multifunctionality.

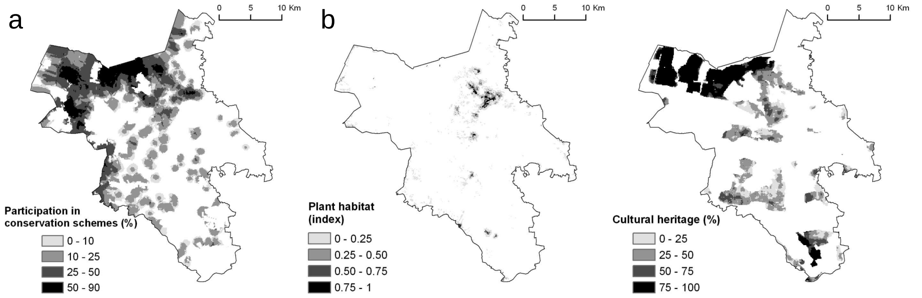

Figure 6.2 Bottom-up and top-down mapping approaches a) Extrapolation of farms participating in nature and landscape conservation programmes in 2005 (Pfeifer et al., 2009) b) Landscape functions maps as presented in Chapter 2.

Modelling changes in landscape functions

After the quantification and mapping of the current state of multifunctional landscapes, future changes can be modelled. Four methodological challenges to model changes in landscape functions are identified in Chapter 1. The first three all relate to the availability of quantified landscape functions and multifunctionality maps. The fourth challenge is that the landscape changes in the model should be driven by a societal demand and related management actions for both commodity and non-commodity landscape functions (Figure 6.3). These challenges resulted in methodologies that address questions in relation to monitoring landscape services as a result of new landscape management strategies and the dynamics of multifunctional landscapes in space and time

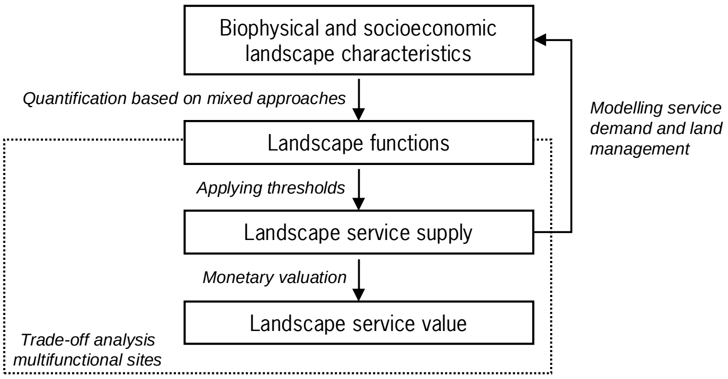

Figure 6.3 Methodological sequence for spatially explicit modelling of landscape functions, services and multifunctionality.

Overview of modelling approaches

Until recently, methodologies to measure changes of multiple landscape functions in space and time were scarce. However, research efforts on ecosystem and landscape service modelling are growing increasingly in numbers (e.g. Helming et al., 2008; TEEB, 2009, www.es-partnership.org). In general, modelling approaches to describe multifunctional landscapes can be subdivided in (i) location-oriented, (ii) market–oriented, (iii) actor-oriented or (iv) governance-oriented approaches (Caron et al., 2008; Renting et al., 2009). The first three modelling approaches strongly relate to the mapping approaches described in the previous section. The governance-oriented approach is assumed to be an integration of these three approaches. In this section the different modelling approaches are evaluated based on their general objectives and spatial and temporal modelling methods. In Table 6.1 the three modelling approaches and their characteristics are summarised.

Location-oriented modelling approaches are based on the land-unit concept from landscape ecology (Zonneveld, 1989). These modelling approaches quantify and map landscape (or ecosystem) services based on ecological process-based models (e.g. Metzger et al., 2006; Tallis et al., 2008b). So, location-oriented models describe multifunctional landscapes by ecological processes that result in biophysical functions. To capture all ecological processes fine scale data and analysis are required. As an illustration, Konarska and others (2002) showed that based on data at a 30-meter resolution the total value of landscape services in their study area was twice that as compared to a resolution of 1-kilometer. In location-oriented models temporal changes in landscape service supply are driven by changes in land management, climate and population. Location-oriented models are strong in addressing spatial and temporal aspects of landscape functions. However, these models are limited in their ability to include socio-cultural functions and have heavy data requirements. Additionally, human decision making processes are only included in an exogenous and aggregated form (Renting et al., 2009).

Actor-oriented models primarily describe decision-making processes. These models represent actors (individual or groups) which interact both with each other and with their environment, and make decisions and change their actions as a result of this interaction (Matthews et al., 2007). Actor-oriented models include information on the needs, values, norms and behaviours of individuals, institutions and organisations in an area (Cowling et al., 2008). An overview of change agents can be presented in a so-called actor framework, indicating interaction and responsibilities of the different stakeholders (Rindfuss et al., 2004). Actor-oriented models include social and non-economic processes that influence on decision-making (Matthews et al., 2007). In many studies related to multifunctionality, these actors are farmers. Based on farming styles, including the motivation and activities in social and market networks, management actions can be modelled in such way that temporal changes in multifunctional landscapes are mimicked (Van der Ploeg et al., 2009; Valbuena et al., 2010). For example, Yadav and others (2008) developed an agent-based model to simulate agricultural decision making and biochemical changes related to land fertility. Actor-based models are strong in representing the diversity of individual decision making, but are weak in describing multifunctionality at aggregated and regional scales because of their lack of spatial scale.

In market-oriented approaches, landscape functions are modelled based on multi-objective optimisation (Rossing et al., 2007). Market-oriented approaches, like actor-oriented models, mostly focus on farms. In this approach, activity and consumption choices are maximised by a farm household for expected utility (Pfeifer et al., 2009). The maximisation is subject to a set of constraints such as available budgets (also comprising farm profits), time, production technologies for commodity and non-commodity outputs (Jongeneel et al., 2008). Market-oriented approaches include economic models in which changes in optimisation, and therefore land management decisions, drive temporal changes in landscape functions. However, these models also mostly lack complete spatial coverage in regional studies, as the unit of analysis is normally farms. Additionally landscape functions without a market mechanism cannot be directly included in these models.

Because of the holistic aspect of multifunctional landscapes, most modelling efforts combine characteristics of the different modelling approaches (Wiggering et al., 2006). Such combinations can result in an integrated governance-oriented modelling approach. This approach aims at supporting policy planning and decision making by including location-specific ecological, social and economic processes. In an integrated approach interactions between society and their environment can explicitly be modelled. By including these interactions, trade-offs can be made explicit as a result of changes in the socio-economic states on ecological processes and vice versa. Socio-economic and ecologic processes take place at different spatial levels (Veldkamp et al., 2001; GLP, 2005; Overmars and Verburg, 2006). Therefore, an integrative approach should have a multi-level structure to capture all these processes. Temporal changes in landscape services supply are driven by changes in these processes or are defined in scenarios. In land-use change studies, different integrative models are present (see overviews by Verburg, 2006; Parker et al., 2008). Land-use models can describe complex feedbacks between society and environment. However these models are less suited to explicitly deal with the multifunctional character of a landscape and the different consequences of management decisions on the service supply of landscape functions (Pinto-Correia et al., 2006; Daily et al., 2009; Verburg et al., 2009). Currently, integrative modelling approaches to support governance dealing with multifunctional landscapes generally combine only two of the above-mentioned modelling approaches. For example, by including spatial aspects in actor and market-oriented approaches (Bockstael et al., 1995; Yadav et al., 2008) or by integrating a participatory valuation in location-oriented models (Gund Institute, 2009). Most integrative landscape function research, however, focus on combining location-oriented ecological production functions including some kind of market mechanism of landscape services (MA, 2003; Tallis et al., 2008b; Villa, 2009). Only a few studies consider social processes that affect the management of ecosystem services (Cowling et al., 2008).

The methodologies developed in this thesis aim to follow an integrative governance-oriented approach. As identified earlier, integrated approaches are currently lacking to support the application of the concept of landscape functions in planning, management and decision making (Cowling et al., 2008; Daily et al., 2009; Fisher et al., 2009; De Groot et al., 2010; Turner II, 2010). To develop a fully integrated approach, existing modelling approaches need to be improved in order to capture ecological processes, individual decision-making processes and market mechanisms. To avoid integrated models becoming complex models with unfeasible data requirements, selection of the core aspects of these processes is needed. This thesis presents a modelling approach in which simplified societal demand for services, multi-level interactions and management actions are combined to simulate changes in landscape service supply. This approach, therefore, presents a first step towards an integrated governance-oriented modelling approach.

Table 6.1 General methodological characteristics of the four modelling approaches to study and support management of multifunctional landscapes.

Characteristic	**Location-oriented**	**Actor-oriented**	**Market-oriented**
Input/output maps			
Unit of analysis	Pixel	Individuals, groups	Enterprise/Farm
Quantification measure	Provided services	Cultural values	Economic values
Modelling			
Objective	System description and understanding	System description and understanding	Optimisation of joint-production
Drivers of temporal changes	Ecological processes	Social processes	Supply-demand functions
Strengths	Spatially explicit	Decision making processes included	Market mechanism included
Limitations	Data availability, aggregated decision making, lack of social processes and preferences	Lack of spatial scale	Market-failure non-commodity services, lack of spatial scale
	Integrated governance-oriented		
Input/output maps			
Unit of analysis	Pixel, management unit		
Quantification measure	Multiple		
Modelling			
Objective	Facilitating and monitoring policy making, explorative and goal optimisation		
Drivers of temporal changes	Multi-level processes, Scenarios		

Methodological contributions and research findings

The assumption that biophysical and socioeconomic characteristics of the landscape define landscape service supply is strongly embedded in this thesis. The challenge in quantification and mapping efforts of multiple landscape services arises from a lack of knowledge on landscape functioning (i.e. the processes behind the service supply) and data availability. In Chapter 2 these relations are quantified using different methods. Key to all methods is an effective use and integration of data sources that describe multiple scales of the landscape. In this thesis existing spatial data on biophysical and socioeconomic landscape properties are used to quantify and map landscape functions. Even though landscape functions cannot be directly observed from the land cover, landscape functions strongly relate to indictors of land cover or its derivative, e.g. distance to a land cover (Chapter 2). However, the combination of land-cover characteristics together with socioeconomic and underlying biophysical factors explains better the spatial variation of landscape functions (Chapter 2). Land-cover data, as presented in the traditional land-cover based land-use maps, therefore remains an important source of information to define the spatial distribution of landscape functions. But, the spatial patterns of land cover, rather than the presence of land cover at a specific location, contributes most to spatial description of landscape functions.

Very few studies have aggregated spatial information on individual landscape functions to multifunctionality maps (Gimona and Van der Horst, 2007; Alessa et al., 2008; Nelson et al., 2009; Reyers et al., 2009, are the few exceptions). This thesis shows that multifunctionality is an important aspect of the landscape. In fact, 75% of our study area is multifunctional because of the spatial overlap of at least two of the eight studied landscape functions. Multifunctionality is promoted in policy making because the total benefits of a location as compared to mono-functional locations is assumed to be higher (Brandt and Vejre, 2004; De Groot, 2006). We observed indeed an increase in total service supply at multifunctional locations, however, at these locations the average service supply per landscape function decreases. Multifunctionality is in this thesis seen as an emergent property of landscapes arising out of the interaction and linkage between the environment and society (Haines-Young and Potschin, 2004). However, multifunctionality can besides a description also be seen as an objective. The normative value of multifunctionality can play a role in defining the pathway of rural development (Van Huylenbroeck et al., 2007; Wilson, 2008; Renting et al., 2009). In Chapter 5 the potential to explicitly identify these pathways is briefly touched upon.

The importance of identifying landscape characteristics to explain the spatial variation of landscape functions is discussed in Chapter 3. Indicators of landscape characteristics cannot only be used to map and quantify landscape functions, but these indicators can also be used to assess the potential of a location to contain multiple functions. Based on a comparison of these landscape characteristics, compatibility of different landscape functions at a single location can be indentified (Table 3.2). As spatially explicit data are used, locations at which possible interactions between landscape functions occurs can easily be mapped. This adds an innovative spatial dimension to the present studies on landscape function interactions and multifunctionality (Gomez-Sal et al., 2003; O'Rourke, 2005; Sattler et al., 2006).

The methodologies presented in the Chapters 4 and 5 are used to assess future states of multifunctional landscapes. Using the methodologies that describe the relations between landscape characteristics – landscape functions – landscape services (Figure 6.3), changes in landscape functions and services are assessed. The methodology in Chapter 4 describes an impact assessment of spatial policies, while the methodology in Chapter 5 provides insight in how the trajectory towards a specific policy objective could look like. Changes in land-use or management normally affect not just one specific landscape function but change multiple landscape functions (Schröter et al., 2005; Diaz et al., 2007; Carpenter et al., 2009). The methodological approaches as presented in the Chapters 4 and 5 both quantify and map trade-offs between landscape functions as a result of management actions. Making changes of landscape functions spatially explicit thus allows for identifying trade-offs in space. An example of this ability is shown in Figure 4.2. Even though a management action can lead to an overall increase in landscape service supply in a region, the service supply in some areas within that region can actually decrease at the same time.

Implications for sustainable landscape management

Landscape services are directly linked to human well-being (MA, 2003). Because of this explicit link, the concept of multifunctionality is generally perceived as a means towards the broad objective of sustainable development (Bastian et al., 2006; Clark, 2007; Renting et al., 2009). So, within the normative framework of sustainable development, the concept of multifunctionality can be used to explore how sustainability goals can be met. In the context of safeguarding the flow of landscape services, good governance plays an important role (Biermann, 2007; Daily and Matson, 2008). According to Cowling and others (2008) three phases are required to ensure landscape service supply in a dynamic but resilient social-ecological system. These phases include (i) assessment, (ii) planning and (iii) management of the landscape. In the assessment phase information on landscape service supply and stakeholder characteristics is collected. In the second phase planning strategies are defined that determine the pathway towards policy goals. These strategies are based on stakeholder needs and biophysical landscape constraints. The last phase focuses on the coordination and execution of management actions to protect key locations that provide landscape services and to ensure the flow of landscape service to the beneficiaries. In the context of these three phases, the presented methodologies in this thesis contribute mainly to the first two actions and thus create a necessary basis for the management phase. More explicitly, this is achieved by visualising landscape functions, by making the effects of human actions explicit and by creating an understanding of functions and dynamics of the landscape system.

Visualising the spatial distribution of landscape functions is needed to be able to adequately manage the landscape. As shown in Chapter 1, not all landscape functions are directly observable and are therefore generally excluded from the widely available land-cover maps. Making regional landscape service supply and values visible therefore contributes directly to the assessment phase of the landscape. Strategies and management actions can be put in place, when conflicting landscape functions are spatially overlapping. Regulations can in this regard be effective tools to mitigate landscape function losses by spatially separating conflicting functions. For example, in many countries drinking water extraction zones are strictly protected by numerous rules concerning land-use in these areas. Additionally, spatially explicit information on landscape service supply has improved the efficiency of payment schemes and subsidies to landowners for managing and maintaining the landscape functions (Barton et al., 2009).

In the process of planning sustainable development strategies, the effects of changes in the landscape on the supply of landscape services need to be explored (Kates et al., 2001). In order to avoid mismatch in governance actions, processes at both environmental and management spatial levels need to be considered (Cash et al., 2006; Ostrom and Nagendra, 2006). In this thesis different methodologies are presented to assess possible future states of landscape functions as a result of changes in multi-level biophysical and socioeconomic processes. The interactions of biophysical limits and social and economic values of a landscape indicate the 'sustainability choice space' in which the final management actions could take place (Potschin and Haines-Young, 2006a). In the planning phase, spatial models can also contribute to a better understanding of the relevant landscape processes. By visualising different futures and creating an understanding on landscape system functioning, models thus can be used as policy discussion tools (Haines-Young et al., 2006; Groot et al., 2007; Claessens et al., 2009).

Nowadays, the concepts of landscape functions and multifunctionality are increasingly being included in policy strategies (see e.g. FAO, 1999; OECD, 2001; EC, 2004; VROM, 2006). In addition many scientists have called for explicitly incorporating landscape functions into policies and practices (MA, 2005; Cowling et al., 2008; Daily et al., 2009; Fisher et al., 2009; De Groot et al., 2010; Turner II, 2010). In order to successfully transfer knowledge from science to society three issues play a role (i) credibility, (ii) salience, and (iii) legitimacy of the research to society (Cash et al., 2003; Tuinstra et al., 2006). Here, credibility relates to the scientific adequacy, salience deals with the relevance of the research to the needs of decision makers, and legitimacy reflects the perception that the generation of information has taken into account values and interests of all stakeholders (Cash et al., 2003). Based on these requirements, two main constraints of the presented methodologies in their ability to support landscape management can be identified.

First, the credibility of the research methods of this thesis cannot yet be defined, as the accuracy of the produced maps and model predictions have not been tested. Therefore the contribution of the presented methodologies and their outcomes to support sustainable landscape management remains uncertain. This is a major constraint that should be addressed with future research (see Future Research section).

Second, the legitimacy of the research presented in this thesis to policy makers could be enhanced by involving stakeholders in the definition and valuation of landscape services. The selection of stakeholders of different landscape functions is, however, challenging. Not all services are provided at the same location where the benefits are realised. Often a discrepancy exist between these locations and therefore stakeholder groups (Hein et al., 2006; Fisher et al., 2009). With the current global markets for agricultural products, stakeholder groups that provide landscape services could be located in a different continent than the beneficiaries. However these different stakeholder groups should be involved to address sustainable landscape management and comprehensively assess trade-offs (Giller et al., 2008).

Future research

Future research should address methodologies to validate (predicted) landscape function maps. Mapping methodologies can be tested for sensitivity to the selection of input data (like in Chapter 2), but this does not cover a complete validation. The validity of the resulting landscape function maps is hard to quantify for several reasons. First, not all landscape functions can be directly observed. Landscape functions have different spatial and temporal scales, which makes the collection of field observations for validation in some cases not feasible. For example, point observations of recreation do not describe the full spatial extent of the recreation landscape function (as this depends on the temporal aspects and surrounding landscape characteristics). This lack of adequate 'reality data' makes the use of spatial models or landscape proxies unavoidable and the validation of the function maps based on field samples complicated. Second, in contrast to biophysical landscape services, socio-cultural functions are stakeholder, location and time specific (Hein et al., 2006). This makes the validation of qualitative measures of, for example, cultural heritage and landscape aesthetics difficult. So far, very few studies have validated landscape function maps using independent data sources. Through validation the uncertainty of maps can be made explicit to policy makers (Heuvelink, 1999; Rae et al., 2007). A clear communication to possible end-users regarding the different dimensions of uncertainty could avoid misinterpretation of the maps (Walker et al., 2003; Janssen et al., 2005). Communicating these uncertainties can be done by presenting additional maps with ranges in which landscape services are likely supplied. Methods that can quantify the uncertainty and validity of landscape functions maps should therefore be further explored.

Additionally, a better and more explicit integration of ecological, social and economic processes is needed to describe and model landscape functions. This includes an improved stakeholder involvement in identifying and valuing landscape functions and better assessments of the underlying individual decision-making processes. Stakeholder involvement should not only focus on farmers, but also on other land owners. For example, companies which strongly depend and have an impact on landscape services, should also be included (Hanson et al., 2008). Next, including (simplified) ecological models could reduce uncertainties in the causality of empirical relationships. Furthermore market mechanisms should be explicitly included in the valuation. To fully integrate these processes, problems with matching information from different disciplines need to be overcome. Therefore additional work is needed to develop mechanisms that enable integrative use and exchange of disciplinary information (like in CFIR, 2004; Ostrom and Nagendra, 2006; Tapio and Willamo, 2009).

Furthermore the saliency of the selection and the definition of landscape services needs to be further explored in the perspective of regional policy support. To better match policy needs, more attention seems needed to landscape functions providing services that are directly being used by society. For example, 'food provisioning' is a very broad concept to be distinguished as a single service. Types of food produced and therefore the underlying processes and interactions can differ strongly per location. The same is true for the 'recreation function'. Many types of recreations and recreation needs are present and this landscape function strongly depends on personal preferences. Additionally, for integrated assessments of regional landscape dynamics, one can wonder if it is appropriate to only consider landscape *benefits*. To make an overall assessment of all trade-offs also the opposite of landscape services should be defined; landscape harms or damages (or dis-services, Zhang et al., 2007). Humans have domesticated and continue to change landscapes and ecosystems in ways that reduce exposure to natural dangers (e.g. predators of livestock, floods). Protecting against such harms has influenced and improved human well-being (Kareiva et al., 2007). Enhancing natural habitats to create a unique biodiversity that happens to be dangerous or otherwise harmful to humans creates additional conflicts. In the current framework of landscape or ecosystem services these trade-offs cannot be taken into account.

A final future research challenge relates to analysing changes in multifunctionality on a larger scale such as continents or the globe. Importing agricultural products creates opportunities for increasing the level of multifunctionality of rural areas but in the same time decreases multifunctionality elsewhere. In many regions in The Netherlands the mono-functional agricultural landscapes of the twentieth century are now being changed into multifunctional areas with less agricultural focus. The current decrease in agricultural area can partly be explained by an increased intensification of agriculture, but there is also a trend of an increased import of agricultural goods from elsewhere (CBS, 2008a). The opposite function-changes are occurring in areas where agricultural areas are rapidly expanding. Here multifunctional (semi)-natural landscapes are converted into mono-functional production areas (Foley et al., 2005). In the scope of global sustainability, these multi-scale and inter-regional trade-offs should also be made explicit.

Conclusions

The objective of this thesis was to develop methodologies to analyse and quantify spatial aspects of both landscape functions and multifunctionality and to model future landscape function dynamics. Based on the work presented in this thesis it can be concluded that:

1. Different landscape functions require specific methods and approaches for their spatial quantification, as result of inherent underlying processes and available data.
2. Based on the identification of landscape characteristics that spatially define landscape functions and their service provision, it is possible to indentify synergies and conflicts at multifunctional locations.
3. Assessments of changes in landscape services in both service supply quantities and monetary values lead to complementary information that contributes to a more comprehensive evaluation of landscape management strategies.
4. Dynamic multifunctional landscapes models should account for multi-scale interactions and feedbacks in relation to landscape service supply and societal demands, in order to realistically explore the dynamics of landscape functions in space and time.

Research conducted to quantify and model landscape services often lacks a spatial component. This thesis presents a first step in the necessary methodological development to map and model current and future spatial variability of landscape functions. Accounting for this spatial and temporal variation has a large potential to improve future landscape studies. The identification of the possible combinations of different landscape functions and the identification of potential conflicts between users, can lead to an improved planning of sustainable use of natural resources in a region. The presented research therefore contributes to the development of an integrated policy support approach, which aims at strengthening the sustainable management of landscape functions.

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Summary

Society benefits from a wide range of services provided through landscape functions. These landscape services include, for example, food and timber, fresh water supply, climate regulation, landscape aesthetics and recreational opportunities. To enhance the supply of some services, people selectively modify landscapes. A good example is the long history of agriculture in which people have converted natural areas into productions fields. These actions strongly improved the landscape function of 'food production', but decreased other functions like climate regulation, erosion protection, and the provision of genetic diversity. However, well-being of people depends on all landscape functions. Therefore effective management of all landscape functions is needed to safeguard landscape service supply to society. At many locations more than one landscape function is present. Especially on these multifunctional locations the choice of land management is important, since changes in the landscape will influence each of these landscape functions differently.

Landscape functions are unequally distributed over the landscape. In order to adequately manage landscape functions policy makers need to know where landscape functions are located and how much landscape services are being provided. The problem is that currently no maps are available that contain this information for complete regions. This relates to the lack of general methodological frameworks to map different landscape functions. Additionally, there is limited knowledge on where and how much landscape functions change as a result of interactions with each other and with their surroundings. The objective of this thesis is therefore to develop methodologies to quantify and map the current state and to model future changes of landscape functions. Although the general focus of this thesis is on the development of methodological approaches, the outcomes are expected to have the potential to support decision-making on landscape management. This potential is illustrated by applying the developed methodologies to data of the Gelderse Vallei region in The Netherlands. The Gelderse Vallei is selected as case study area because of its prominent agricultural character within the highly populated Netherlands. Policy makers of the Gelderse Vallei have to deal with multiple, and often conflicting, claims on land resourses.

The objective of this thesis is addressed in successive steps. After presenting the current state of knowledge on landscape functions in *Chapter 1*, a methodological framework to quantify and map landscape functions is presented in *Chapter 2*. This framework includes three methods that describe the quantitative relations between landscape characteristics, functions and services. Inherent to underlying processes and available data, different landscape functions require specific methods for their spatial quantification. By applying the methodological framework to the study area, we successfully quantified and mapped

eight selected landscape functions: residential use, intensive livestock husbandry, drinking water supply, information on cultural heritage, habitat provision for rare, endemic and indicator plant species, arable agricultural production, and attractiveness for overnight tourism and leisure cycling. By mapping landscape functions that are not directly observable from landscape, the landscape functions maps better represent the different benefits of the land than the traditional land-cover based maps. Additionally, the supply of landscape services is quantified and included in the maps. Therefore spatial variation of the 'productivity' in the region is also made explicit.

In *Chapter 3* we use the defined spatial relations between landscape characteristics and landscape functions to study why some locations are more multifunctional than others. Our results indicate that favourable biophysical and socioeconomic conditions as well as the interactions between landscape functions explain these differences. When different landscape functions are enhancing each other, multifunctional hot-spots emerge. We also related the number of landscape functions to the total service supply of multifunctional locations. For our study area we found that mainly locations with landscape functions that sub-optimally provide services are strongly multifunctional.

In *Chapter 4* we show how a set of policy measures can be evaluated by assessing changes in landscape functions. In policy making, cost–benefit analyses play an important role. Therefore changes in landscape functions are not only assessed based on an index related to the level of service supply but also on an estimation of the value of these services in monetary terms. By linking economic values to landscape functions maps, the relation between the spatial distribution of landscape service supply and their value can be explored. As the quantification and valuation methods describe different dimensions and variations in space and time of landscape services, they can be considered as complementary. For the study area, the evaluation of a set of regional development policies indicated a strong increase in services supply in rural areas while the strongest increase in value is expected in and around urban areas. It is also shown that even though a policy action leads to an overall increase in landscape service supply in a region, the service supply in some areas within that region can actually decrease at the same time.

Based on the insights gained in the previous chapters a modelling approach to analyse the dynamics in landscape service supply is proposed in *Chapter 5*. While the methodology in Chapter 4 is used to assess the impact of policies, the modelling approach in Chapter 5 provides insight in how the trajectory towards a specific policy objective could look like. The modelling approach relates shifts in regional societal demand for landscape services to local land management actions. Consequently, we explicitly include different spatial levels at which interactions and feedbacks occur between landscape service demand, land management and service supply. Modelling these multi-level interactions and feedbacks allowed for an exploration of landscape function dynamics in space and time for our study area. Additionally, the outcomes of the modelling approach visualised trade-offs between different landscape functions as a result of management actions.

To conclude, in *Chapter 6* the methodological contributions of this thesis are discussed in relation to other approaches and sustainable landscape management. As research conducted to quantify and model landscape services often lacks a spatial component, this thesis presents a first step in the necessary methodological development to map and model current and future spatial variability of landscape functions. The quantification and the improved understanding of landscape function interactions can help to design and evaluate spatial policies for multifunctional landscapes. The presented research therefore contributes to the development of integrated policy support approaches, which aim at strengthening the sustainable management of landscape functions. The presented applications can provide guidance in future research efforts to further quantify, map and model landscape functions and multifunctionality.

Samenvatting

Karteren en modelleren van multifunctionele landschappen

De maatschappij profiteert van een grote verscheidenheid van diensten die door landschapsfuncties geleverd worden. Deze diensten zijn onder andere de productie van voedsel en hout, de levering van drinkwater, klimaatregulatie, landschapsbeleving en recreatiemogelijkheden. Mensen gebruiken deze landschapsdiensten en veranderen het landschap dusdanig dat de levering van deze diensten versterkt wordt. Een goed voorbeeld hiervan is de lange geschiedenis van de landbouw waarin de mens de natuurlijke staat van het landschap heeft getransformeerd naar productievelden. Deze acties hebben ertoe geleid dat de voedselproductie sterk is gestegen maar tegelijkertijd zijn andere diensten zoals klimaatregulatie, bescherming tegen erosie en de diversiteit van genetische bronnen, sterk afgenomen. Het welzijn van mensen is echter afhankelijk van al deze landschapsdiensten. Daarom is een adequaat management van het landschap noodzakelijk om de toevoer van alle landschapsdiensten aan de maatschappij veilig te stellen. Op veel locaties in het landschap is meer dan één landschapsfunctie aanwezig. Juist op deze multifunctionele locaties spelen keuzes in landschapsmanagement een belangrijke rol, aangezien veranderingen in het landschap elke aanwezige landschapsfunctie op een andere wijze zal beïnvloeden.

Landschapsfuncties zijn ongelijk verdeeld over het landschap. Om het landschap goed te kunnen managen is het belangrijk om te weten waar en hoeveel landschapsdiensten er geleverd worden. Het probleem is dat er op dit moment geen kaarten zijn die deze informatie voor volledige regio's laten zien. Dit komt doordat geschikte methoden hiervoor ontbreken. Daarbij is ook de kennis beperkt over in hoeverre landschapsfuncties door elkaar en door hun omgeving worden beïnvloed. De doelstelling van dit proefschrift is daarom het ontwikkelen van methoden om de huidige en toekomstige staat van landschapsfuncties te kunnen kwantificeren en karteren. Hoewel de algemene focus van dit proefschrift ligt op het ontwikkelen van methodologische aanpakken, kunnen de uitkomsten van dit proefschrift mogelijk ook gebruikt worden om discussies over landmanagement te ondersteunen. Deze mogelijkheden worden geïllustreerd door toepassingen van de ontwikkelde methoden op een gebied in de provincies Gelderland en Utrecht, de Gelderse Vallei. We hebben de Gelderse Vallei uitgekozen als studiegebied, omdat deze regio een sterk agrarisch karakter heeft terwijl het in het dichtbevolkte Nederland ligt. De beleidsmakers van deze regio hebben hierdoor te maken met verschillende, vaak conflicterende, claims op land en bestaansbronnen.

De doestelling van dit proefschrift wordt behandeld in een reeks van opeenvolgende stappen. Ik begin met het beschrijven van de huidige kennis van landschapsfuncties in *Hoofdstuk 1*. In *Hoofdstuk 2* presenteren we vervolgens een methodologisch raamwerk over hoe men kaarten kan maken van landschapsfuncties. Dit raamwerk bevat drie verschillende methoden om landschapsfuncties te kunnen kwantificeren en karteren, afhankelijk van de onderliggende processen en de beschikbare data. Door het raamwerk toe te passen op het studiegebied is het gelukt acht geselecteerde landschapsfuncties te karteren: wonen, intensieve veehouderij, drinkwatertoevoer, cultureel erfgoed, aantrekkelijk landschap voor toerisme, habitat voor zeldzame en endemische planten, akkerbouw, en een aantrekkelijk landschap voor fietsrecreatie. Door alle landschapsfuncties te karteren, ook deze die niet direct te zien zijn in het landschap, laten de nieuwe kaarten beter dan de traditionele kaarten, die slechts gebaseerd zijn op de landbedekking, zien wat de verschillende kwaliteiten van het landschap zijn. Doordat ook de hoeveelheid van de geleverde diensten in kaart zijn gebracht, is er ook meer inzicht gekomen in de 'productiviteit' van de regio.

In *Hoofdstuk 3* gebruiken we relaties tussen landschapskarakteristieken en landschapsfuncties om te onderzoeken waarom sommige locaties meer multifunctioneel zijn dan andere. Onze resultaten laten zien dat zowel gunstige biofysische en sociaaleconomische omstandigheden, als de interacties tussen landschapsfuncties hierin een belangrijke rol spelen. Als verschillende landschapsfuncties elkaar versterken, ontstaan er zogenaamde hot-spots van multifunctionaliteit. We hebben in dit hoofdstuk ook gekeken naar de relatie tussen het aantal landschapsfuncties en de hoeveelheid geleverde diensten op multifunctionele locaties. In ons studiegebied bleek dat voornamelijk locaties waar de individuele landschapsfuncties weinig landschapsdiensten leveren, juist heel multifunctioneel zijn in termen van het aantal aanwezige functies.

In *Hoofdstuk 4* laten we zien hoe beleidsplannen geëvalueerd kunnen worden op basis van de verwachte veranderingen in landschapsfuncties. In beleid spelen kosten-baten analyses vaak een belangrijke rol. Daarom hebben we de veranderingen in landschapsfunctie niet alleen geschat met een index die de levering van landschapsdiensten laat zien, maar ook aan de hand van een schatting van de monetaire waarden van deze diensten. Door deze economische waarden te koppelen aan de landschapsfunctiekaarten, kan de relatie tussen de ruimtelijke verspreiding van landschapsdiensten en hun waarden onderzocht worden. Doordat de evaluatiemethoden verschillende ruimtelijke en temporele dimensies beschrijven, kunnen de methoden als complementair worden gezien. Voor ons studiegebied liet een evaluatie van een pakket van regionale beleidsplannen zien dat in de landelijke gebieden een grote toename van landschapsdiensten verwacht kan worden, terwijl de sterkste toename van de waarden van de landschapsdiensten rond de steden verwacht wordt. We laten in dit hoofdstuk ook zien dat ondanks dat beleidsplannen over het gehele gebied genomen tot een groei van landschapsdiensten zullen leiden, er binnen de regio gebieden zullen zijn waar de landschapsdiensten juist af zullen nemen.

Gebaseerd op de nieuwe inzichten uit de voorgaande hoofdstukken, presenteren we in *Hoofdstuk 5* een model waarmee de mogelijke dynamiek van landschapsdiensten geanalyseerd kan worden. In tegenstelling tot de methode uit Hoofdstuk 4, waarmee de effecten van beleid geschat worden, geeft het model uit dit hoofdstuk inzicht in hoe het traject tot een specifiek beleidsdoel eruit kan zien. Het model relateert veranderingen in de regionale vraag naar landschapsdiensten aan lokale landmanagementacties. Hierdoor houden we expliciet rekening met interacties die plaatsvinden tussen verschillende niveaus die de vraag naar landschapsdiensten, landmanagement en de levering van diensten door het landschap beïnvloeden. Door al deze interacties mee te nemen, kunnen we een schatting maken hoe de veranderingen in landschapsfuncties in ruimte en tijd plaatsvinden. De uitkomsten van het model laten ook zien waar en hoeveel de levering van landschapsdiensten afneemt als resultaat van landmanagementacties die gericht zijn op het verbeteren van andere landschapsfuncties.

Ten slotte worden in *Hoofdstuk 6* de methodologische bijdragen van dit proefschrift besproken in relatie tot andere wetenschappelijke methoden en duurzaam landbeheer. In onderzoek dat zich richt op het kwantificeren en modelleren van landschapsdiensten, ontbreekt vaak de ruimtelijke component. Dit proefschrift laat zien hoe methoden ontwikkeld kunnen worden om de ruimtelijke verspreiding van landschapsfuncties beter te kunnen karteren en modelleren. De kwantificatie en het verbeterde begrip van de interacties tussen landschapsfuncties kan helpen om ruimtelijk beleid voor multifunctionele gebieden beter te ontwerpen en te evalueren. Dit proefschrift draagt daardoor bij aan de ontwikkeling van methoden om geïntegreerd ruimtelijk beleid dat zich richt op het versterken van duurzaam beheer van landschapsfuncties te ondersteunen. De toepassingen van de methoden kunnen gezien worden als een richtlijn voor toekomstig onderzoek dat zich nader richt op het kwantificeren, karteren en modelleren van landschapsfuncties en multifunctionaliteit.

Samenvatting

Epilogue

In this thesis I have tried to find patterns and regularities to describe our landscape, continually realising that one will never be able to find that one formula that describes the complex world we are in. When I started my PhD research I expected to learn sophisticated analytical methods and complex theories and maybe even generating hard data on causal relations. If I can take the liberty to generalise my person experiences, I now think that obtaining a PhD isn't about difficult methods, theories or finding that one formula, it is all about skills. In the last years I was trained to think logically, write consistently, identify problems, and learn from mistakes. I did this for four years all by myself but not alone. Therefore I would like to thank everyone who contributed to this thesis.

First of all want to express my gratitude to my absurdly interdisciplinary group of colleagues I worked with during the last four years. You all proofed to be a valuable source of knowledge on agricultural practices, economics, GIS tricks, programming, statistics, soils, system dynamics, didactic methods, writing styles and administrative issues. Also I really appreciated your uncensored opinions on concepts, papers, presentations, politics, propositions and thesis lay-outs. Working on my PhD research wasn't lonely at all, thanks to you.

Next, I would like to acknowledge my team of supervisors for their initiative to combine land use modelling with the ecosystem services concept, and thank them for guiding me through all phases of growing-up in the academic world. During these phases I needed guidance, independence, discussion, time for personal development, and unfortunately for you, quite often all at the same moment. Peter Verburg, as daily supervisor you have been my first stop when getting into troubles. Thanks for your quick replies and ever to-the-point feedback. Lars Hein, you showed me around in the, to me new, world of environmental-economics. Thank you for your clear explanations, good listing and critical comments on my manuscripts. Tini, *Meneer*, van Mensvoort, besides being a supervisor you have been my mental coach during these years. I have really appreciated your unlimited capacity of putting things into perspective. Tom Veldkamp, the discussions with you helped me a great deal in framing my research and finding the crux of landscape function dynamics. Rik Leemans, thanks for the freedom you gave me, the introduction to the academic world and thorough reviewing of my work. Besides my supervisors, I would also like to thank Dolf de Groot for giving me the opportunity to collaborate in different projects and initiatives and for introducing me to the ecosystem services community.

In this thesis information from many different sources was used. I would like the acknowledge the people and institutes who provided data or expert knowledge; C. de Bont, S. Hennekes, E. Gies, J. Neefjes, Alterra, LEI, Provincie Gelderland, Provincie Utrecht, Kenniscentrum Recreatie, Stichting Vernieuwing Gelderse Vallei, and the municipalities of Amersfoort, Ede, Eemnes, Leusden, Putten, Soest and Wageningen.

My *paranimfen* deserve special thanks. Diego, from the beginning to the very end of this PhD journey you have been the best companion one could wish for. I hope our paths will keep on crossing. Erika, from Alora to the Aula it was great to have you as partner-in-creative fieldwork/ exotic voyages/ strange obsessions/ PhD-ing. What's next? And also thanks to Kathleen, for being a great all-in-one package deal and back-up.

Furthermore, I would like to thank my family, (too far away) friends and *amiguitos* for their support and pleasant distraction. My house mates did a great job listing and commenting on my daily PhD woes. Ruud, fijn dat ik bij jou heb kunnen afkijken hoe je dat nu doet, dat promoveren. Diana, bedankt voor je hulp bij het maken van de omslag van dit proefschrift.

En dan de man. Roel, het is goed te realiseren dat ik iemand heb die zo dicht bij me staat dat hij me alle ruimte kan geven. Bedankt voor het mee-lezen, mee-denken en mee-leven de afgelopen vier jaar.

Ten slotte wil ik mijn ouders bedanken. Zij creëerden de basis voor dit proefschrift. PaMa, grote dank voor jullie cruciale levenslessen in 'plezier te hebben in de zoektocht naar nieuwe kennis' en 'eeuwig te vertrouwen in *eigen-wijsheid*'.

About the author

Louise (Wieteke) Willemen was born on the 20th of June, 1979 in ‘s-Hertogenbosch, The Netherlands. After finishing her secondary education at the Sint-Janslyceum in ‘s-Hertogenbosch in 1997, Louise started her studies at Wageningen University, The Netherlands. With the modest ambition of ‘saving the world’, she enrolled in the BSc/MSc programme Tropical Land Use. Her major MSc thesis addressed land-use changes in northern Vietnam and was conducted at the Soil Inventory and Land Evaluation group in collaboration with the Plant Production Systems group. Being intrigued by the power of spatial analysis, she started her minor MSc thesis on uncertainty analyses of remotely sensed maps at the chair group of Geographic Information Systems and Remote Sensing. Louise did two research internships during her studies. First, she worked on a project on watershed dynamics at the Smithsonian Environmental Research Center in the United States, supervised by the Nature Conservation and Resource Ecology group. Her second internship on knowledge transfer techniques took place in Ecuador at the International Network for Bamboo and Rattan institute in collaboration with the Communication and Innovation Studies group. In 2000 Louise interrupted her studies for one year to work as a full-time board member in a commission responsible for organising the introduction period for first-year students at Wageningen University.

With this multi-disciplinary background Louise graduated in 2003. In the same year she was selected for the Associate Expert programme of the Dutch Ministry of Foreign Affairs. Within this programme she worked as a GIS expert at Bioversity International (formerly IPGRI) based in Cali, Colombia. For over two years Louise worked on agro-biodiversity issues in Latin America in relation to human-well being. Although she enjoyed her work in Cali, Louise decided to go back to university to pursue a PhD degree. In 2006 she started her PhD project at Wageningen University of which this thesis is the result.

From February 2010 onwards Louise works as a post-doctoral fellow at the Joint Research Centre of the European Commission in Italy. In her current research she focuses on the effects of land-use changes on ecosystem service supply in Africa.

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The SENSE Research School declares that Ms. Louise Lucia Johanna Maria, Willemen has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 42 ECTS, including the following activities:

SENSE PhD courses:

- Environmental Research in Context
- Research Context Activity: Organizing MSc course on Multifunctional Land Use
- Land Science: Bringing concepts and theory into practice
- Agent-Based Modelling and Natural Resources Management

Other Phd and MSc courses:

- PhD Competence Assessment
- Techniques for Writing and Presenting Scientific Papers
- Advanced Analysis with ArcGIS
- Executive education course on Sustainable Development Diplomacy

Research and Management Skills:

- Advisory and participation in the launching workshops of the development project 'Realizing the agricultural potential of inland valley lowlands in sub-Saharan Africa while maintaining their environmental services
- Organizing and activating the SENSE e-Network as on-line exchange community
- Symposium coordinator/chair: 'Ecosystem Services at a Landscape Scale' at the European International Association of Landscape Ecology Conference, Salzburg, Austria
- Panel member of a round table discussion on Cultural Diversity with Mr Kofi Annan, Wageningen, The Netherlands

Oral Presentations:

- Framing Land Use Dynamics II, April 2007, Utrecht, The Netherlands
- Landscape Ecology World Congress, July 2007, Wageningen, The Netherlands
- Conference on the Science and Education of Land Use: A transatlantic, multidisciplinary and comparative approach, September 2007, Washington DC, USA
- Conference of Impact Assessment of Land Use Changes, April 2008, Berlin, Germany
- European International Association of Landscape Ecology Conference, July 2009, Salzburg, Austria

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This research has been funded by the ‘Spatial development and society: integrating analytical and design approaches' programme of Wageningen University. This programme is a collaboration between the Wageningen graduate schools of Production Ecology and Resource Conversation (PE&RC), Mansholt, and Wageningen Institute for Environment and Climate Research (WIMEK).